Geospatially Enabled Modelling of Values and Decisions in Agro-ecological Landscapes

by

Juwairia Mahboob

MS (Remote Sensing and GIS)

(National University of Sciences and Technology)

A thesis submitted in fulfilment of the requirements for a Degree of

Doctorate of Philosophy at the School of Land and Food,

University of Tasmania

June, 2017

Declaration of Originality

This thesis contains no material which has been accepted for a degree or diploma by the University or any other institution, except by way of background information and duly acknowledged in the thesis, and to the best of my knowledge and belief no material previously published or written by another person except where due acknowledgement is made in the text of the thesis, nor does the thesis contain any material that infringes copyright.

Authority of Access

This thesis may be made available for loan and limited copying and communication in accordance with the Copyright Act 1968.

Statement of Ethical Conduct

The research associated with this thesis abides by the international and Australian codes on human and animal experimentation, the guidelines by the Australian Government's Office of the Gene Technology Regulator and the rulings of the Safety, Ethics and Institutional Biosafety Committees of the University.

Juwairia Mahboob

University of Tasmania

26 August 2016

Acknowledgements

All praise is due to Allah, the most Gracious, and the most Merciful, who aided me with eloquence and provided me with guidance to complete this research. Prophet Muhammad (Peace Be Upon Him) and his contributing prayers and exemplary lifestyle cannot be forgotten, that has guided me throughout my life and particularly in my research journey.

Pursuing a PhD project is just like climbing a high peak, step by step, accompanied with bitterness, hardships, frustration, small successes, encouragement, trust, and kind help from so many people. When you find yourself at the top of the peak enjoying a beautiful scenery, you realise that it was, in fact, teamwork that got you there.

My team consisted of myself and my diligent supervisors, who no matter where I was stuck, encouraged and motivated me to move forward. That encouragement and persistent push has led me to this day of completing my research. I wholeheartedly express my gratitude to my inspiring supervisors Dr Jon Osborn, Dr Jagannath Aryal and Associate Professor Rohan Nelson for their support and help throughout my research. I am grateful for the opportunity of being supervised by knowledgeable and passionate supervisors. I give my special thanks to my primary supervisor Dr Jon Osborn for accepting me as his PhD student and for offering his kind advice, patience and guidance throughout my PhD.

I would also like to thank my research advisors Dr Kathleen Broderick and Dr Richard Mount for their kind support during my PhD. I express my appreciation to all those organisations and individuals who offered me their time and support, specifically participants of research survey and staff members of Department of Primary Industries, Parks, Water and Environment, Greening Australia, Southern Midlands Council, Tasmanian Land Conservancy, Natural Resource Management North and Natural Resource Management South.

My acknowledgement is also due to Endeavour Scholarships and Fellowships Program for providing me the scholarship to carry out this research work at University of Tasmania, and their case managers for providing their support and assistance.

Finally yet importantly, I am grateful to all my family and friends who encouraged and supported me throughout the research duration.

Abstract

Effective resource management requires an understanding of the complex associations that occur between natural ecosystems, managed environments, and a variety of economic, social and political factors. Many aspects of resource management are location dependent. They are defined or affected by physical aspects of the immediate or proximal environment, by statutory or legal aspects of the immediate or proximal location, by economic values associated with location, or by relationships with a potentially large variety of policies that create rights, responsibilities, risks or opportunities associated with location, or with attributes that are associated with location.

This study explores the capacity to geospatially enable the representation, analysis and communication of values that inform decision making in complex agro-ecological landscapes. Integrated ecological economic modelling provides one framework in which to account for these various and complex interconnections and interactions. The research begins with a review of spatially enabled integrated ecological economic models to reveal the state-of-the-art and the knowledge gap. A spatially enabled model is then designed to close the knowledge gap and is implemented in software as a tool for application to the analysis of decisions that lead to changes in land use.

In order to demonstrate the utility of the model and its implementation, a model farm is constructed that exhibits a variety of land capability and land uses typical of a farm within the Tasmanian Midlands region, Australia. Historical land use records are used to develop land use change scenarios and a semi-structured stakeholder survey is used to collect stakeholder values with which to populate the model. Outputs from this case study show how complex spatial and a-spatial data can be integrated and modelled, to deliver valuable information, to communicate that information both quantitatively and visually in order to inform policy processes.

This study contributes to the development of models that can be used in complex decision-making settings and that have capacity to be catalysts for, and to inform and support, dialogue between interested parties, supported by spatial reasoning, and providing for socially, environmentally and economically acceptable solutions.

Contents

List of Figures

List of Tables

List of Charts

List of Abbreviations

ABM	Agent Based Modelling
AHP	The Analytical Hierarchy Process
ALUM	The Australian Land Use and Management Classification
ANN	Artificial Neural Networks
ARIES	Artificial Intelligence for Ecosystem Services
ASC	Australian Soil Classification System
CA	Cellular Automata
CAQDAS	Computer Aided Qualitative Data Analysis Software
CEcol	Conventional Ecology
CEcon	Conventional Economics
CI	Consistency Index
CLAC	Crown Land Assessment and Classification Project
CR	Consistency Ratio
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CVI	Conservation Value Index
DEM	Digital Elevation Model
DPIPWE	Department of Primary Industries, Parks, Water and Environment
DSS	Decision Support System
EE	Ecological Economics
EPM	Ecosystem Portfolio Model
ES	Ecosystem Services
ESP	Ecosystem Services Partnership
ESR	Ecosystem Services Review
ESV	Ecosystem Services Valuation
ESVD	The Ecosystem Services Valuation Database
EVRI	Environmental Valuation Reference Inventory
FCF	Forest Conservation Fund
GA	Greening Australia
GIS	Geographic Information Systems
GRB	Great Barrier Reef region of Australia
GUI	Graphical User Interface
IBRA	Interim Biogeographic Regionalisation for Australia

IDE	Integrated Development Environment
InFOREST	Investing in Forests
InVEST	Integrated Valuation of Ecosystem Services and Trade-offs
MA	Millennium Ecosystem Assessment
MBHP	Midlands Biodiversity Hotspot Project
MBHT	Midlands Biodiversity Hotspot Tender
MCA	Multi-criteria Analysis
MIMES	Multiscale Integrated Models of Ecosystem Services
NFVP	Non-Forest Vegetation Program
NRM	Natural Resource Management
PAPL	Protected Areas on Private Land Program
PFRP	Private Forest Reserve Program
RF	Random Forest
SDSS	Spatial Decision Support Systems
SEEA	System of Environmental-Economic Accounting
SEEM	Spatially enabled Ecological Economic Model
SERVES	Simple Effective Resource for Valuing Ecosystem Services
SMC	The Southern Midlands Council
SolVES	Social Values for Ecosystem Services
SRIAS	State-wise Resource and Information and Accounting System
TEEB	The Economics of Ecosystems and Biodiversity
TEV	Total Economic Value
TLC	The Tasmanian Land Conservancy
UNCBD	United Nations Convention on Biological Diversity
WAVES	Wealth Accounting and the Valuation of Ecosystem Services
WTA	Willing To Accept
WTP	Willing To Pay

1 Introduction

1.1 Preface and Background

Land capability and land use play a major role in shaping agro-ecological landscapes and influence the sustainability of farming lands, biodiversity, and the provision of ecosystem services in a landscape (Poppenborg and Koellner 2013). Decisions made by farmers play a pivotal role in shaping rural landscapes. These decisions are influenced by values that the farmers hold for their land (Hansen and Greve 2014, Lamarque *et al.* 2014). In agricultural landscapes, changes to land use are a consequence of decisions made by farmers, influenced by the values they hold for the land together with a variety of economic, socio-political and environmental considerations (Guillem *et al.* 2015).

Agro-ecological land management involves environmental decision-making at diverse levels, ranging from the individual decisions made by farmers to decisions made by organisations at the policy level (Kleindorfer 1999), with agro-ecological policies influencing farmers' decisions at the landscape level. Policy makers concerned with land use can influence the decisions of farmers and private landholders through the selection and application of policy mechanisms such as regulation, education, subsidies, or research (Pannell and Roberts 2009). Conversely the actions of farmers have capacity to influence political and policy agendas (Aerni *et al.* 2009, Celio *et al.* 2014). For example, if particular farming practices are shown to have detrimental effects on the landscape, then organisations with a capacity to set policy may respond in order to influence decisions. Thus, all stakeholders are involved in a cycle that is influenced by the actions and decisions of other stakeholders.

Policy makers are interested in understanding the ways in which decisions can be influenced. In the case of agro-ecological land management, policy makers are interested in how humans value and interact with their environment (Freudenburg 1999), and particularly in understanding the decision making process of farmers (Marsh and Smith 2000).

A large and rapidly growing body of research seeks to model the decision-making processes of farmers, and to ascertain the drivers of their decisions (e.g. Edwards-Jones 2006, Gibbons and Ramsden 2008, Lamarque *et al.* 2013, Seitz and La Torre 2014, Elsawah *et al.* 2015, Fowler *et al.* 2015, Guillem *et al.* 2015, Roberts 2015, Malawska and Topping 2016). Various methodologies including agent based modelling (Elsawah *et al.* 2015, Guillem *et al.* 2015, Malawska and Topping 2016), multi criteria analysis (Seitz and La Torre 2014), role playing games (Serrao *et al.* 2005, Lamarque *et al.* 2013), construction of mental cognitive maps (Elsawah *et al.* 2015) and participatory approaches (Edwards-Jones 2006, Lamarque *et al.* 2013, White and Selfa 2013, Roberts 2015, Guillaume *et al.* 2016, Voinov *et al.* 2016) have been utilised to model farmers' decision making processes. Some have attempted to combine biophysical models with decision models (Bergez *et al.* 2010), others have attempted to combine crop optimisation models with decision models (Fowler *et al.* 2015, Malawska and Topping 2016) and to focus on optimisation techniques. Only a few have taken into account spatial and temporal considerations (Gibbons and Ramsden 2008, Dury *et al.* 2013).

Much work is still needed to understand and interpret farmers' land-use decisions and to unwrap the complexity of decision-making processes (Wilson and Hart 2000, Karali *et al.* 2011) and to further develop tools that can inform this process.

Financial considerations drive many farming decisions, but they are not the only influences (Austin *et al.* 1996, Wilson 1997, Wilson and Hart 2000, Defrancesco *et al.* 2008). Decisions made in socio-economic systems may not primarily aim at profit maximisation (Beratan 2007) but depend on a complex web of values and factors, including economic, demographic, social, cultural, psychological, technological, biophysical and ecological (Austin *et al.* 1996, Wilson 1997, Wilson and Hart 2000).

A recent review of the use of the ecosystem services model in natural resource management (Plant and Ryan 2013) provides an insight into future trends in Australia. The study was based on a literature review, interviews and observations from Australian regional planning, and offers a snapshot of resource managers' experiences in engaging with the ecosystem services concept, highlighting factors that may have enabled or prevented participation in ecosystem services projects. One of the major barriers for adoption of the ecosystem services concept gathered through these interviews was the lack of an ecosystem services toolkits (Plant and Ryan 2013). Several studies have highlighted the management applications of the ecosystem services concept, however the lack of studies linking theory to practice has been identified (Seppelt *et al.* 2011, Plant and Ryan 2013). Recently efforts have been made, particularly in the area of developing toolkits that provide information on the value of ecosystem services, that can in turn support decision making for policy and management (Daily *et al.* 2009, Bagstad *et al.* 2013). Although some of these tools are available (Bagstad *et al.* 2013), there is little evidence of implementation at the local level (Plant and Ryan 2013), which suggests lack of awareness or possibly difficulty in access or implementation of tools. This indicates the need to not only focus on the utility of tools, but also on the means by which they can be implemented.

The environmental decisions faced by organisations are complex and it is to be expected that tools will be used in order to assist the decision-making process, particularly through systematically gathering and analysing information (English 1999). Environmental problems are all spatial in the sense that they are bounded by certain geographical boundaries and scale. Problems that address environmental externalities have, almost by definition, a strong spatial component (Tietenberg and Lewis 2015c). Agro-ecological land management problems are location specific and cannot be isolated from locality: spatial boundaries, spatial adjacency/proximity, and spatially associated attributes. Decision-makers need to understand the spatial context of a problem. Geographic Information Systems (GIS) have been shown to be a useful interdisciplinary tool that has capacity to support decision making through improved data integration, analysis and visualisation, and a capacity to make transparent assumptions that might otherwise go unexamined, and to facilitate forecasting (Couclelis 1991, Burrough and Frank 1995, Kliskey 1995, Wright *et al.* 1997, English 1999, Brodnig and Mayer-Schonberger 2000, Bateman *et al.* 2002, Tietenberg and Lewis 2015c).

Substantial work has previously been carried out to spatially present ecosystem services values using Geogaphical Information Systems (Troy and Wilson 2006, Chen *et al.* 2009, Häyhä and Franzese 2014, Sherrouse and Semmens 2014, Sherrouse *et al.* 2014). A variety of studies has been undertaken in different areas and at different scales (Chan *et al.* 2006, Naidoo and Ricketts 2006, Egoh *et al.* 2008, Maynard *et al.* 2010, Raudsepp-Hearne *et al.* 2010) that demonstrate how spatial mapping and modelling have potential to support decision making that involves ecosystems and human livelihoods (Cork *et al.* 2012).

For modelling agricultural land use dynamics, Lambin *et al.* (2000) proposed a spatially explicit and integrated modelling approach, suggesting that an improved understanding of land use change processes can be achieved by approaches involving multidisciplinary and cross sectoral modelling techniques. Lambin *et al.* (2000) emphasised the need for future land use models to consider the geographic and socio-economic context of the study including issues dealing with spatial and temporal scale. Following the approach of Lambin *et al.* (2000), Gibbons and Ramsden (2008) introduced a framework that involves integrating different modelling approaches that are spatially and temporally multi-scale, in their case in the context of farmers' adaptation to climate change. The framework includes farm structure and models farmers' decision making to improve understanding of important components of climate change adaptation at catchment level. Gibbons and Ramsden (2008), Dury *et al.* (2013) and Daloğlu *et al.* (2014) also highlight the spatial and temporal dimensions of farmers' decision making.

Among the few studies that have focussed on the spatial and temporal dimensions of farmers' decision-making process is Dury *et al.* (2013), who surveyed 30 farmers to study the dynamics of their decisions. Their study demonstrated the extent to which a cropping plan is a complex decision making process and does not emerge from a single decision. A formal representation of the spatio-temporal interactions of farmers was derived from the study, with the intent that it be useful in developing future cropping plan models at farm level. The capacity of geospatial models to integrate socio-economic variables is recognised (Lambin *et al.* 2003) together with the need for integrated spatial modelling approaches (Lambin *et al.* 2000).

Changes to land use are practical decisions that farmers make on a regular, often annual, basis. Yet there are few studies (e.g. Gibbons and Ramsden 2008, Dury *et al.* 2013) that investigate the decision-making process of farmers, and the factors that influence these decisions, in a spatially enabled manner. Many environmental decisions are inherently spatial, and yet many methods used to support them are not devised to accommodate the spatial aspects.

The multi-dimensionality of decision making has been expressed by the incorporation of ecological economics, as a field of research, into the study of decision making in these complex environments. Ecological economics is introduced in Chapter 2 of this thesis. Additionally, there is a need to better understand decision making through integration of ecological and economic systems within a geospatial framework.

This study endeavours to integrate ecological and economic values that influence farm land use decisions within a spatially enabled model and to develop a tool that is flexible and that analyses and presents information in a way that can assist in more-informed and better decision making.

1.2 Research Aim and Objectives

The aim of this research is to advance the development of spatially enabled integrated ecological economic modelling and, particularly, the capacity of such modelling to inform and influence decision making. The overarching research question for the thesis is: **How can geospatial capabilities be integrated into ecological economic modelling and applied to land use decisions in agricultural landscapes?**

The objectives of the research are to:

i. review existing methods of integrated ecological economic models, focussing on spatially enabled integrated models and their strategies

ii. develop a spatially enabled ecological economic model that is capable of modelling land use decisions made by farmers

iii. implement and assess the model using a case study drawn from the Midlands region of Tasmania, Australia.

1.3 Thesis Structure

This Introduction, Chapter 1 provides some background to the study, states the aims and objectives of the study, and outlines the structure of the thesis. Chapters 2, 3 and 4 present a detailed review of ecological economics and the utilisation of geospatial methods. Chapter 2 specifically addresses the history and concepts of ecological economics, to provide the foundation knowledge required by a non-specialist (such as the author). Chapter 3 concentrates on the history of spatial studies in ecological economics through a bibliographic review of published research over the last decade on the spatial valuation of ecosystem services. Chapter 4 provides a detailed review of spatial modelling strategies in the field of ecological economics, and describes the various tools, models and strategies currently prevalent in the spatial domain of ecological economics.

Chapter 5 (Case Study Area) describes the region used as a focus for this study – the Midlands region of Tasmania, Australia – detailing its characteristics and particularly the conservation initiatives and land use change patterns occurring in the region during the period 2001 to 2010.

Chapter 6 (Construction of Maryland Farm) explains why a model farm was constructed for the research, and details the procedure undertaken to construct the model farm.

Chapter 7 (Methods) presents the methods employed in the study. In this chapter the algorithm used as a basis of a spatial decision support system is described in detail, and the model inputs and outputs specified. This chapter explains the method developed to collect information from stakeholders and required as input by the model.

Chapter 8 (Data Assumptions and Model Calibration) addresses data assumptions made in the case of the model farm (Maryland Farm) together with calibration of the survey data into the Spatially enabled Ecological Economic Model (SEEM). This chapter provides a description of each of the implemented software interfaces.

Chapter 9 (Results and Discussion – Maryland Farm) describes the model outputs for the case study, and the results obtained from calibration of the SEEM model with the survey interview data obtained for the model Maryland Farm. The results are presented in the form of detailed and individual summary graphs. This chapter also presents and discusses the results obtained from semi-structured survey questions.

Chapter 10 (Discussion and Conclusion – Spatially Enabled Ecological Economic Modelling) highlights the model characteristics and model caveats. It summarises and reflects upon the findings of this research, and provides suggestions for future research directions. It also provides short concluding remarks on the study.

2 History and Concepts in Ecological Economics

Ecological economics provides a relevant and useful framework for positioning the modelling methods developed and tested in this thesis. This chapter reviews the concepts and theories of ecological economics, describing its history and primary characteristics. The field of ecological economics is illustrated by comparing it to conventional ecology and conventional economics as well as with the related fields of environmental economics and natural resource economics. The review represents the author's journey in understanding the field of ecological economics, and is intended as an introduction to the field for non-specialist readers. It is not intended to be a critique of ecological economics.[1]

2.1 Introduction

There was a dawning recognition in the late 1980s that conventional ecological and economic models were inadequate for solving global ecological problems (Costanza *et al.* 1991). The perceived need for a more integrated approach led to the emergence of a trans-disciplinary academic research area known as ecological economics. Ecological economics aims to address the interdependence and co-evolution of human economies and natural ecosystems over time and space (Xepapadeas 2008). Integrated ecological economic modelling recognises that there are often strong dependencies between ecological and economic variables, that independent models cannot reliably lead to optimum solutions that capture important intersystem feedbacks, and that integrated models can lead to insights and policy decisions that are not available from less integrated models (Settle *et al.* 2002, Tilman *et al.* 2005, Eichner and Pethig 2006, Wätzold *et al.* 2006, Tschirhart 2009). Ecological-economic modelling has demonstrated its capacity to be an effective method for integrating ecology and economics; for example with studies involving the design of cost-effective market-based instruments for the conservation of biodiversity (Drechsler *et al.* 2007).

Researchers from a variety of fields were involved in the formation of the trans-disciplinary field of ecological economics, with contributions from systems ecology, different strands of economics (biophysical economics, environmental and resource economics, agricultural economics, socio-economics), biology, natural sciences, physics and engineering, and general system theory (Røpke 2005). Developments in the theoretical as well as methodological dimensions of ecological economics can be traced from the inception of the International Society of Ecological Economics in 1988 followed by its first conference in 1990 (Røpke 2005a). The breadth of research disciplines contributing to ecological economics is considered one of its strengths (Røpke 2005a, Silva and Teixeira 2011).

[1] In this chapter, although ecological economics has been differentiated from environmental economics (Costanza et al. 1991), it is recognised that these two fields share many similarities and there are not distinct boundaries to differentiate them (van den Bergh 2001, Beder 2011b). Therefore, literature referred to in this chapter as part of ecologial economics may also fall within the domain of environmental economics.

2.2 The Advent of Ecological Economics

To trace the history of ecological economics, it is necessary to explore its emergence alongside broader economic thinking. In this section the primary concepts of economics, classical economics, neo-classical economics, environmental economics, and natural resource economics are described, together with the contribution they made to the formation of ecological economics as a new discipline.

'Eco' is derived from the Greek word 'oikos' meaning household. Ecology is the study of nature's housekeeping (Common and Stagl 2005), or more formally can be defined as the study of the relation of animals and plants to their organic and inorganic environments (Common and Stagl 2005). The term economics also derives from the Greek 'oikos' and is the study of housekeeping in human societies (Common and Stagl 2005) – how humans make their living and satisfy their needs and desires (Common and Stagl 2005) – although is more usually defined as for example the allocation of scarce resources among competing ends (Anderson 2010).

Economic writings date from earlier Greek, Roman and Arab civilizations. Notable writers from ancient times through to the 14th century include Aristotle, Xenophon, Thomas Aquinas and Ibn Khaldun (Kula 1998).

Gray (1931) in his writings about the development of economics during the period of the Greek empire predominantly refers to three prominent Greek writers: Plato, Aristotle and Xenophon. Greece was founded on slavery and a caste system and the types of work undertaken by slaves would fall into disrepute and become disrespectable. Plato considered division of labour as the basis of social organisation. Gray (1931) writes: 'When we turn to Aristotle, we are confronted with a writer who, by his analytical frame of mind, went further than any other thinker in antiquity in the direction of detaching a separate science of economics'. The third Greek writer who contributed to certain embryonic elements of economic science is Xenophon. Regarding Xenophon, Gray (1931) suggests that in a '... fumbling way, he approached or seems to have approached the "Law of Diminishing Returns"'.

The Romans contributed to economics through their renowned philosophers (Cicero, Seneca, Pliny among others), the agricultural writers (e.g. Cato, Varro and Columella) and the jurists (Gray 1931). With abundant natural resources, the Roman Empire ruled for centuries (Kula 1998). Slavery, which existed in earlier Greek times, continued as an institution through Roman times (Gray 1931).

Ibn Khaldun (1332-1406) was a renowned economist who wrote on aspects of political economy. Soofi (1995) writes: 'Ibn Khaldun's rationalistic approach to economic reasoning, his power of abstraction, and his pioneering work in developing economic models are unparalleled among the writers of medieval times. In fact, his theories bear striking resemblance to those later developed independently by Thomas Malthus and John Maynard Keynes, among others'.

2.2.1 Classical Economics

Economics as a distinct field of study, or its effective birth as a separate discipline, began in 1776 when Adam Smith (1723-1790) published 'An Inquiry into the Nature and Causes of the
Wealth of Nations' (Common and Stagl 2005). The book identified land, labour, and capital as the three factors of production and the major contributors to a nation's wealth, as distinct from the physiocratic idea that only agriculture was productive (Common and Stagl 2005). Smith was critical of the mercantilists[2] but described the physiocratic[3] system as perhaps the purest approximation to the truth with all its imperfections. Adam Smith's ideas about 'division of labour' and the self-adjusting market or the 'invisible hand', that produces the best possible outcome for society as a whole, and David Ricardo's principles of international trade are still influential concepts. At the time Smith wrote his book, the discipline was referred to as 'Political Economics', but is now known as 'Classical Economics' (Söderbaum 2000, Common and Stagl 2005).

Utilitarianism[4], proposed and advocated by Jeremy Bentham (1748–1832), had a significant impact on 19th century economics (Read 2004). Bentham defined utility as the tendency of an object or action to increase or decrease overall happiness (Read 2004). He proposed that people ought to desire those things that will maximise their utility, where positive utility is defined as the tendency to bring pleasure, and negative utility is defined as the tendency to bring pain (Read 2004). Over the last 100 years, the term utility has come to be understood as 'decision utility'[5] in contrast to Bentham's

[2] Mercantilism was an economic doctrine that flourished from the 16th to 18th century. It held that a nation's health depended on profitable trade with other countries and the accumulation of State wealth, particularly State owned metals such as gold and silver, and with government regulation used to meet these ends. Nations without access to mines could obtain gold and silver from trade only by selling goods abroad and restricting imports other than of gold and silver. The doctrine called for importing cheap raw materials to be used in manufacturing goods, which could be exported. Famous mercantilists include Bacon, Miselden, Hornick and Serra (Kula 1998).

[3] The Physiocrats, a group of 18th century French thinkers and writers, developed the idea of the economy as a circular flow of income and output. Physiocrats believed that only agricultural production generated a clear surplus over cost, so that agriculture was the basis of all wealth. Thus, they opposed the mercantilist policy of promoting manufacturing and trade at the expense of agriculture. The most famous of physiocrat writers were Quesnay and Turgot (Gray 1931). Physiocrats advocated replacing administratively costly tax collections with a single tax on income of land owners. In reaction against copious mercantilist trade regulations, the physiocrats advocated a policy of laissez-faire, which called for minimal government intervention in the economy (Kula 1998). The 18th century physiocrats were the first to explicitly recognise the importance of nature for the economy (Bergh 1996).

[4] Utilitarianism is a theory in normative ethics holding that the best moral action is the one that maximises utility (Sen 1979).

[5] Decision utility is the utility reflected in choices, or revealed preferences. Kahneman and co-authors distinguish between experienced utility and decision utility (Read 2004, Kahneman and Sugden 2005).

concept of utility, which is now often referred to as 'experienced utility' (Kahneman and Sugden 2005).

Classical economics was widely regarded as 'the dismal science' according to Thomas Carlyle, a 19th-century Scottish writer because it took the view, particularly associated with Thomas Malthus (1766-1834), that the long-run prospects of improving living standards were poor. This view was based on the assumed fixity (unchanging) of the supply of agricultural land, together with the propensity of the human population to grow in size.

2.2.2 Neoclassical Economics

The era of neoclassical economics began around the 1870s and is associated with the writings of Karl Menger, William Stanley Jevons and Leon Walras among others (Söderbaum 2000). While building on many of the classical economists' ideas about markets and international trade, these writers attempted to refine the theory and to express it in mathematical terms (Söderbaum 2000). Neoclassical economics largely ignored the relationship between "human housekeeping" and "nature's housekeeping". In the 1950s and 1960s, economists developed theories of economic growth in which the natural environment largely did not figure (Common and Stagl 2005) and which proposed that, given proper economic management, living standards could go on rising indefinitely (Common and Stagl 2005).

Value was connected with an individual's utility of commodity in use and not exclusively, as in classical theory, with the labour input necessary to produce that commodity (Söderbaum 2000). As part of this approach to economics, a principle of 'diminishing marginal utility' was formulated, which stated that a consumer who, for instance, considers how many slices of bread he or she may consume per day, is expected to enjoy the first slice of bread more than the ones that follow. At some stage the utility added by consuming an additional slice decreases (Söderbaum 2000). The concept of market equilibrium was introduced and elaborated by Alfred Marshall in the late 19th century. Thinking in terms of a balance or 'equilibrium' between the supply of and demand for a commodity became an important part of neo-classical theory. It is on ideas of this kind that present-day neo-classical economics is built (Söderbaum 2000). Neo-classical economics represents a more restricted approach towards studying economic phenomena without taking into account the natural environment (Bergh 1996).

In neoclassical economics, the contributions of Nicholas Georgescu-Roegen's (1906–1994) are also acknowledged for mathematical refinement of standard neoclassical economics in the areas of utility and consumer choice, production theory, input–output analysis, and development economics, although he is most noted for his contributions in the area of entropy and economics (Costanza, Norgaard, *et al.* 2014).

By 1950, the ideas of classical economics were taught to students of economics only as part of the history of the subject (Common and Stagl 2005). Modern mainstream economics builds on neoclassical economics but with many refinements that either supplement or generalise earlier analysis, such as econometrics, game theory, analysis of market failure, and imperfect competition.

2.2.3 Environmental Economics and Natural Resource Economics

At the beginning of the 1970s, neoclassical economists began to show renewed interest in the natural environment, which led to the two important sub-disciplines of environmental economics and natural resource economics (also called resource economics). Environmental economics mainly deals with the *economy's insertions into the environment*, and with problems of environmental pollution (Common and Stagl 2005). Natural resource economics concerns itself mainly with the *economy's extractions from the environment*, and with problems associated with the use of natural resources (Common and Stagl 2005). Costanza (1989) argued that environmental and resource economics covers only the application of neo-classical economics to environmental and resource problems.

2.2.4 Ecological Economics

Ecological economics is based on the idea that any study of how humans make their living has to include the study of the relations of the human animal to its organic and inorganic environment. For ecological economics, the study of economy-environment interdependence is foundational (Common and Stagl 2005). Allocation is the process of apportioning resources to the different goods and services. Neoclassical economics focuses on the market as the mechanism of allocating scarce resources. Ecological Economics takes the view that the market is only one possible mechanism of allocation (Daly and Farley 2011).

To understand sustainability using neoclassical economic reasoning, the distribution of resources between generations, or intergenerational equity, must be central. The conventional stance of neoclassical economists remains that economic growth will provide the conditions to resolve these inequities; however this is being increasingly questioned as inequality has increased over time (Costanza, Norgaard, *et al.* 2014). A growing view that economics must work within more sophisticated social, political, and ecological understandings is consistent with research emerging in ecological economics.

2.2.5 Ecological Economics Vs Main Stream Ecology and Main Stream Economics

Costanza *et al.* (1991) explores the similarities amongst, and differences between, mainstream disciplines and presents his findings as a comparison of ecological economics with conventional (mainstream) economics and conventional ecology, highlighting the intrinsic characteristics of ecological economics such as a dynamic systems view and co-evolutionary perspective of ecological economics (Røpke 2005a). Quoting from Costanza *et al.* (1991): 'Ecological Economics differs from both conventional economics and conventional ecology in terms of the breadth of its perception of the problem, and the importance it attaches to environment-economy interactions. It takes this wider and longer view in terms of space, time and the parts of system to be studied'.

Table 2.1: Comparison of Conventional Economics and Ecology with Ecological Economics

	Conventional Economics	Conventional Ecology	Ecological Economics	Explanation
Basic World View	**Mechanistic, Static, Atomistic** Individual tastes and preferences taken as a given and the dominant force. The resource base viewed as essentially limitless due to technical progress and infinite substitutability.	**Evolutionary, Atomistic** Evolution acting at the genetic level viewed as the dominant force. The resource base is limited. Humans are just another species but are rarely studied.	**Dynamic, Systems, Evolutionary** Human preferences, understanding, technology and organisation co-evolve to reflect broad ecological opportunities and constraints. Humans are responsible in understanding their role in the larger system and managing it sustainably.	Basic World View is similar to CEcol in which the resource base is limited and humans are just another specie, whereas, it differs from CEcol in the importance it gives to humans as a specie and its emphasis on the mutual importance of cultural and biological evolution.
Time Frame	**Short** 50 yrs max, 1-4 yrs usual	**Multiscale** Days to eons, but time scales often define non-communicating sub-disciplines	Multi-Scale Days to eons, multiscale synthesis.	Time frame, Space frame and Species frame off EE all tend to be broader than CEcon and are more similar to CEcol, but there is an explicit realisation of the need for integrated, multiscale analysis. This view is now being recognised by CEcol, but is completely being ignored by CEcon. CEcol all but ignores human, CEcon ignores everything but humans and EE tries to manage whole system and acknowledges interactions between humans and the rest of nature.
Space Frame	**Local to International** Framework invariant at increasingly spatial scale, basic units change from individuals to firms to countries	**Local to Regional** Most research has focused on smaller research sites in one ecosystems, but larger scales have become more important	**Local to Global** Hierarchy of scales	
Species Frame	**Humans Only** Plants and animals only rarely included for contributory value	**Non-Humans Only** Attempts to find “pristine” ecosystems untouched by humans	**Whole Ecosystem Including Humans** Acknowledges interconnections between humans and rest of nature	
Primary Macro Goal	**Growth of National Economy**	**Survival of Species**	**Ecological Economic System Sustainability**	Macro Goal of EE is sustainability, CEcol goal is similar to sustainability but confined to single species and not the whole system, whereas, goal of CEcon is growth rather than sustainability.
Primary Micro Goal	**Max Profits (firms)** **Max Utility (indivs)** All agents following micro goals lead to macro goal being fulfilled. External costs and benefits given lip service but usually ignored.	**Max Reproductive Success** All agents following micro goals lead to macro goal being fulfilled.	**Must Be Adjusted to Reflect System Goals** Social organisation and cultural institutions at higher levels of the space/time hierarchy ameliorate conflicts produced by myopic pursuit of micro goals at lower levels.	EE is unique in acknowledging the two-way interdependencies between the micro and macro levels.
Assumptions About Technical Progress	**Very Optimistic**	**Pessimistic or No Opinion**	**Prudently Sceptical**	CEcon believes in unlimited resources to continued economic growth, CEcol tends to ignore humans altogether, but has a view that humans are resource limited. EE assumes that technology will not be able to remove resource constraints to be on the safe side.
Academic Stance	**Disciplinary** Monistic, focus on mathematical tools	**Disciplinary** More pluralistic than economics, but still focused on tools and techniques. Few rewards for integrative work.	**Trans-disciplinary** Pluralistic, focus on problems.	EE states that tools are important but secondary to the goal of solving critical problems.

Source of the table: Modified from Costanza et al. (1991)

Costanza *et al.* (1991) compare conventional ecology (CEcol) and conventional economics (CEcon) with ecological economics (EE) in Table 1, highlighting their views of the major differences between ecological economics, conventional ecology and conventional economics. Similarly, a study by Gowdy (1991) explores the relationships between mainstream and integrative approaches to economics, with a focus on bio-economic and post-Keynesian economics. It is now generally acknowledged that integration of disciplines is essential because of the associations and relationships that exist between ecological processes, economic activity and human values (Reagan 2006).

2.2.6 Environmental Economics and Ecological Economics

Some experts argue that researchers dealing with the role of economy and the environment have settled in two camps over the past decade, namely ecological economics and environmental economics (Campiglio 2011; Tietenberg & Lewis 2015a) and note the tendency for these two camps to develop their own journals and organise their own conferences. While the two fields share many similarities, ecological economics is consciously more methodologically pluralistic and likely to criticise the reductionaist approach adopted by environmental economics, whereas environmental economics is more deliberately based on the foundations of standard neoclassical economics, which emphasises maximising human welfare and using economic incentives to minimise human destructive behaviour (Tietenberg and Lewis 2015a). Much work has been carried out to distinguish and differentiate these two disciplinary approaches by authors such as Van den Bergh (2001), Illge and Schwarze (2009), Beder (2011), and Nadeau (2011).

2.2.7 Ecological Economics

In the last three decades of the twentieth century it became increasingly evident that human economic activity was having detrimental effects on the natural environment. This led to establishment of the International Society for Ecological Economics in 1988, demonstrating a conviction by researchers of various disciplines to study economy-environment interdependence (Common and Stagl 2005). The inception of ecological economics is described by Petit & Vivien (2015) as follows: ‘Ecological economics essentially grew out of economists working in the environmental field and growing dissatisfied with the way that standard economics saw interactions between nature and societies and ecologists anxious to take human activities (including economic) into account in a much more direct way, within the dynamic of the ecosystems on which they depend.’

Ecological economics is co-evolutionary (Holling 1978, Kallis and Norgaard 2010) and transdisciplinary in nature (Costanza 1991, Common and Stagl 2005, Norton and Noonan 2007) and is intended to address the relationships between ecosystems and economic systems in the broadest sense. These relationships are central to many of humanity’s current problems and to building a sustainable future but many argue are not well covered by other existing scientific disciplines (Costanza 1991). Costanza *et al.* (1991) in his view reflects that ecological economics attempts to focus more directly on the problem, rather than the tools and models used to solve them, by ignoring the academic boundaries.

2.3 Characteristics of Ecological Economics

Costanza (2003) describes ecological economics as a meta-paradigm which, rather than adopting a single discipline, seeks to allow broad, pluralistic viewpoints and models. Therefore, ecological economics adopts an integrative approach towards modelling real world problems and is co-evolutionary in nature, as ecology and economics are inseparable and need to be understood together. Although ecological economics as a discipline may be contested, its intentions provide a strong platform for the work reported in this thesis. The characteristics of ecological economics are that it is: transdisciplinary, pluralistic, integrative and focused on problems rather than on tools (Costanza *et al.* 1991). These characteristics are explained in the following sections.

2.3.1 Transdisciplinary

The prefix 'trans' means across, over, beyond, on the far side of, through. 'Trans' has disparate features and differs in meaning from 'multi' (many, more than two) and 'inter' (among, between, mutual, mutually) (Common and Stagl 2005). Transdisciplinarity means to reach out beyond science and to include aspects of practical contexts and values or normative judgements (sustainability, good-practice), as well as to feed back results into practical (politics, management) actions (Baumgartner *et al.* 2008). Transdisciplinary research is issue-oriented, and ideally involves stakeholders as well as scientists from relevant disciplines. To say that ecological economics is transdisciplinary does not only mean that it is concerned with ecological and economic phonomena, but also that there are phenomena that cross, or are beyond, the disciplinary boundaries (Common and Stagl 2005). Ecological economics, being transdisciplinary, means that it is intended to go beyond the normal conceptions of scientific disciplines and to attempt to integrate and synthesize many different disciplinary perspectives. It is intended to ignore the 'arbitrary intellectual turf boundaries' (Costanza *et al.* 1991).[6]

2.3.2 Pluralistic

Costanza (1989) and Norgaard (1989) both argued for conceptual and methodological pluralism and new ways of thinking about the linkages between ecological and economic systems in Ecological Economics (Söderbaum 2000), given that there is unlikely to be a 'best' methodology when dealing with complex issues, that no single insight can be expected to lead to an optimal solution, and that there will be emergent methodologies and so the choice of methods should not be bounded. Norgaard's view of complex systems being better understood by multiple methodologies is supported in a study by Niraj, Dayal & Krausman (2010) through an application of the concept of methodological pluralism to wildlife and economy. Norton and Noonan (2007) provide a note of caution that, instead of embracing a pluralistic approach, it is too often the case that a more monoistic approach is used to value ecosystem services with contingent pricing that

[6] These principles are sufficient for the puposes of this thesis, but it needs to be noted that in continuing debates some researchers (e.g. Sarewitz 2010; Spash 2012) assert there are limits to the disciplinary boundaries of ecological economics and that transdisciplinarity can not be advocated, arguing the field is holistic in nature and that holistic approaches to scientific inquiry are inherently interdisciplinary with familiar limits of disciplinary science (Sarewitz 2010).

monetises values implemented by ecological economists and a quantitative analysis of non-market values.

In contrast, some argue against methodological pluralism (Bergh 1996, Spash 2012). For example, Spash (2012) suggests that methodological pluralism harms ecological economics more than it serves it (Petit and Vivien 2015) because having an unstructured knowledge base, and no accepted general framework or methodology weakens the identity of the field (Røpke 2005a, Faber 2008). Spash (2012) quotes from Dow (2007) that 'There is a limit to how far there can be plurality of understandings of the nature of reality, approaches to knowledge, and meaning, when knowledge needs to be developed within groups of researchers and communicated to others. Plurality in practice cannot be infinite.'

2.3.3 Integrative and Focused on Problems

Ecology is inherently cross-disciplinary, drawing together and integrating many types of information to address questions about the natural world (Madin *et al.* 2008). Quoting from Jopp *et al.* (2012) 'Ecological modelling makes use of a wide variety of techniques imported from various sources, among which there are numerous mathematical methods, but also techniques from computer science and operations research. In addition, systems theory, quantitative methods from geography, and methods from a variety of other fields have helped supply formal methods to solving ecological problems.' Ecological models can be applied as integrators between disciplines (Muller *et al.* 2012).

2.4 Ecosystem Services

Over the past three decades, the concept of ecosystem services has contributed very significantly to the development of ideas linking ecosystems, biodiversity, economics and human well-being (Gómez-Baggethun *et al.* 2010, Plant and Ryan 2013). The concept of human-environment interactions and the effect of the environment on human welfare can be traced from Roman writings on the increase in population, and from Plato (400 BC) who understood that deforestation could lead to soil erosion and eventual drying of springs (Daily 1997, Fisher *et al.* 2010). Baveye, Baveye & Gowdy (2013) describe attempts carried out in the late 1950s to assign monetary values to ecosystem services (e.g. Osborn (1948), Vogt (1948) and Leopold & Schwartz (1989)). In 1977, Westman published a paper in Science observing the link between ecological and economic systems (Westman 1977), although it is widely accepted that the term ecosystem service was later coined by Ehrlich & Ehrlich in 1981 (Liu *et al.* 2010, Baveye *et al.* 2013).

In the late 1970's, a seminal work on adaptive environmental management was published by Holling (1978). Adaptive management was introduced as an approach to managing complex and changing systems, and to a greater extent than before the vision was introduced of involving the broader public – interested in environmental issues – in monitoring and managing the environment. The work redrew conventional boundaries by integrating science and management in a dynamic view of adaptive environmental management (Holling 1978, Costanza, Cumberland, *et al.* 2014).

A substantial amount of research was undertaken during the 1990's on ecosystem services, devising methods to estimate their economic value (Costanza 1989, 2001, Bergh and Nijkamp 1991, Bockstael *et al.* 1995, Bergh 1996, Daily 1997, Daily *et al.* 1997, Perrings 1997, Costanza *et al.* 1998, Voinov *et al.* 1999, Turner *et al.* 2000). The following paragraphs illustrates how the idea of ecosystem services was brought into mainstream thought and practice through contributions of Millennium Ecosystem Assessment (MA), United Nations Convention on Biological Diversity (UNCBD) and The Economics of Ecosystems and Biodiversity (TEEB)[7].

In the early 2000's the concept of ecosystem services found its way into the policy arena, with a milestone laid by the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005), which constituted the foundation for ecosystem services in policy regimes. The need for valuation of ecosystem services had been voiced internationally under diverse platforms, notable amongst which were the United Nations Convention on Biological Diversity (UNCBD) and The Economics of Ecosystems and Biodiversity (TEEB). The national and international platforms on climate change (e.g UNFCCC) and carbon accounting had also promoted valuation of ecosystem services worldwide. Research on ecosystem services and their relationship with economics has continued to progress (Gómez-Baggethun *et al.* 2010, Plant and Ryan 2013). Several studies have highlighted management applications of the ecosystem services concept. An identified gap between theory and practice (Seppelt *et al.* 2011) was bridged by the introduction of a conceptual framework and methodology for the UK National Ecosystem Assessment (Bateman *et al.* 2011) and its implementation in various case studies (Dugdale 2010, Bateman *et al.* 2014, Sen *et al.* 2014).

2.4.1 Millennium Ecosystem Assessment – Contributions and Proposals

The Millennium Ecosystem Assessment (MA) conducted under the auspices of the United Nations aimed 'to assess the consequences of ecosystem change for human well-being and to establish the scientific basis for actions needed to enhance the conservation and sustainable use of ecosystems and their contributions to human well-being' (Millennium Ecosystem Assessment 2005). The MA advanced a powerful vision for 'a world in which people and institutions appreciate natural systems as vital assets, recognise the central roles these assets play in facilitating human beings and routinely incorporate their material and intangible values into decision making' (Daily *et al.* 2009).

The MA contributed substantially to bringing forward the ecosystem services concept as a policy tool and presenting a holistic research approach, proposing to integrate ecological, economic and institutional perspectives to produce insights into ecosystem impacts on human well-being and vice versa (Seppelt *et al.* 2011). The MA also highlighted the significance of ecosystem services provision to biodiversity conservation (Haslett *et al.* 2010). The momentum that followed from the MA resulted in research publications addressing the theoretical basis of ecosystem services, and involving both ecological and economic perspectives. The MA provided a standard definition of

[7] TEEB was launched by Germany and the European Commission in response to a proposal by the G8+5 Environment Ministers in Potsdam, Germany in 2007, to develop a global study on the economics of biodiversity loss.

ecosystem services and proposed a common international classification scheme for ecosystem services.

However, the MA did not deliver a fully operational method to implement the concept, which resulted in a range of studies with widely differing aims and producing difficulty in practically applying the concept (Seppelt *et al.* 2011).

2.4.2 Ecosystem Services, Structure and Function – Definitions and Concepts

An ecosystem has been defined as 'the dynamic complex of plant, animal and micro-organism communities, and the nonliving environment interacting as a functional unit'. Ecosystems range from as small as a tree or pond to as large as a natural forest. They can be intensively managed agricultural land or an urban area, or a mixed pattern of human use landscapes. Humans are an integral part of ecosystems (Millennium Ecosystem Assessment 2005, Williams 2009).

An ecosystem is composed of two components: ecosystem structure and ecosystem function. 'The structural elements of an ecosystem act together to create a whole that is greater than the sum of the parts. These emergent phenomena in ecosystems are known as ***ecosystem functions*** and they include energy transfer, nutrient cycling, gas regulation, climate regulation, and the water cycle' (Daly and Farley 2011). 'The individuals and communities of plants and animals of which an ecosystem is composed, their age and spatial distribution, and the biotic resources present are termed as ***ecosystem structure'***. The elements of ecosystem structure interact to create ecosystem functions as emergent properties of a complex system and in turn those functions provide services (Daly and Farley 2011).

Numerous definitions for ecosystem services exist in the literature (Costanza 1991, Daily *et al.* 1997, Costanza *et al.* 2002, Millennium Ecosystem Assessment 2005), however, the most widely cited is the MA definition, describing ecosystem services as 'the benefits that people obtain from ecosystems' (Millennium Ecosystem Assessment 2005, Williams 2009, Morse-Jones *et al.* 2011, Seppelt *et al.* 2011).

A more elaborated definition is presented by Troy & Wilson (2006) in which direct and indirect benefits are included as ecosystem services. Their definition states that 'ecosystem services are the benefits people obtain either **directly or indirectly** from ecological systems'. Others have described ecosystem services as goods and services that are of 'value' to human beings in order to allow their valuation in monetary terms (Spash 2008, Daly and Farley 2011). The Economics of Ecosystems and Biodiversity (TEEB) goes a step further and refines the definition of MA by defining ecosystem services as 'the direct and indirect contributions of ecosystems to human well-being', thus distinguishing between services and benefits and clarifying that services can benefit people in multiple and indirect ways (Fisher *et al.* 2010).

Table 2.2: Classification of Ecosystem Services Proposed by The Economics of Ecosystems and Biodiversity (TEEB) as compared to Costanza *et al.* (1997) and de Groot *et al.* (2002), MA (Millennium Ecosystem Assessment) (2005), Daily *et al.* (2009)

Various sources*	MA (Millennium Ecosystem Assessment) (2005)	Daily et al. (2009)		TEEB Classification
PROVISIONING	**PROVISIONING**			**PROVISIONING**
Food (fish, game, fruit)	Food	Seafood, game	1	Food
Water availability [RS] **	Fresh water		2	Water**
Raw materials (e.g. wood)	Fibre	Timber, fibers	3	Raw materials
Fuel & energy (fuel-wood, organic matter, etc.)	„ ?	Biomass fuels		
Fodder & fertilizer	„ ?	Forage		
Useful genetic material,	Genetic resources	- industrial products	4	Genetic resources
Drugs & pharmaceutical	Biochemicals	Pharmaceuticals	5	Medicinal Resources
Models & test organisms	- ?	Industrial products		
Resources for fashion, handicraft, decorative, etc.	Ornamental resources	?	6	Ornamental resources
REGULATING	**REGULATING**	**-**		**REGULATING**
Gas regulation/ air quality	Air quality regulation	Air purification	7	Air purification
Favourable climate (incl. C-sequestration)	Climate regulation	Climate stabilization	8	Climate regulation (incl. C- sequestration)
Storm protection	-?	Moder. of extremes	9	Disturbance prevention or moderation
Flood Prevention	Water regulation	Flood mitigation		
Drainage and natural irrigation (drought prevent.)	‘’	Drought mitigation	10	Regulation of water flows
Clean water (waste treatment)	‘’	Water purification	11	Waste treatment (esp. water purification)
Erosion prevention	Erosion regulation	Erosion protection	12	Erosion prevention
Maintenance of productive and clean soils	Soil formation [supporting service]	Soil regeneration and preservation	13	Maintaining soil fertility
Pollination	Pollination	Pollination	14	Pollination
(biol. control)		Seed dispersal	15	Biological control
Pest & disease control	Pest regulation	Pest control		
	Human disease regulat.			
HABITAT/SUPPORT	**SUPPORTING**	*******		**HABITAT**
Nursery-service	e.g. Photosynthesis, primary production, nutrient cycling		16	Lifecycle maintenance
Maintenance of biodiversity		Maintenance of biodiversity	17	Gene pool Protection
CULTURAL (& Amenit.)	**CULTURAL**			**CULTURAL & AMENITY**
Appreciated scenery (incl. tranquility)	Aesthetic values	Aesthetic beauty	18	Aesthetic information
Recreation & tourism	Recreat. & eco-tourism		19	Recreation and tourism
Inspiration for art etc.	-?		20	Inspiration for culture, art and design
Cultural heritage	Cultural heritage			
Spiritual & religious use	Spirit. & religious val.		21	Spiritual experience
Use in science & education	Knowledge system Educational values	Intellectual stimulation	22	Information for cognitive development

Source: Directly taken from (Fisher *et al.* 2010)

1) * Mainly based on/adapted from Costanza et al. (1997) and De Groot, Wilson & Boumans (2002).

2) ** Water is often placed under Regulating Services [RS] but in TEEB the consumptive use of water is placed under provisioning services.

3) ***Daily et al. (2009) do not use main categories and also included detoxification and decomposition of waste, nutrient cycling, and UVb-protection as services.

In the context of direct and indirect benefits, the terms have been more clearly classified as intermediate and final services for the purpose of valuing ecosystem services (Morse-Jones *et al.* 2011):

> 'An **intermediate service** is one which influences human well-being ***indirectly***, whereas a **final service** contributes ***directly***. This classification depends upon context, for example, clean water provision is a final service and benefit (i.e., direct change in human welfare) to a person requiring drinking water, but it is an intermediate service to a recreational angler, who requires it for the fish population in order to get recreation enjoyment' (Morse-Jones *et al.* 2011).

2.4.3 Classification of Ecosystem Services

Ecosystem services have been classified to describe and value ecosystem services depending on the specific context, with the most commonly used classification developed by Millennium Ecosystem Assessment in 2005 (Williams 2009). According to Millennium Ecosystem Assessment (2005) and accepted widely (Tschirhart 2009, Morse-Jones *et al.* 2011), the values of ecosystem services can be classified into provisioning, regulating, supporting, and cultural.

Table 2.3: Classification of Ecosystem Services

Provisioning Services ***Products obtained from ecosystem***	Regulating Services *Benefits obtained from regulation of ecosystem processes*	Cultural Services *Nonmaterial benefits obtained from ecosystems*
• **Food** • **Fresh water** • **Fuelwood** • **Fiber** • **Biochemicals** • **Genetic Resources**	• Climate regulation • Disease regulation • Water regulation • Water purification • Pollination	• Spiritual and religious • Recreation and ecotourism • Aesthetic • Inspirational • Educational • Sense of place • Cultural heritage
Supporting Services ***Services necessary for the production of all tother ecosystem services*** • **Soil formation** • **Nutrient cycling** • **Primary production**		

Source: Directly taken from (Millennium Ecosystem Assessment 2005)

Provisioning services provide direct benefits such as water, timber, food and fibre (Millennium Ecosystem Assessment 2005, Tietenberg and Lewis 2015b). ***Regulatory services*** keep different elements of the natural world 'running smoothly'; they filter pollutants to maintain air and water quality, moderate the climate, sequester and store carbon, recycle waste and dead organic matter, and serve as natural controls for agricultural pests and disease vectors (Holzman C. 2012). ***Supporting services*** are the ecosystem services "that are necessary for the production of all other ecosystem services" (Millennium Ecosystem Assessment 2005). They can be thought of as the services that maintain the provisioning and regulatory services. These services include soil formation, photosynthesis, nutrient dispersal and cycling, seed dispersal and provision of habitat. Healthy habitats preserve both species diversity and genetic diversity, which are critical underpinnings of all provisioning and regulatory services (Millennium Ecosystem Assessment 2005, Holzman C. 2012). ***Cultural services*** are defined as the intangible benefits obtained from contact with nature, and include the aesthetic, spiritual, and psychological benefits that accrue from culturally important or recreational activities such as hiking, bird watching, fishing, hunting, rafting, gardening, and even scenic road trips (Holzman C. 2012).

The scheme proposed by the Millennium Ecosystem Assessment in 2005 for classifying ecosystem services is summarised in Table 2.3, where the supporting services are regarded as the basis for the services of the other three categories.

Alongside the classification proposed by the Millennium Ecosystem Assessment 2005, there are other classifications proposed by various researchers, and including The Economics of Ecosystems and Biodiversity (TEEB). TEEB proposed another classification of ecosystem services and created a comparison table to view the differences between the classifications proposed by Costanza *et al.* (1997) and de Groot *et al.* (2002), MA (Millennium Ecosystem Assessment) (2005), Daily *et al.* (2009) and TEEB in 2010 (Fisher *et al.* 2010). Table 2.2 shows the classification of ecosystem services proposed by TEEB as compared to others.

Table 2.4: Proposal for a Common International Classification of Ecosystem Services (CICES).

	Ecosystem Services Theme (Type)	Class	Group
1	Provisioning	Nutrition	Terrestrial plant and animal foodstuffs
			Freshwater plant and animal foodstuffs
			Marine plant and animal foodstuffs
			Portable water
		Materials	Biotic materials
			Abiotic materials
		Energy	Renewable biofuels
			Renewable abiotic energy sources
2	Regulation and Maintenance	Regulation of wastes	Bioremediation
			Dilution and sequestration
		Flow regulation	Air flow regulation
			Water flow regulation
			Mass flow regulation
		Regulation of physical environment	Atmospheric regulation
			Water quality regulation
			Pedogenesis and soil quality regulation
		Regulation of biotic environment	Lifecycle maintenance & habitat protection
			Pest and disease control
			Gene pool protection
3	Cultural	Symbolic	Aesthetic, Heritage
			Religious and spiritual
		Intellectual an Experiential	Recreation and community activities
			Information & knowledge

Source: (Potschin and Haines-Young 2011)

Table 2.5: Illustrative example of relationships between some intermediate services, final services and benefits

Abiotic Inputs	Intermediate Services	Final Services	Benefits
Sunlight, rainfall, nutrients etc.	Soil formation Primary productivity Nutrient cycling	Water regulation	Water for irrigation Drinking water Electricity from hydro-power
	Photosynthesis Pollination Pest regulation	Primary productivity	Food Timber Nontimber products

Source: Taken directly from (Fisher and Turner 2008)

Potschin & Haines-Young (2011) propose a common international classification of ecosystem services, modifying and extending the scheme proposed by MA for linking service assessments to other data related to economic activity. They propose that the supporting services brought forward by MA may be regarded as a synonym for ecological functions and processes. The classification proposed by Potschin & Haines-Young (2011) is illustrated in Table 2.4.

Fisher & Turner (2008) propose that multiple relationships between ecosystem processes and human benefits for the sole purpose of valuation can be described by dividing and distinguishing them into intermediate, final services and benefits as shown in Table 2.5.

Among the major international studies of MA and TEEB, a recent initiative has been carried out by Wealth Accounting and the Valuation of Ecosystem Services (WAVES) programme for developing a standardized framework for natural capital accounting focussing on ecosystem services. The tool used for the WAVES programme is called System of Environmental-Economic Accounting (SEEA), and provides a framework for development and implementation of internationally accepted and standardized approaches to natural capital accounting and classification of ecosystem services. This framework has also been adopted on a trial basis by the Australian Bureau of Statistics for development of national environmental-economic accounts (Cork *et al.* 2012). SEEA has differentiated various types of accounts on the basis of their perspective (economic or environmental), scale (regional or national), measurement units (monetary or physical) and account organisational units (e.g. grid cells, catchment or ecosystem)(Cork *et al.* 2012). Table 2.6 lists examples of operational and experimental environmental accounts using SEEA framework (Bureau of Meteorology 2013).

A similar endeavour, with a different scale and purpose, has been made to develop an ecosystem services framework through a mapping approach for South East Queensland by Maynard, James & Davidson (2010). In this study, relationships between ecosystem functions and ecosystem services were developed through expert panels and the ecosystem functions were mapped. The feedback from the study showed that the ability to consider ecosystem processes spatially increased awareness among stakeholders about human-ecosystem interrelationships for better planning and management (Maynard *et al.* 2010, Cork *et al.* 2012).

Table 2.6: Illustrative example of environmental accounts using SEEA framework

Account	Agency	Account Subject	Scale	Measurement Units	Account Statistical Units
Energy Account	Australian Bureau of Statistics	Energy supply and use within the economy	National	Petajoules, Dollars	Enterprises, households
Land Account	Australian Bureau of Statistics	Land use and land cover	Regional, state	Area, dollar	Cadastral parcels, grid cells, NRM regions
Waste Accounts	Australian Bureau of Statistics	Generation and disposal of waste and waste management services industry activity	National	Tonnes, dollars	Enterprises, households
Water Accounts	Australian Bureau of Statistics	Water supply and use within the economy	National, state	Giga-litres, dollars	Enterprises, households
Victorian Experimental Ecosystem Accounts	Department of Environment and Primary Industries, Victoria	Ecosystem functioning	Local, state-wide	Environmental benefits index, spatio-temporal hectares per year	Grid cells, spatial unit
Measuring Ecosystem Goods and Services	Statistics Canada	Ecosystems	Local, national	Area, dollars	Grid cells, spatial units, ecosystems
WAVES	World Bank	Monetary valuation of ecosystem services	National	Monetary	various

Source: Taken directly from Bureau of Meteorology (2013)

Ecosystem services are highly interdependent and often overlap. The different ecosystem service classifications arise from the fact that classification schemes are developed in a particular context or for a particular purpose (Fisher and Turner 2008). The classification adopted for the Millennium Ecosystem Assessment (2005) was framed around making the link between human well-being and services provided by ecosystems. In contrast, Boyd & Banzhaf (2007) established their classification for the purpose of devising an accounting system for ecosystem services, while Wallace (2007) focused on managing landscapes and ecological processes to deliver ecosystem services for the purpose of valuation (Fu *et al.* 2011).

Despite the great amount of work carried out on understanding ecosystem services, there is still debates about definitions and classifications (Millennium Ecosystem Assessment 2005, Boyd and Banzhaf 2007, Wallace 2007, Costanza 2008, Fisher and Turner 2008, Fisher *et al.* 2010, Fu *et al.* 2011). Therefore, it is reasonable to accept that none of the definitions actually capture the countless ways in which ecosystem services contributes to human well-being (Fisher *et al.* 2010).

3 Values and Valuation – Spatially Explicit Valuation of Ecosystem Services

This review chapter introduces, for the non-expert reader, the concept of valuation of ecosystem services and the principal methods by which values may be ascribed to these services. The chapter then reviews previous research that has employed spatially explicit valuation studies.

3.1 Introduction

The concept of allocating a value to ecosystem services emerges from the intersection of ecology and economics, and in recent times has been attributed to the inception of the Millennium Ecosystem Assessment (MA) (Millennium Ecosystem Assessment 2005) and the growing need for biodiversity conservation policies and strategies that involve the sustainable provision of ecosystem services (Haslett *et al.* 2010).

The Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005) defines value as 'the contribution of an action or object to user-specified goals, objectives or conditions' and valuation as 'the process of expressing a value for a particular action or object'. As such, ecosystem valuation signifies 'the process of expressing a value for ecosystem goods or services (… [e.g.] biodiversity, flood protection, recreational opportunity), thereby providing the opportunity for scientific observation and measurement' (Farber *et al.* 2002). Value cannot be assessed without stating the goal being served, the goal either being individual welfare (as in conventional economics), ecological sustainability, social fairness or efficient allocation of resources (Costanza 2001). While acknowledging that different authors, from a variety of disciplinary viewpoints, will hold a variety of views about ecosystem valuation (Kettunen *et al.* 2012), these MA definitions are sufficient for the purposes of the research reported here.

The challenge is to incorporate valuation strategies into modelling or into decision or policy making processes that usefully reflect the decision making processes of individual stakeholders (e.g. (Daily *et al.* 2009)). The valuation process is complicated because many of the goods and services to be considered are public and non-market (Gascoigne *et al.* 2011): it is difficult to obtain meaningful values for goods that are intangible and have no formal market (Morse-Jones *et al.* 2011).

This following sections review the concepts and tools applied to valuation of ecosystem services and particularly any studies that incorporate spatially explicit valuation. The review employed a systematic process of content analysis using computer aided qualitative data analysis software (CAQDAS) QSR NVivo 10 and the open source referencing software Mendeley.

3.1.1 Types of Values

Values determine decisions: they define and prescribe what is important to stakeholders and decision makers (Reagan 2006). Values are an integral part of our culture and derive from worldviews and associated fundamental perceptions (Brondízio *et al.* 2010) and are expressions of preferences that reflect an individual psychology (Spash 2008). As noted above, value cannot be determined without knowing the goal being served. From a utilitarian perspective, the economic value of an asset lies in its role in attaining human goals, whether the goal is spiritual enlightenment, aesthetic pleasure or production of market commodity (Pascual *et al.* 2010). While economists are likely to advocate an anthropocentric perspective that focuses on instrumental values, ecologists in contrast are more likely to take a bio-centric perspective based on intrinsic ecological values (Pascual *et al.* 2010).

With the two different approaches by ecologists and economists towards value, some authors consider the rationales to be complementary and see no conflict in their simultaneous use (Costanza 2006, Pascual *et al.* 2010).

Table 3.1: A typology of values with definitions (from DEFRA (2007) and Pascual et al. (2010))

Value type	Value sub-type	Meaning
Use value	Direct use value	Results from direct human use of biodiversity (consumptive or non-consumptive). Where individuals make actual or planned use of an ecosystem service.
	Indirect use value	Derived from the regulation services provided by species and ecosystems. Where individuals benefit from ecosystem services supported by a resource rather than directly using it.
	Option value	Relates to the importance that people give to the future availability of ecosystem services for personal benefit (option value in a strict sense).
Non-use value	Bequest value	Value attached by individuals to the fact that an ecosystem resource will be passed on to the future generations (inter-generational equity concerns).
	Altruism value	Value attached by individuals to the fact that other people of the present generation have access to the benefits provided by species and ecosystems (intra-generational equity concerns).
	Existence value	Value related to the satisfaction that individuals derive from the mere knowledge that species and ecosystems continue to exist.

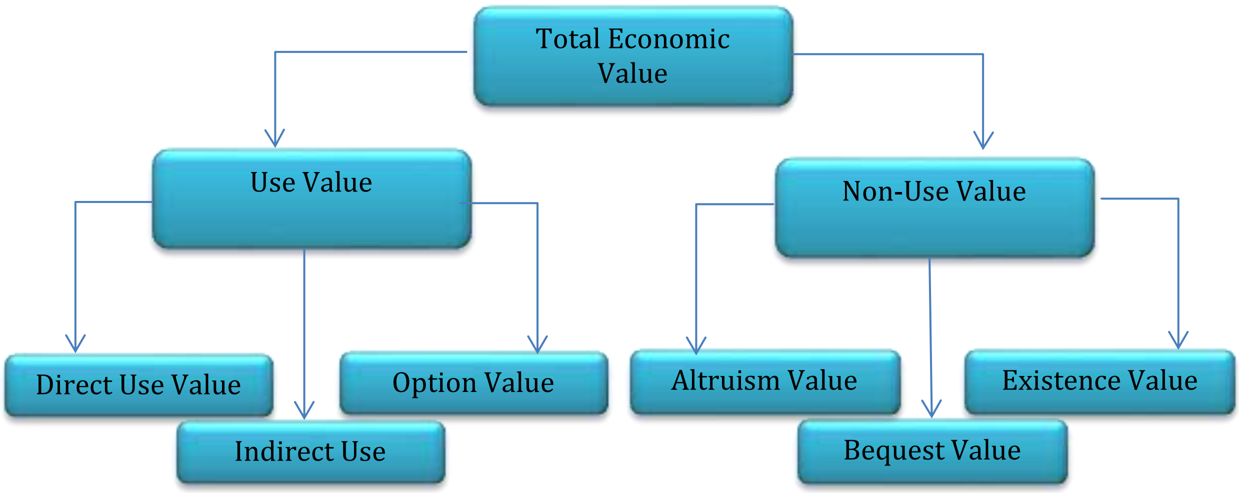


Figure 3.1: Total Economic Value Framework taken from (DEFRA 2007, Pascual *et al.* 2010)

The economic value of an ecosystem is generally considered in terms of two main types: use value and non-use value. Use value is further divided into direct use, indirect use, and option value. Non-use value (or passive use value) may be categorized into bequest value, altruistic value and existence value (DEFRA 2007, Pascual *et al.* 2010). Use value is generally associated with goods for which market prices exist, whereas non-use value may be, for example, the satisfaction derived from the knowledge that a natural environment is maintained (DEFRA 2007, Pascual *et al.* 2010). The typology of values is given in Figure 3.1, with their descriptive definitions in Table 3.1.

3.1.2 Methods of Valuation of Ecosystem Services

Valuation methods fall broadly into two main categories: economic and non-economic valuation approaches. Economic approaches attempt to value ecosystem services in terms of monetary values, whereas non-economic approaches attempt to value ecosystem services in units other than monetary. Some approaches (e.g. stated preference methods) use a combination of economic and non-economic approaches (DEFRA 2007). This section categorises valuation methods based on studies of Pascual *et al.* (2010), DEFRA (2007), Liu *et al.* (2010) and Farber *et al.* (2002). Table 3.2 describes the categories in detail.

A. Economic Valuation Methods

Economic valuation or monetary valuation methods can be divided into the following subcategories (Pascual *et al.* 2010) based on a total economic value[8] framework:

i. direct market approaches
ii. revealed preference approach
iii. stated preference approach
iv. benefit transfer

i. Direct Market Approaches

Direct market approaches use data from actual markets and thus represent actual preferences or costs (Pascual *et al.* 2010). They can be further divided into further three main approaches:

a. market price based approaches
b. cost based approaches
c. production function based approaches

a. Market price based approach

The market price of a commodity can be used for valuation, as the market price in a well-functioning market reflects the market preferences as well as the marginal cost of production (Pascual *et al.* 2010). Market price based approaches use observed market prices as a direct measure of the economic value of an ecosystem service (DEFRA 2007). The price of a commodity multiplied by the marginal product of the ecosystem service is an indicator of

[8] **Total economic value** is defined as the sum of the values of all service flows that natural capital generates both now and in the future – appropriately discounted (Pascual *et al.* 2010).

the value of the service, therefore, market price can be a good indicator of the value of an ecosystem service (Pascual *et al.* 2010).

b. Cost based approach

In this method, observed market prices are used as a proxy for the value of ecosystem services. This is based on estimates of costs that would be incurred in order to recreate ecosystem service benefits through artificial means (DEFRA 2007, Pascual *et al.* 2010). Cost based approaches have further been divided into the following categories:

i. *Avoided cost:* It is the cost that would have been incurred in the absence of the ecosystem service (DEFRA 2007, Pascual *et al.* 2010).
ii. *Replacement cost:* It is the cost incurred by replacing the existing ecosystem service with artificial technology (DEFRA 2007, Pascual *et al.* 2010).
iii. *Mitigation or Restoration cost:* It is the cost that would be incurred in order to restore an ecosystem service after its loss (DEFRA 2007, Pascual *et al.* 2010).

c. Production function based approach

Production function estimates 'how much a given ecosystem service contributes to the delivery of another service or commodity which is traded in an existing market' (Pascual *et al.* 2010). Valuation is applied to the outcome quantified through this approach. It focuses on the relationship that may exist between a particular ecosystem service and the production of a market good. Ecosystem services valuation is carried out by inferring their value through a consideration of the changes in production process of market goods that result from an environmental change (DEFRA 2007).

ii. Revealed preference approach

This method is based on data representing an individual's preferences for a marketable good which includes environmental attributes (DEFRA 2007). This approach relies on 'observation of individual choices' in an actual existing market products (Pascual *et al.* 2010). As the name indicates, preferences are revealed through choices. There are two main types of revealed preferences:

a. *Travel cost:* This is often applicable to recreational values and is based on recreational experience associated with direct expenses incurred on visiting the site (Pascual *et al.* 2010).
b. *Hedonic pricing:* This is based on the 'demand for an environmental attribute of a marketed commodity'(Pascual *et al.* 2010). For example, the change in value of a property near a forest when the forest is logged will indicate the forest's value.

iii. Stated preference approach

This approach is used when no surrogate or proxy market exists from which the value of an ecosystem service can be deduced, thus market and demand for ecosystem services are simulated by means of surveys of hypothetical policy-induced changes in the provision of ecosystem services (Pascual *et al.* 2010). In this method, carefully structured questionnaires are used to obtain individuals' preferences for a given change in natural resource or environmental attribute. It is the only method that can estimate non-use values (DEFRA 2007, Pascual *et al.* 2010). Two main type of stated preference approach are:

a. *Contingent valuation:* which uses questionnaires to illicit how much individuals would be 'willing to pay' (WTP) to increase the provision of an ecosystem service or, alternatively, how much they would be 'willing to accept' (WTA) for its loss or degradation.
b. *Choice modelling:* which attempts to model the decision process of an individual in a given context (Hanley and Wright, 1998; Philip and MacMillan, 2005). Individuals are faced with two or more alternatives with shared attributes of the services to be valued, but with different levels of attribute (one of the attributes being the money people would have to pay for the service).

iv. Benefit transfer

The actual process of valuation is both costly and time consuming. Benefit transfer, also known as value transfer, is a method used in cases where a primary valuation study cannot be carried out due to cost or time constraints. It is widely used to inform policy decisions, though it has its own limitations and its validity continues to be a point of academic debate (Barton 1999, Boyle *et al.* 2010, Windle *et al.* 2013). It is the process of taking evidence on the value of ecosystem services from one context and study site and transferring it to another context and policy site (DEFRA 2007). For example, ecosystem service values obtained from tourists viewing protected natural habitats at one park may be used to estimate that for tourists viewing a different park. It is 'the adaptation of existing valuation information or data to new policy contexts that have little or no data available' (Pascual *et al.* 2010).

Benefit transfer values can be acquired from all types of valuation approaches. The steps involved in undertaking a benefit transfer include an appropriate literature review of valuation studies relevant and applicable to the policy context, selection of a study site as close a match as possible, adjustment of willingness to pay (WTP) values using income or function transfer and aggregation. For example, the Environmental Valuation Reference Inventory (EVRI) coordinated by Environment Canada comprised over 1900 valuation studies, and The Ecosystem Services Valuation Database (ESVD) compiled by the international Ecosystem Services Partnership comprised 1310 valuation studies for benefit transfer (DEFRA 2007, de Groot, Alkemade, *et al.* 2010a, de Groot, Fisher, *et al.* 2010, Van der Ploeg and De Groot 2010, Van der Ploeg *et al.* 2010).

Where primary valuation research will always be considered a first-best strategy, benefit transfer represents a robust and meaningful second-best strategy for the evaluation of environmental management and policy alternatives. A value transfer strategy, although having limitations, is certainly preferred to assigning a zero value to ecosystem services (Troy and Wilson 2006). In the context of value transfer, value transfer databases including numerous studies around the world developed by various organisations including Environment Canada and The Economics of Ecosystems and Biodiversity (TEEB[9]) are of high significance (DEFRA 2007).

Table 3.2: Categorization of Valuation Methods based on the studies of Pascual *et al.* (2010), DEFRA (2007), Liu *et al.* (2010) and Farber *et al.* (2002).

Economic Valuation Methods	**Direct Market Approaches**	Market Price Based Approach	
		Cost Based Approach	Avoided Cost
			Replacement Cost
			Mitigation or Restoration Cost
		Production Function Based Approach	
	Revealed Preference Approaches	Travel Cost	
		Hedonic Pricing	
	Stated Preference Approaches	Contingent Valuation	
		Choice Modeling	
	Benefit Transfer		
Non-Economic Valuation Methods	Deliberate Group Valuation		

B. Non-economic Valuation Methods

Non-economic valuations tend to value ecosystem services in units other than monetary. It is a method in which participatory or deliberative techniques are used to understand people's opinion and preferences. Different techniques have been used to capture value through people's preferences including discourse based deliberate group valuation (Jacobs 1997), contingent ranking (Beggs *et al.* 1981), citizens' juries (Coote and Lenaghan 1997, Blamey and James 1999), citizen participation (James and Blamey 1999) and Q-methodology (Stephenson 1953, Barry and Proops 1999). Among these, some are used in combination with economic valuation methods, and have been tested to provide improved understanding of values (DEFRA 2007).

Each of these methods has advantages and disadvantages; these are extensively described by Pascual *et al.* (2010), although special considerations are required while dealing with benefit transfer. Benefit transfer encounters challenges of dealing with nonlinearities in benefits and with threshold effects (Morse-Jones *et al.* 2011). A significant limitation of benefit transfer valuation studies is considered to be availability of valuation data (Troy and Wilson 2006). As benefit transfer should use economic studies whose valuation coefficients were derived in a similar context to the policy site,

[9] TEEB was launched by Germany and the European Commission in response to a proposal by the G8+5 Environment Ministers in Potsdam, Germany in 2007, to develop a global study on the economics of biodiversity loss.

benefit transfer faces shortages in finding studies that are comparable to the policy site, thus resulting in compromise on specificity and reliability. Troy & Wilson (2006) emphasize that lack of sufficient data will result in biased outcomes.

3.1.3 Complications in Valuing Ecosystem Services

Regardless of whether ecosystem services are valued through monetary or non-monetary measures, there exist issues of validity and accuracy in every method. In general, there are several identified concerns when carrying out valuation studies: these include double counting, aggregation, availability of data and procedural errors in benefit transfer (Troy and Wilson 2006, Spash 2008, Morse-Jones *et al.* 2011, Wainger and Mazzotta 2011).

Double counting of ecosystem services may occur because some services (in particular supporting and regulating services) are inputs to the production of other services (Boyd and Banzhaf 2007, Wallace 2007, Fisher and Turner 2008, Fisher *et al.* 2010). A common confusion is between ecosystem functions and ecosystem services and this confusion can also lead to errors in ecosystem services valuation (Fu *et al.* 2011). It is apparent that, for assessing the overall value of ecosystems, a simple summation of ecosystem services and ecosystem functions would result in double counting. It is necessary to understand that ecosystem functions are the intermediate processes that are necessary and conducive to the end product of ecosystem services. To avoid this confusion and double counting, Wallace (2007) introduces a classification scheme that divides services into ***intermediate (indirect) services*** and ***final (direct) services***, now adapted by others (Morse-Jones *et al.* 2011), as well to reduce redundancy found in the MA definition and classification. It has been suggested that the value of the intermediate service should be incorporated into the value of final service for best results (Fu *et al.* 2011).

To eliminate double counting in ecosystem services audits, The Economics of Ecosystems and Biodiversity (TEEB) has revised the MA definition and classification. The MA definition of ecosystem functions was redefined as 'a subset of the interactions between ecosystem structure and processes that underpin the capacity of an ecosystem to provide goods and services.' Moreover, TEEB replaced "Supporting Services" in the MA classification with "Habitat Services and Ecosystem functions" (Fisher *et al.* 2010). TEEB describes ecosystem functions as the interactions that occur between structure and processes, whether being physical, chemical or biological. Fisher *et al.* (2010) clarifies that functions represent the potential that ecosystems have to deliver a service and illustrates it by an example of primary production being a process, which is needed to maintain a viable fish population i.e. function, which in turn can be used to provide food i.e. a service.

TEEB also argued that a clear delineation between ecosystem functions, ecosystem services (their direct and indirect contribution to human welfare) and benefits (the welfare gains they generate) is indispensable in avoiding the problem of double counting. Boyd & Banzhaf (2007) pressed the need for clear and standardized definitions and indicated that loose definitions undermine accounting systems. It is out of the scope to describe here the detailed reasons for, and the possible strategies to eliminate, double counting; the study by Fu *et al.* (2011) is recommended in this context.

A collective measure of total benefit from ecosystem services including both use and non-use values are referred to as total economic value (TEV). While summing up the values and benefits of ecosystem services, challenges arise when dealing with monetary and non-monetary values and converting them into a similar aggregatable unit. Also, there exist multiple competing and complementary services which can increase the potential for double counting (Wainger and Mazzotta 2011).

3.1.4 Valuation and its Significance

For about the last three decades, nature has been conceptualised as natural capital in environmental and ecological economics, with this terminology often attributed to the late Professor David Pearce from his writings in and around 1988 (Sullivan 2014). To acknowledge the value of this natural asset and to facilitate decision making in policy and management it is essential to carry out ecosystem services valuation (Costanza *et al.* 1997, Serafy 1998, Toman 1998, Liu *et al.* 2010).

Valuation also plays an important role in creating markets for the conservation of biodiversity and ecosystem services through Payments for Ecosystem Services (Brondízio *et al.* 2010). The concept of payment for ecosystem services date to the 1980's when the United States implemented wetland and stream credit banking. The idea was implemented in Costa Rica in 1996, where landowners were paid $42 per hectare per year to preserve forest (Holzman C. 2012). The concept of Payment for Ecosystem Services is well defined in Walls & Riddle (2012) (taken from Wunder (2007)) as:

> 'a voluntary, conditional agreement between at least one seller and one buyer, over a well-defined environmental service—or a land use presumed to produce that service. The agreement is conditional in that payment should be made only if it can be verified that the environmental service is continually provided. Sellers are typically private landowners and buyers may be the government, conservation agencies or other nongovernmental organizations, or private individuals or firms'.

An upsurge in research literature addressing the methodological perspective of ecosystem services valuation occurred in the late 1990's with the launch of a special issue of Ecological Economics addressing Values of Ecosystem Services (published in 1998) (Fu *et al.* 2011). Researchers have attempted to value ecosystem services from different perspectives in the past two decades. In Australasia, Lockwood (1997, 1998) published articles on integrated value theory. Other papers published in Australasia that address ecological economics and valuation include but are not limited to the work of Blamey *et al.* (2000) and Rolfe *et al.* (2002), who applied choice modelling methods as an alternative to contingent valuation methods (Patterson 2006). An extensive literature published in the past two decades includes research on valuing urban ecosystems (Kreuter *et al.* 2001), landscapes (Willis *et al.* 2012), catchments (Kragt *et al.* 2011), agricultural land (Bastian *et al.* 2002) and wetland ecosystems (Woodward and Wui 2001); valuing various services such as biodiversity (Lewandrowski *et al.* 1999), carbon sequestration (Gascoigne *et al.* 2011), and investigating valuation strategies from mixed valuation (Aznar *et al.* 2011) using GIS and artificial neural networks (Dai *et al.* 2005) through to hedonic models (Bastian *et al.* 2002).

Table 3.3: List of ecosystem services tools and models based on studies conducted by Bagstad *et al.* (2012)and Bagstad *et al.* (2013)

Tool	Brief Description	Authority	GIS compatibility	Availability & Link
Ecosystem Services Review (ESR)	Publicly available, spreadsheet-based process to qualitatively assess ecosystem services impacts	World Resources Institute	No	Open Source http://www.wri.org/
InVEST	Integrated Valuation of Ecosystem Services and Trade-offs – Ecosystem services mapping and valuation models accessed through ArcGIS	Natural Capital Project	Yes	Open Source http://www.naturalcapitalproject.org
ARIES	Artificial Intelligence for Ecosystem Services – Modelling framework to map ecosystem services flows; online interface; web based	University of Vermont	Yes	Open Source http://www.ariesonline.org
MIMES	Multiscale Integrated Models of Ecosystem Services – Map and value ecosystem services Simile Modelling Platform	University of Vermont	Yes	Open Source http://www.afordablefutures.com
EcoMetrix	Measuring ecosystem services at site scale using field surveys, Spreadsheet based approach	Consulting Firm	No	Proprietary http://www.parametrix.com
EcoAIM	Mapping ecosystem services and stakeholder preferences by weighing or aggregating scores	Consulting Firm	Yes	Proprietary
ESValue	Mapping ecosystem services and stakeholder preferences literature-derived data and experts	Consulting Firm	Yes	Proprietary
NAIS	Natural Assets Information System– Database Valuation database paired with GIS mapping of land cover types for point transfer	Consulting Firm	Yes	Proprietary http://www.sig-gis.com
SERVES	Simple Effective Resource for Valuing Ecosystem Services–Economic valuation	NGO	Yes	Subscription based
SolVES	Social Values for Ecosystem Services – ArcGIS toolbar Mapping social values of ecosystem services based on public attitude and preference survey	USGS and Colorado State University	Yes	Publicly Available http://solves.cr.usgs.gov
EPM	Ecosystem Portfolio Model – web accessible tool to model economic, environmental and quality of life impacts of alternative land use choices (Florida)	Miami Dade County, Florida	Yes	Publicly Available http://geography.wr.usgs.gov

Tool	Brief Description	Authority	GIS compatibility	Availability & Link
EcoServ	web based – region based	USGS and Chinese Academy of Sciences	In development	Publicly Available
InFOREST	Investing in Forests –– web based Quantify ecosystem services in Virginia	Virginia Tech University	Yes	Publicly Available http://inforest.frec.vt.edu/
ESR	Ecosystem Services Review –(business spreadsheet), For corporations	-	No	Publicly Available
Ecosystem Services Toolkit	A framework	UNEP	No	Publicly Available
Ecosystem Valuation Toolkit	Valuation database paired with GIS mapping of land cover types for point transfer	-	Yes	Subscription based http://www.esvaluation.org
Benefit Transfer and Use Estimating Model Toolkit	Spreadsheet, uses function transfer to value changes in ecosystem services in the US.	US	No	Publicly available http://www.defenders.org
Co$ting Nature	Web accessible tool to map ecosystem services and conservation priority areas	-	Yes	Publicly available http://www1.policysupport.org/cgi-bin/ecoengine/start.cgi?project=costingnature
LUCI (Polyscape)	GIS toolkit to map areas providing services and potential gain and loss of services under management scenarios	-	Yes	Open source http://www.polyscape.org
Envision (Evoland)	Integrated urban growth ecosystem services modelling system based on agent based modelling	Oregon State University	Yes	Open source http://envision.bioe.orst.edu

3.1.5 Databases and Tools for Ecosystem Services Valuation and Quantification

There are numerous valuation studies that have been carried out with a wide range of strategies used. In this context, Pascual *et al.* (2010) carried out a review of 314 peer reviewed valuation case studies, which showed interesting patterns of valuation techniques and types of ecosystem services. The strategies used for valuation methods were diverse and included (Pascual *et al.* 2010): avoided cost (Gunawardena and Rowan 2005), benefit transfer (Stuip *et al.* 2002), bio-economic modelling (Brown and Hammack 1973), choice modelling (Carlsson *et al.* 2003), consumer surplus (Bergstrom *et al.* 1990), contingent ranking (Emerton 1996), conversion cost (Abila 1998), contingent valuation method (Hanley and Ruffell 1993), factor income or production function (Barbier *et al.* 1991), hedonic pricing (Mahan 1997), market price (Pattanayak and Kramer 2001), mitigation cost (Van Kooten *et al.* 2007), opportunity cost (Dixon *et al.* 1990), replacement cost (Stuip *et al.* 2002), restoration cost (Emerton 2005), and travel cost (Whitten and Bennett 2002).

Among the studies, various types of provisioning, regulating, habitat/support and cultural services were represented in a conceptual matrix for wetlands and forests with the studies carried out under various strategies. Here, a detail reference of studies on valuation strategies used for various ecosystem services is not provided, as a detailed referenced database of valuation studies already exist in Pascual *et al.* (2010).

Costanza *et al.*'s paper in 'Nature' (Costanza *et al.* 1997) was the first of its kind in which an endeavour was made to estimate the total economic value of all the ecosystems throughout the world. Constanza classified the global bioshpere into 16 types of ecosystems and 17 types of services with ecosystem services value of each type (Hao *et al.* 2012). The controversial paper stirred an academic debate in a special issue of Ecological Economics on 'The Dynamics and Value of Ecosystem Services: Integrating Economic and Ecological Perspectives'. As stated by Howarth & Farber (2002), 'the paper is both widely cited and widely criticized'. Some critics have identified flaws in estimates (Toman 1998), while others have focused on problems in methods and assumptions in the paper (Bockstael *et al.* 2000, National Research Council 2004). Critics challenge the notion of attaching a price tag or a dollar value on the world's ecosystems (Serafy 1998), but there is no doubt that the paper contributed significantly to the worldwide propagation of the ecosystem services concept.

Based on Costanza *et al.'s* (1997) paper, Xie Gaodi and others derived an equivalent weight factor matrix for ecosystem services in China based on a survey of more than 200 Chinese scholars (Xia *et al.* 2003, Xie *et al.* 2003). This followed a series of value transfer valuation studies conducted in China based on the equivalent weight factor derived per hectare for estimating ecosystem services contributions from landuse (Zhao *et al.* 2004, Amut *et al.* 2006, Hao *et al.* 2012, Liu *et al.* 2012). Similarly, Alessa, Kliskey & Brown (2008) outlined perceived landscape services associated with Kenai Peninsula, Alaska and designed a random household survey to gather data on values of the landscape services, and mapped these using a spatial approach. In another study, 2-D cellular automata and the CORINE 2000 land cover classification model were used to obtain value contributions for each land cover class under ecosystem services of CC mitigation, bio-resource provision, ecological functioning, economic wealth, aesthetic value and human well being (Fürst *et al.* 2010). Similarly, Troy & Wilson (2006) conducted a spatial analysis of ecosystem services values by using value transfer method for assigning values to various land cover types and further linking it to GIS. Their study carried out valuation of land cover types for Massachusetts, California and Maury Island through value transfer and estimated total ecosystem services value (ESV) flow in units of average dollars per hectare per year.

Table 3.4: A list of databases available for ecosystem services valuation with their authority and number of studies

S#	Valuation Database	Owned by	No. of Studies
1	Environmental Valuation Reference Inventory (EVRI)	Environment Canada	1900
2	The Ecosystem Services Valuation Database (ESVD)	TEEB & ESP	1310
3	New Zealand Non-Market Valuation Database in	New Zealand	100
4	EnValue	New South Wales, Australia	400
5	Ecosystem Services Database	Gund Institute of Ecological Economics, University of Vermont	-
6	ValuebaseSWE	Sweden	200
7	Beneficial Use Values database	University of California	-
8	Biodiversity Economics	IUCN and WWF	100
9	Ecosystem Valuation	US Dept. of Agriculture	-

Various valuation databases have been developed to assist with value transfer thus providing a suite of valuation studies carried out in various parts of the world. In this context, the Environmental Valuation Reference Inventory (EVRI) is a useful database coordinated by Environment Canada and comprising 1900 valuation studies for benefit transfer (DEFRA 2007). Also, The Ecosystem Services Valuation Database (ESVD) compiled by international Ecosystem Services Partnership (ESP – www.es-partnership.org) is a valuable database consisting of 1310 studies, developed by ESP members and TEEB researchers and initially released as TEEB Valuation Database in 2010. This database is publically accessible for non-commercial, non-profit and educational use (De Groot *et al.* 2010, de Groot, Fisher, *et al.* 2010, Van der Ploeg and De Groot 2010, Van der Ploeg *et al.* 2010). There are a number of other databases available with different characteristics, including the New Zealand Non-Market Valuation Database; EnValue in New South Wales, Australia; Ecosystem Services Database of Gund Institute of Ecological Economics, University of Vermont; ValuebaseSwe comprising 200 primary studies from Sweden; the Beneficial Use Values database from University of California; Biodiversity Economics by IUCN; and WWF and Ecosystem Valuation from the U.S Deaprtment of Agriculture (Iovanna 2005, McComb *et al.* 2006, Liu 2011).

Development of tools and models in order to represent ecosystem services in terms of ecology and economics has continued since the inception of the field of ecological economics. Bagstad *et al.* (2013) presented a comparison of 17 such ecosystem services tools and models against eight set of evaluative criteria. These tools are listed in Table 3.3.

3.2 Content Analysis of Spatial Valuation Studies

Considerable progress has been made over the past decade towards improved tools and techniques in ecological economics. In this context, the role of GIS (Geographical Information Systems) technology and the associated spatially explicit data and tools have played a key role in enhancing the modelling strategies (Holzman C. 2012). Publications on ecosystem services in scientific journals and international conferences increased markedly since 2000 (Williams 2009). Among recent publications, substantial work has been carried out to present these values spatially using GIS and mapping methods and technology (Troy and Wilson 2006, Chen *et al.* 2009). Values associated with land and its associated ecosystem services are inherently location specific and can best be understood and valued within a locational context (Anderson and West 2006, Alessa *et al.* 2008b, Sherrouse *et al.* 2011, Bastian *et al.* 2015). A diverse range of studies has been undertaken in different areas and at different scales (Chan *et al.* 2006, Naidoo and Ricketts 2006, Egoh *et al.* 2008, Maynard *et al.* 2010, Raudsepp-Hearne *et al.* 2010), indicating that spatial mapping and modelling are vital tools for ecosystem services valuation (Cork *et al.* 2012).

In view of the above, an in-depth literature review was carried out to ascertain the trends in the spatial valuation paradigm in the past decade; the foci being on methods of valuation and tools used for valuation purposes, and the current trends in publishing activity.

3.2.1 Method and Design

The literature review concentrated on the decade: 2004 to 2014. Bibliographic tools used were QSR NVivo 10, Mendeley and the Scopus search engine.

A. Data Source and Datasets

Scopus served as a search engine in this study, where the primary keywords given for search were 'Spatial', 'Ecosystem Services', Valuation' in order to identify articles that dealt with valuation of ecosystem services in a spatially explicit manner. The search result resulted in 199 studies in total. After viewing all 199 studies, there were 7 studies that were duplicates, either having a duplicate 'in press' copy followed by a published one, or a working paper followed by a journal article. Therefore, the 7 duplicates consisting of 'in press' articles and working papers were removed and the remaining 192 were analysed further. These 192 articles were imported into QSR NVivo 10 software for further scrutiny and analysis.

QSR NVivo is computer aided qualitative data analysis software and was used to represent, manage, explore, and analyse the data. QSR NVivo can be used to analyse and import data of different formats; for this study, data sources were mainly the downloaded pdf format articles, Excel .xls sheets and a Mendeley literature database in

RIS format. The three types of datasets were initially imported into NVivo software for further scrutiny and analysis.

Mendeley is an open access reference manager and an academic social network. It is a combination of Mendeley Desktop and Mendeley Web. Mendeley Desktop is a desktop application that provides reference management tools and allows users to manage PDFs and generate citations and bibliographies using different styles; Mendeley Web is a website that helps to manage, share and discover both content and contacts in research. In this paper, Mendeley's ability as a reference manager was utilised for developing a complete and relevant literature database. This was then imported into NVivo in RIS format.

The imported literature database was exported as an Excel worksheet and edited to retain the columns that were of interest for this study, such as 'Name of Article', 'Journal', 'Author(s)', etc., which was then imported into NVivo as a Classification Sheet.

B. NVivo For Data Management

Data Management in NVivo

Sequential steps are used in NVivo to manage, track and organize the datasets. This involves creation of an NVivo project, a project log and importing datasets into NVivo as Internals and Externals.

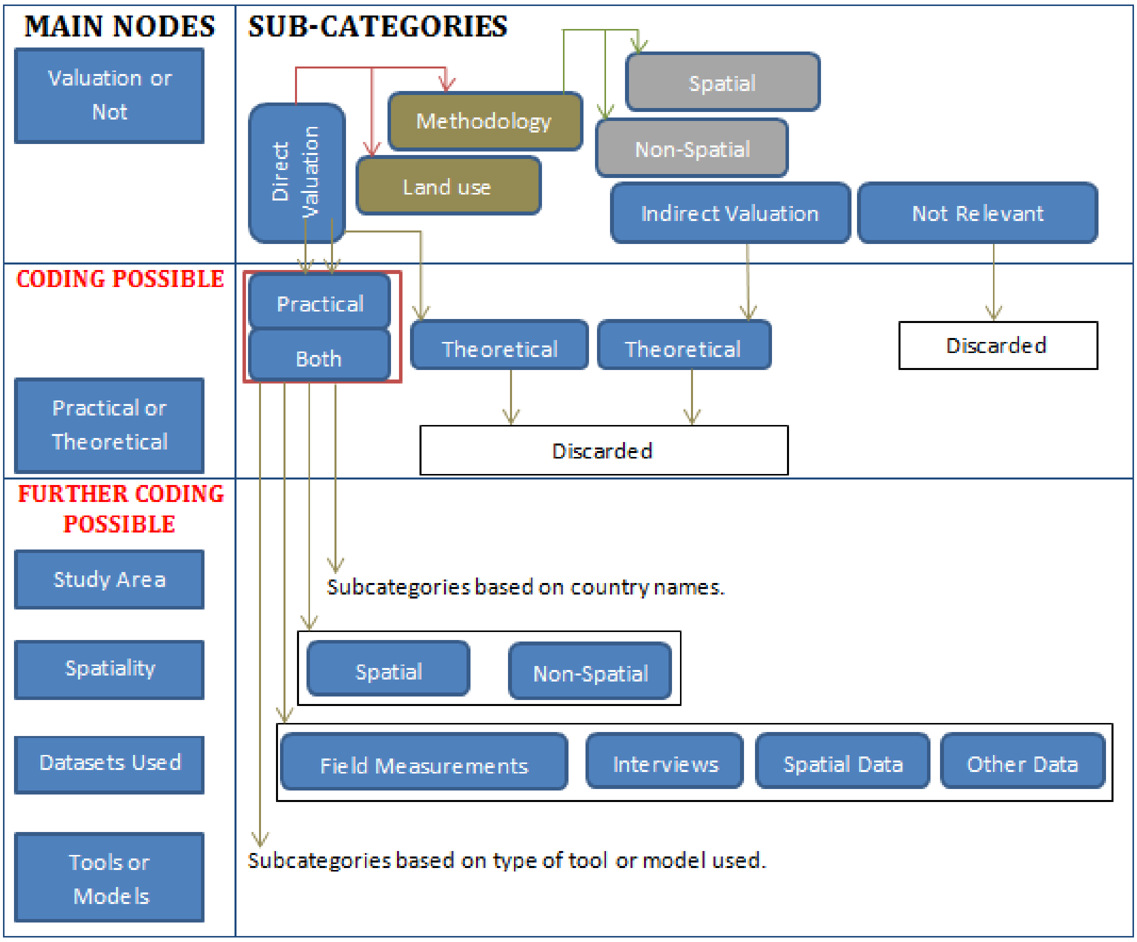


Figure 3.2: Representation of nodes and their sub-categories

Creating Nodes, Node System and Coding Rules

In order to categorize data into different themes, nodes are created in NVivo and relevant data are coded into these nodes. To start the work on NVivo, a deductive approach towards formulating nodes was undertaken initially based on the information required for this study followed by an inductive approach to categorize the data and formulate concepts and links from the data to generate further nodes. This approach was adopted because the author was aware of the precise information that needed to be extracted from the input data (article pdfs in this case). It was necessary to write the description of each node in order to give a specific Coding Rule to a node. The coding rule describes the extent of the information that needs to be coded into the node. Therefore, precise and clear coding rules were assigned to all the nodes and coding was carried out for every imported pdf.

Nodes and their Subcategories

The first node in the categorization was 'Valuation or Not'. Initially this first node and its subcategories were coded by either deciding whether the article was a study that was directly dealing with the valuation of any ecosystem services, or was a study indirectly related to valuation, or furthermore, was a study not relevant to valuation. This process resulted in identifying 129 studies that were directly valuation studies, 58 were indirect valuation studies, and 5 were not relevant to valuation.

The second node in the sequence was 'Practical or Theoretical', and had three subcategories: 'Practical', 'Theoretical' and 'Both'. 'Direct Valuation' studies were the only ones allowed to be coded further into any of the subcategories as illustrated in the diagram below, although, only the 'Practical' or 'Both' valuation studies were further coded into the subsequent node categories of 'Study Area', 'Spatiality', 'Datasets Used' and 'Tools and Models'. There were 50 articles that were classified as Theoretical containing review papers, proposed frameworks for valuation and concepts and views in relation to valuation.

3.2.2 Analysis and Results with NVivo

While analysing the various nodes and coding within those nodes, some of the trends in valuation studies were visually evident by viewing the 'sources' coded into each node.

A. Trends Visually Observable

Out of 129 Valuation studies, 108 articles were found to be spatial, i.e. having spatial datasets used for analysis or spatial representation of results, while 21 articles were found to have no substantial spatial analysis. 108 articles had spatial datasets; 19 articles used interviews, surveys, questionnaires or group discussions as source of data collection. The spatial studies were double checked by the Node Valuation or Not -- Direct Valuation – Methodology -- Spatial; which also showed 108 spatial method articles, whereas, the Non-Spatial Method node had 21 articles coded into the node. Therefore, it can be said that the keywords used resulted in 83.7% of spatially explicit articles and 16.2% of non-spatial articles.

It was also observed that in the Node 'Landuse', forest had the highest number of articles quoted (18). The nodes were reclassified into Terrestrial Land use, Marine or Water

Land use, and Regional Division. This analysis showed that Marine and Water had the highest number of articles (58), followed by Regional Division having 44 articles, and Terrestrial having 30. This showed that most of the valuation research in the past decade has been focussed on marine, coastal regions, wetlands, mangroves and watersheds.

B. Trend by Study Area

In terms of Study Areas, the analysis showed that valuation studies framed in the context of Chinese landscapes had the highest number of publications over the past decade. A series of value transfer valuation studies were published out of China. These studies were based on equivalent weight factor derived per hectare for estimating the ecosystem services contributions of land use (Zhao *et al.* 2004, Amut *et al.* 2006, Hao *et al.* 2012, Liu *et al.* 2012). These published articles drew largely on the concepts provided in Costanza's paper (Costanza *et al.* 1997) and an equivalent weight factor matrix derived from a survey of more than 200 Chinese scholars (Xia *et al.* 2003, Xie *et al.* 2003). A number of review papers had also been published in relation to ecosystem services valuation studies conducted in China (Bao *et al.* 2007, Zhang *et al.* 2010, Zhang and Liu 2011, Siew and Döll 2012). These reviews highlighted discrepencies in the separate studies and indicated a need for development in the areas of uncertainty analysis, standardization of methods and advancement of spatially and temporally explicit tools and models. One of the reviews (Zhang *et al.* 2010) noted that, despite having discrepencies, the studies conducted raised public awareness of the value of ecological and biological resources in China.

The second largest contributor of spatial ecosystem valuation studies was USA with 19 articles followed by Spain with 9. Chart 3.1 shows the number of published articles on spatial valuation based on country names from 2004 to 2014. The chart shows only those countires having more than three published articles.

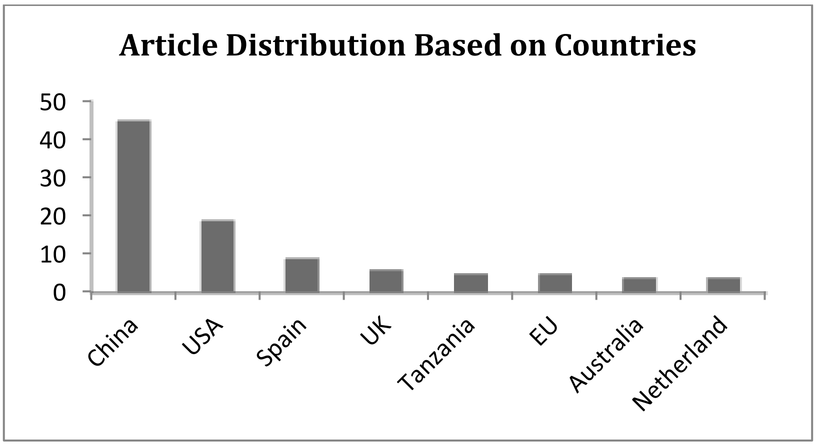


Chart 3.1: Distribution of number of articles published from 2004 to 2014 specifically on spatial ecosystem services valuation based on study areas presented by country of origin

This analysis was also carried out on the basis of continents. Asia had the highest number of studies, followed by Europe and then North America. In turn, the highest contributor of articles in Asia was China and that of Europe was Spain as shown in Chart 3.2.

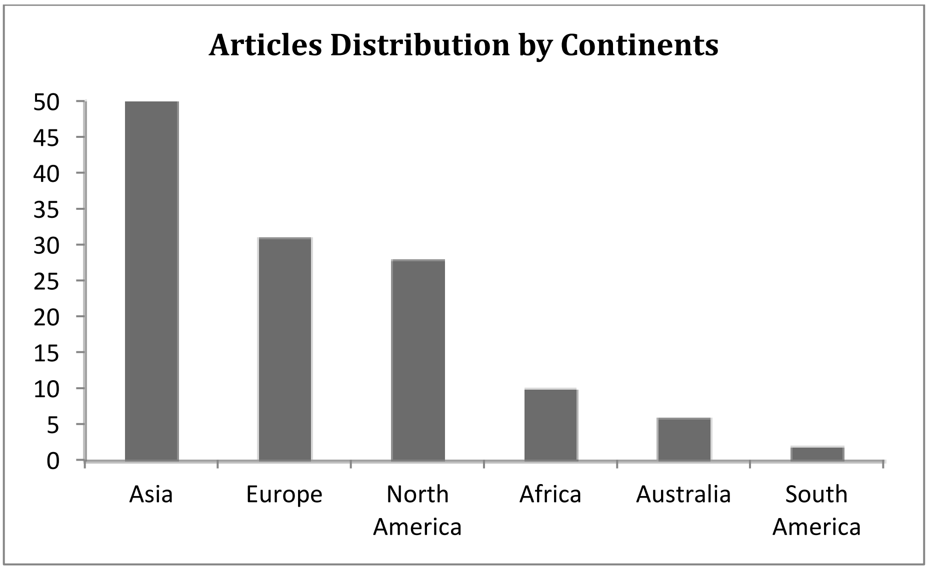


Chart 3.2: Distribution of articles on the basis of continents

C. Trend by Methods

A similar analysis was carried out for methods used. Valuation methods were classified into monetary and non-monetary approaches which were further classified into subcategories. Alongside this classification, the studies were divided into spatial and non-spatial categories based on their usage of spatial or non-spatial strategies.

Chart 3.3 shows that the highest number of articles that were spatial in nature were direct market approaches including different market price and cost based approaches. The spatial studies show a higher trend towards benefit transfer approach than the non-spatial ones. In the case of a stated preference approach, contingent valuation was the primary method, contributing almost equally to the spatial and non-spatial studies. The direct market based approach and benefit transfer approach were more prevalent and dominated the spatial valuation studies. Benefit transfer is more dominant in spatial studies than non-spatial studies.

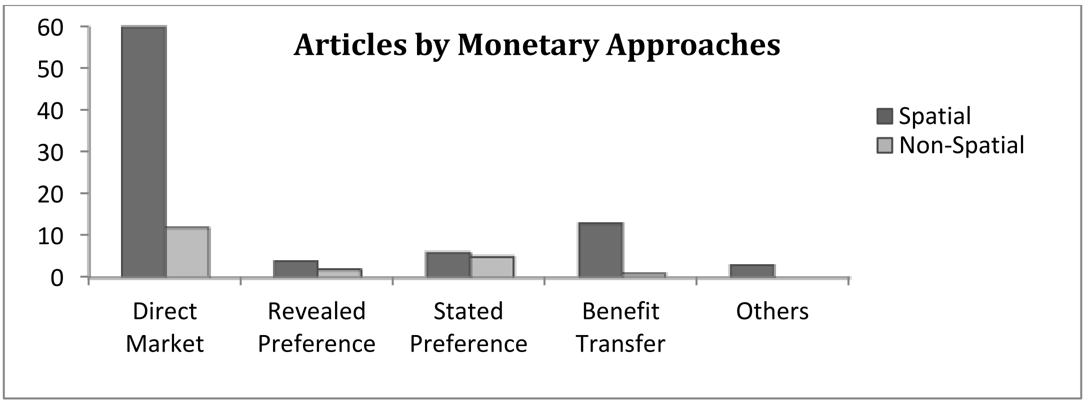


Chart 3.3: Distribution of articles on the basis of monetary approaches and spatial characteristics

Chart 3.4 shows the number of published articles employing spatial and non-spatial methods in relation to non-monetary valuation approaches. The public preference and expert panel approaches have predominantly been used in the spatially enabled studies.

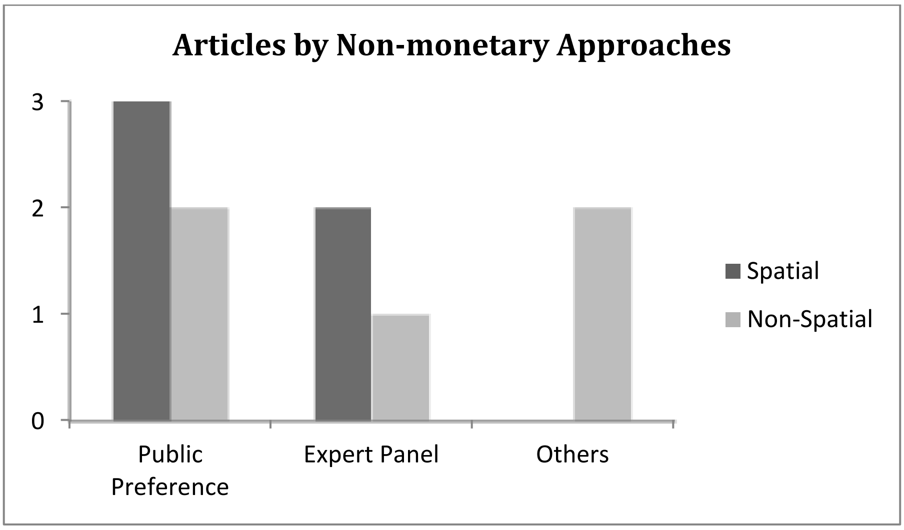


Chart 3.4: Distribution of number of articles on the basis of non-monetary approaches and spatial characteristics

D. Trend by Tools

Similarly, the trend of studies was analysed in relation to the tools and models used, thus indicating the types of tools more or less common in spatially enabled methods. A large number of studies used GIS tools for analysis purposes, although the published papers often did not provide specific details (software etc) of the tools used and thus here have been included in a general category of 'GIS Tools'. Several valuation studies used various valuation methodologies, but a specific software or tool was not utilised in the study, thus including it in the 'None' category. The None category studies were all non-spatial in nature. Out of 129 direct valuation articles, 69 used some kind of GIS tool, 17 of these used ArcGIS (ESRI) for analysis, and 14 used InVest as a valuation tool. There were 14 non-spatial studies that did not use any software or tools for analysis. In the case of analysing remote sensing data, ENVI (Harris GeoSpatial Solutions) was used the highest number of times, followed by IDRISI (Clark Labs). Chart 3.5 shows the trend of tools being used in the articles during the period of 2004 to 2014.

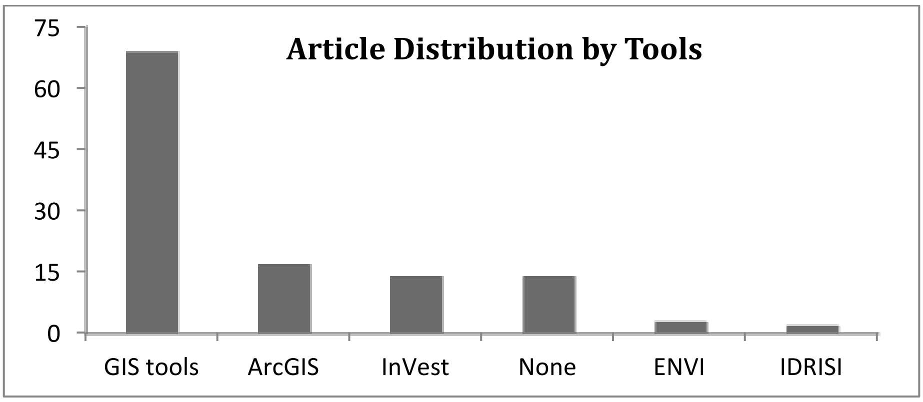


Chart 3.5: Distribution of articles on the basis of tools used for the period '2004 to 2014'

Figure 3.3 provides a deeper insight into a valuation tool known as InVest that has been widely used in the past decade. InVEST is used as an ecosystem service valuation tool. Initially, the tool was developed by the Natural Capital Project within the ArcGIS platform, but was later released as open source, requiring any GIS based desktop tool to operate and so making it more convenient to use. InVest contains sub-models addressing habitat quality, habitat risk assessment, marine water quality, carbon sequestration, blue carbon, water yield model and others that are diverse and cover various ecosystems, thus enabling it to be used by a larger scientific community.

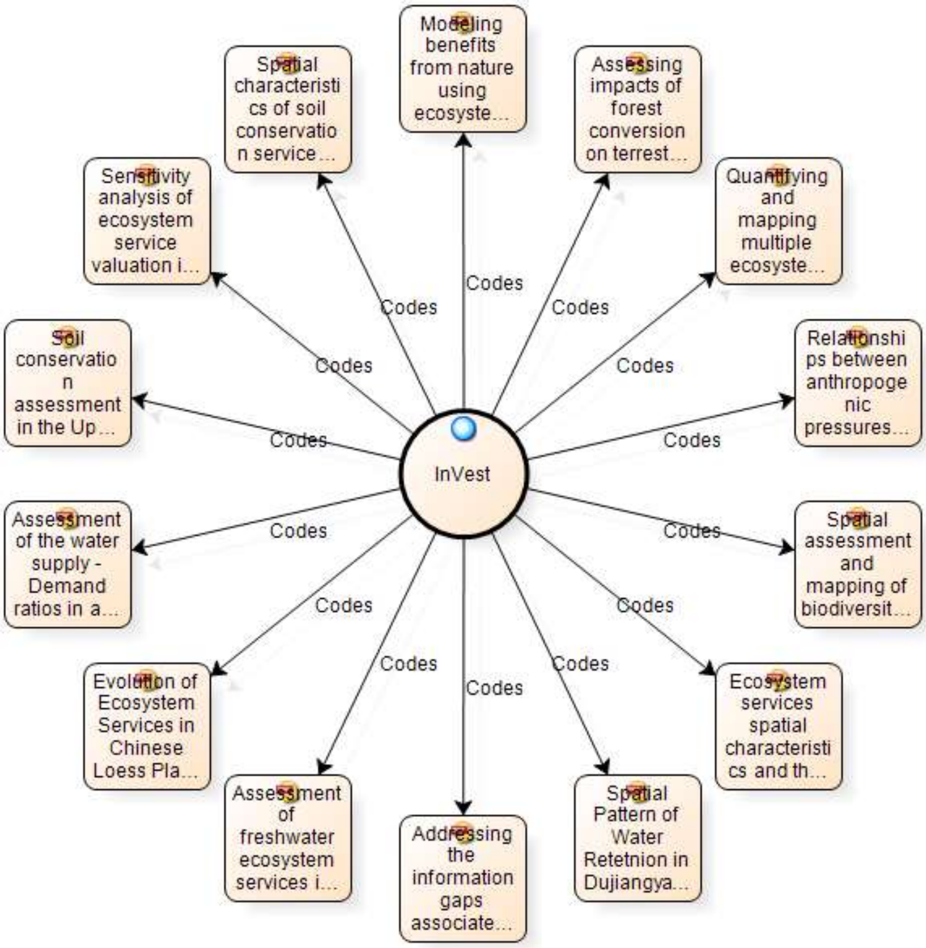


Figure 3.3: The InVest node linked with 14 articles that used InVest as a valuation tool

An analysis of the studies using InVest indicated that out of 14 total articles that used InVest for ecosystem services valuation, 13 were journal articles while one of them was a conference proceeding published in 2011. The studies were published from 2010 to 2014, and showed an increasing trend with time as shown in Chart 3.6. The highest numbers of publication were in journal Acta Ecologica Sinica as shown in Chart 3.8, followed by International Journal of Biodiversity Science, Ecosystem Services and Management and The Science of the Total Environment. Five articles reported study areas in China, two articles each reported study areas Spain and USA as shown in Chart 3.7.

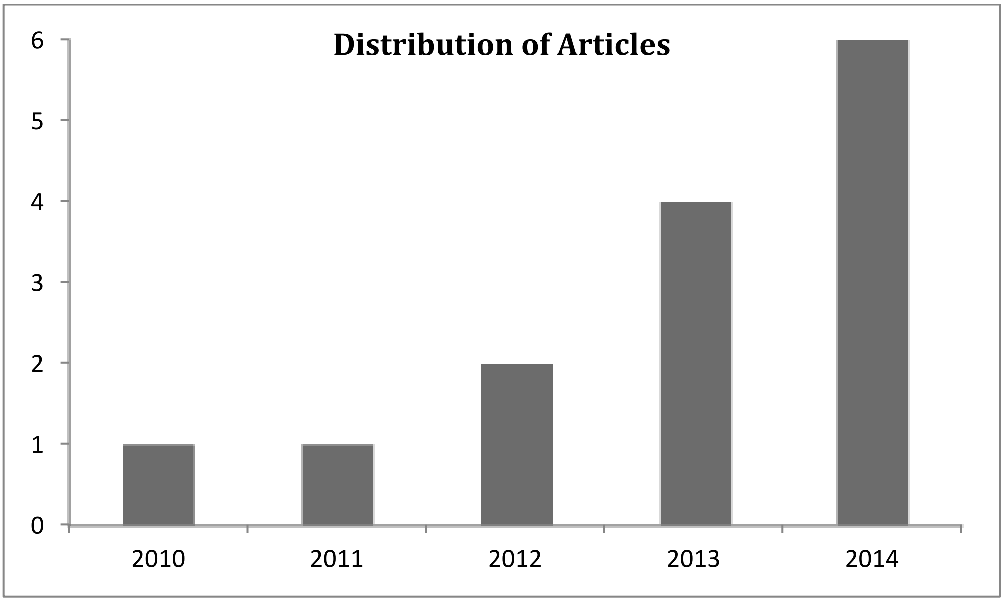


Chart 3.6: Distribution of InVest articles from 2004 to 2014

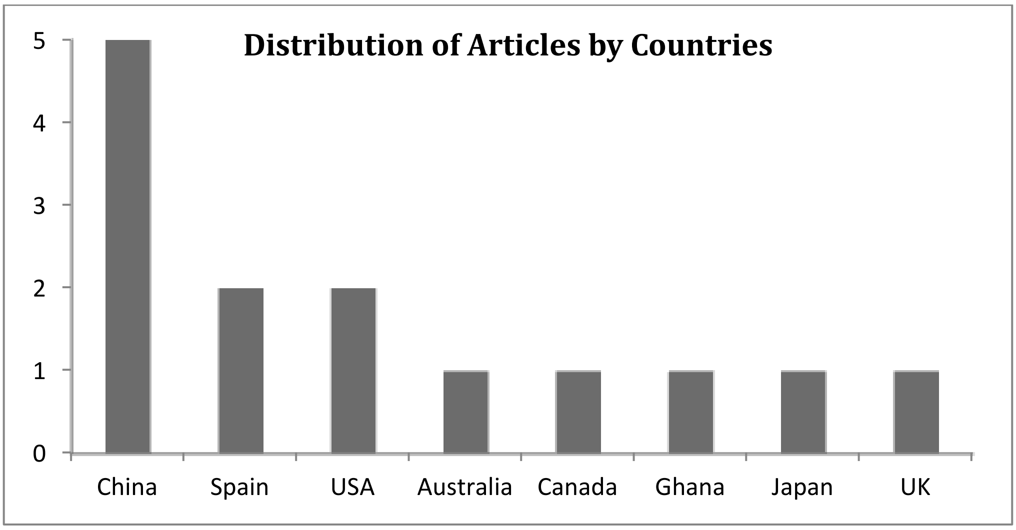
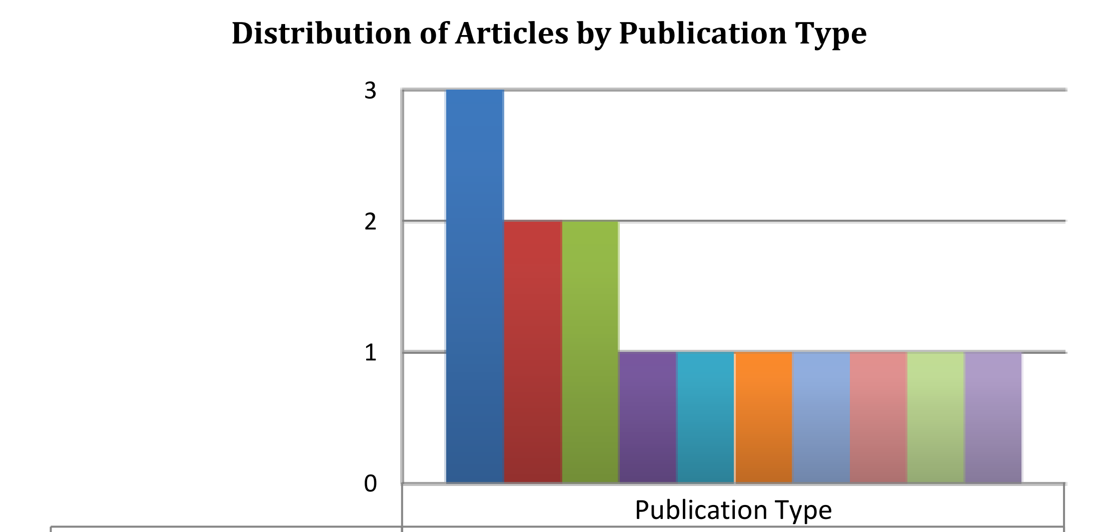


Chart 3.7: Distribution of InVest articles on the basis of study area

E. Trend of Publishing Activity

In order to review the overall trend of the publishing activity over the past decade, an analysis was first carried out on the number of articles and their distribution. Chart 3.9 shows that the trend has generally increased from 2004 to 2014 from less than 5 publications per year to more than 35 per year. Of 192 articles published, 174 were journal articles and 13 were conference proceedings. A list of journal titles in ascending order of number of publications was generated along with their impact factor. Table 3.5 shows that the highest publication rate (19) was in the journal Ecological Economics, followed by 17 in Acta Ecologica Sinica (having mainly studies with China as their study area), and 10 in Ecosystem Services. Chart 3.9 indicates the increasing number of articles in journals along with increasing number of cumulative impact factors for those articles from 2004 to 2014, which indicates the growing importance of these valuation studies over time.



	Publication Type
Acta Ecologica Sinica	3
International Journal of Biodiversity Science, Ecosystem Services & Management	2
The Science of the total environment	2
Agriculture, Ecosystems & Environment	1
Applied Geography	1
Ecological Indicators	1
Ecosystem Services	1
Global and Planetary Change	1
Journal of Great Lakes Research	1
ISWREP 2011 Proceedings	1

Chart 3.8: Distribution of InVest articles on the basis of publication type

Table 3.5: Distribution of articles on the basis of journal titles along with their impact factors

Journal Title	No of Articles	Impact Factor as per 2014
Ecological Economics	19	2.517
Acta Ecologica Sinica	17	0
Ecosystem Services	10	4.307
Chemistry and Ecology	5	1.047
Ecological Indicators	5	3.23

Journal Title	No of Articles	Impact Factor as per 2014
Ecology and Society	5	3.31
Ocean & Coastal Management	5	1.769
Annals of the New York Academy of Sciences	4	4.375
Environmental and Resource Economics	4	1.703
International Journal of Biodiversity Science, Ecosystem Services & Management	4	0.77
Journal of environmental management	4	3.188
Environmental management	3	1.648
Environmental Science & Policy	3	3.514
International Journal of Sustainable Development & World Ecology	3	1.771
Landscape and Urban Planning	3	2.606
PloS one	3	3.53
The Science of the total environment	3	3.163

Charts 3.10 and 3.11 show an analysis of publication activity from 2004 to 2014. Chart 10 shows the distribution of articles on the basis of publication type. Chart 3.11 shows the frequency of words used in articles and illustrates the significance of the words ecosystem, services, value, water, land, use etc. in spatial valuation studies and therefore, can be suggestive keywords for publication purposes.

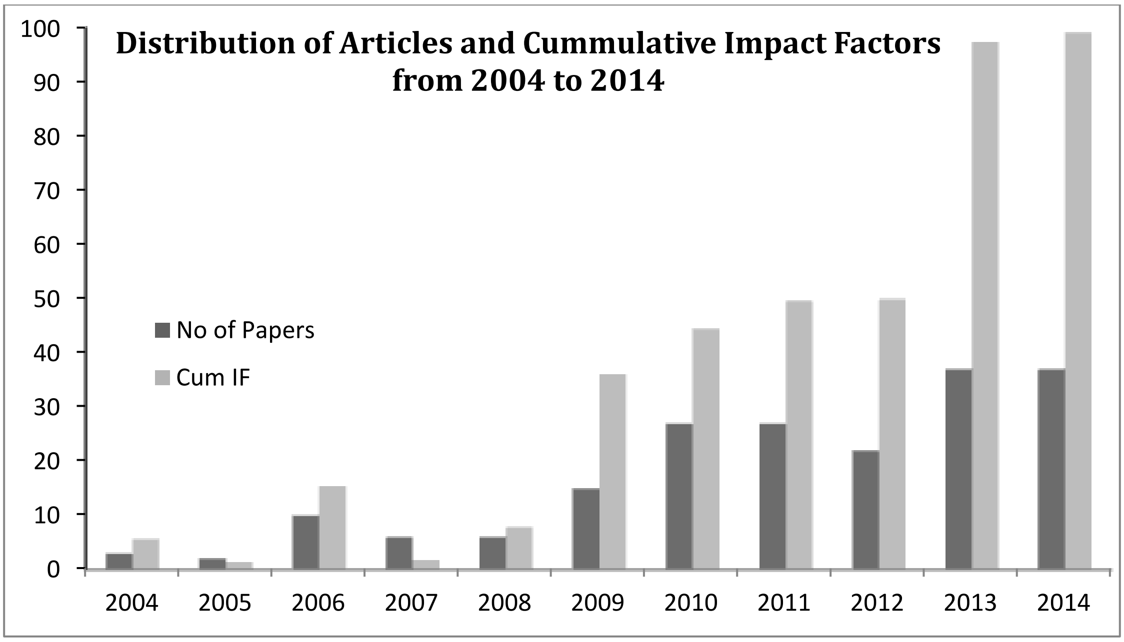


Chart 3.9: Distribution of spatial ecosystem services valuation articles and their cumulative impact factors from 2004 to 2014

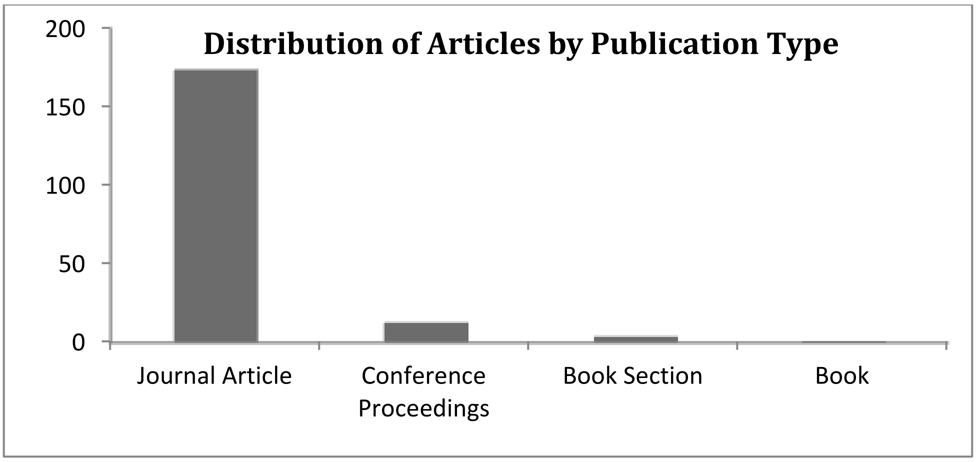


Chart 3.10: Distribution of articles on the basis of publication type for the period ‘2004 to 2014’

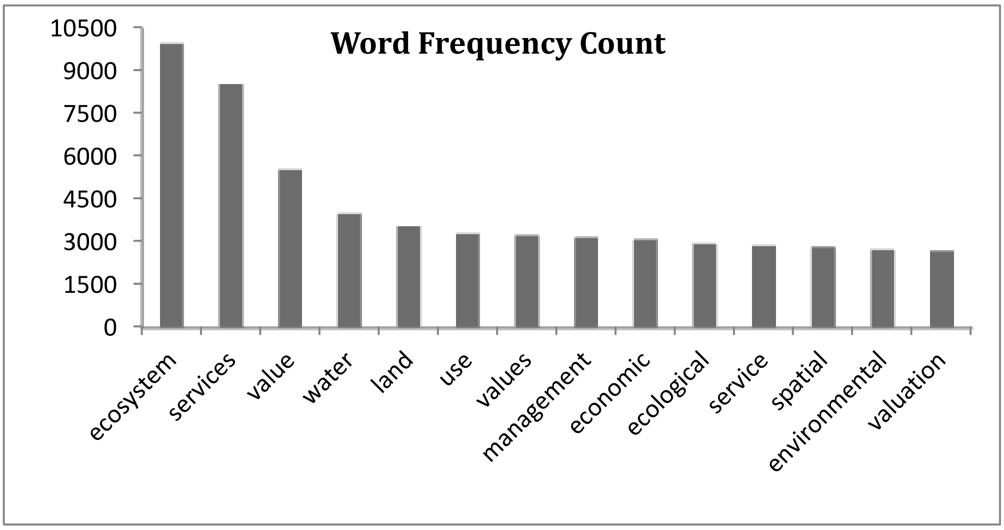


Chart 3.11: Frequency of words used in all published articles along with their number for the period ‘2004 to 2014’

3.3 Discussion

With the help of bibliographic content analysis, various trends in spatial valuation studies in the last decade from 2004 to 2014 have been highlighted. It is evident from the increasing trend of publication of ecosystem services valuation studies ranging from less than 5 in 2004 to more than 35 in 2014 that there has been an increase in interest among researchers and scientists of valuation of ecosystem services. The study also indicates that the focus of valuation studies has largely been on marine, coastal regions, wetlands, mangroves and watersheds during this decade and, for the terrestrial studies, a primary interest in forest valuation studies in terrestrial land uses.

The study also shows that valuation studies framed in the context of Chinese landscapes were highest in number. The Elsevier journal of Acta Ecologica Sinica is authorised by the China Association for Science and Technology and sponsored by the Ecological Society of China, and is shown to be a preferred journal for these Chinese studies. Most of these studies used benefit transfer method, and were based on Constanza’s highly cited paper (Costanza *et al.* 1997), together with an equivalent weight factor matrix derived from a survey of more than 200 Chinese scholars (Xia *et al.* 2003, Xie *et al.* 2003, Zhao *et al.* 2004, Amut *et al.* 2006, Hao *et al.* 2012, Liu *et al.* 2012). These studies have received some criticism due to discrepencies in uncertainties and standardisation of methods; nevertheless they have also been praised for raising public awareness in China of the value of ecosystem services (Bao *et al.* 2007, Zhang *et al.* 2010, Zhang and Liu 2011, Siew and Döll 2012).

The study also indicates that overall the highest number of publications have been reported in the journal of Ecological Economics, followed by in Acta Ecologica Sinica and Ecosystem Services. Focussig on the tools, models and softwares used in spatial valution studies, it is evident that ArcGIS as a GIS software has been widely used; followed by InVEST, an ecosystem services valuation tool; ENVI and IDRISI as remote sensing software. As a valuation tool, InVEST is employed in the highest number of studies and has been used in many countries. This is at least in part likely to be due to its multi-functional models, its generalizability, and because it is a free, open source modular software, where the users can employ the sub-models they are interested in, instead of the whole package (McKenzie *et al.* 2012).

The study has highlighted that, for monetary valuation methods, the direct market based approach and the benefit transfer approach are more prevalent, and dominate the spatially enabled valuation studies, whereas revealed preference studies are the least prevalent in terms of the number of publications. The stated preference approach was the third in prevalence. Within the stated preference approach, the contingent valuation method was found to be the primary method adopted. In case of non-monetary valuation methods, public preference and expert panels are the two approaches most commonly used in the spatial realm.

3.4 Conclusion

This study highlighted trends in valuation studies over the decade 2004-2014, and particularly the methodological trends for studies that employed spatially enabled methods. The findings of this study informed the methodology adopted for the model used in the research reported here.

This research employs a non-monetary valuation method utilising a value typology and a survey conducted with diverse stakeholders involved with the Midlands region of Tasmania. This review presented in this Chapter informs that research, and particularly the design of a value typology. A modified form of a widely used value typology (Clement and Cheng 2006, Sherrouse et al. 2011, 2014, van Riper et al. 2012, Sherrouse and Semmens 2014) has been selected. This work reported in this chapter contributes to the research by providing foundational knowledge of ecosystem services values, valuation methods and their trends in the geospatial realm in order to make informed decisions on selection of valuation method and value typology.

The purpose of this chapter is not to critique current trends in spatial valuation studies within a broader context of ecology or economics, but to describe and summarise current prevalent valuation methods that have capacity for spatial analysis and capacity for capturing non-monetary values. This has informed the methodology developed for the research, particularly the decision to employ a semi-structured questionnaire and the choice of value typology.

4 Spatially Explicit Modelling Strategies in Ecological Economics

Geospatial methods provide an opportunity to model processes in a spatial and temporal context. Geospatial analysis is widespread in both the ecological and economic domains and substantial work has been carried out to develop integrated ecological economic models that are geospatially enabled. This work needs to be understood and communicated, not only to recognise the ways in which geospatial methods have the capacity to integrate diverse disciplines and combine inter-disciplinary tools, but also to focus on future directions for advancement in the field of spatially explicit ecological economic modelling strategies. This chapter provides a systematic literature review, carried out to identify and describe studies utilising spatially explicit modelling strategies in ecological economics, and therefore also the tools that can prospectively be utilised in such modelling.

4.1 Introduction and Background

The conservation of biodiversity and the maintenance of ecosystem services have both ecological and economic dimensions. Deriving strategies to conserve biodiversity and use the resources of ecosystems in a sustainable manner requires the integrated consideration of both dimensions. Natural resource management has to manage various stakeholders and their interests, policies, politics, geographical boundaries, alongside economic considerations. It is difficult to satisfy all aspects at the same time and the process calls for negotiation and compromise (Opdam *et al.* 2006), with communication a key factor of success in multi-actor decision making (Treu *et al.* 2000, Von Haaren 2002, Opdam *et al.* 2006). Any decision related to ecosystem services requires the integration of knowledge from many fields, along with the active and transparent engagement of stakeholders (Guswa *et al.* 2014).

In the same way, many of the natural resource management concerns of local land users require a collective approach. This may be because individualized actions are either ineffective (cannot solve the problem) or inefficient (requiring efforts that are greater than the benefits they yield in return), or because local stakeholders have divergent interests that hinder easy solutions (German *et al.* 2012). Additionally, most natural resource management scenarios involve a close and complex interaction between natural ecosystems and social, economic, political and psychological factors, making them complex to model.

Wätzold *et al*. (2006) assert that it is not sufficient to first develop disciplinary models and then combine results from each discipline because each discipline poses the problem in its own way and comes up with its own most appropriate solution rather than a combined result. Further, this approach does not address the feedback loops between ecological, social and economic systems, either positive feedback loops in which secondary effects tend to reinforce a trend, or negative feedback loops which may be self limiting (Tietenberg and Lewis 2015a).

Geospatial methods provide an opportunity to model processes in a spatial and temporal context. Geospatial methods are widespread in both the ecological and economic domains and substantial work has been carried out to develop integrated ecological economic models that are geospatially enabled. Buenemann *et al.* (2011) argue that integrative geospatial approaches should be implemented for decision making processes as such approaches promote communication among scientists of diverse backgrounds, facilitate inter-institutional knowledge sharing and create synergy between government and private organisations working in the area.

Research has been carried out on integrating similar types of models e.g. integrating environmental models (Peach *et al.* 2011) or integrating two interrelated models (hydrologic and environmental models) to focus on a single issue, such as for addressing water resource management (Liu *et al.* 2008), hydrology (Kralisch *et al.* 2009), ground water studies (Steward and Bernard 2006), infrastructure management (Au-Yeung *et al.* 2009), or human health risk studies (Stewart and Purucker 2006). An integrated approach has also been carried out to analyse ecosystem interactions with land use change (Goetz *et al.* 2004). Similarly, the use of integrated geospatial models to assist policy making (Piorr *et al.* 2009) for urban planning (Chen *et al.* 2011) is not new. A diverse range of studies has been undertaken in different areas and at different scales (Chan *et al.* 2006, Naidoo and Ricketts 2006, Egoh *et al.* 2008, Maynard *et al.* 2010, Raudsepp-Hearne *et al.* 2010), indicating that spatial mapping and modelling can be vital tools for aligning land management with human needs (Cork *et al.* 2012). However, the development of decision support tools that integrate ecology, economics, and geography to support decision making is a more recent phenomenon (Söderbaum 2006, Daily *et al.* 2009, Bagstad *et al.* 2013).

Considerable progress has been made over the last decade in developing tools and models in order to represent ecosystem services in terms of ecology and economics. Although the theoretical foundations of ecological economics are still being debated (including for example whether ecological economics is transdisciplinary or interdisciplinary[10]) (Spash 2012), there clearly exists a crucial challenge of developing tools based on comprehensive interdisciplinary knowledge (Alberti *et al.* 2003, Lehtonen 2004, Daily *et al.* 2009, de Groot, Alkemade, *et al.* 2010b). The advancement of GIS technologies and the utilisation of spatially explicit data in the ecological economics of ecosystem services has provided a boost to the field (Holzman 2012). Current tools are of diverse nature, and extend from simple spreadsheets and databases, to frameworks, mapping tools, web-based tools and complex software packages (Bagstad

[10] Transdisciplinary research is defined as research conducted by investigators from different disciplines working jointly to create new conceptual, theoretical, methodological, and translational innovations that integrate and move beyond discipline-specific approaches to address a common problem (Aboelela *et al.* 2007). In Ecological Economics, transdisciplinarity has been defined as some kind of interrelationship between science and society ((Baumgärtner *et al.* 2008) following (Norgaard 1989, Costanza 1992, Røpke 2005b)). Interdisciplinary research is any study or group of studies undertaken by scholars from two or more distinct scientific disciplines. The research is based upon a conceptual model that links or integrates theoretical frameworks from those disciplines, uses study design and methodology that is not limited to any one field, and requires the use of perspectives and skills of the involved disciplines throughout multiple phases of the research process (Aboelela *et al.* 2007). In Ecological Economics, interdisciplinarity has been broadly understood as some kind of cooperation between scientific disciplines ((Baumgärtner *et al.* 2008) following (Norgaard 1989, Costanza 1992, Røpke 2005b)). There is also ongoing discussion of the meaning and intent of ‘interdisciplinarity’ and ‘transdisciplinarity’ (Stock and Burton 2011, Mauser *et al.* 2013).

et al. 2013). Some developers are attempting to integrate public and private sector decision making processes into ecosystem services analysis (e.g. SERVES, SolVES) (Sherrouse *et al.* 2011, 2014, van Riper *et al.* 2012, Sherrouse and Semmens 2014, Earth Economics 2016), while others are focussing on mapping the values of ecosystem services – usually through web-based tools (e.g. ARIES, EPM, EcoServ, InFOREST) (Bagstad *et al.* 2013, Baral *et al.* 2013, Jackson *et al.* 2013, Guswa *et al.* 2014, Häyhä and Franzese 2014, Boumans *et al.* 2015). The tools differ in varying ways, including their approaches to economic valuation, spatial and temporal representation of services, and incorporation of existing biophysical models (Bagstad *et al.* 2013).

4.2 Ecological Economics including Models and Tools

Although there has been increasing development of geospatial models in the field of ecological economics, there are a limited systematic reviews of these models, their respective characteristics, their capabilities and their scope of application. Some previous reviews focussed on detailed descriptions of a limited number of tools (e.g. Nelson & Daily 2010), some on one-off modelling approaches not intended for broader applicability (Tucker and Braat 2009, Egoh *et al.* 2012, Martínez-Harms and Balvanera 2012), and others on tools for ecosystem services excluding studies and models that encompass the broader perspective of ecological economics (Bagstad *et al.* 2013). Silva and Teixeira (2011) present a review describing the evolution of ecological economics over the previous two decades (1989 to 2009) and addressing research topics such as methodological issues, policies, governance and institutions, and valuation and discussing the emergent themes of research in this area.

Among other reviews, Power (2010) discusses ecosystem services in agricultural landscapes and notes the need for spatially and temporally explicit tools to tackle trade-offs in agricultural systems and to improve the ability to estimate the provision of ecosystem services. A review by Häyhä and Franzese (2014) addresses methodological approaches used to assess the flows of ecosystem services and highlights the need for interdisciplinary and systems perspectives when dealing with ecological and socio-economic problems, although they do not target spatially explicit tools or studies. While focussing on methodological approaches in ecological economics, another review by Wätzold *et al.* (2006) is noteworthy as it addresses potentials and pitfalls in the field of ecological economics and suggests future directions.

In the context of reviews, certain non-peer reviewed publications warrant mentioning, including Turner *et al.* (2014) who assess existing data, models, and knowledge, and review the availability of data in a spatially explicit form for integrated modelling and analysis. Their paper also reviews computer models that could be useful for analysing and valuing land management options including farm and site scale models (CropSyst, DNDC, APSIM, CENTURY, DAYCENT and EPIC), watershed and regional scale models (APEX, DSSAT, STICS, LPJmL, ORCHIDEE, Biome-BGC and CARAIB), climate change models (INFFER and CLUE), integrated global models (World3, IMAGE, IMAGE-2, Ifs, DICE, TARGETS and GUMBO), and ecosystem services models (ESR, Integrated Valuation of Ecosystem Services, InVEST, ARIES, LUCI Polyscape, MIMES, EcoServ, Co$ting Nature, SolVES, Envision, EPM, InFOREST, EcoAIM, ESValue, EcoMetrix, NAIS, Ecosystem Valuation Toolkit, Benefit Transfer and Use Estimating Model Toolkit).

Other reviews highlighting ecosystem services tools include Bagstad *et al*. (2012), who also compare results from InVEST and ARIES in a case study, and BSR (2011) that carries out a comparative tools assessment of ecosystem services. Bagstad *et al*. (2013) present a comparison of 17 ecosystem services tools and models against eight evaluative criteria. Some of these tools were only spreadsheets (ESR, EcoMetrix) having no spatial capabilities, others were databases of ecosystem services valuation useful for the purpose of transferring these values to other studies and locations through a benefit transfer method (Benefit Transfer and Use Estimating Model Toolkit). Their study was based on a review of tools for ecosystem services including studies having multiple ecosystem services, and so excluded studies lacking a central focus on ecosystem services or from a larger domain of ecological economics, thus excluding many models this study was interested in from a geospatial perspective.

In Australia, a very comprehensive and a detailed description and review of ecosystem services has been written in a report by Cork *et al.* (2012) that highlights various aspects of ecosystem services including related activities currently underway in Australia and worldwide. Baral *et al.* (2013) review approaches for measuring and managing ecosystem goods and services in the changing landscapes of south-east Australia by analysing two case studies. The case studies involved three stage scenarios of land use change, with change in production of ecosystem services analysed at three time frames i) pre-European condition native vegetation, ii) pre 1970s or conversion to pasture and iii) recent conditions or post 1970s conversion to plantations. The study also identifies important ecosystem services in the study area and provides a summary of recent studies carried out on ecosystem services in south-east Australia.

4.3 Objective of the Study

The aim of this chapter is to review diverse spatially enabled models and tools that have application to the field of ecological economics. This review includes valuation studies that have a spatial means of representing the valuation of ecosystem services. Studies addressing theoretical arguments or conceptual frameworks that did not support practical implementation of a spatially explicit model were excluded. The purpose was to conduct a detailed review of spatially explicit modelling strategies to identify patterns and to propose future directions for research. The review draws from existing research, publications, and case studies to summarise current practices for a proposed spatially explicit integrated model.

4.4 Methods and Design

A systematic procedure was employed to analyse the accumulated literature. Only published peer reviewed research papers were considered, and these were further scrutinised to extract those studies that were application papers and had significant spatial, ecological and economic components. To seed the search, literature databases were searched using various keywords (e.g. spatial, model, ecological economics, ecosystem services, land use, landscape, integrated, decisions, ecosystems, ecology, economics etc.) and using the Windows Explorer search tool and the PDF-XChange

Viewer 3.0 (Tracker Software Products)[11]. In this way, 186 studies were selected in total for further examination.

All the papers were thoroughly examined and studies that were application papers were set aside, removing papers dealing only with theory, conceptual frameworks, arguments, and judgements. Again, only papers that had significant components of the three key aspects, i.e. spatial, ecological and economic, were considered. This reduced the literature to 65 candidate studies that were application papers, and involved all three of spatial, ecological and economic components.

A reference database of these 65 studies was created in Mendeley (version 1.15)[12]. It was then exported from Mendeley in RIS format to be used as a classification sheet in NVivo (version 10 developed by QSR International)[13]. NVivo was then used to export the classification sheet as a .xlsx format Microsoft Excel based spreadsheet. The spreadsheet was then extended with criteria columns, across which each research paper was analysed.

The intention of this review was to capture the main aspects of spatially enabled models, particularly the type of strategy used, software utilised, analysis scales and model/study outputs. For this purpose, the studied models were classified under a set of criteria variables, out of which the variables listed in Table 4.1 were of prime importance.

Table 4.1: Criteria variables used for analysing literature

• Strategy used	• Analysis scale
• Software used	• Model type
• Management problem	○ Applied as software
• Output accomplished	○ Not applied as software
• Study area	

Appendix A 'Detail Charts' provides the classification sheets extracted from the review. Appendix B: 'Detailed Descriptions of Models' provides a detailed description of the models, their mode of implementation (whether 'standalone' independent models, models that are extensions of ArcGIS or ArcView, or whether the models are web-based), details of where they have been applied, and to what types of management problem. Appendix C: 'Modelling Strategies undertaken in Tasmania' reports a short review of research undertaken in Tasmania. The detailed descriptions in these Appendices inform the summaries and conclusions provided below.

4.5 Results and Analysis

An overview of all the 65 studies was carried out in order to identify patterns in the collated studies. Patterns were observed in relation to modelling strategies, software used, management problem, study area, and analysis scale. This section summarises the findings, supported by the detailed information provided in Appendices B and C.

[11] PDF-XChange Viewer is an open source pdf file viewer that has a powerful search tool that can search within pdf files.
[12] Mendeley is an open source reference manager.
[13] NVivo is a computer aided qualitative data analysis software (CAQDAS).

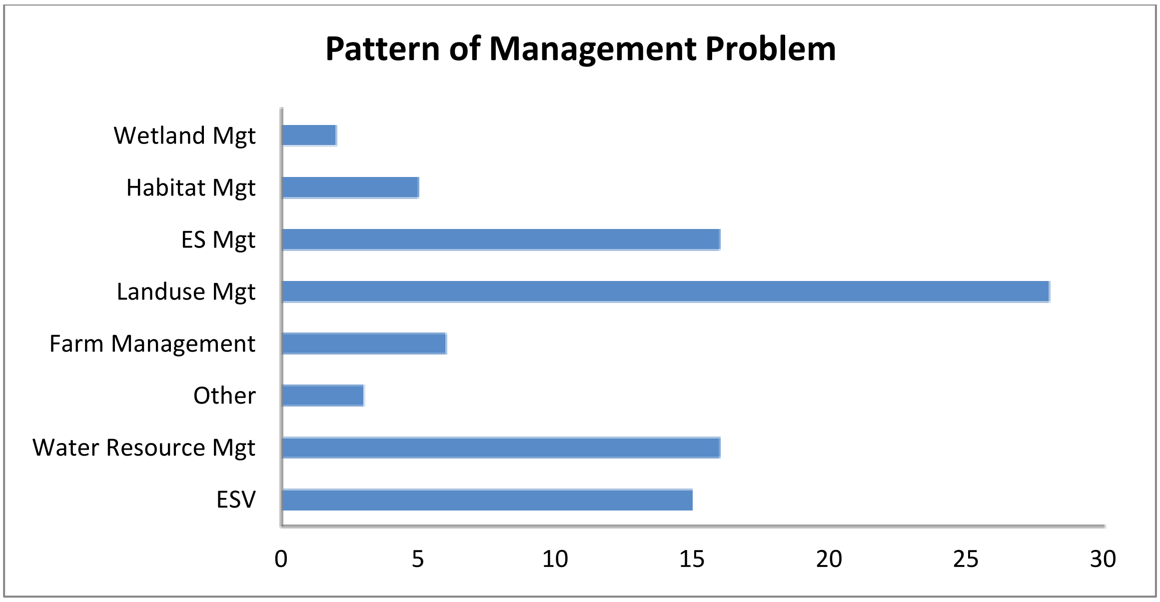


Chart 4.1: Pattern of management problems in the collated studies, where x-axis represents the total number of publications over the period '1995 to 2015'

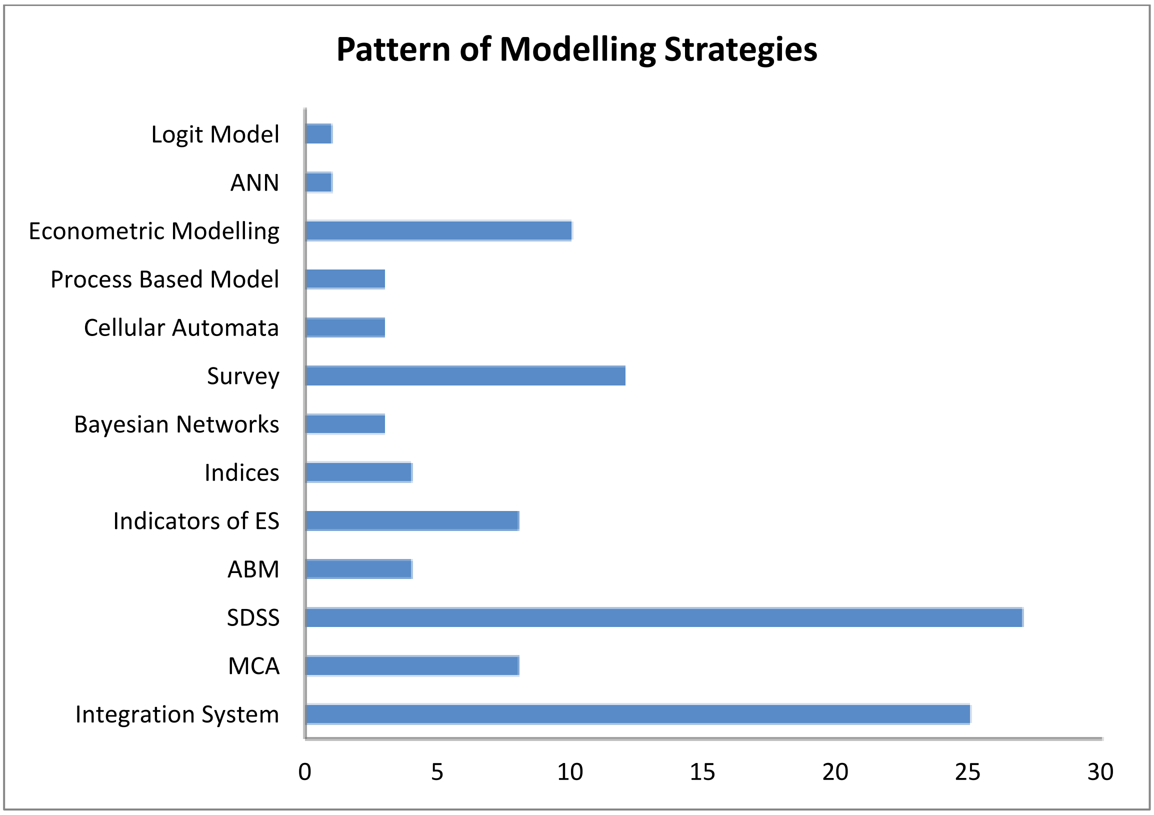


Chart 4.2: Pattern of modelling-strategies in the collated studies, where the x-axis represents the total number of publications over the period '1995 to 2015'. ANN stands for Artificial Neural Networks, ES stands for Ecosystem Services, ABM stands for Agent Based Modelling, SDSS stands for Spatial Decision Support Systems and MCA stands for Multi-criteria Analysis

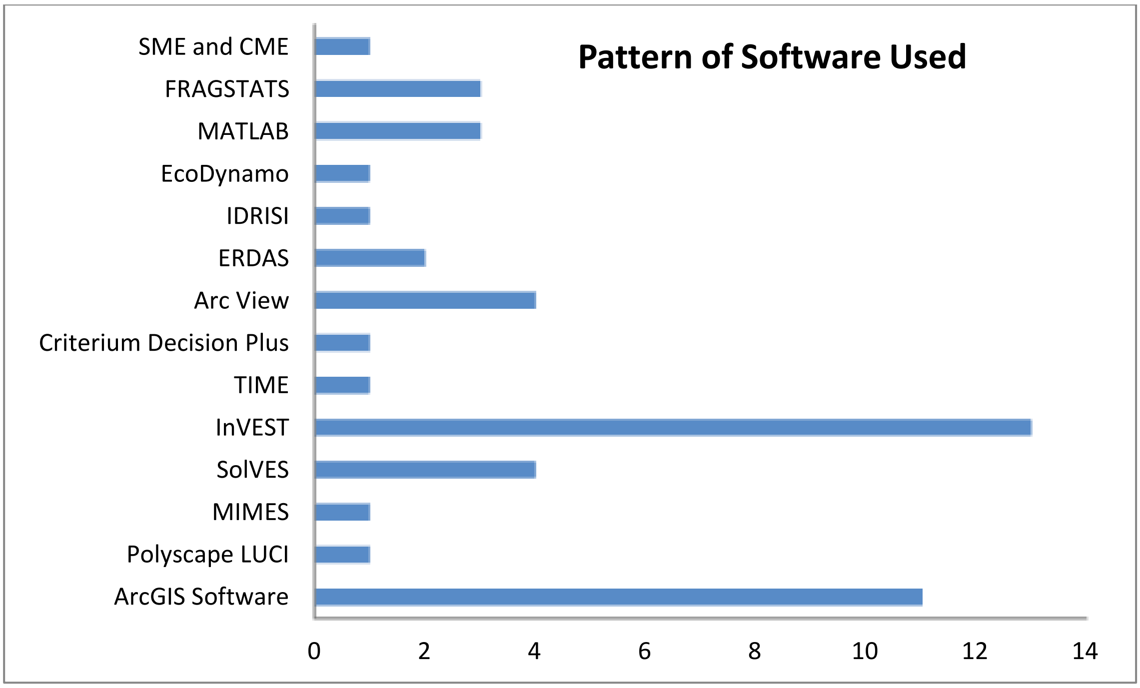


Chart 4.3: Pattern of software used in the collated studies, where the x-axis represents the total number of publications over the period '1995 to 2015'

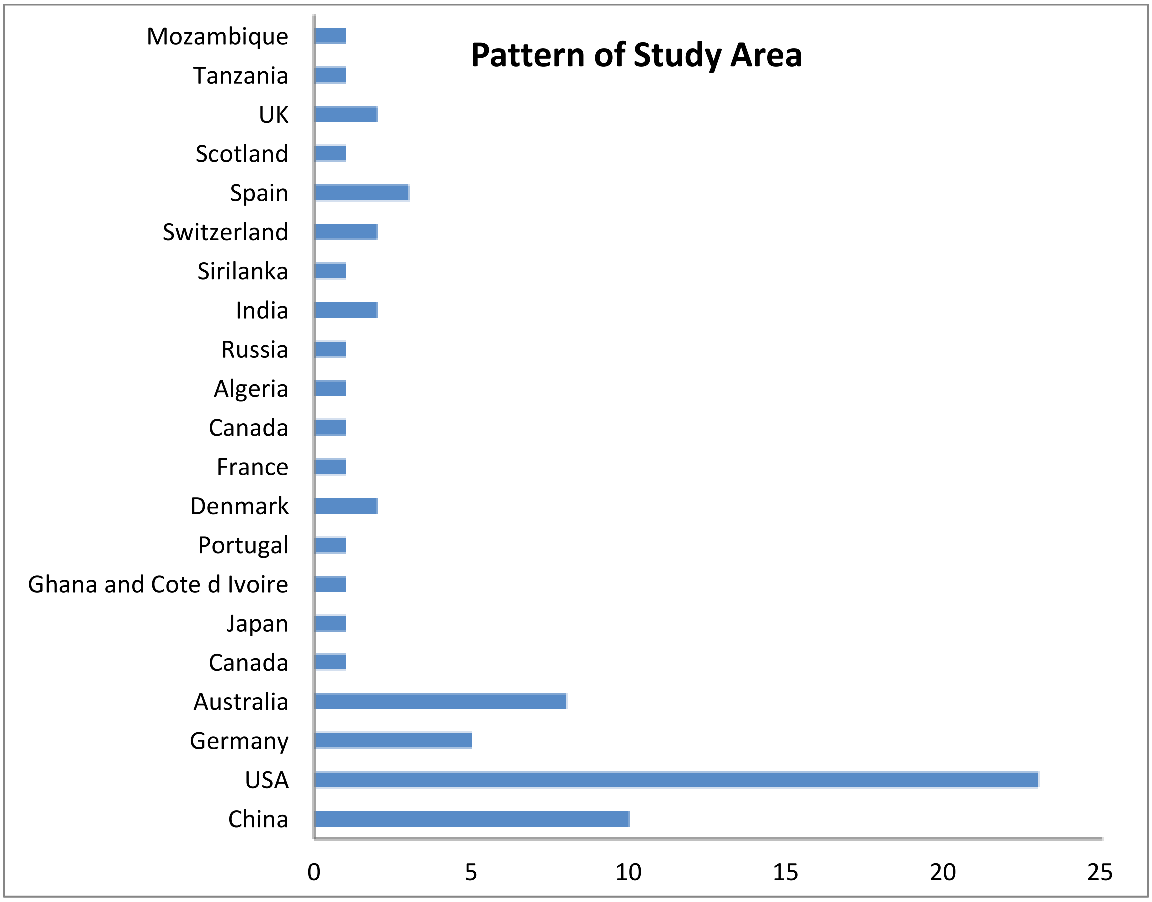


Chart 4.4: Pattern of study areas in the collated studies, where the x-axis represents the total number of publications over the period '1995 to 2015'

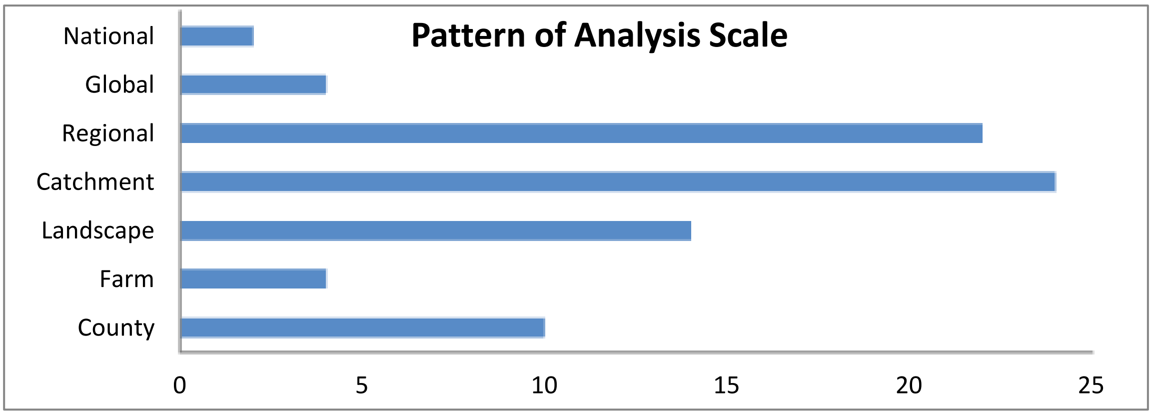


Chart 4.5: Patterns of analysis scale in the collated studies, where the x-axis represents the total number of publications over the period '1995 to 2015'

Chart 4.1 indicates a pattern of management problem in the studies, with land use and ecosystem management dominating others. Chart 4.2 describes the pattern of modelling strategies in the selected 65 studies, where the number of publications for each modelling strategy is reported over the period '1995 to 2015'. The chart indicates that Spatial Decision Support Systems (SDSS) were most common in these studies, followed by Integration Systems (i.e. studies that developed a system, a platform or a framework to integrate various disciplinary models or to integrate useful information from spatial, ecological or economic domains). Econometric modelling, surveys, indicators of ecosystem services (ES), and Multi-criteria Analysis (MCA) also figure prominently. Chart 4.3 shows the software preferences across the collated studies, with high use of InVEST and ArcGIS software, but noting also common use of SolVES and ArcView (previous version of ArcGIS).

Chart 4.4 illustrates the pattern of the studies on the basis of study area and shows that the majority of studies were carried out on U.S. landscapes, followed by Chinese, Australian and German landscapes. Chart 4.5 illustrates the pattern of the studies based on the scale of analysis, and shows that regional and catchment level studies dominated others, whereas, farm level and national level studies were least evident.

The review data consisted of 65 spatially explicit application papers. The data was divided into two high level categories:

1. A category of papers that were based on ecosystems services valuation (ESV)
2. A category of papers that were intended for applications other than ecosystem services valuation (Others).

This categorisation was carried out to set aside studies that involved ecosystem services valuation from those that involved other aspects of ecological economics. The intent was to capture other aspects of ecological economics; the models in the ESV category studies were primarily intended for ecosystem service valuation and had been addressed in Chapter 3. The focus of the work reported in this chapter was to investigate other modelling strategies applied in the spatial domain. Out of the 65 studies, 15 were from the ESV category, and the remaining 50 were studies that did not involve ecosystem services valuation. Eight studies from the total of 65 were framed in the context of Australian landscapes.

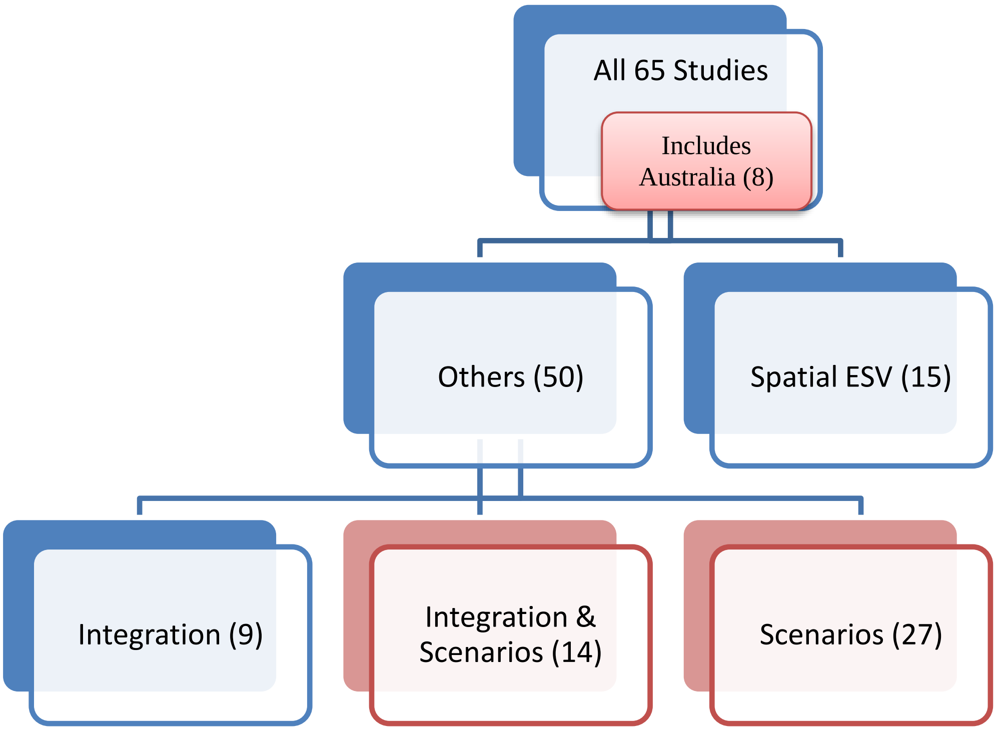


Figure 4.1: Categorisation of studies on the basis of generated output

As the intent of this review was to inform a proposed spatially explicit model that describes farmers' land use decisions, a model that was flexible and followed a scenario-based approach[14] was required. Following Kelly *et al.* (2013), who categorised models on the basis of generated outputs, the 50 studies in the category 'Other' indicated in Figure 4.1 were further categorised on the basis of generated outputs. From the 50 studies, the scenario-based studies were separated to examine in detail. The remaining studies were categorised as primarily addressing integration of either knowledge or models in a geospatially enabled manner, or a combination of this and scenario generation. Thus, to summarise, the categories adopted were 1) Scenarios[15], 2) Integration, and 3) Integration & Scenarios, indicating whether the study resulted in:

i. the generation of scenarios
ii. an integration of knowledge or models through spatial means, or
iii. both integration of knowledge or models along with scenario generation.

The review particularly focused on studies that involved scenario-based approaches; that is the categories Scenarios (27) and 'Integration & Scenarios' (14) (highlighted in red in Figure 4.1), together with a particular interest in those studies that addressed Australian landscapes: 'Australia' (8). The findings for each of the highlighted categories

[14] **Scenario-based** approaches consider the impacts of implementing management interventions or decision options involving 'what if' analysis. The approach allows the users to explore the results of various actions or policies and the effect and associated trade-offs (Kelly *et al.* 2013). Scenario based outputs have been recognised as one of the types of outputs generated in integrated modelling environments (Kelly *et al.* 2013).

[15] **Scenarios** are storylines that describe possible futures. They explore aspects of and choices about, the future that are uncertain. To tell the story, scenarios can include qualitative descriptions of changes and quantitative representations (McKenzie *et al.* 2012).

shown in Figure 4.1 are discussed under their respective headings; detailed analysis charts for all the categories are provided in Appendix A.

4.5.1 Modelling that generated scenarios

This category comprised studies that were not ESV, and generated outputs in the form of scenarios, as shown in Figure 4.1. In this category there were 27 studies. The patterns that emerged from these studies were that:

- they are predominantly based on SDSS rather than other strategies
- they are all dynamic rather than static
- the models developed are largely implemented as software rather than not implemented as software
- they mostly work on runtime, and show temporal variations
- they mainly use mainstream spatial analysis software (such as ArcGIS) along with other specialised ecological economic software (InVEST and SolVES)
- they were applied to problems at the catchment or regional scale rather than the local or landscape scale
- the predominant management problems were land use, ecosystem and water resource management
- they were conducted primarily in USA and China

It has been found that SDSS based strategy is widely prevalent among scenario based outputs generated from models implemented as softwares, whereas, SDSS as a strategy has not been used for the purposes of integration.

Table 4.2: Modelling strategies with their references in the Scenarios category

	Modelling Strategy	No. of papers	References
1	MCA	1	(Liacc 2007)
2	SDSS	21	(Lant *et al.* 2005, Liacc 2007, Isely *et al.* 2010, Ren *et al.* 2011, Sherrouse *et al.* 2011, 2014, Sánchez-Canales *et al.* 2012, van Riper *et al.* 2012, Guerry *et al.* 2012, Leh *et al.* 2013, Rao *et al.* 2013, Su and Fu 2013, Fu *et al.* 2013, Jackson *et al.* 2013, Baral *et al.* 2014, Boithias *et al.* 2014, Sherrouse and Semmens 2014, Wang *et al.* 2014, Dhakal *et al.* 2014, Hoyer and Chang 2014, Boumans *et al.* 2015)
3	ABM	2	(Liacc 2007, Karali *et al.* 2011)
4	Indicators	1	(Wang *et al.* 2014)
5	Indices	2	(Dai *et al.* 2005, Dhakal *et al.* 2014)
6	Survey	2	(Lubowski *et al.* 2006, Karali *et al.* 2011)
7	Process Based Modelling	1	(Voinov *et al.* 1999)
8	Econometric Modelling	4	(Lubowski *et al.* 2006, 2008, Radeloff *et al.* 2012, Sherrouse and Semmens 2014)
9	ANN	1	(Dai *et al.* 2005)

4.5.2 Modelling that involved integration and that generated scenarios

This category comprised studies that were not ESV, involved integration of knowledge or models, and generated scenarios, as shown in Figure 4.1. In this category, there were 14 studies. The patterns that emerged from these studies were that:

- they predominantly use integration as a strategy as opposed to other strategies, followed by SDSS
- they are predominantly dynamic rather than static
- some models were implemented as software (6 out of 14); others were not (8 out of 14)
- they were applied to problems at the catchment or landscape scale rather than the regional scale
- they were conducted predominantly on German and Australian landscapes.

Table 4.3: Modelling strategies with their references in the Integration & Scenarios category

	Modelling Strategy	No	References
1	Integration system	14	(Schou *et al.* 2000, Costanza *et al.* 2002, Goetz *et al.* 2004, Münier 2004, Hill, Braaten, *et al.* 2005, Volk *et al.* 2007, 2008, Rudner *et al.* 2007, Chen and Wu 2009, Meyer *et al.* 2009, Wang *et al.* 2010, Li *et al.* 2010, Bohnet *et al.* 2011, Daloğlu *et al.* 2014)
2	MCA	4	(Hill, Braaten, *et al.* 2005, Chen and Wu 2009, Meyer *et al.* 2009, Wang *et al.* 2010)
3	SDSS	6	(Hill, Braaten, *et al.* 2005, Rudner *et al.* 2007, Volk *et al.* 2007, 2008, Wang *et al.* 2010, Bohnet *et al.* 2011)
4	ABM	2	(Hill, Braaten, *et al.* 2005, Daloğlu *et al.* 2014)
5	Indicators	1	(Meyer *et al.* 2009)
6	Bayesian Networks	2	(Volk *et al.* 2007, Daloğlu *et al.* 2014)
7	Cellular Automata	2	(Goetz *et al.* 2004, Chen and Wu 2009)
8	Process Based Modelling	1	(Costanza *et al.* 2002)
9	Econometric	3	(Schou *et al.* 2000, Volk *et al.* 2008, Wang *et al.* 2010)
10	Logit Model	1	(Daloğlu *et al.* 2014)

4.5.3 Australian Research

This section describes the findings for research that was framed in the context of Australian landscapes. A lesser review was undertaken of research undertaken in Tasmania; this is reported in Appendix C.

Relevant publications on ecosystem services in scientific journals and international conferences increased markedly since 2000 (Williams 2009). Among the publications, a recent review on the practical use of ecosystem services in natural resource management (Plant and Ryan 2013) provides an insight into future trends in Australia. The study was based on a literature review, interviews and observations from Australian regional planning and offers a snapshot of the experiences of resource managers engaging with the ecosystem services concept, highlighting factors that may have enabled or prevented their adoption. One of the major barriers for the adoption of the ecosystem services concept identified through the interviews was the lack of an Ecosystem Services toolkit (Plant and Ryan 2013). Recently efforts have been made, particularly in the area of developing toolkits that provide information on the value of ecosystem services, that can in turn support decision making for policy and management (Daily *et al.* 2009, Bagstad *et al.* 2013). Although some of these tools are available (Bagstad *et al.* 2013), there is little evidence of implementation at the local level (Plant and Ryan 2013), which suggests lack of awareness or possibly difficulty in access or implementation of tools. This indicates the need to not only focus on both the utility and implementation of tools, but also on the means by which they can be implemented.

In this context, recently introduced tools like InVEST, ARIES etc. are worth mentioning. The following section describes eight such studies that were undertaken in Australia using spatially enabled ecological economic models. The trends that emerge from examination of these Australian research studies are that:

- they are predominantly dynamic rather than static
- the models developed are predominantly implemented as software rather than not being implemented as software
- the models are mostly capable of working on runtime
- they mainly use mainstream spatial analysis software (such as ArcGIS) along with other specialised ecological economic software (InVEST and SolVES)
- they were applied to problems at the catchment or landscape scale rather than the farm scale or regional scale
- the predominant management problems were water resource management, land use, and ecosystem management
- they predominantly used SDSS strategy and integration strategy as opposed to other modelling strategies.

Table 4.4: Modelling strategies with their references for studies from Australia

	Modelling Strategy	No, of papers	References
1	ESV	2	(Kragt *et al.* 2011, Butler *et al.* 2013)
2	Integration system	4	(Mallawaarachchi *et al.* 1996, Hill, Braaten, *et al.* 2005, Chen and Wu 2009, Bohnet *et al.* 2011)
3	MCA	2	(Hill, Braaten, *et al.* 2005, Chen and Wu 2009)
4	SDSS	4	(Hill, Braaten, *et al.* 2005, Bohnet *et al.* 2011, van Riper *et al.* 2012, Baral *et al.* 2014)
5	ABM	1	(Hill, Braaten, *et al.* 2005)
6	Indicators	2	(Kragt *et al.* 2011, Butler *et al.* 2013)
7	Indices	1	(Butler *et al.* 2013)
8	Bayesian Networks	1	(Kragt *et al.* 2011)
9	Survey	2	(Kragt *et al.* 2011, van Riper *et al.* 2012)
10	Cellular Automata	1	(Chen and Wu 2009)

Mallawaarachchi *et al.* (1996) present an integrated approach through a State-wise Resource and Information and Accounting System (SRIAS) designed for the analysis of policy options for managing land degradation in New South Wales, Australia. The system was developed for a strategic policy and regional scale and is not suitable for policy implementation requiring detailed information at local scales. In their work, SRIAS was used to assess the opportunity costs of land degradation associated with sheet and rill erosion over farmland in the Lachlan catchment in NSW. SRIAS uses GIS methodology and is capable of addressing broad-scale resource, environmental and economic policy questions through trans-disciplinary modelling. The model generates land use maps, value of production maps, soil erosion estimates and an estimation of the value of foregone productivity. SRIAS was developed in 1991 by the Australian Commonwealth Scientific and Industrial Research Organisation (CSIRO) and was used to address policy questions about sustainability and bio-diversity.

Other studies (Hill, Braaten, *et al.* 2005, Chen and Wu 2009, Baral *et al.* 2014) employ different methods (ASSESS, AHP-CA-GIS and InVEST) for selection of suitable sites (Hill, Braaten, *et al.* 2005, Chen and Wu 2009) and priority areas (Baral *et al.* 2014) and generating spatially explicit maps. Hill *et al.* (2005) and Chen and Wu (2009) uses AHP (The Analytical Hierarchy Process) as a means of multi-criteria analysis. Hill *et al.* (2005) develop a model named ASSESS (A System of Selecting Suitable Sites) that provides an interface in the ArcInfo Grid GIS environment and enables addition and combination of data layers that are quantised and ranked in AHP. ASSESS has been implemented in the Radwaste project for selecting disposal sites for low level radioactive material, in MDBSIS project (Murray Darling Basin Soil Information System) as a soil information system for selecting suitable sites for agriculture, and in CatCon project to assess the biophysical condition of a catchment. Chen and Wu (2009) uses AHP and CA (Cellular Automata) for spatial simulation of multiple information layers for resultant suitability layers. The tool is called AHP-CA-GIS developed using C#.NET computer language. The results represent land use suitability potential for irrigated agriculture in the Macintyre Brook Catchment, Southern Queensland, Australia. Baral *et al.* (2014) measure biodiversity values to identify conservation priority sites in fragmented landscapes and uses, Patch Analyst in ArcGIS to assess landscape fragmentation, and InVEST (Integrated valuation of Ecosystem Services and Tradeoffs) to identify habitat quality, resulting in areas of high biodiversity conservation value at less modified land cover sites and vice versa.

Another study has been undertaken (van Riper *et al.* 2012) on Hinchinbrook Island National Park, Australia, in which the social value of ecosystem services held by outdoor recreationists were mapped with the help of SolVES (Social Values for Ecosystem Services). SolVES uses social values and integrates them to point based spatial locations, resulting in a spatial distribution of point density of social values attached to ecosystem services.

Several Australian studies have employed an integration framework that is intended to link spatially explicit disciplinary models. One such study (Bohnet *et al.* 2011) developed the Landscape Toolkit, which enabled integrated assessment of water quality, biodiversity and economic outcomes of stakeholder-defined land use and management change scenarios through integration of component models. The method was applied at the Tully-Murray catchment in the Great Barrier Reef (GBR) region of Australia. Another study (Butler *et al.* 2013) analyses trade-offs between multiple ecosystem services and stakeholders, also applied to land use and water quality management in the Tully-Murray Catchment and utilising model (e.g. N-SPECT), indicators and indices to value various ecosystem services using the Millennium Ecosystem Assessment. The study used four land use scenarios and the ecosystem services and stakeholders linked to those scenarios to identify cause-effect relationships, trade-offs and thresholds between services and stakeholders.

Kragt *et al.* (2011) describe a study that used an integrated model rather than coupling existing disciplinary models; in this case, knowledge about hydrological, ecological, and economic systems was integrated within a Bayesian Network. In application, this integrated approach is used by Kragt *et al.* (2011) to link spatially referenced economic valuation and catchment modelling in the George Catchment, Northeast Tasmania, Australia. The model utilises biophysical and economic information integration on costs and benefits of catchment management changes together with visual indicators of catchment condition and choice experiment surveys to select management actions and stated preference surveys to value ecosystem goods and services.

4.6 Discussion and Conclusion

Planning for sustainable futures through improved ecosystem management requires spatially explicit approaches that integrate ecological as well as economic dimensions and that allow substantial stakeholder engagement. This chapter and its associated Appendices have reviewed models and tools that allow for spatially explicit modelling.

The findings most relevant to the research reported here were identification of those methods and models that had the potential to be spatially enabled, that were not focussed on ecosystem services valuation that allowed integration of ecological and economic values, that generated scenarios, and that allowed stakeholder engagement.

Appendix B provides a detailed summary of the many models and their software implementation. Of these, the methodology that emerges as the most suitable starting point for the current research was the logistic regression model described in Daloğlu *et al.* (2014). This model has the capacity for explicit spatial analysis, it provides for the integrated analysis of the ecological and economic dimensions, it facilitates the comparisons of trade-offs between diverse stakeholders, and it supports the investigation of potential futures through scenario modelling.

The purpose of this chapter was to examine current models in order to identify characteristics that could be utilised to benefit the methodology developed for this research. The methodology employed a mix of approaches that draws on the strengths of previous studies, particularly the study by Daloğlu et al. (2014), therefore, developing and implementing a modelling approach that extends the field of geospatial science from traditional mapping and modelling to modelling that integrates diverse disciplines and supports spatial reasoning and dialogue.

5 Selection of Study Area and its Characteristics

The global importance of natural resource conservation is widely understood, with substantial investments being made worldwide to conserve natural environments and protect biodiversity. The Midlands region of Tasmania, Australia, is one of 15 nationally recognised biodiversity hotspots (Australian Government 2012a, 2012b). In recent years, this region of mixed land use has been a focus of attention from a variety of local government, commonwealth government, and private organisations. This chapter describes the Midlands region of Tasmania, and particularly a sub-region of the Midlands that is the focus of the research reported here. It describes the conservation strategies that have been implemented within the region in recent years, and shows the spatial and temporal distribution of those strategies.

5.1 Study Area and its Characteristics – An Introduction

The island of Tasmania lies off the south-east corner of the Australian mainland. Tasmania has a population of around 519,100 (Australian Bureau of Statistics 2016). Tasmania's area is 68,401 km2, of which the main island covers 64,519 km2 (Australian Government 2016). Tasmania spans around 500 kilometres from north to south (including offshore islands) and is 400 kilometres from the eastern coast to the western coast (Australian Bureau of Statistics 2011).

Tasmania has a cool temperate maritime climate with four distinct seasons (Australian Bureau of Statistics 2002a, 2011). The normal daily temperature range close to the coast is around 70C but can be double that inland (Australian Bureau of Statistics 2002a). The interaction of airstream and topography is the main factor governing rainfall in Tasmania, and this causes a marked variation in rainfall across the State. Annual averages are less than 600 mm in the Midlands, around 800 mm on the North-west Coast, 1500mm in the North-east Highlands and also on the west coastal strip, but over 3500 mm in some parts of the mountainous west (Australian Bureau of Statistics 2002a, 2011).

Tasmania contains a diverse range of soils due to variations in climate, landscape and geology. Tasmanian soils in their native state are not inherently fertile for productive agriculture (Australian Bureau of Statistics 2002b). Phosphorus and molybdenum are commonly deficient, while soil acidity can limit the growth of some plants. Therefore, adding superphosphate and molybdenum along with liming has been common in Tasmania (Australian Bureau of Statistics 2002b). The soils mostly support productive pasture and large number of grazing sheep prevail in the area. A wide range of vegetable (potato, beans, peas, onions, broccoli), pharmaceutical (poppy), industrial (pyrethrum), fodder (maize) and cereal (wheat, barley) crops are typically grown in rotation with a longer-term pasture phase (Lisson and Cotching 2011).

Tasmania has been promoted as a natural state; almost 45% of Tasmania lies in reserves, national parks, and World Heritage Sites (Parks and Wildlife Service Tasmania 2015). The Tasmanian Wilderness World Heritage Area, which comprises about 20% of the State (1,383,865 hectares), is essentially wild, natural country in central and south-western Tasmania (Australian Bureau of Statistics 2002a).

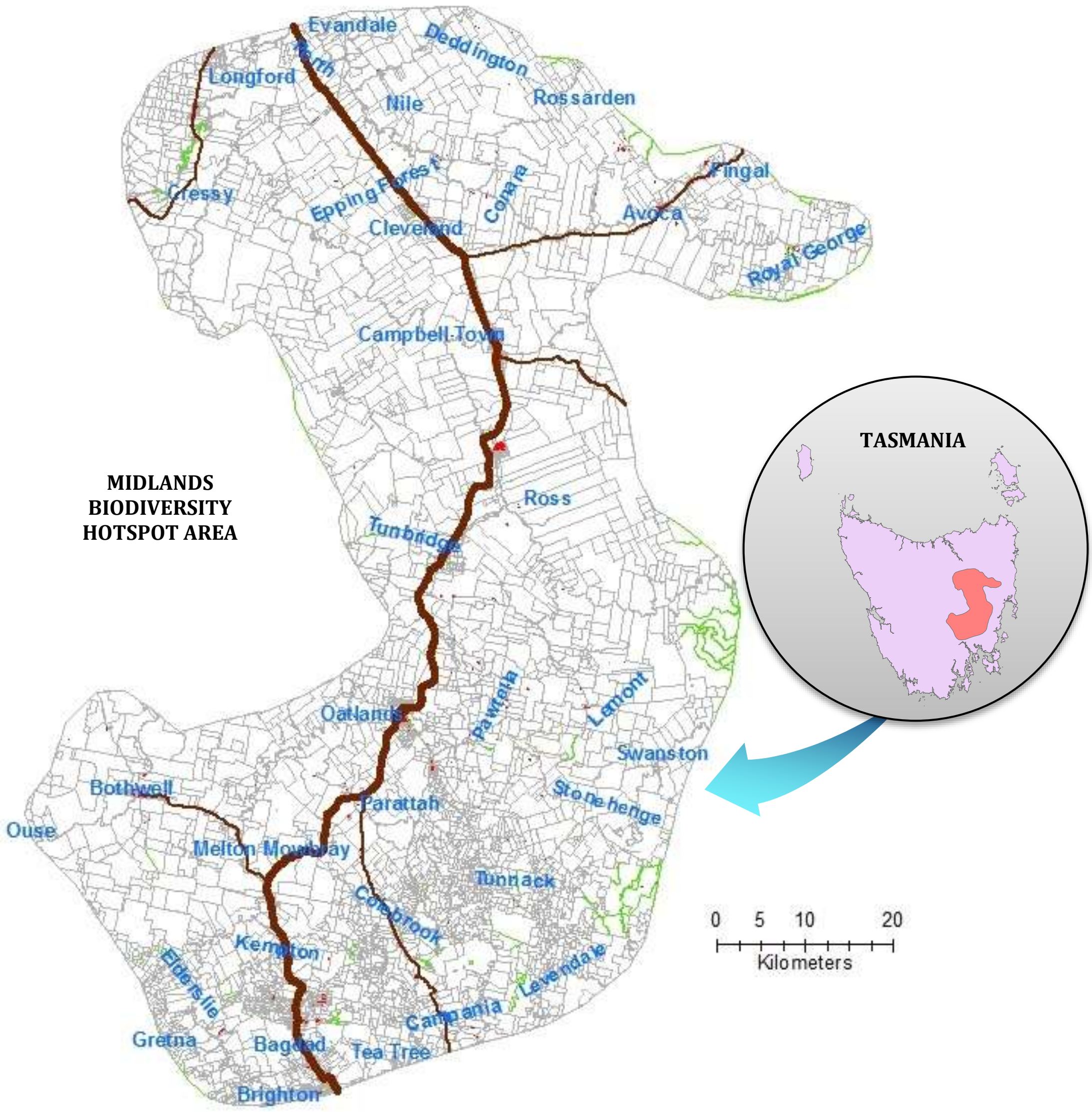


Figure 5.1: Study area showing land parcels and major locations of the biodiversity hotspot area of Midlands Tasmania, also known as the duck area. (Source data obtained from DPIPWE and TLC.)

The Midlands of Tasmania, is a region between the two cities of Hobart and Launceston, having a total area of 7,746 square kilometres (Gadsby et al. 2013). The Midlands region is comprised of the basin in the valley between the major landforms of the Great Western Tiers, the Ben Lomond Plateau and the Eastern Tiers. It is a relatively flat, dry and low plains area that is drained by The South Esk river, Jordan river, Coal river, Prosser and Little Swanport river and their tributaries (Tasmanian Government's Wealth from Water 2012).

The Midlands has the driest climate in Tasmania, with annual rainfalls ranging from 4450 to 600 mm (Tasmanian Government's Wealth from Water 2012). The Midlands has warmest summers and some of the coldest winters in Tasmania. The soils in

Tasmania are very fertile, due to the relatively low rainfalls and the presence of eroded fertile dolerite rocks, resembling the chernozems of Asia and North America (Midlands Initiatives for Local Enterprise Inc 2003). Hence, the Midlands is rich grazing land supporting intensive rearing of sheep, whereas, some grain crops are also grown in the region (Midlands Initiatives for Local Enterprise Inc 2003).

The natural vegetation of this region was predominantly grassland, but is now either grazed by sheep and cattle, cleared for growing better pasture or used for agriculture. Tasmanian Midlands contributes 16 per cent of the total value of agricultural production in the state (Tasmanian Government's Wealth from Water 2012). Many farms in Tasmania are generally thought of as mixed farms, because they combine a range of enterprises including grazing, vegetable growing, grain and oil poppy growing. Farming properties in the Tasmanian Midlands are much larger, with an average size of 1,750.4 hectares, than the state average of 405.1 hectares (Gadsby et al. 2013).

Natural ecosystems provide a variety of benefits in the form of ecosystem services (Walls and Riddle 2012). The wealth of plants, animals and micro-organisms, expressed as biodiversity, play a central role in the provision of ecosystem services. Efforts to better understand the importance of ecosystems and their conservation is evidenced by the high number of recent scientific publications on ecosystem services (Plant and Ryan 2013). However, despite an increasing recognition of the importance of ecosystems and biodiversity for human welfare, ecosystems continue to decline at an unprecedented rate (Morse-Jones *et al.* 2011).

Various initiatives have been taken in Tasmania, Australia to enhance the conservation of native vegetation and reduce the loss of biodiversity. Conservation planning and work has addressed diverse issues including restoration of landscape connectivity (Michaels *et al.* 2008), protecting biodiversity and planting trees for carbon sequestration (Schirmer and Bull 2011), landscape scale planning (Brown 2010), ecosystem services mapping (Williams 2009) and the protection and restoration of riparian habitat (Tasmanian Natural Heritage Trust 2002, 2003, Greening Australia 2003, Bush Heritage 2008, NRM North 2010). A substantial amount of this work has been carried out (by groups such as the Tasmanian Land Conservancy, Greening Australia, Natural Resource Management, Local Government, and the Department of Primary Industries Parks, Water and Environment, Tasmanian Government) in a sub-region of the Tasmanian Midlands known as the 'duck' area because its geographic shape. It has an area of 640,909 Ha. The duck area is part of a nationally recognised biodiversity hotspot area (Iftekhar *et al.* 2014). The study site for the work reported here is within that sub-region of the Tasmanian Midlands and includes much of the Northern and Southern Midlands Council areas, a little of the Glamorgan Spring Bay area and some parts of the Central Highlands. The region is shown in Figure 5.1.

Apart from having more than 180 rare and threatened plant and animal species (Iftekhar *et al.* 2014) and highly significant wetlands, the Tasmanian Midlands Region is uniquely characterised by the extent to which important habitat is located on private lands (Iftekhar *et al.* 2014). Ninety eight percent of the land is privately owned, mostly by families who have lived in the area for generations (Bush Heritage 2008). In the absence of a significant amount of land to expand public reserve networks, the

Australian Government initiated several projects in the region to promote private land conservation. A range of incentive programs was introduced by the Department of Primary Industries, Parks, Water and Environment (DPIPWE), in response to national conservation priorities, to establish voluntary conservation agreements on private lands.

The first such covenanting program operating in Tasmania was the Protected Areas on Private Land Program (PAPL), which commenced in 1999 (Iftekhar *et al.* 2014). The Private Forest Reserve Program (PFRP - 1997), Non-Forest Vegetation Program (NFVP – 2003), Midlands Biodiversity Hotspot Project (MBHP – 2005), Midlands Biodiversity Hotspot Tender (MBHT – 2007) and Forest Conservation Fund (FCF – 2007-2009) were a number of programs initiated to complement the overarching PAPL conservation objectives. Among these, MBHP and MBHT were specifically targeted at the Tasmanian Midlands. A tender-based approach in MBHT followed by a conservation value index (CVI) approach was used to select submitted projects for funding for conservation covenanting in Tasmania until late 2009 (Iftekhar *et al.* 2014).

Spatial data representing programs of conservation activities carried out in the Tasmanian Midlands were collected, in order to establish an overview of activities in the region from a spatio-temporal perspective. Various projects were identified (MBHP, MBHT, FCF, Bushlinks 500, Crown Land Assessment and Classification Project (CLAC), PFRP, PAPL, NFVP, Rivercare and Midlandscapes) that were funded for, or were active during, the period 2000 to 2013, together with information about project duration. These data were sourced from a number of contributing organisations including Natural Resource Management (NRM) North, Natural Resource Management (NRM) South, the Tasmanian Land Conservancy (TLC), Bush Heritage Australia, the Southern Midlands Council (SMC), Greening Australia (GA) and the Department of Primary Industries Parks, Water and Environment, Tasmanian Government (DPIPWE). Table 5.1 shows a summary of projects, their duration, target and policies.

Table 5.1: List of Conservation Initiatives or Policies active during 2000 to 2013 in the Midlands region

S#	Projects	Duration	Target Land	Policies
1	Midlands Biodiversity Hotspot Project (BHP)	2005 - 2007	Land having threatened species	Protection of threatened species - Land owners were given incentive funds and management funds for weed control and fencing. Conservation covenants and vegetation management agreements. -
2	Midlands Biodiversity Hotspot Tender (MBHT)	2007 - 2010	Land having specific natural value	Stewardship Payments – Targeting land having specific natural value
3	Forest Conservation Fund (FCF)	2005 - 2009	Private forests	Protection of forested private land through perpetual and 24 year covenants - Financial incentives
4	Bushlinks 500	2012 to date	Private grazing lands and native vegetation	Protecting and enhancing native vegetation on private land through fencing, weed control, grazing management, replanting. Protecting native grasses. Incentives for

S#	Projects	Duration	Target Land	Policies
				revegetation, seeding, fencing, weeding, spraying etc.
5	Crown Land Assessment and Classification Project (CLAC)	2004 - 2008	-	Proclamation of land for reservation, public ownership or public sale (no incentive found)
6	Private Forest Reserve Program (PFRP)	1997 - 2009	Private Forests	Reserves in perpetuity secured for per hectare incentives
7	Protected Areas on Private Land Program (PAPL)	1999 - current	Land having threatened species, natural values	Land having important natural values (threatened species, freshwater values, geo-conservation areas) - Conservation covenants – incentives given at per hectare value
8	Non-Forest Vegetation Project (NFVP)	2003 - 2010	Land having native grass	Incentives for protecting non forest vegetation particularly native grass - Incentives for fencing and weed control
9	Rivercare Program in Tasmania	1997 - 2003	Rivers and native vegetation	Funding for restoring protecting ecological health of rivers - Some funds for conserving native vegetation
10	Midlandscape	2008	Tasmanian Midlands	Protection and management of target area

Conservation covenants in the region have been funded and co-funded through multi-program investments delivered through partnerships between the Tasmanian Government, private land philanthropy (Bush Heritage Australia and Tasmanian Land Conservancy) and the Australian Government, under the National Reserve System program (Iftekhar *et al.* 2014). The sub-region that is the focus of this study is regarded as a prime biodiversity hotspot within the Midlands region.

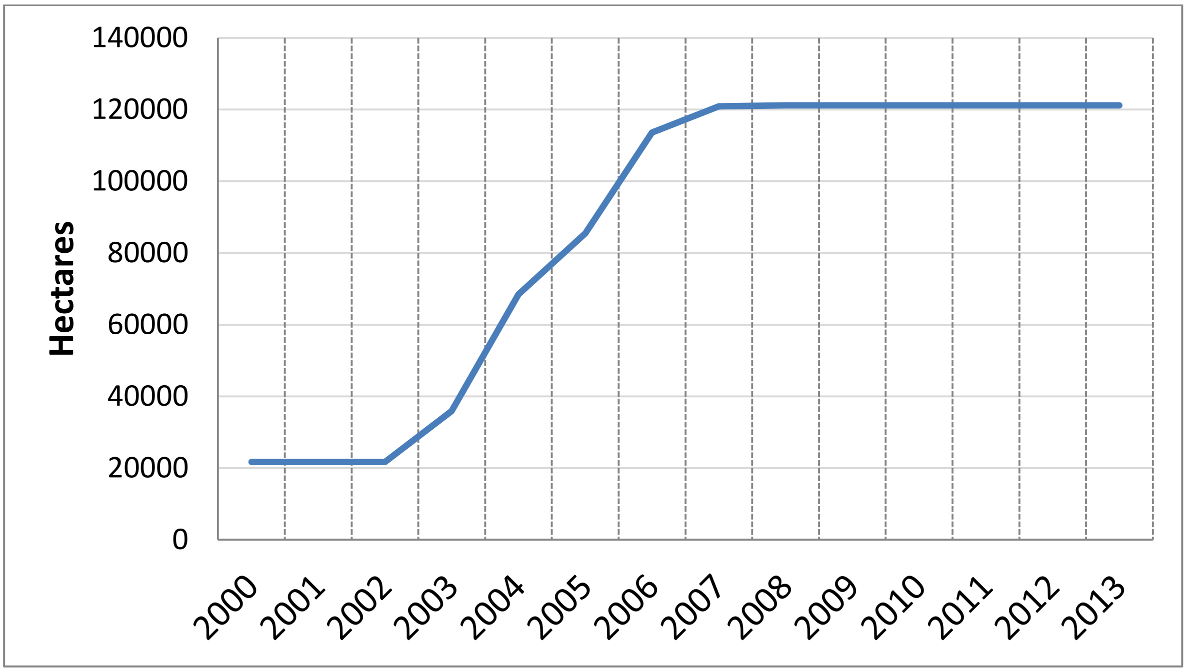


Chart 5.1: Representation of increase in conservation and reserve area from 2000 to 2013, representing area being steady from 2000 to 2002, followed by a rapid increase from 2002 to 2007 and steady during 2008 to 2013

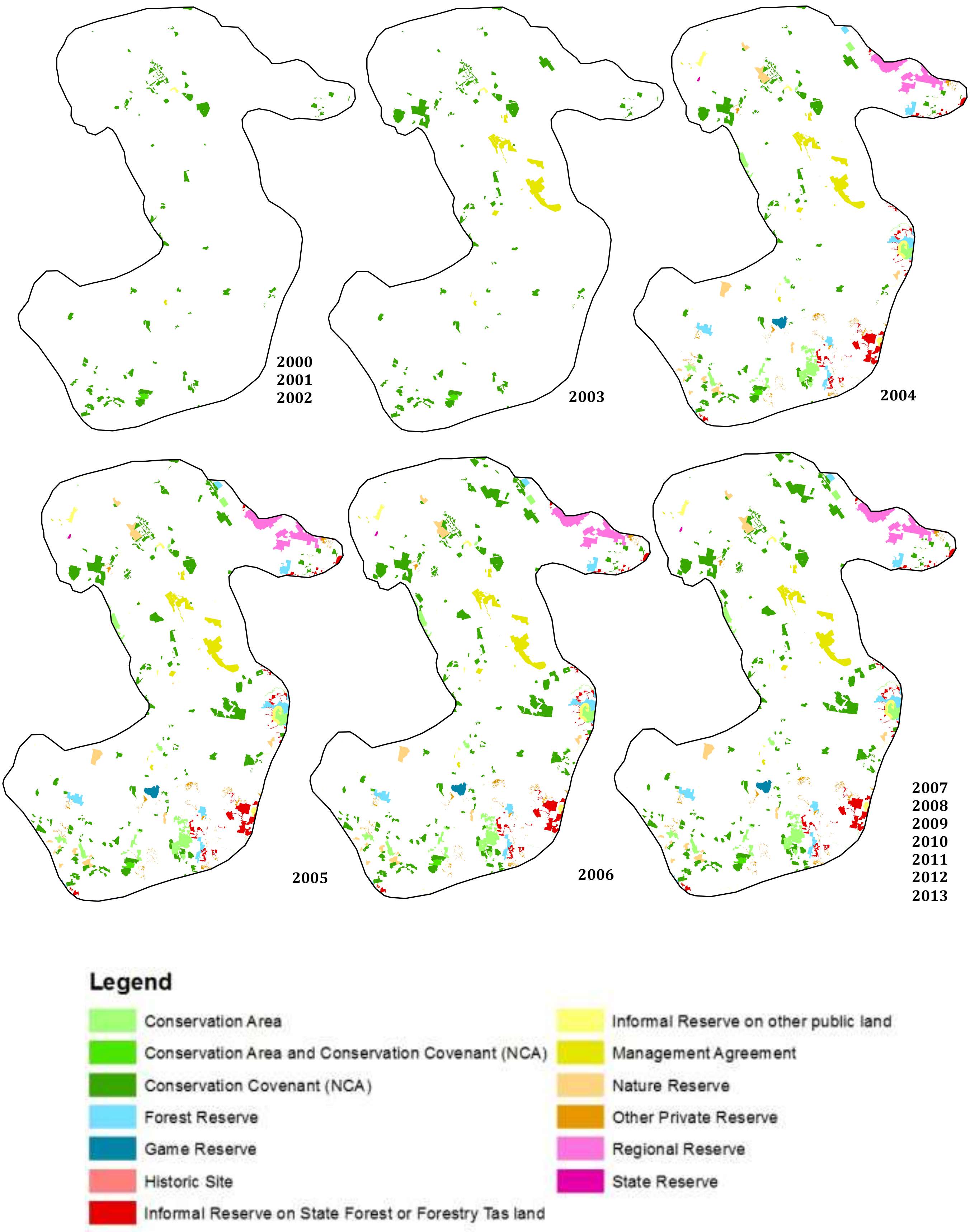


Figure 5.2: Change in conservation and reserve area from 2000 to 2013

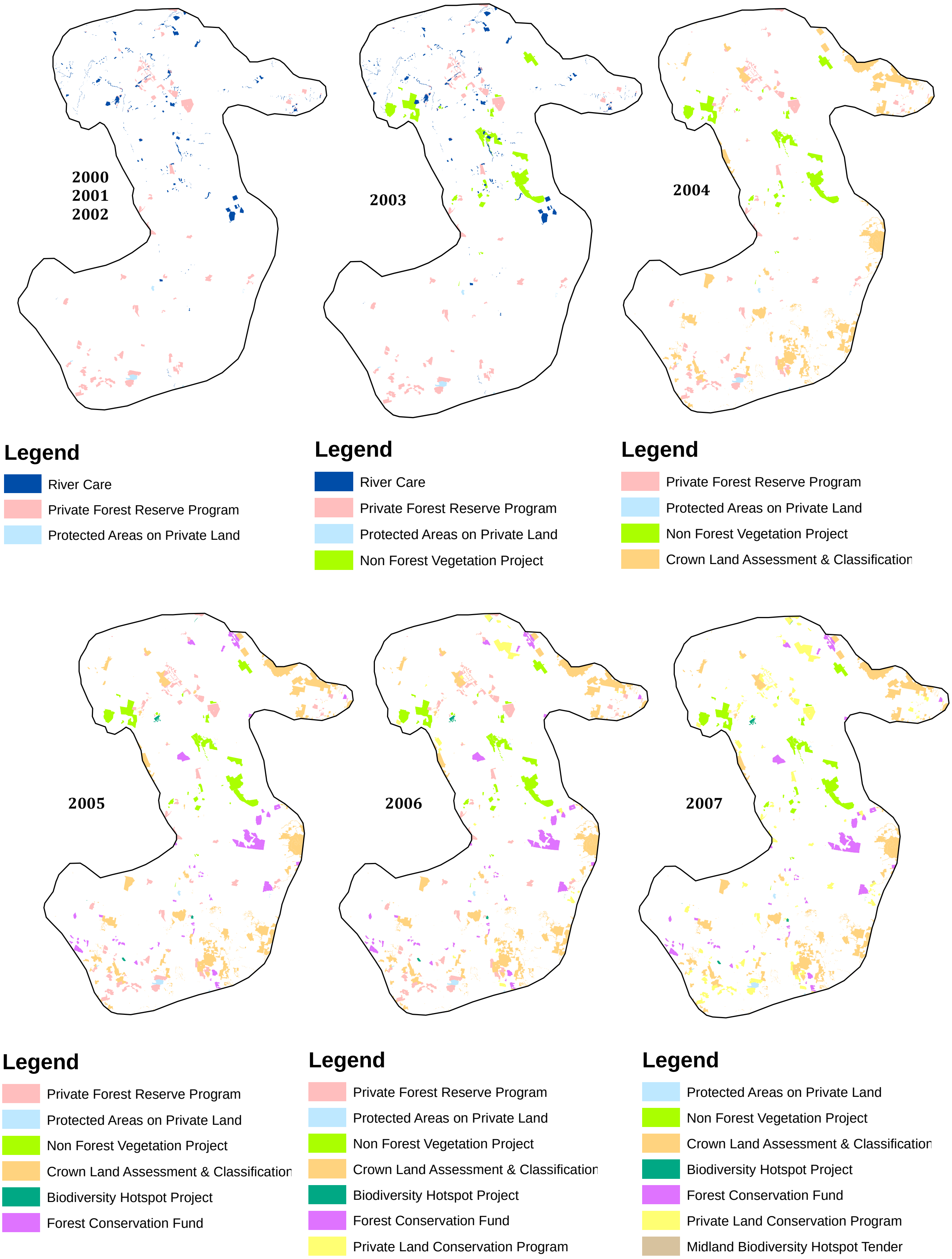


Figure 5.3: Spatio-temporal representation of location of project activities and their duration in Duck area from 2000 to 2007

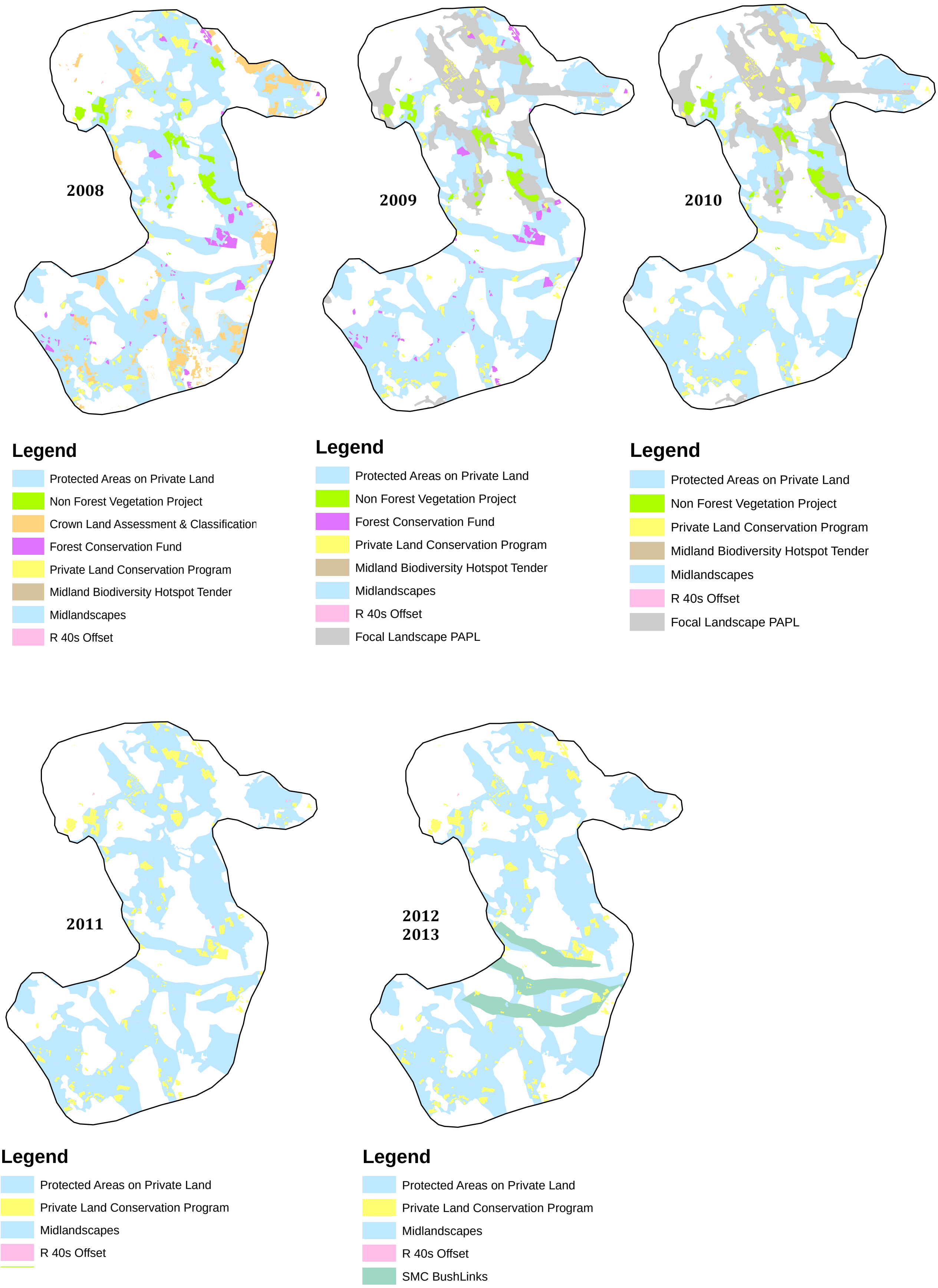
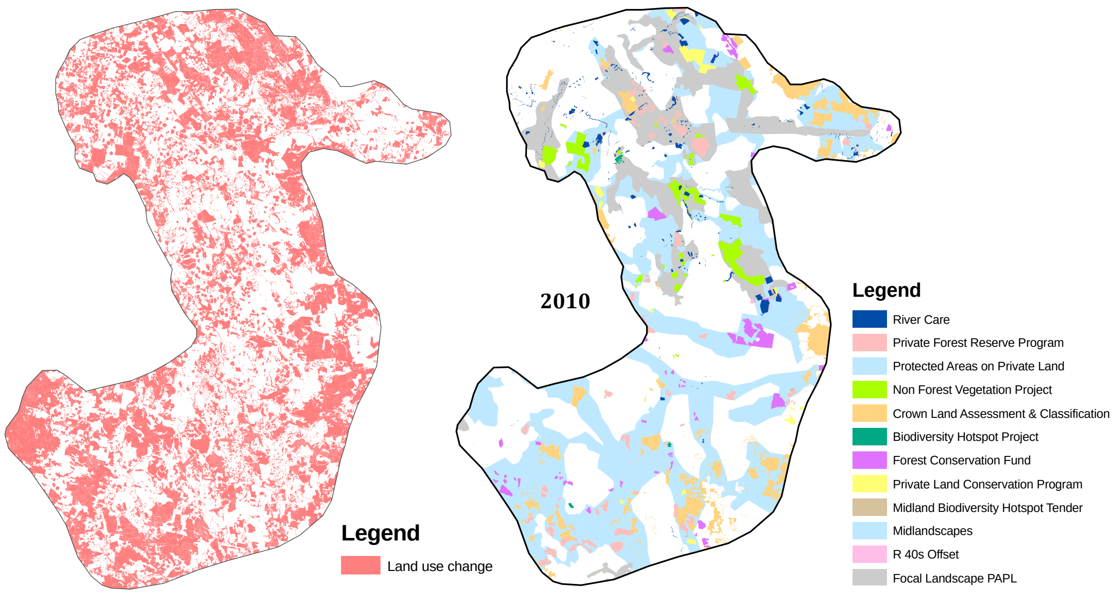


Figure 5.4: Spatio-temporal representation of location of project activities and their duration in Duck area from 2008 to 2013

The data obtained for each of the projects were organised, assembled and analysed using ESRI ArcGIS 10.1 in order to represent change in the conservation and reserve areas in the region in a geospatial and temporal manner for the period 2000 to 2013. Chart 5.1 shows the total annual increase in conservation and reserve area for the period 2000 to 2013, and highlights the cessation of this growth from 2007 to 2013. Chart 5.1 indicates that the conservation and reserve area remained steady during 2000-2002, followed by a rapid increase in the area from 2002 to 2007, which then remained steady during the period 2008-2013.

Figure 5.2 shows the change in conservation and reserve areas for the period 2000 to 2013, highlighting the periods in which particular practices commenced and were implemented. As Figure 5.3 suggests, only a few projects were active during the period from 2000 to 2002. Projects were introduced each year from 2003 to 2007, increasing the overall conservation and reserve area until 2007. This was followed by a steady state in the conservation and reserve area from 2008 to 2013.



Change in Land use from 2001 to 2010 *Project activities until 2010*

Figure 5.5: A comparative spatial view of land use change from 2001 to 2010 and project activities in the 'duck' area until 2010. High concentration of change in land, in the land use change map shows occurrence of project activities in conservation status map

Data collected on the duration (implementation period) for each project was used to map project activities on a timeline that showed their geographic location only for the period in which they were active. Figures 5.3 and 5.4 show, on an annual basis, the activities carried out in the 'duck' area, with project names and their respective locations. It is apparent from Figures 4 and 5 that the highest project activity was in the period 2008 – 2009, followed by the years 2006, 2007 and 2010. It is also evident that only three projects were underway in the area in the year 2000, namely River Care, Protected Areas on Private Land, and the Private Forest Reserve Program. The number of projects and their activities increased from 2000 to 2009 and then slowly decreased until 2013. The area under conservation agreements did not increase after 2007, although some projects such as PAPL were still operating.

5.2 Effect of Conservation Activities on Land Use Change Dynamics

A land use change[16] dataset obtained from DPIPWE representing change in land use from 2001 to 2010 was visually compared with the spatial view of project activities undertaken in the duck area from 2001 until 2010. These are shown in Figure 5.5. The comparison demonstrates that a substantial change in land use (illustrated by a concentration of red colour in the map) occurred in areas under project activities. About 45% of the total area of the duck (640,909 Ha) incurred a change in land use during the period from 2001 to 2010, which suggests that the projects and their activities had a notable influence on land uses.

5.3 Selection of Study Site

The Australian land mass is divided into 89 bioregions[17] (Australian Government 2012a, 2012b). Each region is a land area made up of a group of interacting ecosystems that are repeated in similar form across the landscape. The bioregions are described in the Interim Biogeographic Regionalisation for Australia (IBRA), which is the National Reserve System's planning framework for identifying reservation targets and setting priorities to meet them (Australian Government 2012a, 2012b). IBRA version 7 (2012) was developed through the co-operative efforts of the Australian Government Department of Sustainability, Environment, Water, Population and Communities and State/Territory land management agencies. IBRA regions represent a landscape based approach to classifying the land surface, including attributes of climate, geomorphology, landform, lithology, and characteristic flora and fauna. Specialist ecological knowledge combined with appropriate regional and continental scale biophysical data sets were interpreted to describe these 89 regions existing across Australia.

The duck area comprises four different bioregions, namely: the Ben Lomond Bioregion, the Central Highlands Bioregion, the Northern Midlands Bioregion and the South East Bioregion as derived from IBRA data. These bioregions are shown in Figure 5.6. In order to appropriately scope our study, a single bioregion having the largest area of the four bioregions was selected. This is the South East bioregion. Therefore, historical data is used to derive a set of potential future land use change pathways. These are illustrated in Figure 5.6. Bioregions are geographically distinct areas (Australian Government 2012a). The duck area comprises four bioregions and so, for this research, the largest bioregion within the duck area (the South East bioregion) was selected. This ensures that the study employs a predominant geology, land forms, climate, ecological features and plant and animal communities.

---

[16] Land use change is measured by measuring changes in area, productivity, intensification, new uses and improvements (Lesslie *et al.* 2011).

[17] Bioregions are large, geographically distinct areas of land with common characteristics such as geology, landform patterns, climate, ecological features and plant and animal communities (Australian Government 2012a).

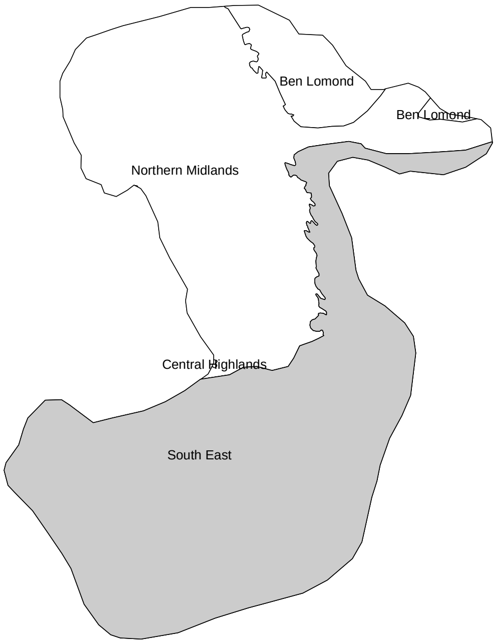


Figure 5.6: Bioregions in the duck region, with South East Bioregion highlighted in grey selected as the study area

5.4 Conclusion

The area selected as the study area is part of a nationally recognised biodiversity hotspot area and has a high level of private land ownership. Land management activities in the region are representative of a mix of interest groups and draw investment from a variety of organisations. The area is characterised by diverse land uses, specific to agro-ecological landscapes, and various land capabilities and soil types. The active interventions, competing stakeholder interests and changing land uses in the region, as evident from the land use change dataset, make this region highly suited as a case study for this research. The South East bioregion of the duck area is historically analysed in Chapter 6, in order to derive future land use change pathways, required for constructing scenarios employed in this research.

6 Construction of a Maryland Model Farm

6.1 Introduction

The aim of the research is to develop a geospatially enabled model that integrates both ecological and economic values and that can operate at the scale of operational land uses. The model is required to have the capacity to contribute to decision making by capturing, computing, and presenting information that describes the trade-offs between the values of diverse stakeholders for a variety of land use and land management options. The intention of the model's design is that it provides an interactive tool that will enable users to understand, compare and contrast the values held by different stakeholders and the influences of those values on possible land use outcomes. The model is intended to enable users to readily manipulate inputs and observe outputs so that it can be deployed as a decision support system[18]

The overarching endeavour is to show that it is possible to develop an integrated method of capturing the spatial, ecological and economic dimensions of on-farm decision making and to construct the model in such a way as to ensure it supports and informs expert decision making processes.

The study required data that fairly represented a typical farm within the Tasmanian Midlands. For a number of reasons, including the need to use data that was appropriately complex and that captured an appropriate range of land capability and land uses, together with ensuring that issues such as privacy or confidentiality did not hinder the research or bias the data collected from stakeholders, the research has employed a fictional model-farm rather than an actual Midlands property. This fictional, or model-farm, is intended to be representative of an actual Midlands property farm for this study.

The model farm, which has been called 'Maryland Farm', facilitated open communication with stakeholders, particularly farmers and land owners, as they were not required to respond to questions about their own, or another person's property.

A methodology was devised to construct a model farm that fairly represented typical on ground situations for a Midlands farm. The methodology utilised and adapted a number of real datasets taken from regions within the Midlands region. Important relationships between attributes such as land use, soil type, land capability, and topography for a location or parcel were selected and represented based on typical and demonstrated patterns in the underlying spatial datasets for the region. Where real data was used to

[18] Decision aids are interventions or tools designed to facilitate shared decision making usually in health care decisions (Stacey *et al.* 2014). Whereas, a decision support system (DSS) is a computer-based information system that supports decision-making activities. Decision aids range from a simple heuristic model to a very complex tool (Power 2016). Complex, sophisticated computerised decision aids are a subcategory of decision support systems. In general, DSS are computerized decision aids, although, some computerized decision aids may be so simple that they cannot be categorised as decision support systems (Power 2016). In this research, decision support system is referred to as a complex, sophisticated, and computerised decision support tool.

inform the design of the model-form, it was extracted from geographically diverse locations, topography was manipulated, and model land parcel boundaries were constructed, so that the model farm had no discernible similarity to any real property in the region, but was nevertheless representative of the prevalent parcel sizes, land capabilities and land uses.

Table 6.1: Datasets used in the study and the authority that provided the data

S#	Datasets	Provided by Authority	Year	Data Type	Map Scale or Spatial Resolution
1	Spatial data on ***Dominant Soil Orders***	Department of Primary Industries, Parks, Water and Environment (DPIPWE)	2005	Vector	1:500000
2	Spatial data on ***Land use*** based on ALUM classification	Natural Resource Management North (NRM North)	2001 and 2010	Vector	Varied scale from 1:25000 to 1: 250000
3	Spatial data of ***Land use Change*** based on ALUM classification	Department of Primary Industries, Parks, Water and Environment (DPIPWE)	from 2001 to 2010	Vector	Varied scale from 1:25000 to 1: 250000
4	Spatial dataset of ***Land Capability***	Department of Primary Industries, Parks, Water and Environment (DPIPWE)	2005	Vector	1: 100000
5	***DEM*** with 25m spatial resolution	LIST data from Department of Primary Industries, Parks, Water and Environment (DPIPWE)	2013	Raster	25 m
6	Spatial data of ***Bioregions*** in the area	Interim Biogeographic Regionalisation for Australia (IBRA) http://www.environment.gov.au /land/nrs/science/ibra	2012	Vector	1:10million
7	Spatial ***Boundary of Duck*** area	Tasmanian Land Conservancy (TLC)	2012	Vector	N.A

6.2 Creation of Maryland Farm and Datasets

Data was obtained from the south east bioregion of the Duck area described in detail in Chapter 5. Maryland farm comprises the following integrated data sets, represented spatially in a GIS:

1. land parcel boundaries
2. topography:
   ii. terrain height (DEM)
   iii. terrain slope
   iv. terrain aspect
3. hydrology: rivers, streams and dams
4. dominant soil type
5. land capability

6. land use

Additionally, land use change data was required in order to formulate potential future land use change patterns and thus derive future change pathways.

The following sections describe the methods used to derive or create each of these data sets:

6.2.1 Creation of the farm area and land parcel boundaries

The land area of Maryland farm was selected to be approximately 1200 hectares, based on a typical or average size for a Midland's farm and a suitable, approximately rectangular, perimeter boundary was constructed. Land use boundaries were constructed within this outer boundary. For ease of analysis, a unit area of 25 m x 25 m was produced by creating a line feature 25 m x 25 m grid in AutoCAD, importing the layer into ArcGIS, and then converting the line features into polygon feature using the Topology and Advanced Editing tools in ArcGIS. The grid polygon was then saved into an already created empty polygon shape file boundary layer to obtain a polygon grid output layer having an area of 1200 hectares as shown in Figure 1. The size of the grid cells remain intact within the farm and along land use/ land capability parcel boundaries, although, they are cropped along the whole farm boundary and may be reduced in size at the edges. There are various methods to ensure the topological integrity of GIS datasets (Hadzilacos and Tryfona 1992, Brisaboa *et al.* 2014). In this research, the geometry of the land parcel boundaries, GIS layers and topography were kept intact by joining the information from these layers to a single grid polygon of 25 m x 25 m. Thus, allowing to share geometry in an integrated way and maintaining the topological integrity in the model farm.

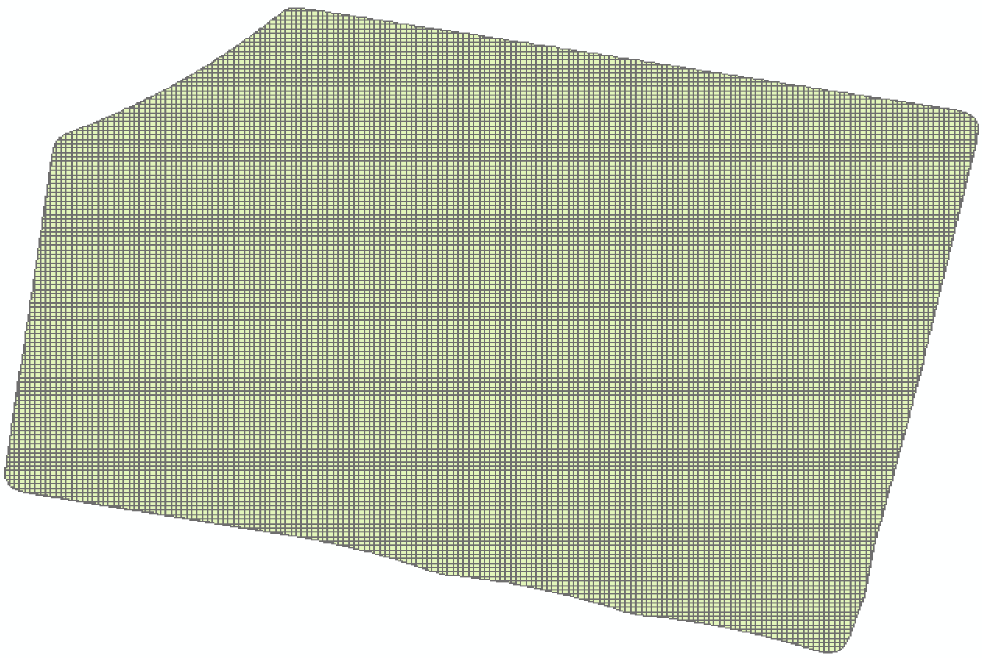


Figure 6.1: Maryland Farm boundary layer having an area of approx. 1200 hectares divided into polygon grids of 25m x 25m

6.2.2 Creation of topography

A Digital Elevation Model (DEM) is a digital geographic dataset of elevations in *xyz* (ENH) coordinates. The term "Digital Elevation Model" has both generic and specific meanings. In general, a DEM is any digital (raster) representation of a terrain (topographic) surface. In this work and the model, the DEM represents terrain elevations sampled at regularly spaced horizontal intervals. Slope and aspect have been derived from the DEM and are utilised in this study. Slope is the incline, or steepness, of a surface. Slope can be measured in degrees from horizontal (0–90), or percent slope (which is the rise divided by the run, multiplied by 100). Here, slope is measured in percentages. Aspect is the direction that a topographic slope faces, usually measured in degrees from north. For this research, a DEM obtained from LIST (Land Information System, DPIPWE, Tasmanian Government) having 25 m spatial resolution was clipped from the south east bioregion and used for the model farm.

6.2.3 Creation of hydrology: rivers, streams and dams

The availability of GIS tools and Digital Elevation Models makes it easy to extract watershed properties. In this case, Arc Hydro Analyst and the DEM were used to derive surface hydrology for the farm, thus defining catchments and streams.

The processing of a DEM to delineate streams and catchments is referred to as terrain pre-processing. The method used in the terrain pre-processing involved (following Merwade 2012) DEM reconditioning, filling sinks, computing flow direction and flow accumulation, defining streams with stream definition function, catchment delineation, converting the raster data to vector through processing of catchment polygon and drainage lines, and generation of drainage points and slope grid.

6.2.4 Creation of dominant soil type

There is a variety of soils across Australia as shown in Figure 6.2. They are broadly grouped into soil orders based on the Australian Soil Classification System (ASC). The soil orders are:

- Organosols
- Rudosols
- Tenosols
- Podosols
- Vertosols
- Hydrosols
- Sodosols
- Chromosols
- Kurosols
- Calcarosols
- Ferrosols
- Dermosols
- Kandosols
- Anthroposols

In the SE Bioregion of Tasmania, the dominant soils include Chromosol, Dermosol, Ferrosol, Kurosol, Rudosol, Sodosol and Tenosol, as derived from the datasets. A Tasmanian dominant soil dataset was clipped to the South East Bioregion of the duck area (total clipped area = 349,827 Hectares) to calculate the relative percentage of the dominant soil using ArcGIS 10.1. Calculations showed that Chromosol soil occupied the highest percentage (41.52% = 145263 Hectares) among other Dominant Soil Order soils (Dermosol, Ferrosol, Kurosol, Rudosol, Sodosol, Tenosol). Thus, a Chromosol soil was selected for the Maryland Farm[19]. Chromosols are texture-contrast soils (abrupt change in texture between the surface and subsoil) and are mildly acidic (pH above 5.5).

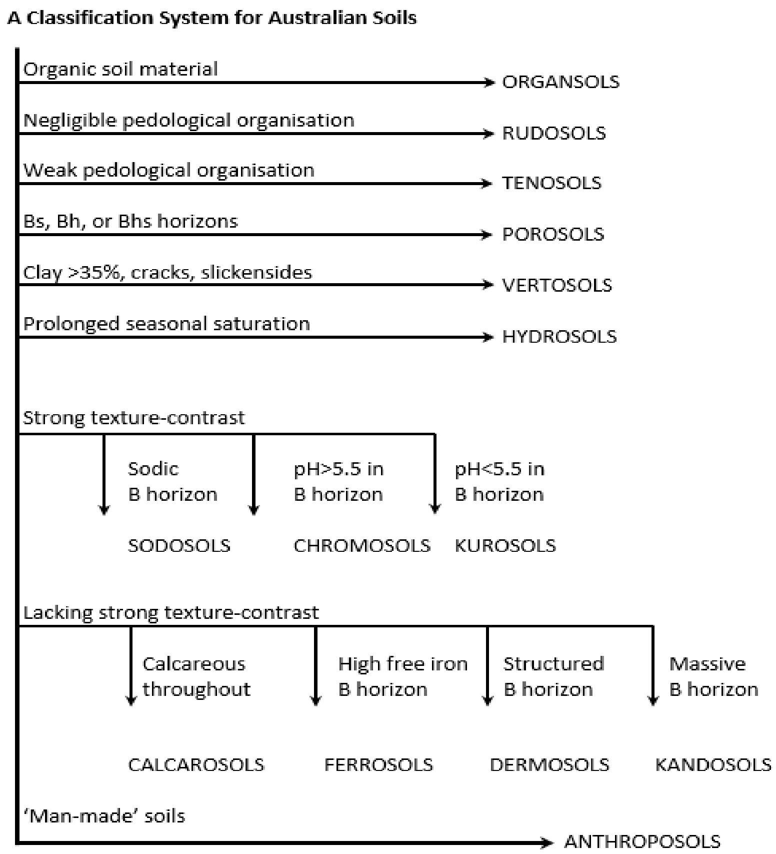


Figure 6.2: Classification of Australian Soils into dominant soil orders adapted from Australian Soil Classification System (Isbell 2002)

6.2.5 Creation of land capability

A land capability classification is used to assess the capability of land to support a range of land uses on a long-term sustainable basis (DPIPWE 1999). Land capability assessments consider a number of factors including geology, soils, slope, climate, land management practice etc. in order to determine land's potential for sustainable agriculture (DPIPWE 1999).

Areas of the southeast bioregion of the duck area having chromosol soil were used in order to extract a land capability dataset from DPIPWE data. An overlay analysis showed that a number of land capabilities existed in the area. The land capabilities occupying maximum area on each selected land use (shown in subsequent Table 6.3) were selected and used as land capabilities of Maryland farm.

[19] The SEEM model has been developed to be used with diverse areas and datasets, and has the flexibility to incorporate different soil types. The model farm was developed to have a single soil type based on the highest occurrence of soil type as explained under section 6.2.4. The scale of the model farm was such that adding various soil types was not considered necessary, and would not have added to the model's development of testing.

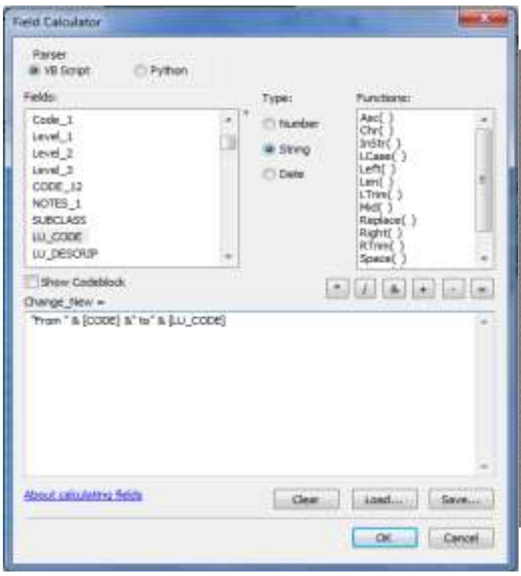
6.2.6 Creation of land use

The Australian Land Use and Management (ALUM) Classification is an Australian standard for collecting and presenting land use information within Australia. In this research, the ALUM classification system was used to assign land uses to land parcels in Maryland farm. The boundaries of land uses were fixed in the current model and the stakeholders did not have the privilege to alter the boundaries. The procedure for selecting and assigning land uses to the land parcels is explained below.

6.3 Creation of land use change pathways

The land use change dataset was further analysed in order to select land uses for parcels on Maryland Farm. Firstly, spatial datasets comprising Land use data 2001, Land use data 2010, Land use change data, Dominant soil order data, and Land Capability data were joined using Spatial Join and cropped for the South East Bioregion and for land with Chromosol as the Dominant Soil. An analysis of this land use change dataset of South East bioregion showed that 47.2% (165163 hectares) of land had undergone a change from one land use to another during the decade from 2001 to 2010. This change data was used to construct 'future change pathways'.

The change dataset showed only data to level 2 of the ALUM classification. In order to provide greater resolution, another field was added into the shape file using the Field Calculator and VB scripting, allowing an analysis of the data at Level 3 of ALUM classification. The script and the resultant field obtained are shown in Figure 6.3. The reason for moving from a lower thematic resolution (ALUM level 2) to a higher thematic resolution (ALUM level 3) was to understand the land use and its management more clearly. For instance, 1.1 (Nature conservation) as a level 2 classification could be a nature reserve, wilderness area, national park, natural feature protection, habitat species management area, protected landscape, or other conserved area. Using, for example the level 3 classification, 1.1.5, clarifies that within the classification nature conservation, the land use is habitat/ species management area.



	FID	Shape *	CODE	LU_CODE	CHANGE	Change_New
	1673	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1673	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1673	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1674	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1674	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1799	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1799	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1799	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1799	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1799	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1815	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1815	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1815	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1815	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1815	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1815	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1829	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1830	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1830	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1830	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1830	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1830	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1830	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3
	1830	Polygon	1.1.1	1.3.3	From 1.1 to 1.3	From 1.1.1 to 1.3.3

Figure 6.3: Field Calculator and VB scripting were used to obtain the resultant field showing ALUM classification Level 3 for change of land use

Table 6.2: Sum area and sum percentages of land use changes in the south east region of the duck area under chromosol soil

#	ALUM Classification	ALUM Code for Land use Change	Sum Area	Sum Percentage
1	From Grazing natural vegetation to Residual native cover	From 2.1.0 to 1.3.3	12734.69	20.49
2	From Grazing natural vegetation to Grazing modified pasture	From 2.1.0 to 3.2.0	10316.15	16.60
3	Residual native cover to Grazing natural vegetation	From 1.3.3 to 2.1.0	9330.607	15.01
4	From Grazing natural vegetation to Native/exotic pasture mosaic	From 2.1.0 to 3.2.1	5768.191	9.28
5	Grazing modified pasture to Grazing natural vegetation	From 3.2.0 to 2.1.0	3783.874	6.09
6	Production forestry to Grazing natural vegetation	From 2.2.0 to 2.1.0	3238.434	5.21
7	From Grazing natural vegetation to Habitat/species management area	From 2.1.0 to 1.1.5	3172.507	5.11
8	Production forestry to Residual native cover	From 2.2.0 to 1.3.3	1940.854	3.12
9	Grazing modified pastures to Residual native cover	From 3.2.0 to 1.3.3	1799.92	2.90
10	Residual native cover to Habitat/species management area	From 1.3.3 to 1.1.5	1372.961	2.21
11	From Grazing natural vegetation to Plantation Forestry (Hardwood production)	From 2.1.0 to 3.1.1	1123.681	1.81
12	Managed resource protection (Biodiversity) to Protected landscape	From 1.2.1 to 1.1.6	707.5084	1.14
13	Grazing modified pasture to Irrigated cropping	From 3.2.0 to 4.3.0	669.0492	1.08
14	Grazing modified pasture to Cropping	From 3.2.0 to 3.3.0	641.7028	1.03
15	From Grazing natural vegetation to Managed resource protection	From 2.1.0 to 1.2.0	607.9773	0.98

Further analysis showed that about 54% (79230 hectares) of the total area of SE bioregion under Chromosol soil (145263.78 Hectares) underwent no change during this decade, whereas, about 45% (66033 hectares) of the area changed from one land use to another land use.

The change data (comprising 45% of the total area) was further summarised to derive sum areas for each change category and to compute the percentage changes in a descending order. Table 6.2 shows the changes from the calculated list; these encompass more than 90% of the total land use changes in the area over the period 2001-2010. The first five of these land use changes represented more than 67% of the total changes and were highly significant to this study. It was also noted that by using thee first five changes, it was possible to include survey data for land uses: Grazing Natural Vegetation (2.1.0), Residual Native Cover (1.3.3), Grazing Modified Pastures (3.2.0/3.2.1). Therefore, all combinations that comprised these land uses (including Habitat Species Management Area (1.5.5), Cropping (3.3.0) and Irrigated cropping 4.3.0) from the list were also selected. These are highlighted in Table 6.2, and account for 79.8% of total changes that occurred in the area from 2001 to 2010.

The ALUM classification description of the selected land uses are shown in Table 6.3. These land uses were assigned to Maryland farm land parcels, without compromising topological integrity and ensuring that all changes highlighted in Table 6.2 are applicable to the farm. A detailed analysis was carried out focussing on land capability of the land use in question, thus assisting in demonstrating Table 6.4 and 6.5 and finally formulating the future change pathways/scenarios in Table 6.6.

Table 6.3: Selected Land uses with their ALUM classification codes and Level 1, 2 and 3 description

	ALUM Classification				
#	Code	Level 1	Level 2	Level 3	Main aspect
1	**1.1.5**	Conservation and Natural Environments	Nature conservation	Habitat/species management area	Maintenance of habitat or species through management interventions Limited human interventions
2	**1.3.3**	Conservation and Natural Environments	Other minimal use	Residual native cover	No prime use native cover land used for any environmental purpose
3	**2.1.0**	Production from Relatively Natural Environments	Grazing natural vegetation	Grazing natural vegetation	Low level of intervention – Grazing of domestic stock on native vegetation
4	**3.2.0 and 3.2.1***	Production from Dryland Agriculture and Plantations	Grazing modified pastures	Grazing modified pastures and Native/exotic pasture mosaic	Production of pasture and forage from dryland through modification - For grazing
5	**3.3.0**	Production from Dryland Agriculture and Plantations	Cropping	Cropping	Production from dryland through cropping
6	**4.3.0**	Production from Irrigated Agriculture and Plantations	Irrigated cropping	Irrigated cropping	Production from irrigated cropping

*From here onwards, 3.2.1 will be merged with 3.2.0 (level 2) calculations for convenience

Table 6.4: Selected Land uses with their Land Capabilities selected for Maryland Farm

#	ALUM Code	Land Capabilities (LCap) found in original data*	Selected Land LCap for Maryland Farm	Selected LCap for Scenarios
1	**1.1.5**	5, 6 and E	5	5
2	**1.3.3**	4, 5, 5+6, 6 and E	4 and 6	6
3	**2.1.0**	4, 4+5, 5, 5+4, 5+6, 6, 6+5, 7 and E	4, 5, 6	6
4	**3.2.0**	4, 4+5, 5, 5+4, 5+6, 6, 6+5, 7 and E	4, 5, 6	4
5	**3.3.0**	4, 5, 6	4	4
6	**4.3.0**	4, 5, 6	4	4

*Original data extent is the south east bioregion of duck area occupied by chromosol soil.

Table 6.5: Percentage change of land use occurred in following land uses in South East Bioregion under Chromosol soil.

#	ALUM Code	Total Area in Hectares	Percentage Area Changed	Percentage Area not changed
1	**1.1.5**	3740.01	< 1%	> 99%
2	**1.3.3**	12686.69	88%	12%
3	**2.1.0**	59850.92	60%	40%
4	**3.2.0**	61932.06	18%	82%
5	**3.3.0**	124.29	> 99%	< 1%
6	**4.3.0**	889.23	89%	11%

Table 6.6: Future change pathways/ scenarios for Maryland Farm

From Land use			Change To Land use							
Current Land use	Lcap	Soil	Most Likely	%	Moderately Likely	%	Likely	%	Least Likely	%
2.1.0	6	Ch	**1.3.3**	40.8	**3.2.0**	34.01	**1.1.5**	13.66		
3.2.0	4	Ch	**1.3.3**	15.6	**3.3.0**	11.14	**2.1.0**	11.03	**4.3.0**	11.7
1.3.3	6	Ch	**2.1.0**	83.5	**1.1.5**	12.92				
4.3.0	4	Ch	**3.2.0**	61	**3.3.0**	4				

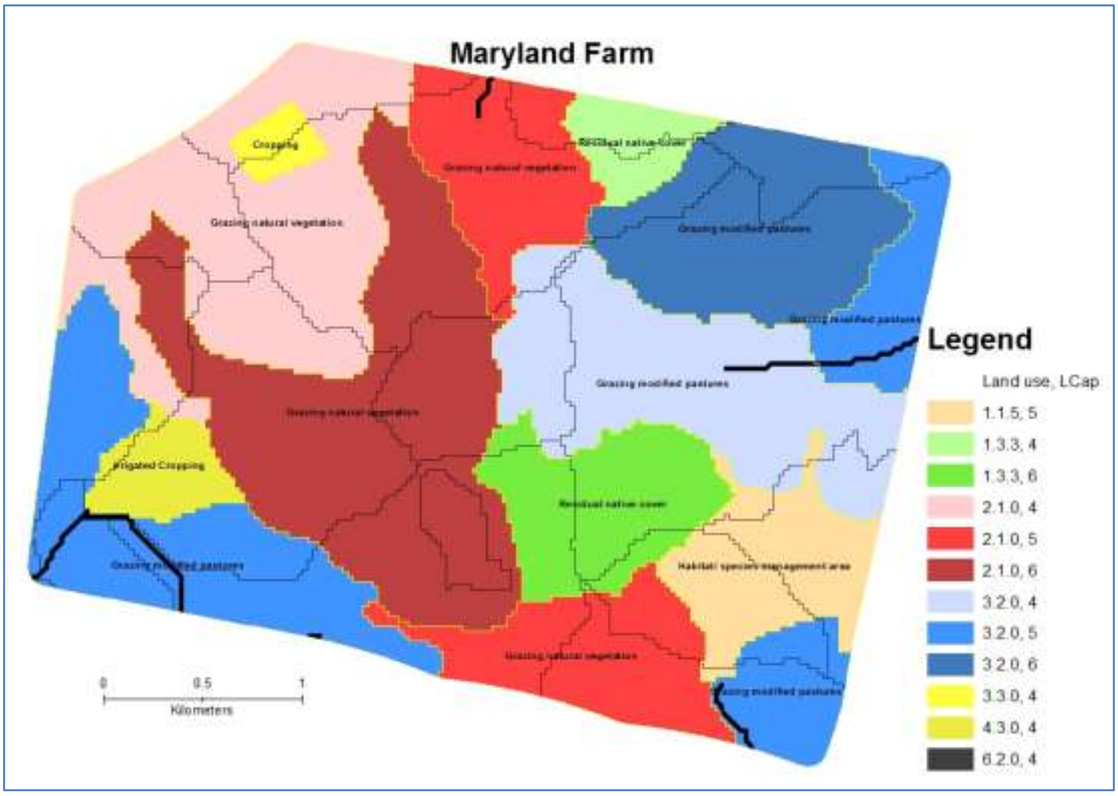


Figure 6.4: Map of Maryland Farm showing various land uses with land capabilities along with catchment boundaries (black lines) and drainage lines (blue lines)

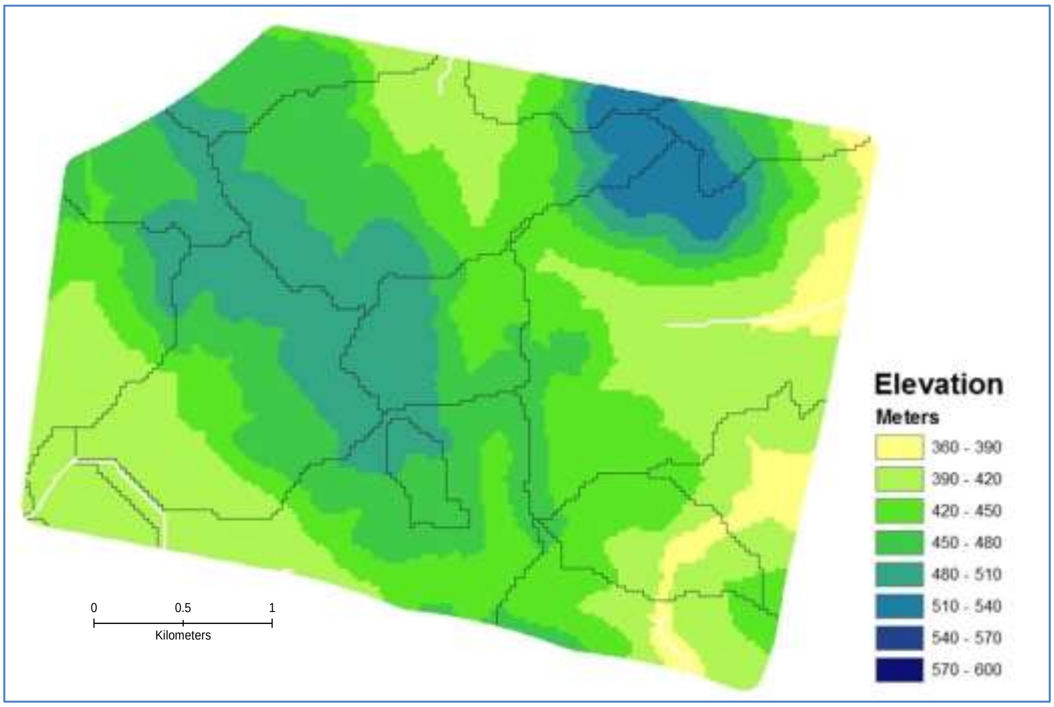


Figure 6.5: Elevation Map of Maryland Farm

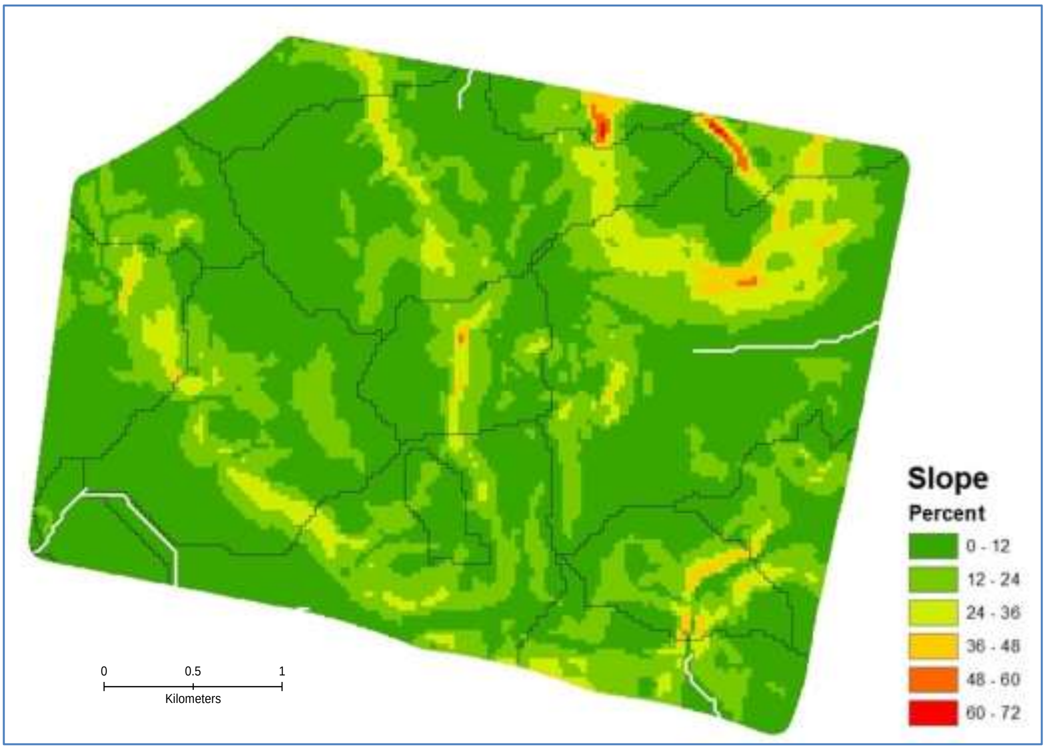


Figure 6.6: Slope map of Maryland Farm

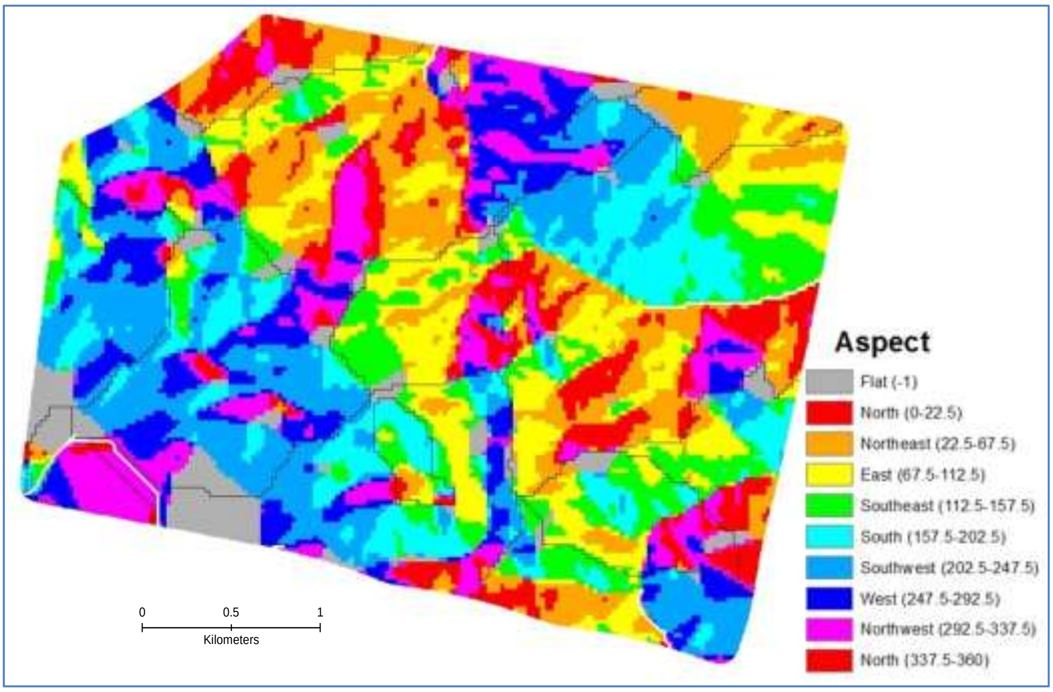


Figure 6.7: Aspect Map of Maryland Farm

Table 6.4 was generated in order to assign land capabilities to Maryland farm land parcels and to allocate a specific land capability for generating change scenarios. Table 6.5, shows the total area under a specific land use in the study area, followed by percentage area 'changed' and percentage area 'not changed' during the change period from 2001 to 2010. Table 6.6 depicts 'change from' and 'change to' land uses with their percentage rate of change observed in historic data, thus representing the future change pathways for Maryland Farm. In this way, the farm included the following *land uses* and *land capabilities* with their respective area (hectares), also shown in the Table 6.7.

Table 6.7: Maryland Farm with land uses, land capabilities and respective area in hectares

Land use Code	Land use	Land Capability	No. of Cells	Area in Hectares
1.1.5	Habitat/ species management area	5	1108	68
1.3.3	Residual Native Cover	4	429	25
1.3.3	Residual Native Cover	6	1245	77
2.1.0	Grazing Natural Vegetation	4	2856	174
2.1.0	Grazing Natural Vegetation	5	2399	146
2.1.0	Grazing Natural Vegetation	6	3682	230
3.2.0	Grazing Modified Pasture	4	2423	150
3.2.0	Grazing Modified Pasture	5	3408	204
3.2.0	Grazing Modified Pasture	6	1860	115
3.3.0	Cropping	4	178	11
4.3.0	Irrigated Cropping	4	451	28
6.2.0	Reservoir	4	1	
TOTAL			20,040	1231

6.4 Maps of Maryland Farm

The geographic data constructed for Maryland Farm was then used to construct the maps shown in Figures 6.4 , 6.5, 6.6 and 6.7. These show land capability, land use, elevation, slope and aspect.

6.5 Conclusion

This chapter has described the methods employed to construct a model farm (Maryland farm) and the maps that show land use, land capability, elevation, slope and aspect. These maps were designed to be used in the stakeholders' survey.

7 Methods

7.1 Introduction

Farmers routinely make practical decisions regarding changing land use, yet there are few studies (e.g. Gibbons and Ramsden 2008, Dury *et al.* 2013) that examine the decision making process of farmers in a spatial context. The literature usually describes the decision making process of a farmer in one of two ways, generally described as either perfect rationality or bounded rationality (Simon 1955, 2000). The notion of perfect rationality proposes that farmers maximise utility, which is mostly confined to economic value, and thus base decision making largely on rational profit maximisation. Bounded rationality, in contrast, includes to a greater extent subjectivity and allowances for behavioural assumptions (An and López-Carr 2012, Holtz and Nebel 2014, Malawska and Topping 2016). Simple observations of land use suggest that maximising profit is not true in most cases and that the decisions that farmers make have a more complex mix of objectives (Edwards-Jones 2006, Holtz and Nebel 2014). A growing interest therefore lies in modelling decision making based on the concept of bounded rationality (Edwards-Jones 2006), with the focus shifting away from maximising profit to a bounded rationality theory that focuses also on behavioural assumptions in the representation of decision-making (Malawska and Topping 2016).

Rationality denotes a style of behaviour that is appropriate to the achievement of given goals, within the limits imposed by given conditions and constraints (Simon 1972). Full or perfect rationality, aiming for profit maximisation, requires unlimited cognitive capabilities, although human beings have limited cognitive capabilities (Selten 1999). The theory of bounded rationality holds that an individual's rationality is limited by the information they have, the cognitive limitations of their minds, and the finite amount of time they have to make a decision. When the decision maker lacks the ability and the resources to make an optimal solution, they often seek a satisfactory solution (Simon 1972, Selten 1999, Kahneman 2003). The decision-maker in this view acts as a satisficer and the strategy is known as satisficing – the adoption of a decision that is <u>satisfactory</u> and will <u>suffice</u> even if it is not optimal. Instead of going for maximisation of utility function, a satisficing strategy is postulated. The concept of bounded rationality was proposed by Herbert Simon as a more holistic way of understanding decision-making, where he applied it on a real world scenario considering the decisions made by real managers in a real organization (Selten 1999).

The concept of bounded rationality has been explicitly applied by Daloğlu *et al.* (2014) in an agent based model designed to elicit and model the decision making of farmers with logit model. Daloğlu *et al.'s* (2014) model constitutes the base model for this study. Daloğlu *et al.'s* (2014) model calculates an overall utility to a farmer for any specified conservation practice and transforms the values of that utility into choice probability using a logical regression, or logit, model. The logit model measures the relationship between a dependent variable and one or more independent variables by estimating probabilities using a logistic function that is the cumulative logistic distribution (Lee 2005, Achmad *et al.* 2015, Schultz *et al.* 2016). Daloğlu *et al.* (2014) use a decision algorithm that integrates available information using a logit framework that allows incorporating uncertainty in decision making and the bounded rationality of the farmers. Daloğlu *et al.* (2014) uses this method to simulate farmer agent in agent based modelling for conservation practice adoption decisions.

Logistic regression was developed in 1958 by a statistician named David Cox (Cox 1958, Strother H. Walker 1967). A logistic regression model, also called a logistic model or logit model, is a method for modelling the relationship between multiple independent variables and a categorical dependent variable. It is commonly used when the dependent variable is dichotomous (Siray *et al.* 2015). A logistic model estimates the probability of occurrence of an event by fitting data to a logistic curve. The logistic function, on which the logistic regression model is based, provides estimates in the range from 0 to 1 making a typical pattern of a logistic curve (Kleinbaum & Klein, 2010) (Park 2013).

The model is useful in two ways; 1) it can predict the value of the dependent variable for new values of independent variable, and 2) it can assist in showing the relative contribution of each independent variable to the dependent variable outcome (Park 2013) (McDonald 2014).

Logistic regression is useful in many research problems of social, educational, medical and psychological sciences where the response variable is dichotomous and multiple linear regression is inadequate (Constantin 2015) (LeBlanc and Fitzgerald 2000). It has also been widely applied in landslide susceptibility assessment and mapping (Akgun 2012, Althuwaynee *et al.* 2014, Kavzoglu *et al.* 2014). Logistic regression does not require many of the assumptions of linear regression models regarding linearity of relationships and normality of error distribution. It can handle non-linear relationships between dependent and independent variables as it applies a non-linear log transformation of the linear regression (Park 2013). The model relationship expresses the predicted dependant variable as the outcome of sum of independent variables, such that each independent variable is multiplied by a coefficient. The coefficients are obtained as best mathematical fit for the model and represent the impact of each variable on the outcome variable (Park 2013).

A series of transformations to the linear regression model, transforms it to a useful logit model. Firstly, as explained by LeBlanc and Fitzgerald (2000) the probability that Y = 1 can be changed to the odds that Y = 1. Odds are the probability of an occurrence divided by the probability of a non-occurrence. The odds ratio can be computed as P/(1-P) where P represents the probability of some defined occurrence. This transformation to an odds ratio will result in predicted probabilities that are not less than 0 but may still exceed 1. Secondly, the transformation needed is to change the odds ratio to a "logit." The logit is formed by taking the natural logarithm (In) of the odds ratio. This transformation results in predicted probabilities that are bounded by zero and one. The transformation is computed as logit (P) = ln (P/1-P). Logistic regression then uses this transformed probability to model the regression equation expressed as Eq. 1.

$$\mathrm{logit(P) = a + bX} \qquad\qquad \dots\dots..\mathrm{Eq.\,1}$$

The Eq. 1 can be simplified for the probability of an outcome with the Eq. 2 given below. In Eq. 2, e = 2.7183, which is the base of the natural logarithm. Eq. 2 can further be simplified as Eq. 3. In Eq. 3, $z = a + b_1X_1 + b_2X_2 + \ldots\ldots\ B_kX_k$ for k predictor variables.

$$P(\text{outcome}) = \frac{e^{a+bX}}{1 + e^{a+bX}} \qquad \ldots\ldots..\text{Eq. 2}$$

$$P(\text{outcome}) = \frac{e^{z}}{1 + e^{z}} \qquad \ldots\ldots..\text{Eq. 3}$$

In another formulation, instead of writing the logit of the probabilities P as a linear predictor, the linear predictor is separated into two, one for each of the two outcomes expressed in Eq. 4 and Eq. 5. In Eq. 4 and Eq. 5, the logarithm of the associated probability as a linear predictor, with an extra term -lnZ at the end serves as normalizing factor and ensures that the result is a distribution. So, exponentiating both sides, gives Eq. 6 and Eq. 7. In Eq. 6 and Eq. 7, $Z = e^{b_0 \cdot X_i} + e^{b_1 \cdot X_i}$, and the resulting equations are Eq. 8 and Eq. 9. Or generally, they can be expressed as Eq. 10.

$$\ln P(Y_i = 0) = b_0 \,.\, X_i - \ln Z \qquad \ldots\ldots..\text{Eq. 4}$$

$$\ln P(Y_i = 1) = b_1 \,.\, X_i - \ln Z \qquad \ldots\ldots..\text{Eq. 5}$$

$$P(Y_i = 0) = \frac{1}{Z} e^{b_0 \cdot X_i} \qquad \ldots\ldots..\text{Eq. 6}$$

$$P(Y_i = 1) = \frac{1}{Z} e^{b_1 \cdot X_i} \qquad \ldots\ldots..\text{Eq. 7}$$

$$P(Y_i = 0) = \frac{e^{b_0 \cdot X_i}}{e^{b_0 \cdot X_i} + e^{b_1 \cdot X_i}} \qquad \ldots\ldots..\text{Eq. 8}$$

$$P(Y_i = 1) = \frac{e^{b_1 \cdot X_i}}{e^{b_0 \cdot X_i} + e^{b_1 \cdot X_i}} \qquad \ldots\ldots..\text{Eq. 9}$$

$$P(Y_i = c) = \frac{e^{b_c \cdot X_i}}{\sum_h e^{b_h \cdot X_i}} \qquad \ldots\ldots..\text{Eq. 10}$$

In this research, the farmer's decision making is being modelled using the theoretical concept of bounded rationality utilising the logistic regression model, as the logit model is useful for dichotomous dependent outcomes. In this case, the dichotomous dependent outcome is defined by: either the land use being selected for change (value 1) or the land use not being selected for change (value 0), for each scenario. In the logit model, the probability ranging from 0 and 1 is transformed to odds ranging from 0 and +∞ , which in turn is transformed to log odds ranging from -∞ to +∞. This transformation is an attempt to get around the restricted range problem (Institute for Digital Research and Education 2016). Therefore, bounded rationality as well as the logit model has been chosen as a method for this research.

The work reported here extends the method reported by Daloğlu *et al.* (2014), by modifying the logit model and framing it to the objectives of a geospatial model that integrates the ecological values of farmers, the ecological values of stakeholders other than farmers, and the economic returns to farmers for different land uses and land use change scenarios.

An overview of the methods and sequence of tasks employed in this research is presented as a flow chart in Figure 7.1.

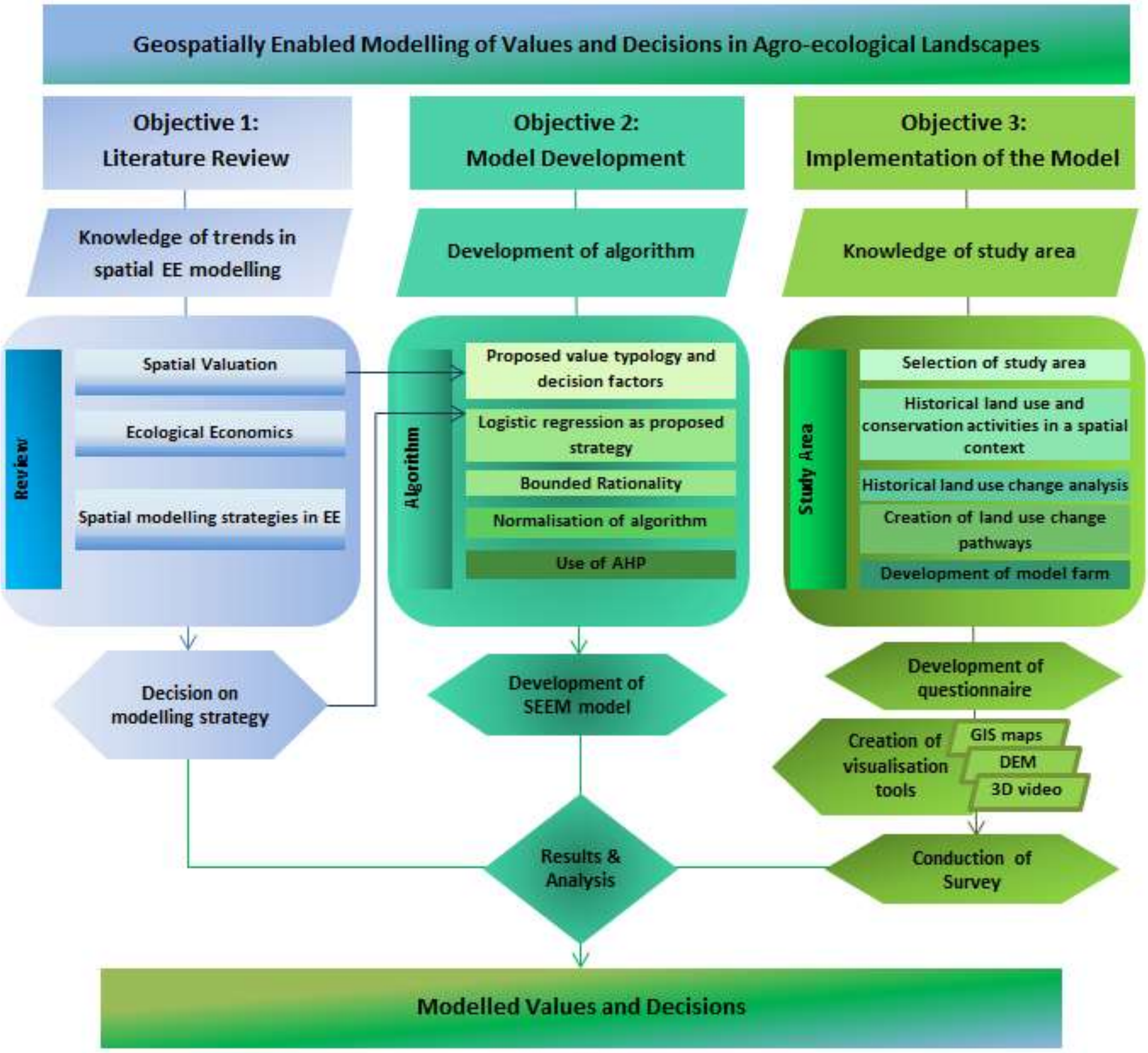


Figure 7.1: Overview of methods employed in this research depicting the tasks undertaken to achieve the research objectives

The modelling tool developed in this research is called SEEM (Spatially enabled Ecological Economic Model). It is intended to provide both the analytical capacity and the interactive functionality required in order to facilitate informed analysis of stakeholder values and an interactive capacity to interrogate the impact of both economic and ecological values attached to alternative land uses in a manner that is geospatially enabled.

SEEM is designed to accept user-entered values and subsequent calculated decisions on land use change scenarios. SEEM is intended to be a generalizable and flexible tool that has the capacity to explore potential futures and management scenarios based on land use and land use change in terms of economic, spatial and ecological aspects and thus provide a platform for investigation and experimentation.

7.2 Model algorithm and its components

In this study, the logit model is used to derive values that represent a combined decision making factor for farmers in a defined study area, in comparison to a single type of farmer as captured in Daloğlu *et al.* (2014). Further, the model is not based on an agent based model simulation of farmer agents, rather it utilises the logit model and uses it in an interactive, spatial decision support tool (SEEM). The logit model proposed by Daloğlu *et al.* (2014) has been altered in order to better reflect that farmers decisions are influenced and effected by certain external pressures including the values of other stakeholders. The overall utility[20] to farmers of a land use decision are calculated for every land use type in question for adoption and are then transformed into choice probability using the logit model. In this case, the overall utility to farmers is calculated by obtaining and aggregating values that farmers and other stakeholders hold for a specific land use in question.

The decision algorithm combines economic returns from land use, government subsidies, covenant programs, and policies, the influence of farmers' values and the values of stakeholders other than farmers for the land use, to formulate an adoption decision. These are combined in a utility function $F_{decide\ (k,\ l)}$ for land use combination *k* and management practice *l*, which is a combination of three sub-functions (Eq. 5). This model assigns probabilities to different options, where the probability of an inferior option could be non-zero (Eq. 6).

$$\mathrm{F_{decide\ (k,l)}} = \mathrm{b_1\ F_{econ\ (k,l)}} + \mathrm{b_2\ F_{value\ (k,l)}} + \mathrm{b_3\ S_{value\ (k,l)}} \quad \ldots\ldots\ldots\ldots\ldots \text{Eq. 5}$$

$$\text{Selection Probability (k, l)} = \frac{e^{\mathrm{F\ decide\ (k,l)}}}{\sum e^{\mathrm{F\ decide\ (k,l)}}} \ldots\ldots\ldots\ldots \text{Eq. 6}$$

The probability of occurrence expressed in Eq. 6 defines the probability of each land use within a scenario resulting in a probability of 0 or 1. If the selection probability for a given land use within a scenario results in 1, it denotes the land use is selected for change, whereas, 0 denotes the land use not being selected.

As the model can be run at any time, the time steps are not considered, although it is to be noted that economic returns are calculated on annual per hectare values. The model is able to represent change scenarios based on changes in the land use, and can represent changes based on changes in management practices carried out on the same land use. The model does not incorporate any biophysical measures of the land use, but generates decisions on the basis of three weighted sub-functions (Eq. 5). The model is

[20] **Overall utility** is the aggregate sum of satisfaction or benefits that an individual gains from using or consuming given amount of goods and services (Satija 2009).

developed and executed in a spatial manner (as explained in the section 'SEEM model development'), thus incorporating spatial dynamics into the study.

The following sections explain the components of the Spatio-Ecological Economic Model (SEEM), the value typology used for the study, the normalisation carried out in the algorithm, and the use of an Analytical Hierarchical Process (AHP) for assigning weights to the sub-functions. While AHP has been used as the weighing method for this study, any other justifiable method of assigning weights could be implemented.

7.2.1 Components of the Spatio-Ecological Economic Model (SEEM)

At every model run, and for every land use k and management practice l, a value for $F_{decide\ (k,\ l)}$ is calculated. The three sub-functions of $F_{decide\ (k,\ l)}$ are $F_{econ\ (k,\ l)}$, $F_{value\ (k,\ l)}$ and $S_{value\ (k,\ l)}$. These sub-functions are multiplied by weights (b) that are determined using an Analytical Hierarchical Process (AHP) pair wise comparison method. In this process, $F_{econ\ (k,\ l)}$ represents the economic return from land use k with a management practice l; $F_{value\ (k,\ l)}$ represents the farmers' value for the land use k and management practice l; and $S_{value\ (k,\ l)}$ represents other stakeholders' value for the land use k and management practice l. The three sub-functions and AHP are explained as follows.

A. Sub-Function - Economic return

$F_{econ\ (k,\ l)}$ represents the economic return obtained from land use *k* under land management practice *l*. The economic returns are calculated for each land use type based on the gross margin per hectare values of crops or livestock that the farmer expects to achieve from their land. The economic returns are therefore the agricultural revenue generated from land production. $F_{econ\ (k,\ l)}$ is calculated using the following formula, following the method of Daloğlu *et al.* (2014):

$F_{econ\ (k,\ l)} = P\ (A\text{-}F\text{-}a) + [(gF + ra) - c]$ ……(Eq. 3)

where,

P = Gross Margin
A = Total Area in Hectares
g = Economic incentive associated with structural practice per hectare
F = Area of land allocated for structural practice
r = Policy adoption incentive payments to farmer per hectare
a = Area of land allocated for policy adoption
c = Cost of allocating land to structural and policy uses

The formula in Eq. 3 includes any incentive payments given to a farmer for a policy adoption 'r' or structural practice 'g' on land use *k* and the area 'a' for policy adoption and associated area 'F' for structural practices within the total area of 'A'. The policy adoption and structural practice may influence the size of the production area, and so the areas of policy adoption and the structural practice are subtracted from the total production area. Policy adoption can include any policy adoption payment incentives such as conservation covenants, whereas structural practices may include filter strips, grassed waterways etc. Agro-ecological policy programs are usually voluntary and farmers are able to engage with or select any policy program that fits their needs. This formula therefore allows the user to include or not include such incentive programs if adopted by the farmer, or if the influence of an incentive program needs to be tested in future scenarios. The model calculates the area allotted for these incentive programs on run time by user input indicating the percentage of the total area allotted for such incentives.

The gross margin depends upon the land use k and management practice l. For example, if the land use is grazing land according to the ALUM classification system, then it can have a variety of management practices including e.g. sheep grazing for wool production or sheep grazing for meat production. Thus, the management practice adopted on that land use determines the gross margin values. For our study in Tasmania, gross margin values were obtained from the Tasmanian Department of Primary Industries, Parks, Water and Environment (DPIPWE) (DPIPWE 2015).

The SEEM model has the capability to incorporate the cost of change from one land use to another through the interface on F_{econ} that includes 'cost of allocating land through structural and policy use per hectare', denoted as 'c'. This can incorporate any costs associated with a proposed land use change. The costs of possible structural and policy interventions were not included in the data modelled on the Maryland farm, in large part because these costs may be included in the modelling process as part of the dynamic use of the model in a consultative process.

B. Sub-Function - Values of farmers

$F_{value\ (k,\ l)}$ represents the values that different types of farmers have for a certain type of land use k under certain management practice l. Representative values were obtained from farmers for different types of land uses under specific management practices for Maryland Farm through a questionnaire survey. The farmers were later categorised into different types based on the information they provided regarding their interests and activities in the region. The values of similar type of farmers were averaged, followed by summation of values of all types of farmers under each value typology category (explained under 'Value Typology'). All the values under value typology categories were further summed to obtain a final $F_{value\ (k,\ l)}$.

C. Sub-Function - Values of other stakeholders

Similarly, $S_{value\ (k,\ l)}$ represents the values stakeholders other than farmers have for a certain type of land use k under a management practice l. The values were obtained from stakeholders for land uses under specific management practices for Maryland Farm through a questionnaire survey. The stakeholders were later categorised into different types based on the information they provide regarding their interests and activities in the region. The values of similar types of stakeholders were averaged, followed by summation of values of all types of stakeholders under each value typology category (explained under 'Value Typology'). All the values under value typology categories were further summed to obtain final $S_{value\ (k,\ l)}$.

7.2.2 Value Typology

A value typology is required in order to capture and represent the values that farmers and other stakeholders hold for each type of land use k. Any reasonable value typology suitable for the study area could be adopted, and a number are presented in the literature (e.g. Brown and Kyttä 2014). In our case, we have applied and modified a widely used value typology (e.g. Clement and Cheng 2006, Sherrouse *et al.* 2011, 2014, van Riper *et al.* 2012, Sherrouse and Semmens 2014). The value typology describes different types of values through a sentence statement. The value typology as given in Table 7.1 and a Likert scale was used in a questionnaire to ascertain the values of farmers and other stakeholders with interests in the Midlands region of Tasmania.

Table 7.1: Value types and description used for the Midlands, Tasmania survey

Value Types	**Value Description**
Aesthetic value	I value this land use because I enjoy the scenery, sights, sounds, smells, etc.
Biological diversity value	I value this land use because it provides a variety of fish, wildlife, plant life, etc.
Cultural value	I value this land use because it is a place for me to continue and pass down the wisdom and knowledge, traditions, and way of life of my ancestors.
Economic value	I value this land use because it provides an economic return.
Future value	I value this land use because it allows future generations to know and experience the land as it is now.
Historic value	I value this land use because it has places and things of natural and human history that matter to me, others, or the nation.
Intrinsic value	I value this land use in and of itself, whether people are present or not.
Learning value	I value this land use because we can learn about environment through scientific observation and experimentation.
Life Sustaining value	I value this land use because it helps produce, preserve, clean and renew air, soil and water.
Recreation value	I value this land use because it provides a place for my favourite outdoor recreation activities.
Spiritual value	I value this land use because it is a sacred, religious, or spiritually special place to me or because I feel reverence and respect for nature here.
Therapeutic value	I value this land because it makes me feel better physically and/or mentally.
Other reason/s	Please mention the reason here:

7.2.3 Decision Factors

Decision factors are a set of factors associated with changing or retaining the land use. In this study a set of decision factors were used. The decision factors used in this study were land capability, elevation, slope, aspect, dominant soil, drainage lines, area of land, surrounding land use, policy and other, where 'other' stands for any other factor not included and that would like to be added and discussed by any participant. The decision factors do not serve as an input into the model algorithm, although the model utilises the decision factors to present the trend of these factors in changing and not changing the land use, facilitating the overall overview of the land use itself while comparing it with other land uses in the scenario.

7.2.4 Land Management

Land management practice describes the way in which land is managed - the means by which a land use outcome is achieved (Australian Government 2010). It can also be described as the way by which land resources are used. Land management largely depends on the land use and land capability of the land. For example, native vegetation being grazed by sheep and having a land use Grazing Natural Vegetation and land capability of level 6, can be managed either for fine wool production or for breeding and trading first cross ewes. Similarly, if the grazing is by cattle, on similar land use and land capability, the land management could be either for beef breeding or beef trading. Therefore, the land management option decided by a land owner/ farmer is based on the resources of the land (in this case, sheep/cattle), and the land use and land capability. The range of economic outcomes from the land largely depends on the resources of the land and the associated land management practice adopted.

7.2.5 Normalisation of algorithm outputs

The output values for each of the three sub-functions can range from a minimum value of 0 to an unknown maximum value that is based on the number of value types in the value typology. As each category in the value typology could obtain a maximum value of 20, and the total number of value types was 13 in this case, the maximum obtainable value was 260 (20x13). In order to preserve the relative scaling of individual values obtained against each value type as derived from the survey of farmers and other stakeholders, each value type was normalised. Further, $F_{econ\ (k,\ l)}$ was also normalised according to the given range i.e. minimum = 0 and maximum = 20 x (number of value types). An equation that linearly rescales data values having observed min and max into a new arbitrary range min' to max' is given below and derived in Appendix D.

$$\text{New Value} = \frac{(\max' - \min')}{(\max - \min)} * (\text{Value} - \max) + \max' \qquad \text{(Eq. 4)}$$

7.2.6 Assigning weights based on AHP

After normalisation, each sub-function is further multiplied by a weight assigned by the user in the model. In this case we used AHP, pair-wise comparison method to weight the three sub-functions.

The Analytic Hierarchy Process (AHP) is a multi-criteria decision-making method that was originally developed by Prof Thomas L Saaty in 1977 (Saaty 1977). AHP is a theory of measurement that uses pairwise criteria comparisons and relies on the judgements of experts to arrive at a scale of preferences among sets of alternatives (Marinoni 2004, Saaty 2008), and so is a method to derive ratio scales from paired comparisons. In using AHP, one needs a hierarchic structure and pairwise comparisons to establish relationships within the structure. In discrete paired comparisons, it leads to dominance matrices that are positive and reciprocal (Saaty and Vargas 2012). Dominance matrices are used to rank individuals that compete against each other and to find which vertex in a matrix is the most dominant.

AHP provides measures of judgment consistency, derives priorities among criteria and alternatives, and simplifies preference ratings among decision criteria using pair-wise comparisons. In this study, AHP is used to develop priority weights among the three sub-functions (F_{econ}, F_{value}, S_{value}) through pair-wise comparison. This is carried out through following steps (Saaty 1977, 1994a, 1994b, 2008, Marinoni 2004, Ishizaka and Labib 2009, Saaty and Vargas 2012):

1. developing a pairwise comparison matrix

2. normalising the matrix
3. calculating a consistency ratio.

A scale of comparison was taken from Saaty (2000, 2008), and is shown in Table 7.2. The first step in the AHP procedure is to make pair wise comparisons between each criterion.

Table 7.2: Example scale for comparison from Saaty (2008)

Degree of importance	Scale
Extremely less important	1/9
	1/8
Very strongly less important	1/7
	1/6
Strongly less important	1/5
	1/4
Moderately less important	1/3
	1/2
Equally Important	1
	2
Moderately more important	3
	4
Strongly more important	5
	6
Very strongly more important	7
	8
Extremely more important	9

A. Step 1: Developing a pairwise comparison matrix:

In this step, a pairwise matrix is generated among the alternatives for each relevant criterion. In each comparison matrix, each row entry (alternatives/criteria) is compared to each column entry by using a scale of (1/9 – 1 – 9) of relative importance. Results of the comparison (for each alternative/criteria pair) were described in terms of integer values based on the scale given in Table 7.2. The values were assigned based on an expert's opinion (a researcher in the field of Agriculture Economics) in order to allocate weights to the sub-functions, and a pair-wise comparison matrix was generated (Table 3).

Table 7.3: Pair-wise comparison matrix based on three sub-functions – Step 1

Sub-Functions	$F_{econ\ (k,\ l)}$	$F_{value\ (k,\ l)}$	$S_{value\ (k,\ l)}$
$F_{econ\ (k,\ l)}$	1	3	7
$F_{value\ (k,\ l)}$	1/3 = 0.333	1	5
$S_{value\ (k,\ l)}$	1/7 = 0.142	1/5 = 0.2	1
Total	1.475	4.2	13

B. Step 2: Normalising the matrix

Normalisation of the resultant pair-wise comparison matrix shown in Table 7.3 was achieved by creating a priority vector, computed using a normalised principal eigenvector. A normalised principal eigenvector is derived by dividing each entry in the column by its column sum.

Table 7.4: Normalisation of values

Sub-Functions	$F_{econ\ (k,\ l)}$	$F_{value\ (k,\ l)}$	$S_{value\ (k,\ l)}$	Total	Average
$F_{econ\ (k,\ l)}$	0.677	0.714	0.538	1.929	0.643
$F_{value\ (k,\ l)}$	0.225	0.238	0.384	0.847	0.282
$S_{value\ (k,\ l)}$	0.096	0.047	0.076	0.219	0.073

The sum of the new column entry should equal one. The average of each row is calculated using the new entry in the matrix, which provides the priority vector. In this way, Table 7.4 is populated.

C. Step 3: Calculating consistency ratio

Calculating a consistency ratio confirms (or otherwise) the consistency of importance of one entry over another in the matrix. A consistency ratio is calculated by first computing a consistency index and then using a random index to obtain the ratio. The first step in calculating the consistency ratio is taking a product of the original pairwise comparison matrix generated through Step-1 and represented in Table 7.3, with the priority vector (average value) calculated in Step-2, represented in Table 4. In this way, a new matrix is generated represented as A. Avg. The second step in calculating the consistency ratio requires that each entry in the new matrix is divided by the priority vector (average value) to create new corresponding matrices represented as a.Avg, b.Avg and c.Avg. Three values of consistency measure were thus computed using this method. The average of these three consistency measures is known as the maximum eigenvector (ƛ max). The closer the value of the maximum eigenvalue to the number of rows/columns in the pairwise comparison matrix, the better the consistency among the entries (Lange *et al.* 2012, Tian *et al.* 2013, Giri and Nejadhashemi 2014).

$$\mathbf{A} \cdot \mathbf{Avg} = \begin{pmatrix} 1 & 3 & 7 \\ 0.333 & 1 & 5 \\ 0.142 & 0.20 & 1 \end{pmatrix} . \begin{pmatrix} 0.643 \\ 0.282 \\ 0.073 \end{pmatrix}$$

$$\mathbf{a} \cdot \mathbf{Avg} = \frac{\left[\begin{pmatrix} 1 & 3 & 7 \end{pmatrix} . \begin{pmatrix} 0.643 \\ 0.282 \\ 0.073 \end{pmatrix} \right]}{0.643}$$

$$\mathbf{b} \cdot \mathbf{Avg} = \frac{\left[\begin{pmatrix} 0.333 & 1 & 5 \end{pmatrix} . \begin{pmatrix} 0.643 \\ 0.282 \\ 0.073 \end{pmatrix} \right]}{0.282}$$

$$\mathbf{c} \cdot \mathbf{Avg} = \frac{\left[\begin{pmatrix} 0.142 & 0.2 & 1 \end{pmatrix} . \begin{pmatrix} 0.643 \\ 0.282 \\ 0.073 \end{pmatrix} \right]}{0.073}$$

Sub-Function	Consistency Measure
$F_{econ\ (k,\ l)}$	3.118

F ***value (k, l)***	3.058
S ***value (k, l)***	3.010
ƛ max	3.062

The third step in calculating consistency index is to subtract the number of criteria (3) from ƛ max and divide it by the number of criteria (3) minus one, as computed by Saaty (2000).

$$CI = \frac{ƛ\,\max - n}{\mathrm{n} - 1}$$

$$\mathrm{CI} = \frac{3.062-3}{3-1} = 0.03134$$

For computing the consistency ratio, the value of the random index was chosen based on the number of criteria (3 = 0.58) using a standard table obtained from Saaty (1977). The random index varies depending on the *n* value. If the consistency ratio is 0.1 or less, then the pairwise comparison matrix formed during the first step is consistent. Otherwise, a rearrangement of entries in the pairwise comparison matrix is performed to ensure the logic and consistency between the alternatives/criteria.

Table 7.5: Random indices from Saaty (1977)

n	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**
RI	0.00	0.00	**0.58**	0.9	1.12	1.24	1.32	1.41

The formula for calculating the Consistency Ratio (CR) is as follows:

CR = CI/RI

CR = 0.03134/0.58

CR = 0.0540

The values of the consistency index and random index resulted in a consistency ratio of 0.0540 (5.4 percent). The suggested value by Saaty for the CR is 0.1 or 10 percent (Saaty 1988, Franek and Kresta 2014, Kułakowski 2015). The value derived here is therefore consistent.

7.3 Collection of data on values

In order to test and calibrate the SEEM model on Maryland Farm, data was required from diverse stakeholders. Using the value typology given in the Section above ('Value Typology') and the table on 'Future change pathways/ scenarios for Maryland Farm' from Chapter 6, six different scenarios were generated for which values of diverse stakeholders were collected. Along with values for these six land use scenarios, a group of decision factors were also tested in order to determine whether the factors promote change or no change in the scenarios. The scenarios and the decision factors are further described in the following sections.

7.3.1 Questionnaire development

A semi-structured questionnaire was constructed in order to capture values from a diverse set of stakeholders and to support model calibration. The questionnaire was approved by the Tasmania Social Sciences Human Research Ethics Committee (Ethics Ref No. H0015188).

When designing the questionnaire, GIS derived maps of Maryland Farm and additional visualisation tools such as 3D flyovers and panoramic views of the farm, together with photographs of particular land uses were used to aid visualisation and understanding of the landscape. Visual images and realistic panoramas are, not surprisingly, considered a preferred mechanism for explaining complex landscape information and more easily understood (e.g. Smith *et al.* 2012). The questionnaires and supporting maps are attached as Appendix E.

The questionnaire was divided into three parts: the first two being informative and the third containing the questions. The first part included descriptions of the terms and concepts used in the questionnaire survey; the second included a description of Maryland Farm; the third contained the survey questions on values and decision factors. The third part was further divided into three sections: Section A dealt with questions about past experiences, Section B addressed the stakeholders' interests in the Midlands region; and Section C described the scenarios and required responses to questions addressing values attached to land uses and addressing decision factors.

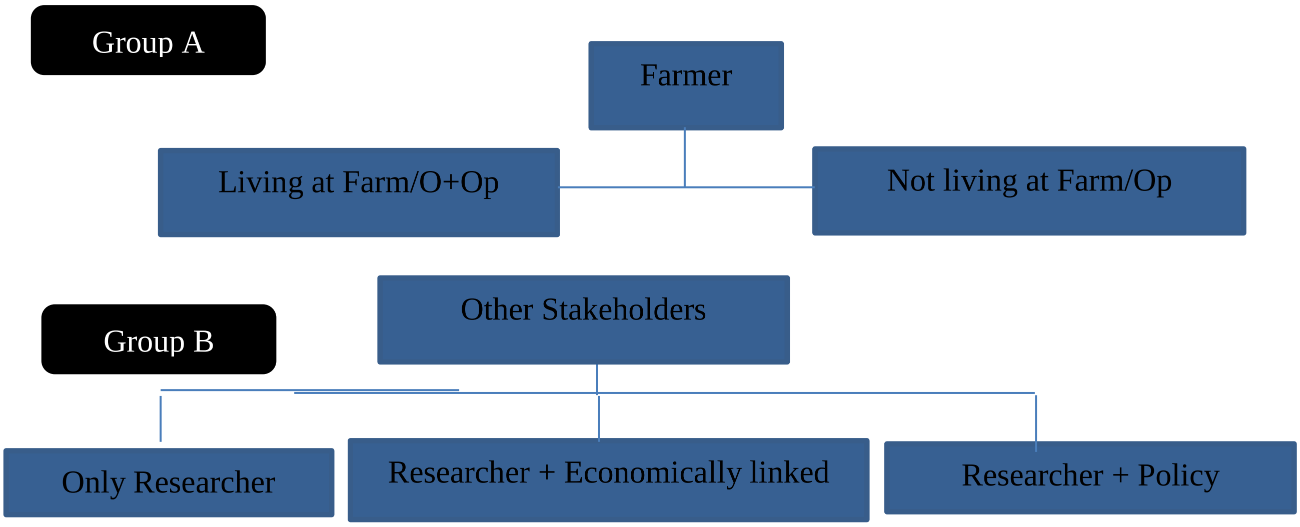


Figure 7.2: Categorisation of participants into groups for model calibration

In this study, the identification and classification of farmers and other stakeholders was carried out manually based on each interviewee's responses to Part 3 Section B of the questionnaire. If there were a larger cohort, then these inputs could be categorised through cluster analysis. The division of land owners could also be based on literature studies of the area (van Ingrid *et al.* 2011) dividing land owners on the basis of their attitudes and preferences, although this method may result in divergent values that may lead to unsatisfactory results. Thus, it was preferable to categorise stakeholders based on responses received through the survey.

7.3.2 Selection of participants and sample size

The questionnaire was used in a series of meetings with individual stakeholders, across a range of participants including landowners, farmers, government officials, representatives of non-government organisations, community members and researchers. The aim was to obtain a small but reasonably diverse sample of potential participants; it was not the expectation that the sample would be a statistically representative sample of the community, but sufficient for testing the SEEM model.

Table 7.6: Group A and B categories of participants classified for model calibration

Group A	Group B
1: Owner + Operator - Living on Farm	1: Only Researcher
2: Operator – living Outside Farm	2: Researcher + Businessperson
	3: Researcher + Policy

Seven participants were interviewed for the survey and these participants were further categorised based on Part 3, Section B of the questionnaire, in order to calibrate the model. The categories are further described in Figure 7.2 and Table 7.6. Owing to the multiple attributes of the selected participants, some of the participants fell into more than one category, and thus, the data obtained from such participants were utilised for more than one category. It is to be noted that this process was undertaken only to create data for a wide range of categories. The number of participants was sufficient for testing of the SEEM model: the data was not meant to be and should not be considered statistically representative.

7.3.3 Coding of data and ethics approval

De-identification of the questionnaire survey responses was ensured by allocating a unique code manually to participants to allow coding and analysis of data. A database of the completed questionnaires was constructed in Microsoft Excel and further utilised to calibrate the SEEM model for analysing results.

7.4 SEEM Model Development

The following section describes the development of the SEEM model and interfaces. The construction of the model is illustrated in Figure 7.3.

7.4.1 Conceptual framework of SEEM model with connections to user interface model

This section describes the input datasets, the runtime processes, and the output data.

A. Model Inputs

There are three types of input data: 1) shape (.shp) files, 2) Excel files, and 3) data input during runtime by the user through the model's software interfaces.

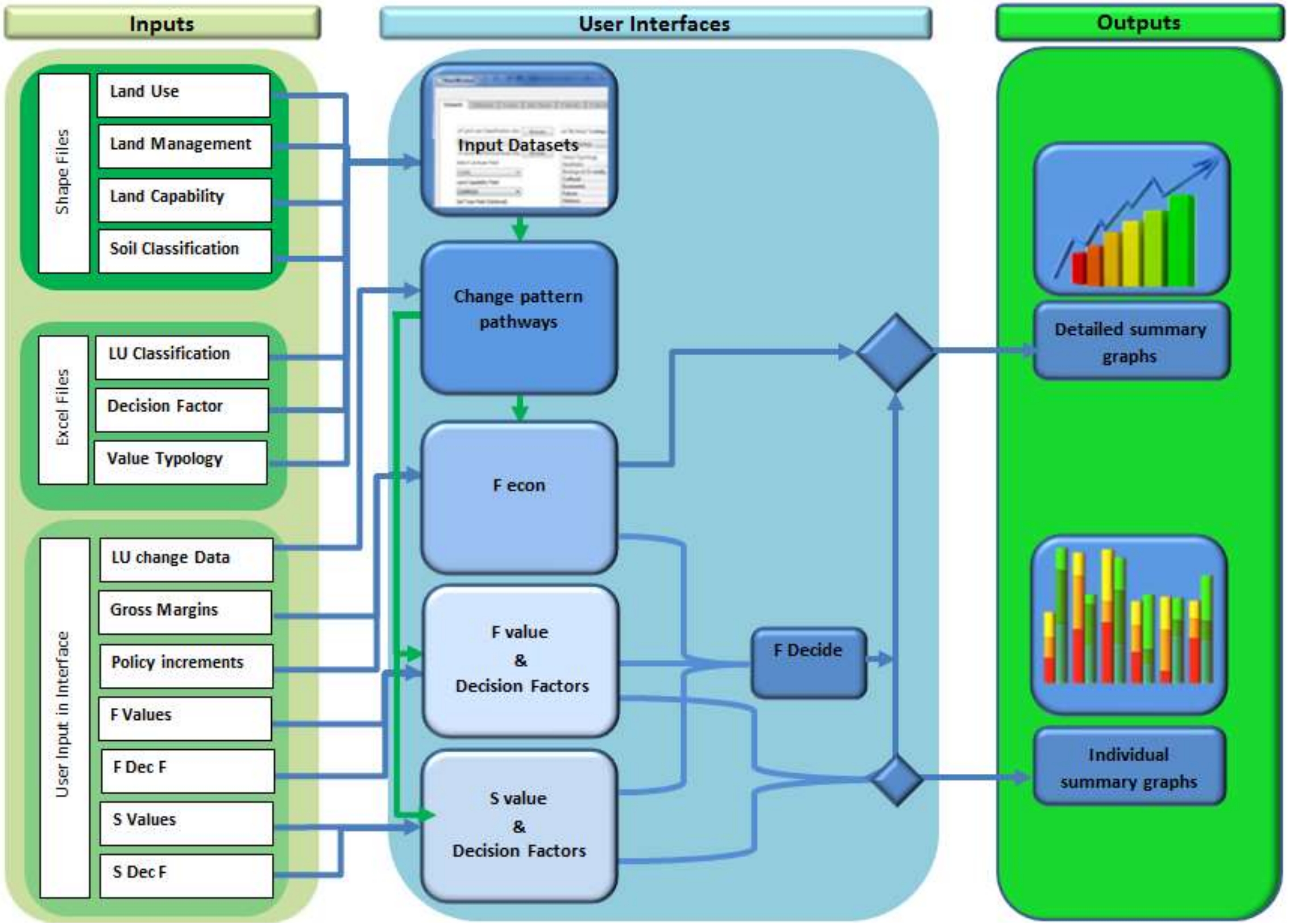


Figure 7.3: An overview of the SEEM conceptual model and its connection to the user interface model

i. Data input as .shp file

Shape files are a geospatial vector data format used by geographic information system (GIS) software including ESRI ArcGIS. The input shape file contains the following attribute information in columns:

- Land use ALUM classification codes
- Land use ALUM classification Level 1 description
- Land use ALUM classification Level 2 description
- Land use ALUM classification Level 3 description
- Land capability classification Level
- Area of land parcels

The SEEM model uses the shape file to compute the area of each parcel and, on the basis of the land use, calculates the results for F_{econ}.

ii. Data input as Excel files

The input Excel files contain a list of values (e.g. Aesthetic, Cultural, Biodiversity etc.), a list of decision factors (e.g. land capability, elevation, slope, aspect, drainage lines etc.), and the coded land use ALUM classification system. The user is able to utilise all, or only a few, of the values in value typology and all, or only a few, of the decision factors, or introduce a different set of values and decision factors tailored to the needs of their study.

iii. Data input by user in model interfaces

a. Change Pathways

The user is required to define a change pathway for each of the selected land uses to be analysed. In this case, we have utilised a change pathway derived from historic land use change data for Tasmanian Midlands, as discussed and provided in Chapter 6 under the Table 'Future change pathways'.

b. Gross margin values and incentives

The user is required to input gross margin values for selected land use pattern together with data for any payments or incentives attached to the land use through policy adoption or through structural practice. In this way, the model calculates the economic return from the land use pattern. The user also has to allocate a percentage area of the total land use area in order to associate certain policy or structural incentives with the land use. The model automatically calculates the area allocated for such incentives and subtracts any costs associated with this, further computing the total economic return achieved from the land use.

c. Using data on values for calibration

The user is required to input 'values data' for the farmers and other stakeholders through the user interfaces. The user initially selects a value typology (Excel file) which has a list of value types. The user can select all, or a few, value types from the list. The user also defines the number of types of farmers and other stakeholders involved in the study. The user then assigns values to each land use type under the value types selected for each type of farmer and other stakeholder. The model compiles and calculates these values and later presents this data in the form of charts. In this case, the value data is obtained from a semi-structured questionnaire survey carried out in the study area.

d. Using data on decision factors for calibration

The user is required to input decision factors associated with changing or retaining the land use. The decision factors are a list of factors predefined and selected initially (from the Excel file) and can be changed on a case by case basis. From the list of decision factors, the user can select all, or only a few, factors. This is followed by entering values on decision factors by the user for each land use for each farmer and stakeholder type selected. The model then compiles this data and presents it in the form of charts. In this case, the data was obtained from the semi-structured questionnaire survey carried out for the study.

B. Model Outputs

The model outputs are described below.

i. Resultant Change Pathway Results

The resultant change pathways will be visible in an interface highlighting the land use with highest value for F_{decide} in each scenario shown in Figure 7.4. The model has the ability to visualise the detailed results that constitutes F_{decide} through Detailed and Individual summaries.

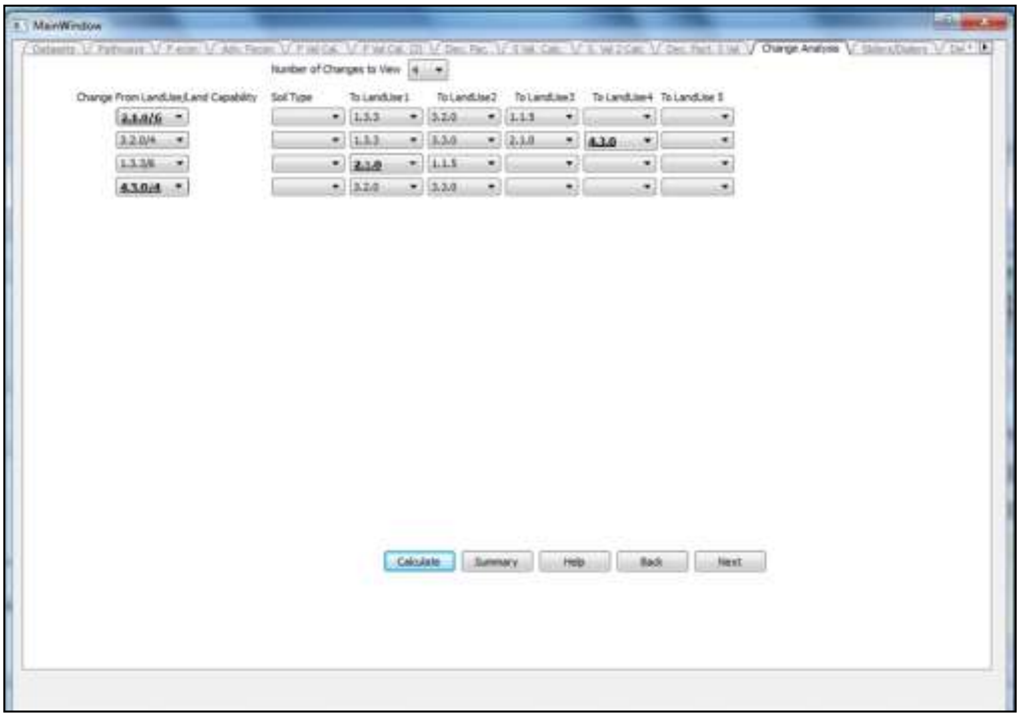


Figure 7.4: Figure indicating resultant change pathways highlighting the land use with probability 1 in each scenario through bold and underlined fonts

ii. Detailed Summary Charts

Detail summary charts are generated, with each chart representing a single scenario or change pathway selected by the user. Each chart shows the values of F_{econ}, F_{value}, S_{value} and F_{decide} for all the land uses selected in the scenario, giving an overall view of rise and fall of F $_{decide}$ in land uses and the sub-function contributing to that rise and fall in charts. An illustrative chart is shown in Figure 7.5.

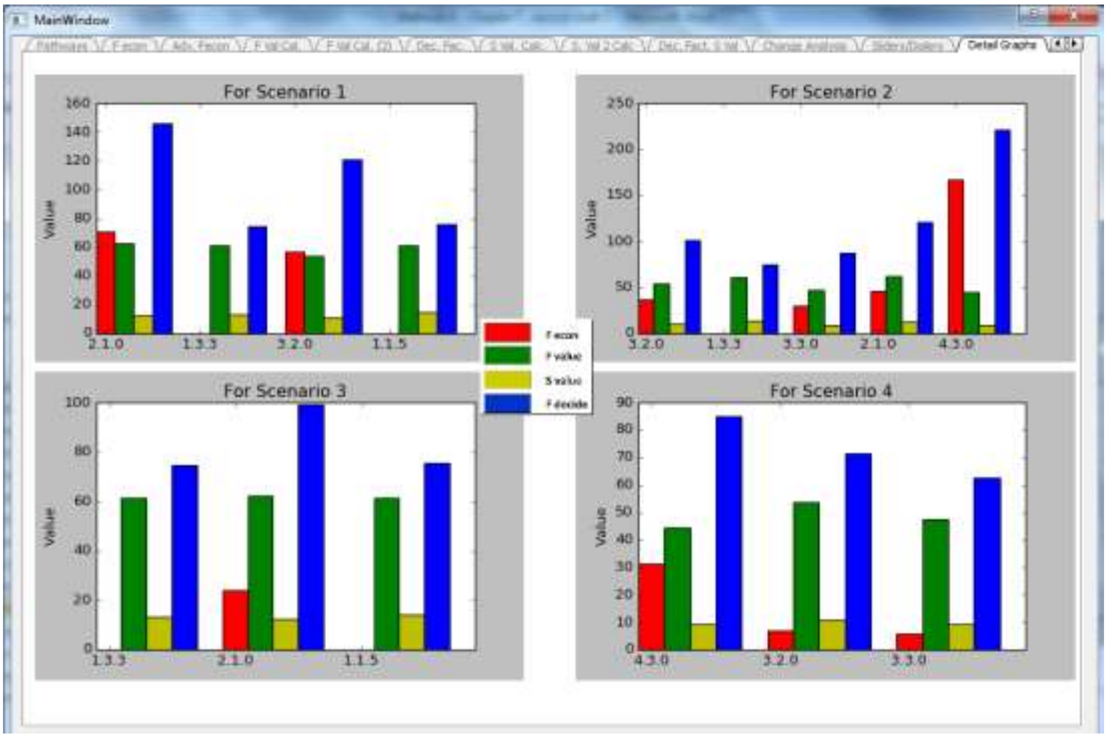


Figure 7.5: Charts showing values of F $_{econ}$, F $_{value}$, S $_{value}$ and F $_{decide}$ for land uses in each scenario

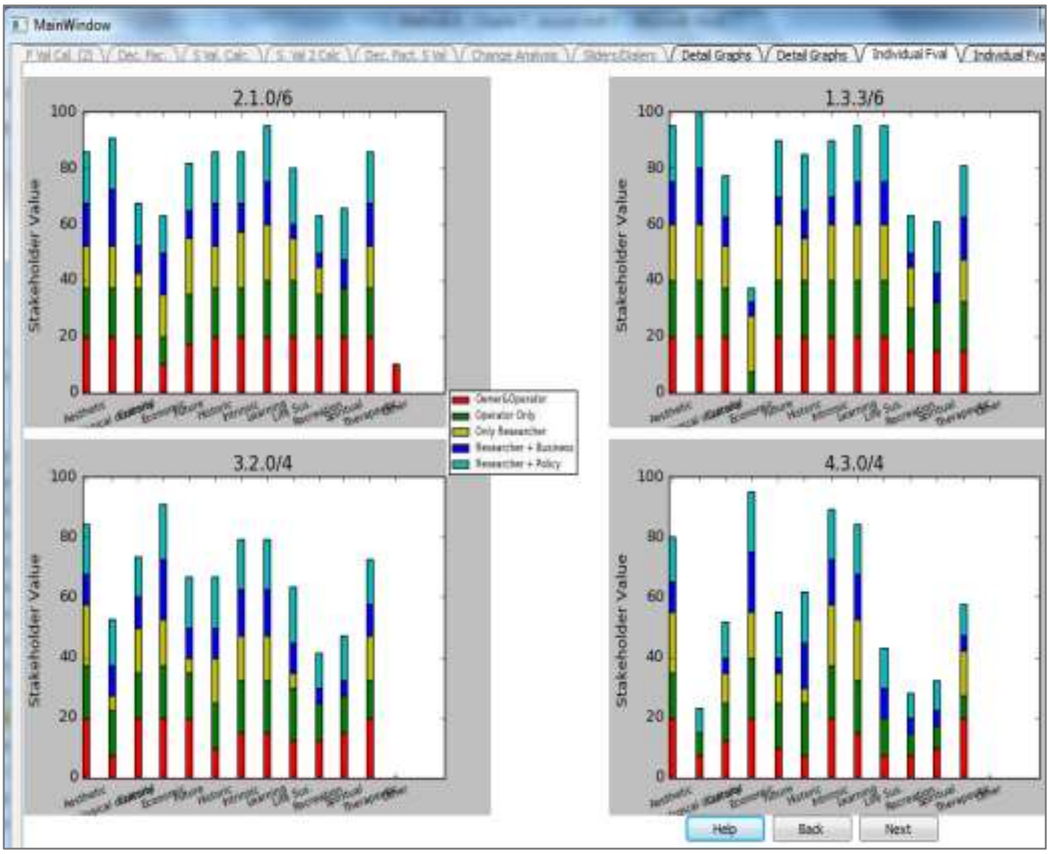


Figure 7.6: Chart showing stakeholder values for each value type for various land uses

iii. Individual Summary Charts

Individual summaries are generated that show, for each land use, the values assigned to the land use along with decision factors contributing to changing or retaining the land use. These charts are generated for all selected land uses in the change pathways/ scenarios, and provide a detailed comparison of values assigned under each value type by each farmer type and stakeholder type, facilitating comparison of values between land uses and between diverse stakeholders. The charts are intended to support interpretation of this detailed data and so support interpretation and discussion of the trade-offs between values. An illustrative chart is shown in Figure 7.6.

iv. Using Python as Programming Language

There are many programming languages currently used for geospatial modelling. Python is one such language and is used and recommended by ESRI (Wunderlich 2012, ESRI 2015), which is the GIS platform used in this research. ESRI is an international supplier of GIS software (Schutzberg 2001, ESRI 2002).

v. Using PyQt Designer to create graphical interfaces

For this study, a customised python tool was developed with its own Graphical User Interface (GUI) utilising customisable GUI Designer known as Qt Designer, PyQt module and Python. Qt Designer is a tool that allows designing and building of graphical user interfaces by designing widgets, dialogs etc through a drag and drop interface (Dalheimer 2002, Molkentin 2007). Qt Designer uses XML .ui files to store designs and does not generate any code itself. PyQt is a set of Python version 2 and version 3 bindings for Qt Desinger PyQt module and includes the uic Python module (pyuic; which is a UI compiler for Qt that comes with the PyQt package) that can load .ui files to create a user interface dynamically from Qt Desinger (Dalheimer 2002, Molkentin 2007). Like the uic utility it can also generate the Python code that will create the user interface. Therefore, Qt Desinger and PyQt makes it easy to add python code having a GUI to a project. As graphical user interfaces provide ease of use, higher productivity and better accessibility, this model applies a GUI that can help users to better understand and visualise the model for easy implementation.

vi. PyCharm for scripting using model libraries

After migrating the code to python, JetBrains PyCharm was utilised as an integrated development environment (IDE) for scripting purposes. PyCharm comprises a source code editor, build automation tools and a debugger. Python 2.7 was used as an interpreter in PyCharm. In this study, python libraries were also imported and utilised including PySide, xlrd (for excel rows handling), numpy, arcpy, pyqtgraph and matplotlib.

vii. Creating a stand-alone application based on ArcPy licence

For developing a stand-alone application, the Arcpy site-package was used to extract data from shape files (.shp). This package requires ArcGIS installed, although it does not require starting any ArcGIS applications. It is currently a stand-alone application and cannot be used within ArcMap, although in future it can be tested to embed it as a custom python script into a toolbox in ArcMap or ArcGIS Pro, and thus by invoking the

PyQt script the tool may be setup and run in ArcGIS (Scheirer 2011, McCune 2012, Tereshenkov 2015).

7.5 Conclusion

This chapter has detailed the methods employed in the research and described the conceptual framework and development of the SEEM model. The chapter has also described the methods employed for data collection for the model Maryland Farm. This study applies the concept of bounded rationality and utilises logistic regression model to capture the decisions of farmers made in regards to changing land uses, by highlighting the values held by farmers themselves, values of stakeholders other than farmers having interest in the farm and the economic returns from the farm. This chapter explains the model algorithm and its sub-functions; economic return, vaues of farmers and values of other stakeholders in detail. For this research, scenarios were generated as explained under section 6.3 and land management activities were assigned to land uses on Maryland farm described as storylines of each scenario in the questionnaire attached as Appendix E. This chapter describes the value typology used in the survey to capture values of participants and the decision factors associated with changing or retaining the land use. It also explains the normalisation procedure carried out to normalise the acquired values and the AHP by which weights were assigned to the sub-functions. The chapter also explains the development of semi-structured questionnaire used to capture values in the survey from participants, and how the participants were then divided into five categories. The data from these five categories was then manually entered into the SEEM model, where, the scripting was carried out to calculate and represent farmer's decisions. The chapter also explains the development of SEEM model and its data inputs, interfaces and data outputs in detail.

The methodology developed for this model extends on previous research in the following ways:

- The model communicates values in a distinctive way, describing and communicating values between diverse stakeholders and across different land uses.
- The model generates flexible and non-optimised results, suitable for decision-making settings, to support dialogue among engaged stakeholders and the negotiation of acceptable decisions. This makes the SEEM different from other existing models (ARIES, ENVISION, Daloğlu et al's model).
- The model and its implementation are not hardwired, but provide considerable flexibility.
- The model utilises and supports stakeholder engagement in capturing non-monetary ecosystem values, unlike many modelling tools (e.g. ARIES, MIMES).
- The model has the capacity to integrate current or proposed policy incentives as a contributing component to F econ. It also has the capacity to compute costs and returns of changing land use in an effective way.

The SEEM employs a distinctive strategy when compared to other existing approaches, as it involves locational context, communicates values in a distinctive manner, generates flexible, non-optimised results, promotes stakeholder's engagement, utilises non-monetary ecosystem values in generating results, has the capacity to integrate policy incentives, is capable of computing costs and returns involved in land use change decisions, and can facilitate decision making in the context of complex trade-offs.

The methodology employed in this research contributes towards development of a model that captures decisions on changing land uses by taking into account values of farmers and other stakeholders in a geospatial manner, and takes into account the economic aspect of the farms, thus, integrating the disciplines of spatial sciences, ecology and economics. The model developed has the capacity to be used in a decision making setting in order to support dialogue, facilitate spatial reasoning and promote consensus among diverse stakeholders in order to approach towards practical solutions.

8 Data Assumptions and Model Population– Maryland Farm

8.1 Introduction

This chapter documents assumptions made when constructing data for Maryland Farm and describes the process used to populate questionnaire survey data and bring that data into the SEEM model.

8.2 Data Assumptions

This section documents assumptions made when constructing data for Maryland Farm and notes consequential limitations.

Maryland Farm comprises six unique land uses, selected using the future change pathways explained in Chapter 6 and shown in Table 3 and Table 6 of that chapter. The selection of land uses was not necessarily representative of a majority of farms in the region, but was intended to provide a sufficient variety of options and possible pathways to ensure that a reasonable number of land use permutations could be described and assessed in the survey of stakeholders.

A variety of stakeholders were invited to complete the questionnaire, with participants selected so as to ensure a broadly representative range of interests and likely viewpoints. The number of stakeholders interviewed was not sufficient to allow collected data to be treated as a reliable indication of population views: the intention of the survey was not to measure population views, but only to demonstrate how such data can be collected and then how such data can be brought into the modelling process, and the results presented and visualised.

Additionally, the following assumptions were made:

- The values (aesthetic, cultural etc.) obtained for any parcel with a particular land use and under a particular management practice existing on Maryland Farm are going to be the same for all such land use types under the same management practice on Maryland farm. Therefore, a single set of values was collected for a unique land use type and management practice on Maryland Farm.
- The land capability of a unique land use under a particular land management practice does not affect the values (aesthetic, cultural etc…) attached to that land use and management practice, and thus will remain the same regardless of land capability.

8.3 Populating the model

The data obtained from the survey was first assembled in an excel file. This file held data from all stakeholders. The stakeholders were categorised and grouped into categories, as explained in section 7.3.2. For a single category, the values were averaged for a number of participants. The average values for each category was then manually entered (populated) into the SEEM model under each land use and each stakeholder category. The SEEM model normalises the values and thus calculates outputs. An overarching view of the data inputs, user interfaces and the resultant outputs is presented in section 7.4 and shown in Figure 7.2. The SEEM model has been implemented as a software application that facilitates data input, computes model parameters in run-time, and presents data in numeric and graphical format. This section describes the model interfaces and explains how the model was populated.

The model requires the following input:

8.3.1 Input Datasets

Input Datasets is the first interface in the model and requires input in the form of Microsoft Excel files and a shape file. Figure 8.1 illustrates the interface used for data input.

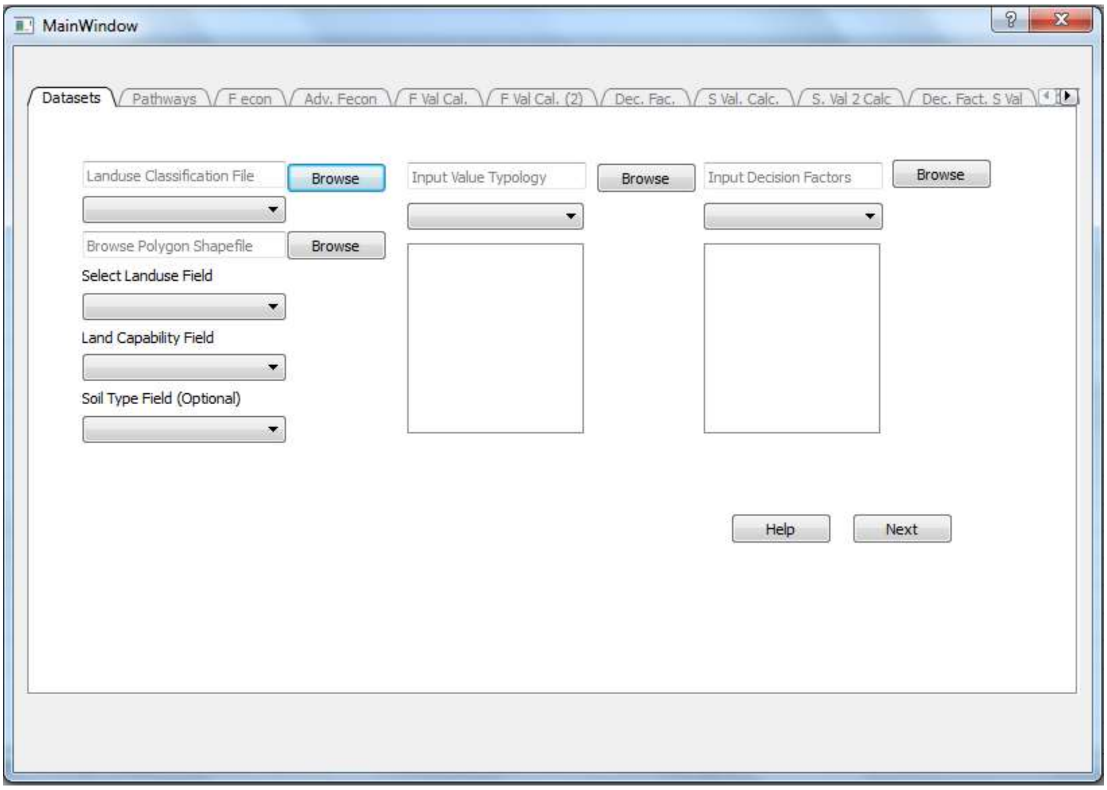


Figure 8.1: Model interface showing dataset inputs: land use classification file, shape file with attributes of land use, land capability and soil type, value typology and decision factors

8.3.2 Land use Classification File

The first input required is a land use classification; this can be based on any preferred classification system. The file should be in Excel format and where a single column can be selected that comprises a list of land use classifications, either descriptive text or a classification code[21]. That list can comprise all land use types or only those that the user needs to analyse for a particular model.

[21] It improves the presentation of graphs if codes rather than descriptive statements are used.

8.3.3 Polygon Shape File

The second input required is a polygon shape file. The user browses to, and loads, a polygon shape file describing the farm or landscape by parcel boundaries, and with attributes that describe land use, land capability, and any other attributes, such as soil type. The attributes (land features) are selected as the shape file is browsed.

8.3.4 Value Typology

The third input required is a value typology. This is loaded from an Excel file. Any value typology appropriate for the study area and site-specific conditions can be used[22].

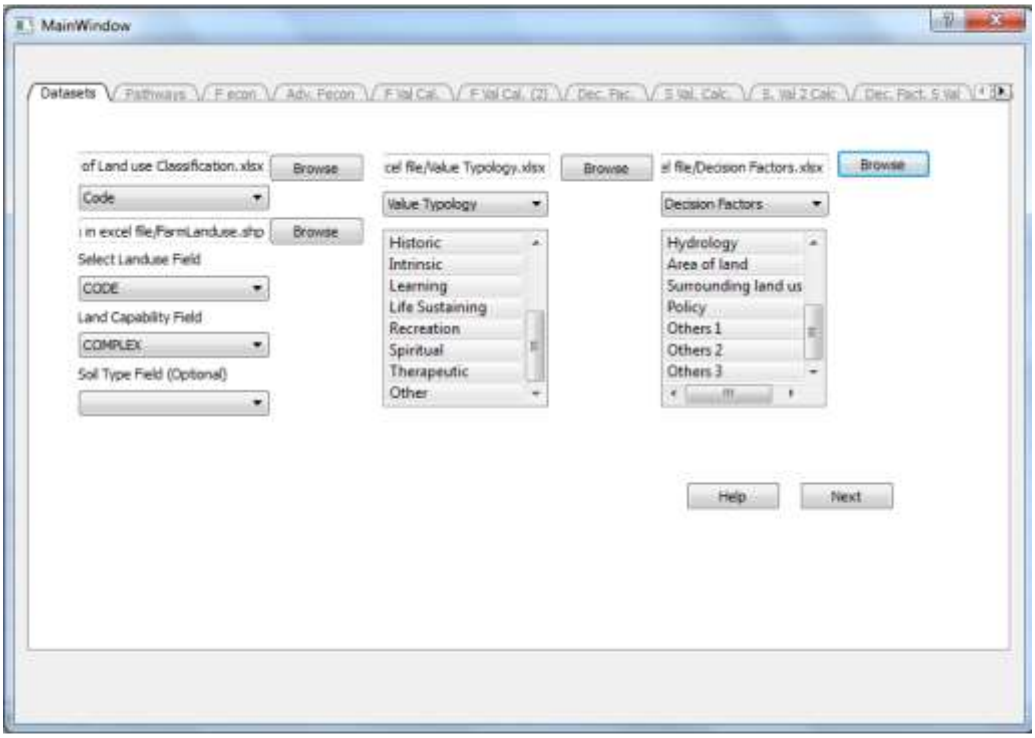


Figure 8.2: Dataset input for Maryland Farm using ALUM land use classification, Maryland Farm shape file, land capability, value typology and decision factors selected

8.3.5 Decision Factors

The fourth input required is the list of decision factors that describe whether a land use is influenced by the factor to change it to any other land use or retain it in its current land use state. Again, this is loaded from an Excel column[23]. The list of factors can be selected or modified by the user. In the case of Maryland farm, the decision factors included in the list were: land capability, dominant soil, topographic factors (slope, elevation and aspect), drainage lines, size of the land use parcel, proximity to similar land use parcels, proximity to transport routes and irrigation infrastructure. The information on decision factors for each one of the Maryland farm land use parcels was gathered through the questionnaire, where the participants – while observing these factors in the context of each parcel and land use – indicated whether the factor would

[22] In the current implementation, a limit of 13 value types can be entered and visualised

[23] In the current implementation, a limit of 12 decision factors can be entered.

influence them to change the land use or retain the land use in its current state or would induce no effect.

8.4 Populating the model with the Maryland Farm input datasets

Figure 8.2 shows the "Datasets" tab of the software, populated with data for the Maryland Farm case study.

8.4.1 Change Pathways

The second interface in the model deals with Change Pathways. In this interface, the user selects the land uses to be analysed. In the first part of the interface the user selects the number of changes to be modelled. Land uses are then selected in the '<u>From Land Use</u>' category (signifying a current land use), and then in the '<u>To Land Use</u>' category, with these two representing one Change Pathway. A number of scenarios can thus be selected, with each single row indicating a single scenario in which a land use is tested for change to other land uses. Figure 8.3 shows the blank interface; Figure 8.4 shows the pathways selected for Maryland Farm as per the 'Future change pathways/ scenarios' described in Chapter 6.

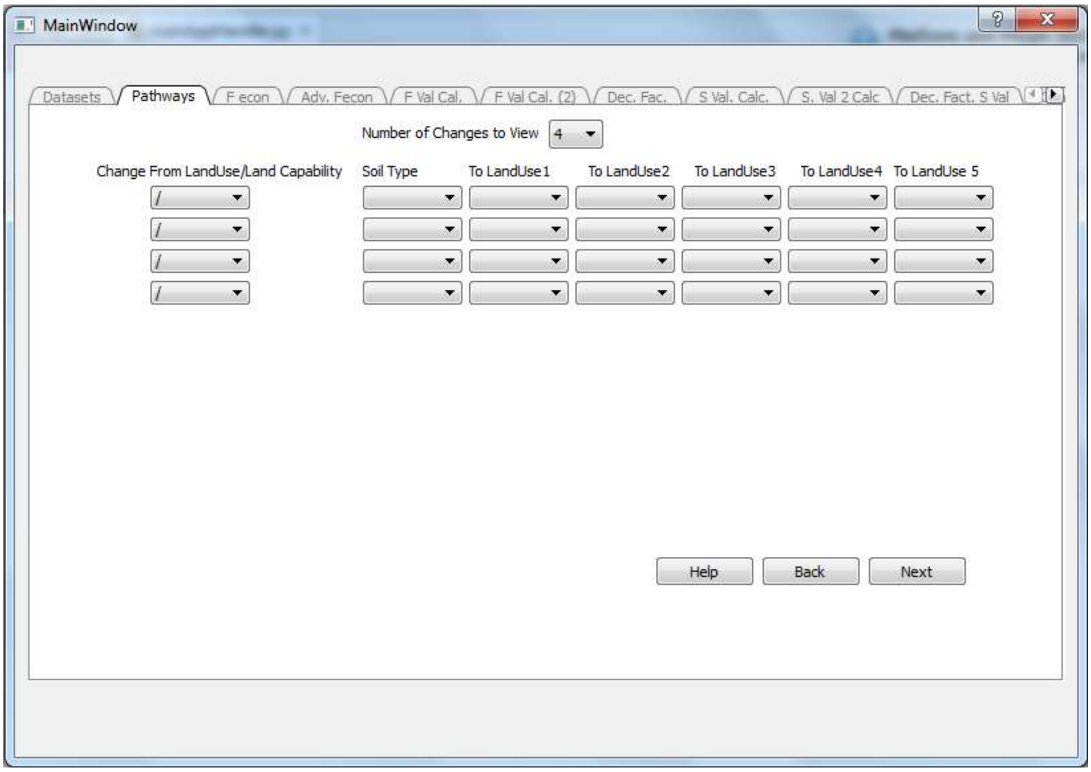


Figure 8.3: Model interface representing change pathways for four scenarios, not yet listed and so showing as blank

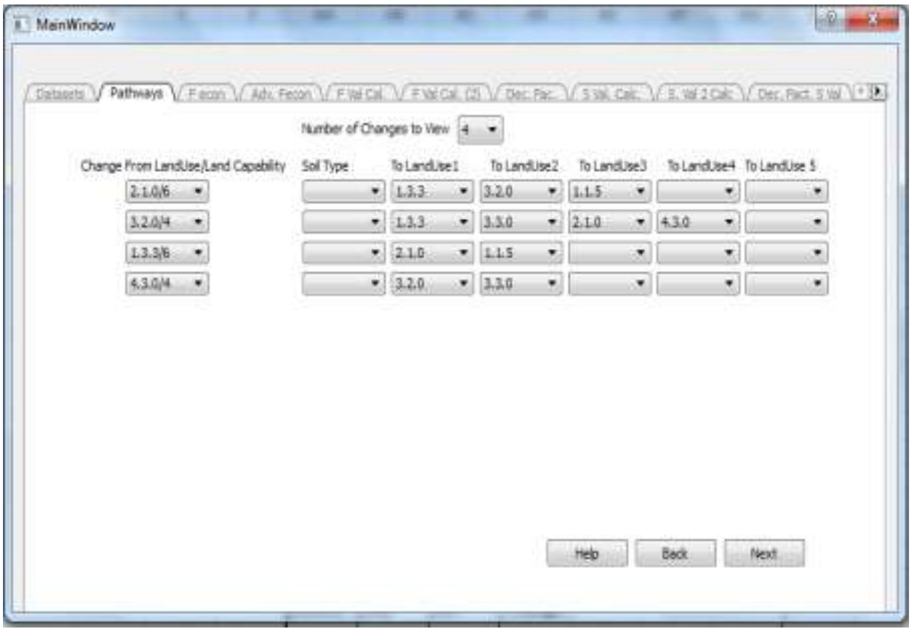


Figure 8.4: Model interface populated with future change pathways derived for Maryland Farm

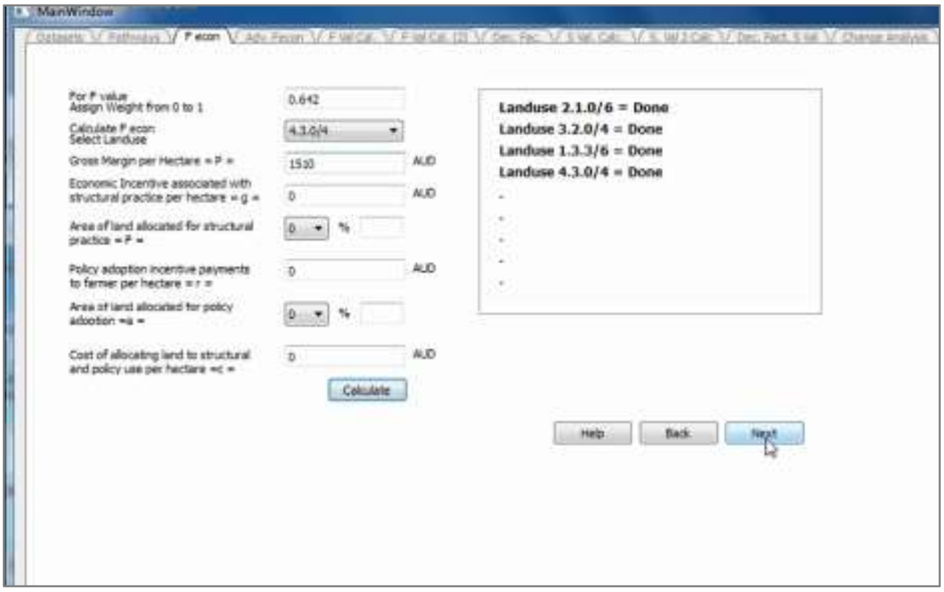


Figure 8.5: Model Interface F $_{econ}$ representing populated data for Maryland Farm for land uses under 'From Land use' category

8.4.2 in the next interface Adv. F $_{econ}$ for land uses selected under the 'To Land uses' category F $_{econ}$

The next interface deals with the sub-function F $_{econ}$. This interface is used to input information on gross margin values for each land use under a management practice, information on policy incentives, structural incentives, and costs associated with implementation of policies or structural practices. The user completes the first interface for F $_{econ}$ for land uses selected under the 'From Land use' category in 'Change pathways', and then completes in 'Change pathways'. Figures 8.5 and 8.6 show interfaces populated for Maryland Farm. In this scenario, gross margin values for specific land uses under management practices were input using the data shown in Table 8.1. In addition, a weight is assigned to the sub-function $F_{econ\,(k,\,l)}$ for the Maryland Farm, which in this case was derived through AHP.

For Maryland Farm, the gross margin values are taken as the average of minimum and maximum obtainable from dryland cropping and irrigated cropping land uses, whereas for other land uses it depends on the management practice carried out on the land use given in Table 8.1.

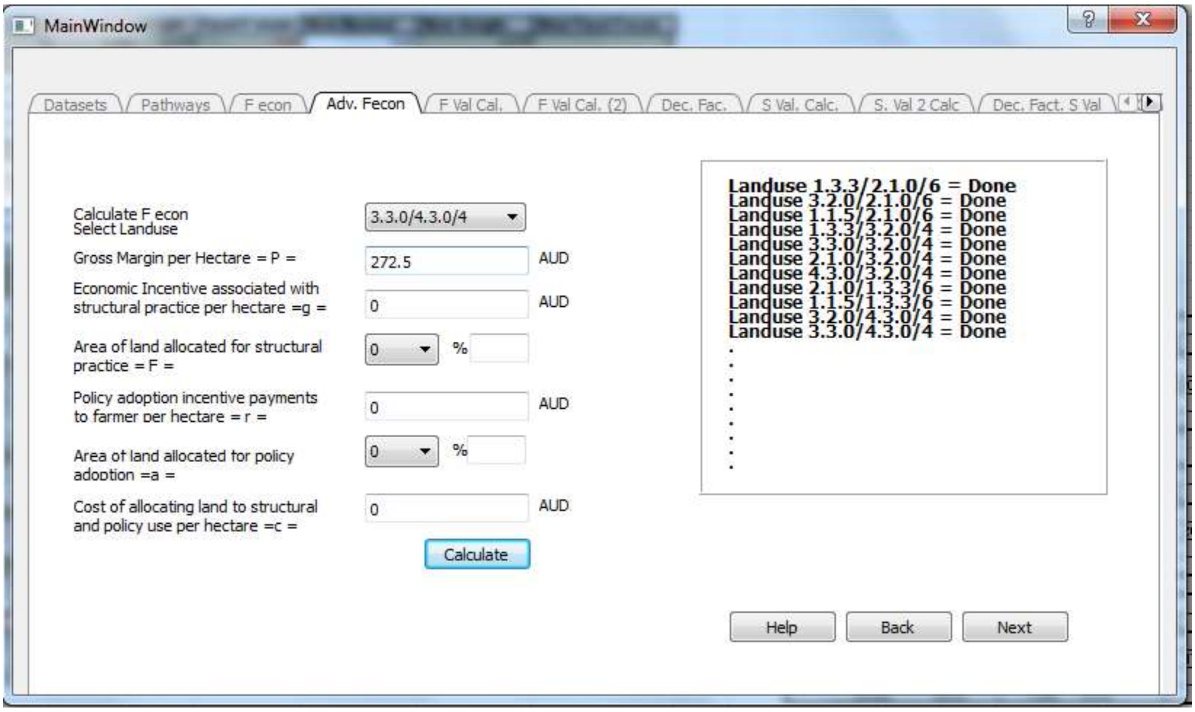


Figure 8.6: Model Interface Adv. F_{econ} representing populated data for Maryland Farm for land uses under 'To Land use' category

Table 8.1: Gross margin values calculated for land uses under specific management practice explained in Chapter 7 - 'Sub-Function Economic return' are obtained from DPIPWE (2015).

S#	Land use	Land Capability	Management Practice	Gross margins obtained from DPIPWE website	Applied Gross margin per hectare
1	Grazing natural vegetation (2.1.0)	6	Fine wool production from grazing sheep	420 $ per hectare	420
2	Residual native cover (1.3.3)	6	No prime use of native vegetation cover	0	0
3	Grazing modified pastures (3.2.0)	4	Sheep grazing for breeding and trading first cross ewes	335 $ per hectare	335
4	Cropping (3.3.0)	4	Dryland cropping	Min Barley – 255$ Max Wheat – 290 $	272.5

S#	Land use	Land Capability	Management Practice	Gross margins obtained from DPIPWE website	Applied Gross margin per hectare
				Average – 272.5 $ per hectare	
5	Habitat species management area (1.1.5)	5	Protected area	0	0
6	Irrigated Cropping (4.3.0)	4	Irrigated Cropping	Min Barley – 710$ Max Poppies – 2310 $ Average – 1510 $ per hectare	1510

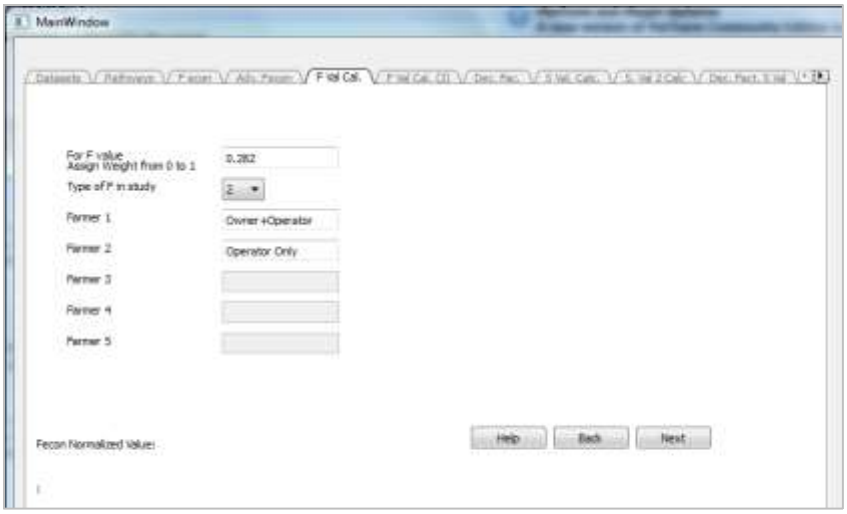


Figure 8.7: Model interface for input of Farmers' values and requiring input of weight assigned to F $_{value}$ and each type of farmer in the study

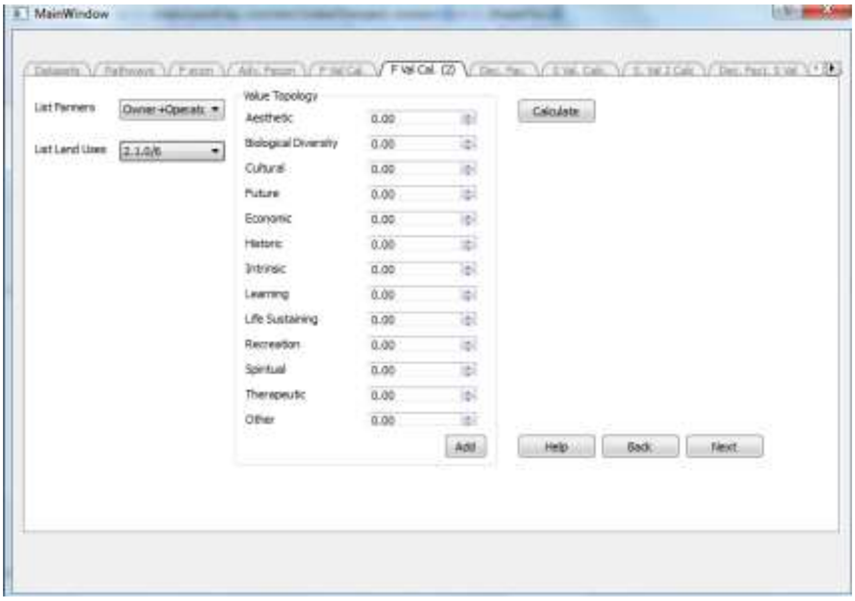


Figure 8.8: Model interface for input of Farmers' values and requiring input on values from each type of farmer for each land use type

8.4.3 F_{value}

The interface for F_{value} deals with farmers' values. For Maryland Farm, the data collected from the questionnaire is used as input for F_{value} calculations. Data were collected using a Likert scale, which is then converted into values ranging from 0 to 20 with an interval of 5. These numeric values are entered into the model. Figures 8.7 and 8.8 show the interfaces used for F_{value}: they require, as input from the user, the types of Farmer, the values the Farmers assign to each value category for land uses, and the weight assigned to sub-function $F_{value\,(k,\,l)}$, which was in this case derived through AHP.

8.4.4 F Decision Factors

The interface for farmers' decision factors deals with decision factors and their influence on the decisions made by farmers. The list in this interface displays the decision factors, as selected by the user in the first data input interface. The factors identified and used for the Maryland Farm case study include land capability, elevation, slope, aspect, dominant soil, drainage lines, area of land, surrounding land uses, and influencing policies. The questionnaire survey conducted with farmers and other stakeholders captured information about whether participants regarded any specific listed decision factor as influential in their decisions about whether a particular land use should change, or not change.

In the stakeholder survey, this data was obtained as participant responses (comprising only farmers for 'F Decision Factors' interface) on a Likert scale, which was then converted into values ranging from -10 to + 10, where the higher positive values indicate that the factor would highly influence a change in the land; the higher negative values indicate that the factor would highly influence *no* change in the land use, whereas a value close to zero indicates a neutral response – the factor would have little influence. Figure 8.9 shows the interface that captures these decision factors for farmers.

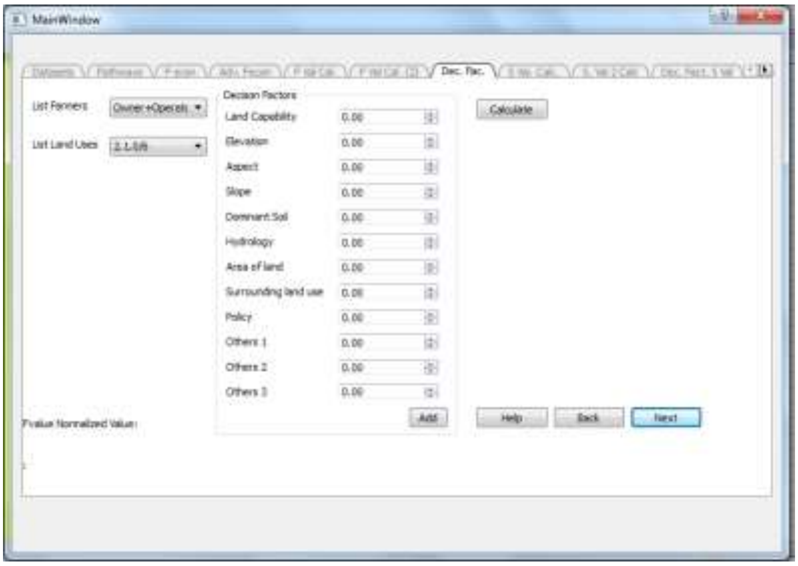


Figure 8.9: Model interface for farmers' decision factors depicting farmers' perceptions in regards to the factors influencing land uses: to change or not to change

8.4.5 S value

The interface for S $_{value}$ deals with values of other stakeholders (stakeholders other than farmers) and how they value the land uses under a selected value typology. For the Maryland Farm, data collected through the questionnaire was used as an input for S $_{value}$ calculations. The data was obtained using a Likert scale, which was then converted into values ranging from 0 to 20 with an interval of 5. These values were entered into the model using the interfaces shown in Figure 8.11. Figure 8.12 and Figure 8.13 are the interfaces that deal with S $_{value}$ and that require input from the user for each type of stakeholders, the values that each type assign to each value category for land uses, and the weight assigned to sub-function S $_{value\ (k,\ l)}$ i.e. derived through AHP in this case.

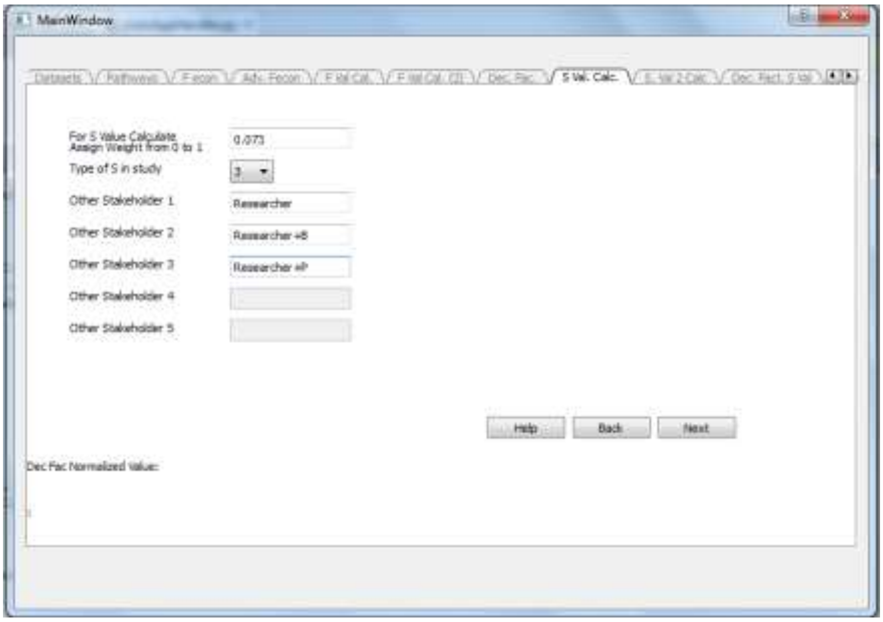


Figure 8.10: Model interface on other stakeholders' values requiring input on weights assigned to S $_{value}$ and types of other stakeholders in the study

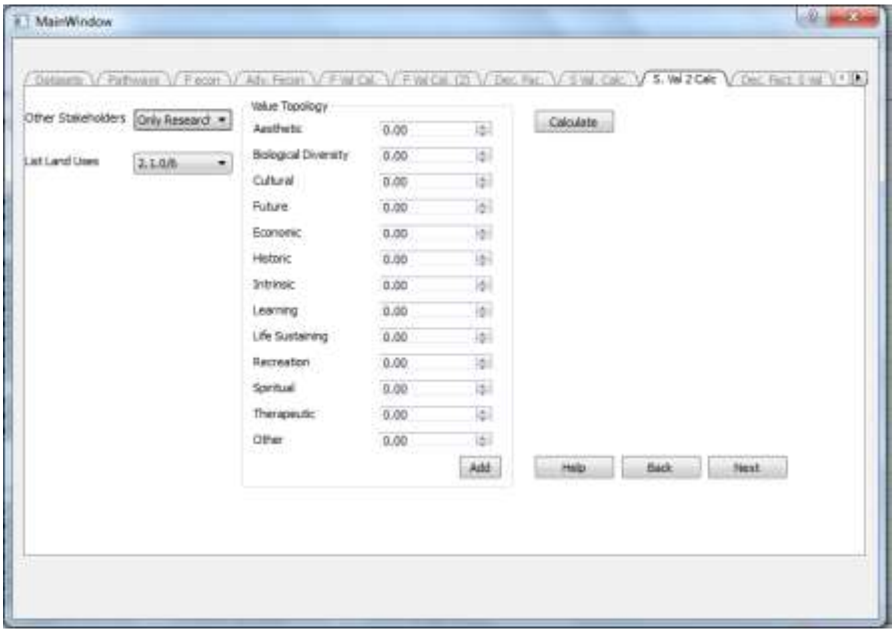


Figure 8.11: Model interface on other stakeholders' values requiring input on values of each type of other stakeholder for each land use type

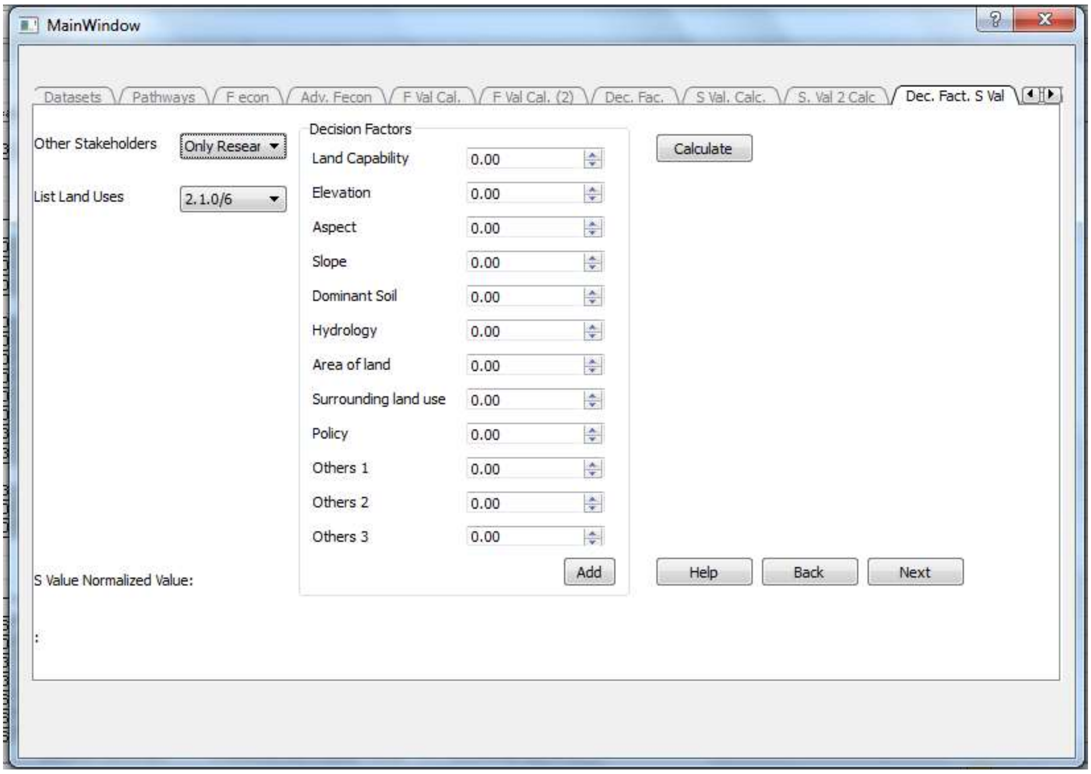


Figure 8.12: Model interface on other stakeholders' decision factors depicting other stakeholders' perceptions in regards to the factors influencing land uses to change or not to change.

8.4.6 S Decision Factors

The interface for other stakeholders' decision factors deals with decision factors and their influence on decisions of stakeholders other than farmers. The list in this interface displays the decision factors as selected by the user in the first interface on data inputs. The factors identified and used for the Maryland Farm case study include land capability, elevation, slope, aspect, dominant soil, drainage lines, area of land, surrounding land uses, and influencing policies.

The questionnaire survey conducted with farmers and other stakeholders captured information about whether participants regard any specific listed decision factor as influential in their decisions regarding whether a particular land use should change, or not change. In the stakeholder survey, this data was obtained as participant responses (comprising only stakeholders other than farmers for the 'S Decision Factors' interface) on a Likert scale, which was then converted into values ranging from -10 to + 10, where the higher positive values indicate that the factor would highly influence a change in the land use; the higher negative values indicate that the factor would highly influence *no* change in the land use, whereas a value close to zero indicates a neutral response – the factor would have little influence. Figure 8.12 shows the interface that captures these decision factors for stakeholders other than farmers.

8.4.7 Change Analysis

Finally, the model displays the change pathways selected in the second interface and highlights the land use in each scenario based on respective probabilities (each scenario row). This interface is shown in Figure 8.13.

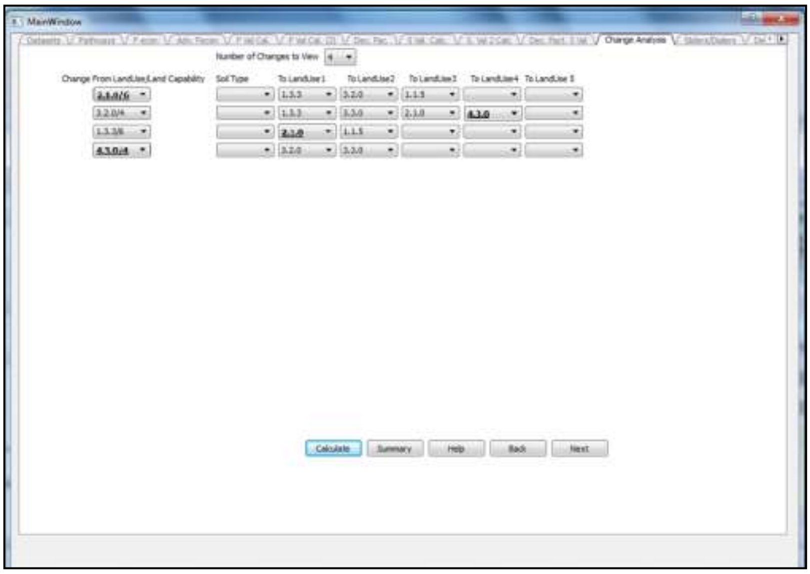


Figure 8.13: Model interface showing change pathways and the resultant analysis by calculations of probability for each land use in a scenario

8.5 Summary

This chapter has documented the assumptions made when constructing data for Maryland Farm and has described the process used to populate survey data and to bring that data into the SEEM model. This chapter has described each step employed in populating data into the SEEM model interfaces, explaining particularly any manual input required to populate the model. This chapter is intended to assist the reader to understand data population and subsequently the charts developed by SEEM model and explained in Chapter 9.

9 Results and Discussion – Maryland Farm

9.1 Introduction

This chapter describes and discusses the outputs and results obtained from calibration of the SEEM model using the survey data obtained for Maryland Farm. The SEEM model presents results in the form of summary charts which can be divided into two types: individual summary charts and detailed summary charts.

9.1.1 Individual summary charts

Individual summary charts are obtained for each type of land use and are entirely based on values assigned by farmers and other stakeholders to various land uses together with values obtained for the decision factors. Two charts are produced for each land use on Maryland Farm: one shows values obtained for each land use, the other shows decision factors influencing each existing land use. The values on decision factors are obtained for six land uses under six scenarios in the questionnaire. Only the values on decision factors for existing land uses on Maryland Farm (the four land uses that were surveyed in the Questionnaire) are analysed. The remaining two land uses (also surveyed in questionnaire) are represented in 'future change pathways' given as a table in Chapter 6. Thus, the values on decision factors for perceived land uses are not included in the model analysis. The charts are stacked bar charts representing the values of each group of participants, using stacks along each value type (e.g. aesthetic, cultural etc.) in the first chart, and values of each group of participants along each decision factor (e.g. land capability, elevation etc.) in the second chart. This assists in visualising the comparative values within a land use from diverse participant groups. An illustrative chart is shown in Figure 9.1.

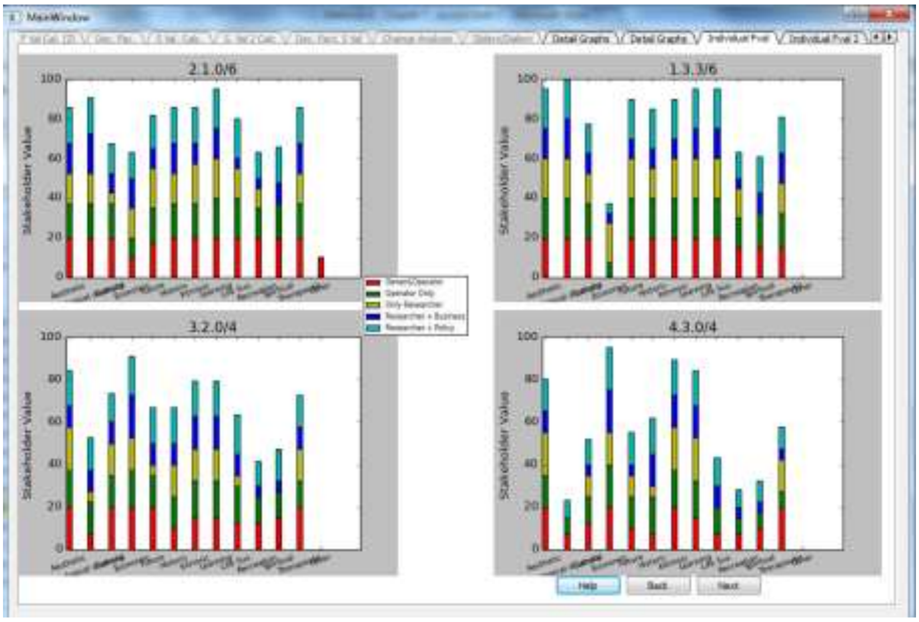


Figure 9.1: An individual summary chart showing stakeholder values for each value type for various land uses

9.1.2 Detailed summary charts

Detailed summary charts are obtained for each scenario calculated, and represent each land use and scenario and the values calculated for sub-functions $F_{econ\ (k,\ l)}$, $F_{value\ (k,\ l)}$, $S_{value\ (k,\ l)}$ and the resultant $F_{decide\ (k,\ l)}$. A single chart is produced for each scenario, showing the calculated values of $F_{econ\ (k,\ l)}$, $F_{value\ (k,\ l)}$, $S_{value\ (k,\ l)}$ and $F_{decide\ (k,\ l)}$ using vertical bar charts for all land uses within each scenario. In this way, a comparative view of sub-functions affecting the resultant $F_{decide\ (k,\ l)}$ in each land use can be observed, together with a comparative view of the sub-functions in each land use within the scenario. An illustrative chart is shown in Figure 9.2.

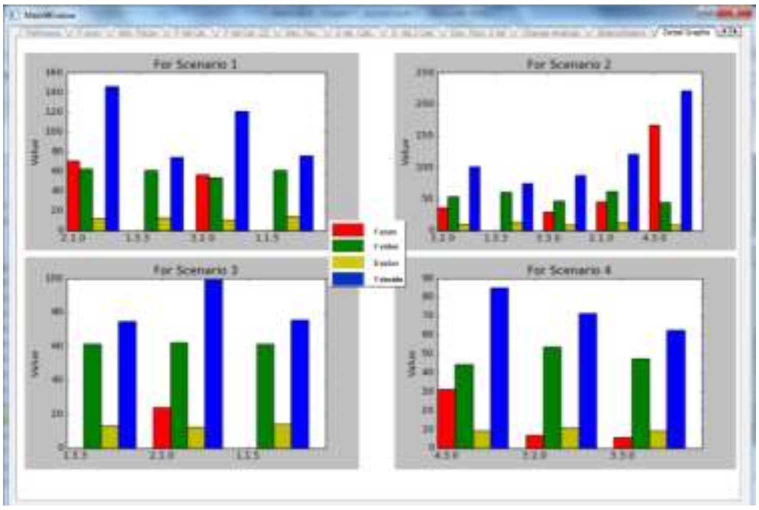
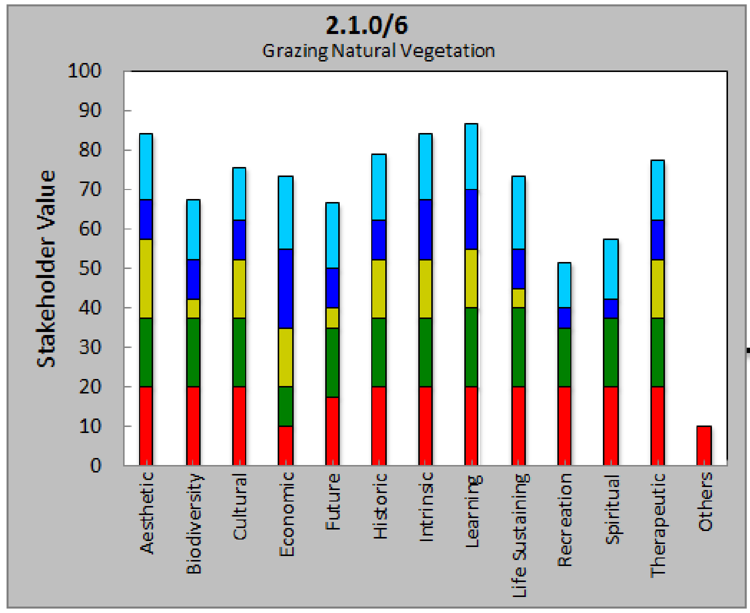


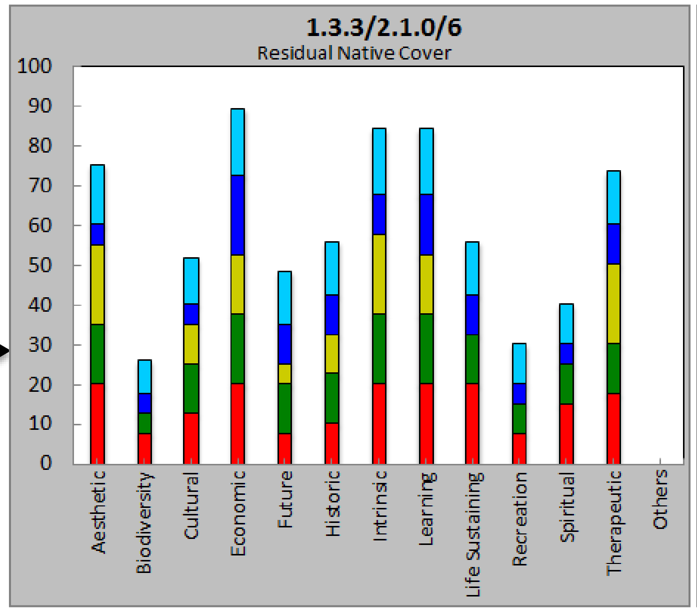
Figure 9.2: A detailed summary chart showing values of F_{econ}, F_{value}, S_{value} and F_{decide} for land uses in each scenario

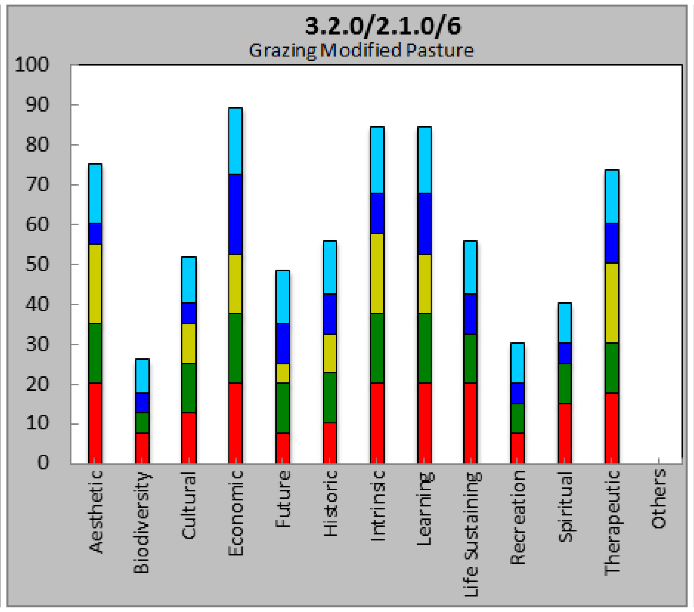
9.2 Results

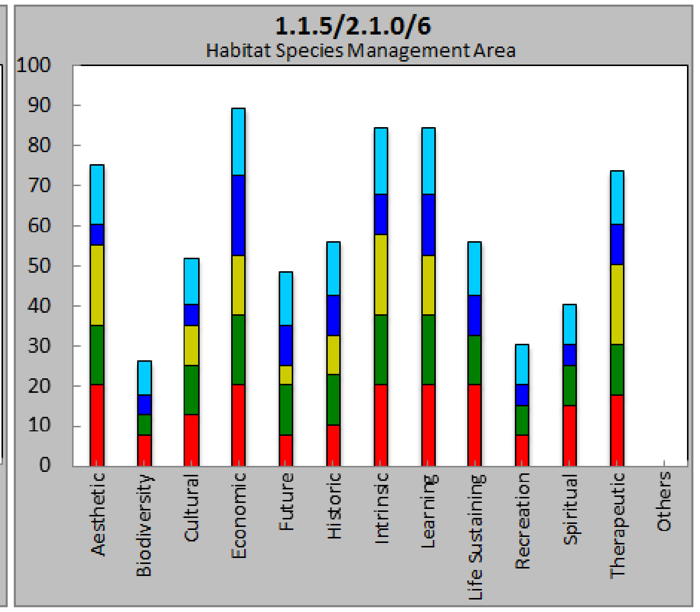
This section describes the results obtained from the SEEM model for Maryland Farm. The results are presented in the form of scenarios. For Maryland Farm, we have four scenarios or future pathways. The storylines presented to the participants, and which describe each scenario in detail, are included in the questionnaire. This is attached as Appendix E. The land use change pathways selected for each scenario are described in detail in section 6.2 ‘Creation of land use change pathways’. Each scenario is described in detail, and the charts produced are presented and discussed. These charts communicate the comparative results of all the land uses for the three sub-functions ($F_{econ\ (k,\ l)}$, $F_{value\ (k,\ l)}$, $S_{value\ (k,\ l)}$) and the resultant function $F_{decide\ (k,\ l)}$. The results obtained from the SEEM model are organised in the order presented in Scenario 1, 2, 3 and 4.

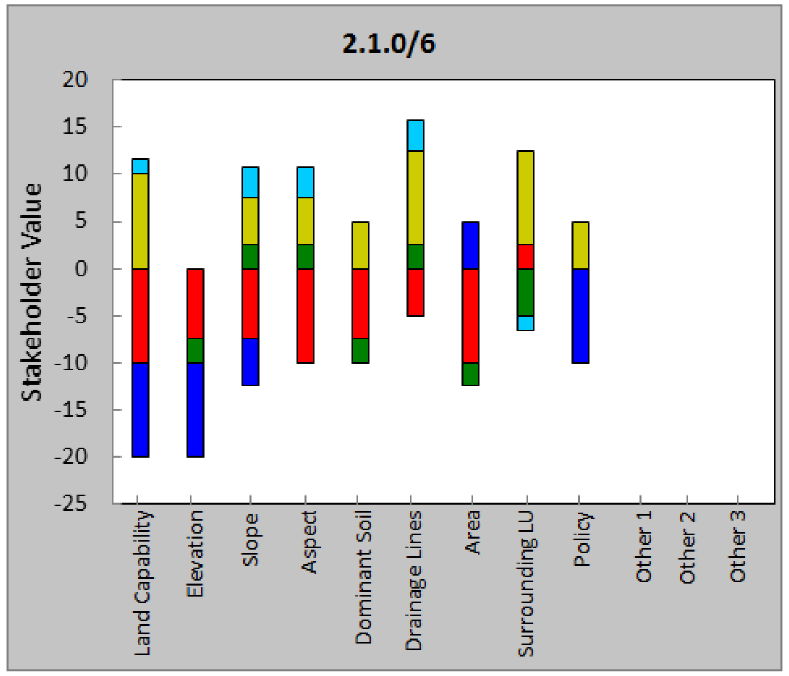
Scenario 1:

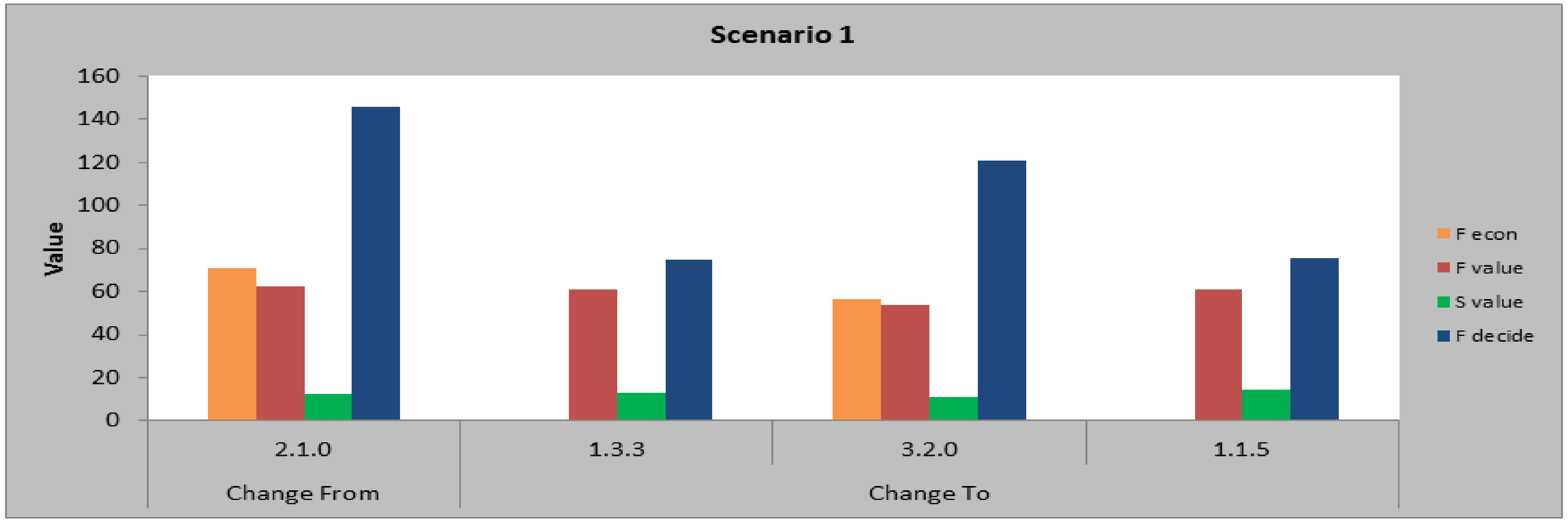










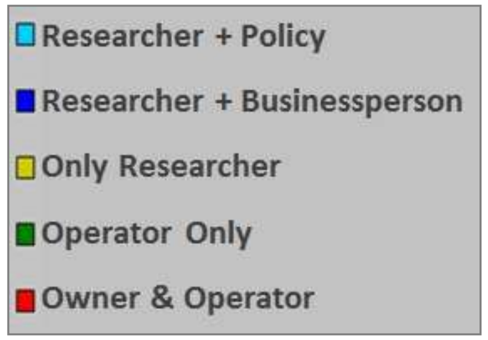


Description of Decision Factors Chart:

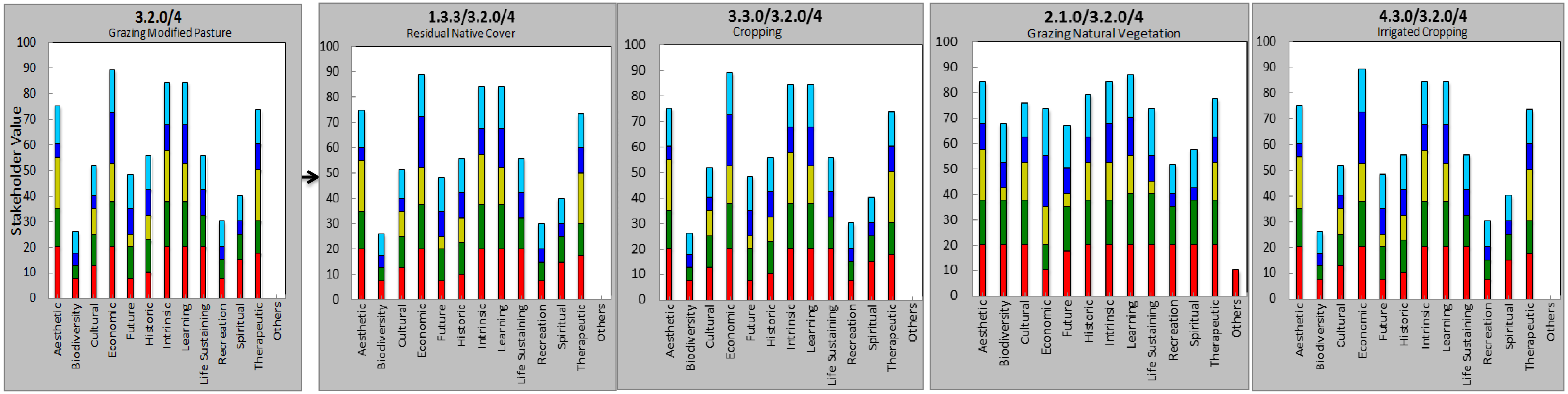
The values above zero indicate support for changing the land use to another land use, whereas values below zero indicate support for retaining the land use in its existing state. Zero indicates no preference for changing or not changing the land use. The factors in the chart include: land capability, elevation, slope, aspect, dominant soil, drainage lines, area of land, surrounding land use, and policy.

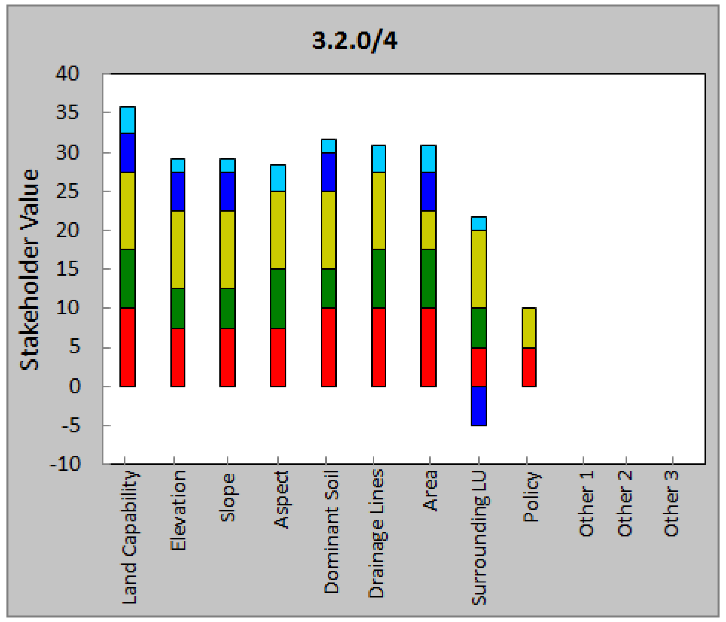
Contextual Comments:

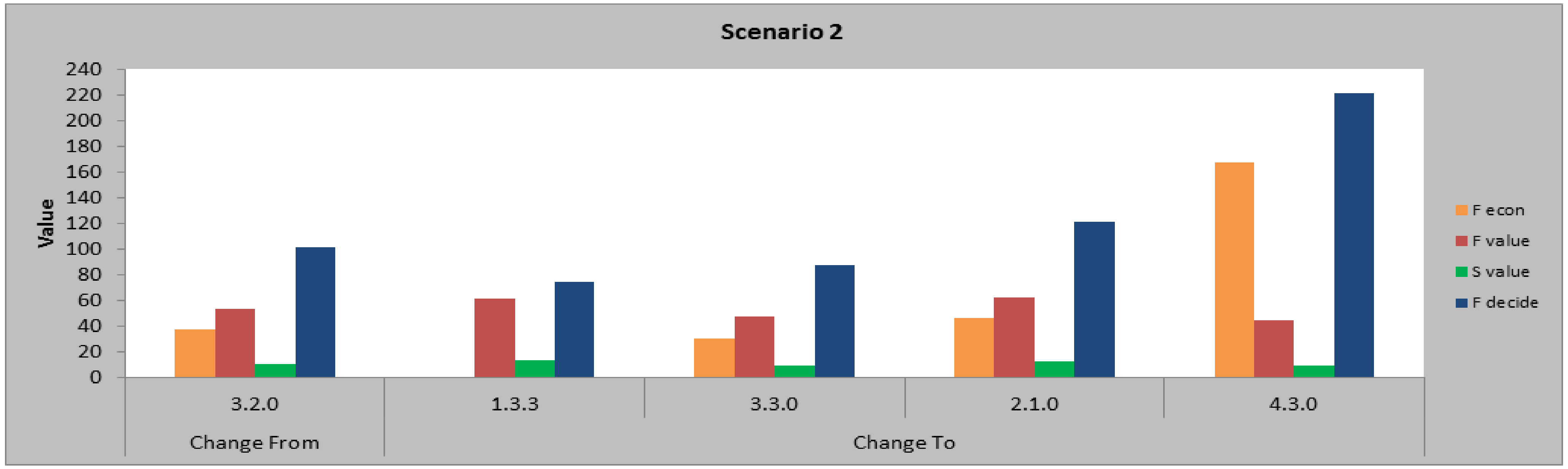
In the survey conducted, participants indicated that the area of this land use was of useful size, i.e. a large intact block of natural bushland, presumably managed for fine wool production for several decades, and that the land still retains important grazing value and can sustain its cultural and environmental values. Some of the participants indicated that the land capability being Level 6 is not suited for intensive use, and the elevation being high makes it cold, aspect being south/ southwest slope makes it less productive and soils usually are fragile on such slopes, and so the land is not suited for irrigation. Where participants commented on wanting to change it, they also wanted to be sure that evidence existed in support of change, and would prefer to lock the land into a protected area in perpetuity. Some wanted it to change to Habitat Species Management Area, others wanted to change to stop grazing for longer periods or graze strategically or a long rotation.



Scenario 2:







Description of Decision Factors Chart:

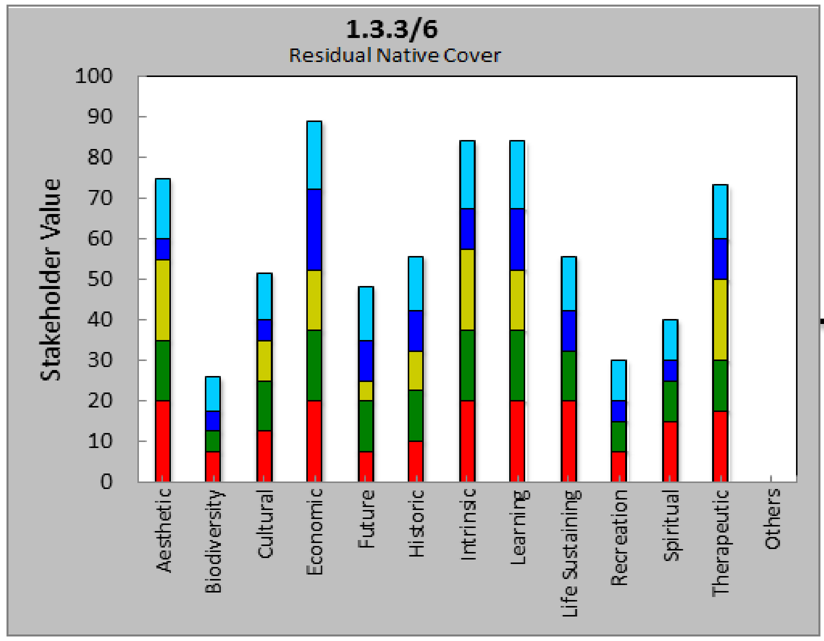
The values above zero indicate support for changing the land use to another land use, whereas values below zero indicate support for retaining the land use in its existing state. Zero indicates no preference for changing or not changing the land use. The factors in the chart include; land capability, elevation, slope, aspect, dominant soil, drainage lines, area of land, surrounding land use, and policy.

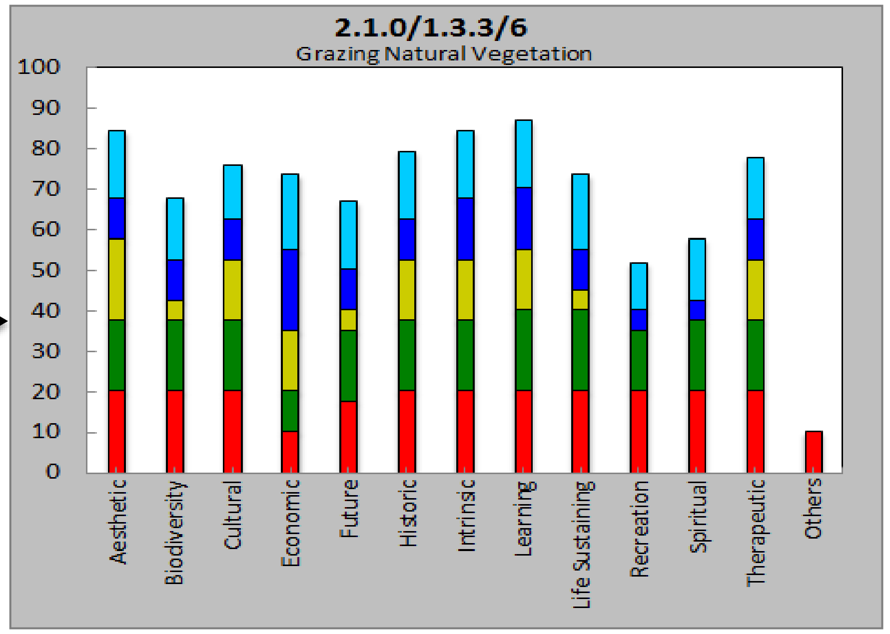
Contextual Comments:

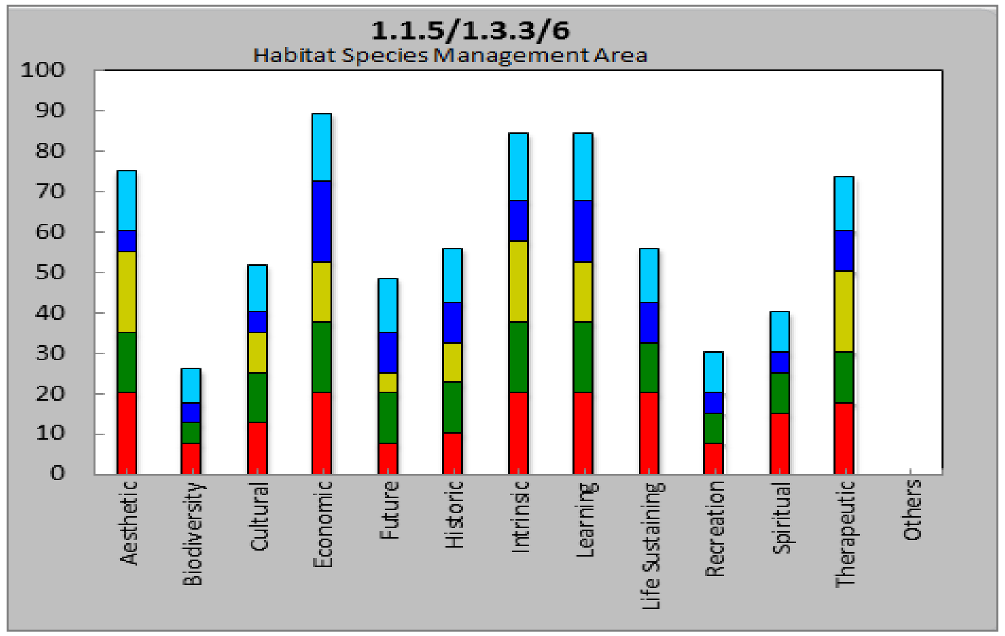
In the survey conducted, participants indicated that the land, being highly modified and versatile with minimal biodiversity and ecosystem services, would therefore be suitable for evidence-based change to other land use. Regarding decision factors, some participants indicated that the factors would influence a decision not to change to any form of cropping. Aspect was regarded as warm and good for sheep, while the drainage lines for some provided an opportunity for dams for irrigation. One participant indicated that a farm development financial assistance package would assist a decision to change land use.

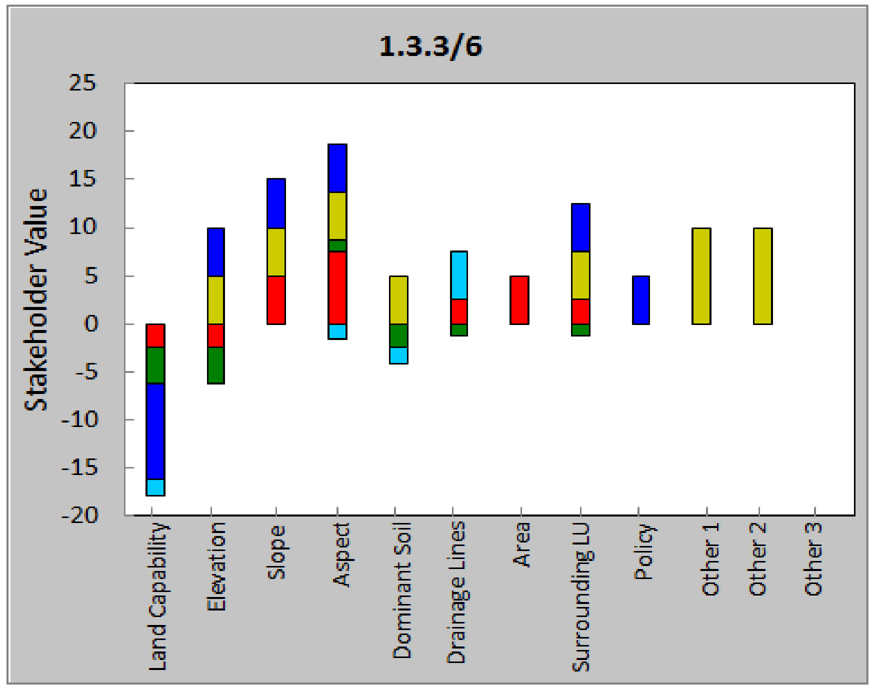
- Researcher + Policy
- Researcher + Businessperson
- Only Researcher
- Operator Only
- Owner & Operator

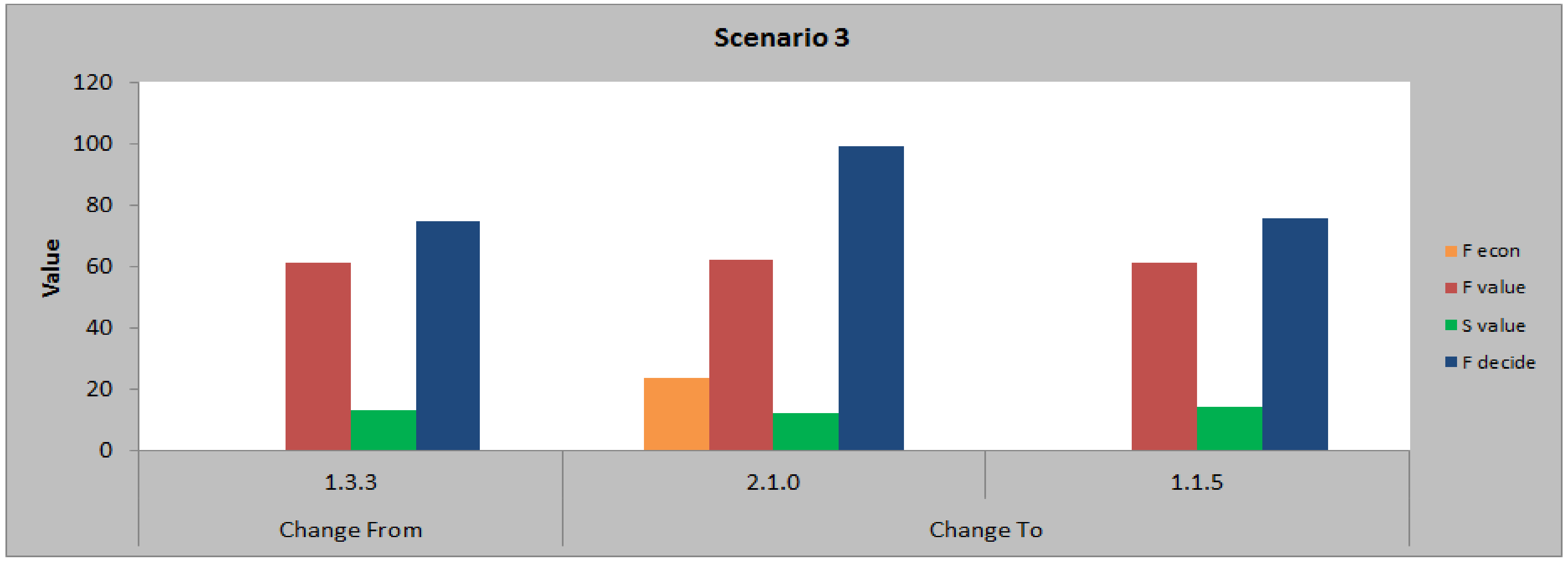
Scenario 3:

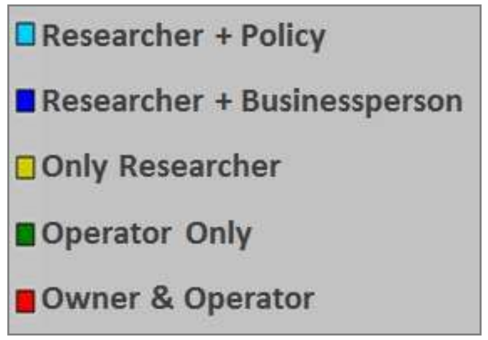












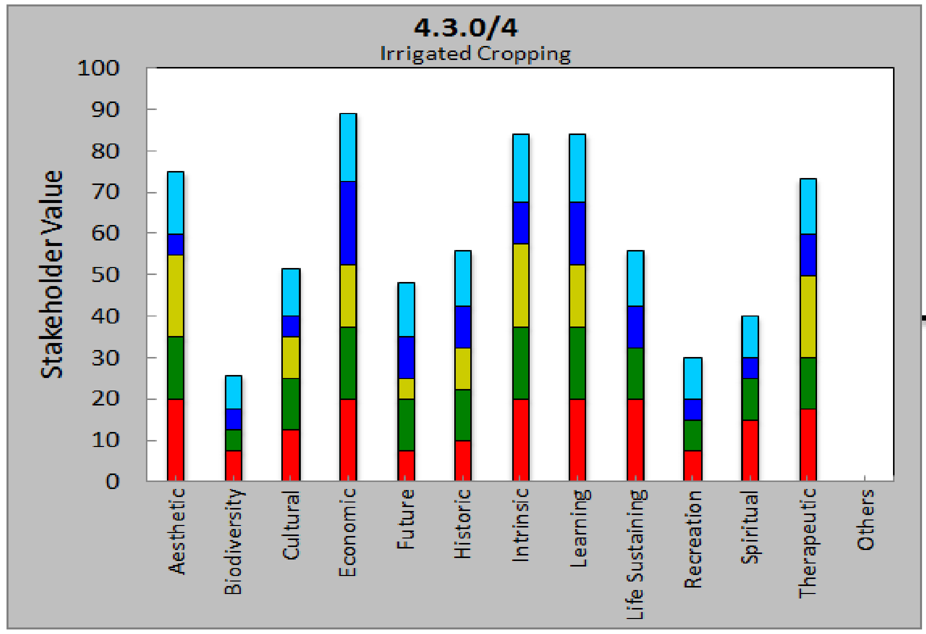
Description of Decision Factors Chart:

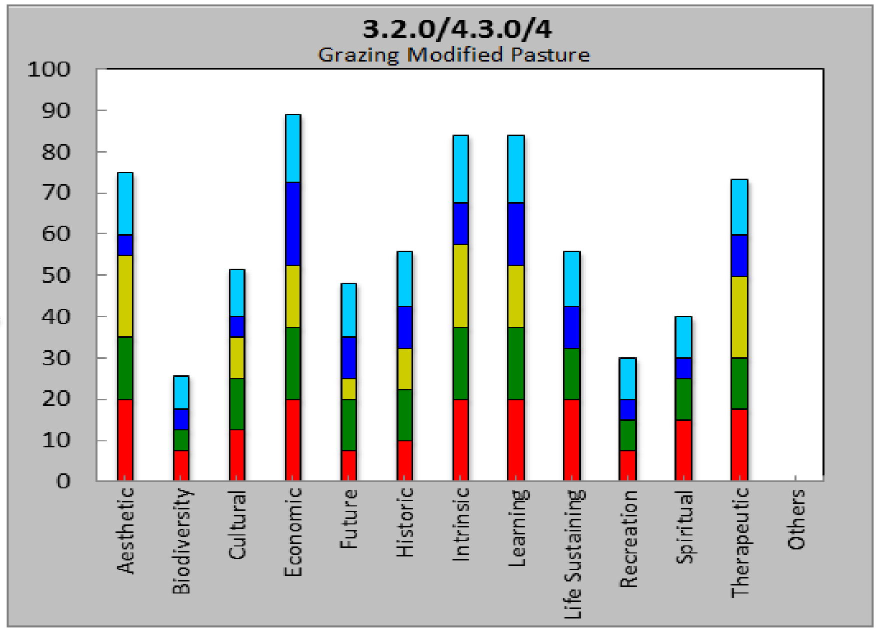
The values above zero indicate support for changing the land use to another land use, whereas values below zero indicate support for retaining the land use in its existing state. Zero indicates no preference for changing or not changing the land use. The factors in the chart include; land capability, elevation, slope, aspect, dominant soil, drainage lines, area of land, surrounding land use, and policy.

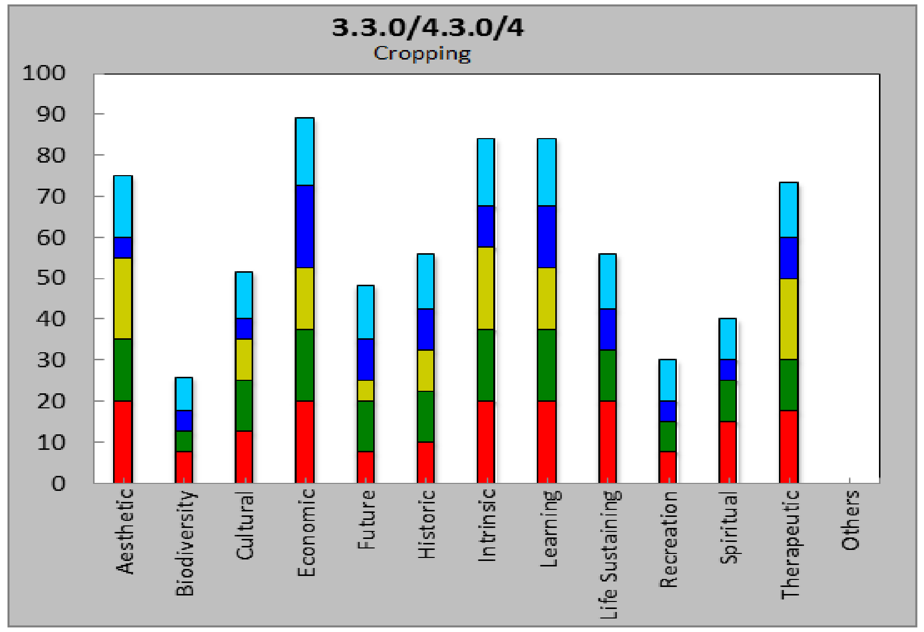
Contextual Comments:

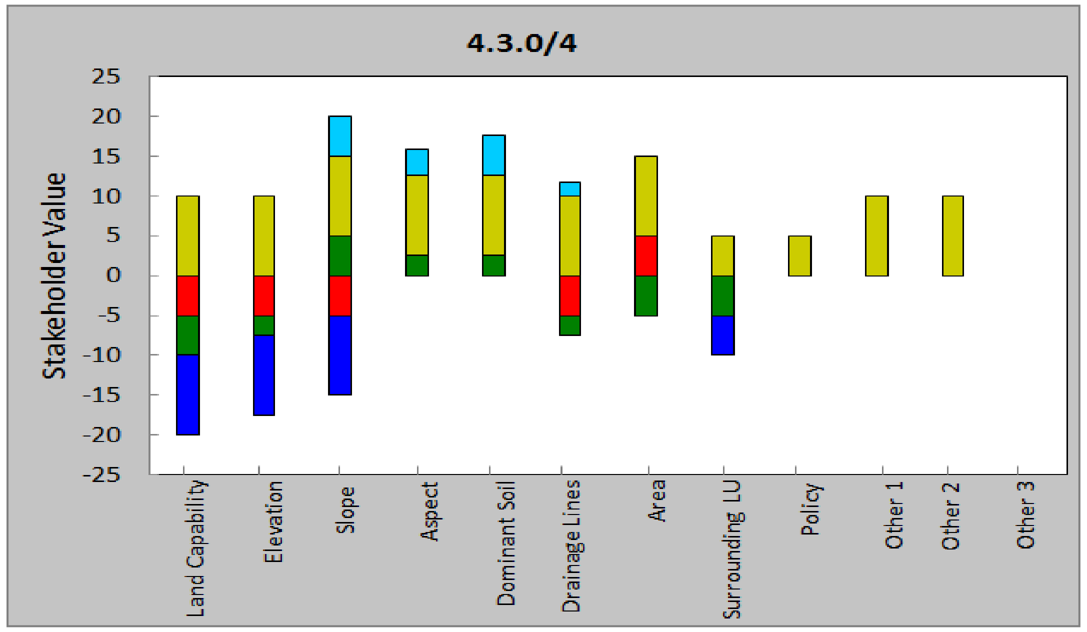
In the survey conducted, a few participants indicated that the area of this land use being large, with no prime use, would mean they were paying rates with no economic return. Some participants indicated that the natural values of the land use should be retained and considered in its natural state, given its large size and proximity to surrounding habitat species management area. Others indicated that change to Habitat species management area is highly preferable as there exists potential incentives to increase protection of the land.

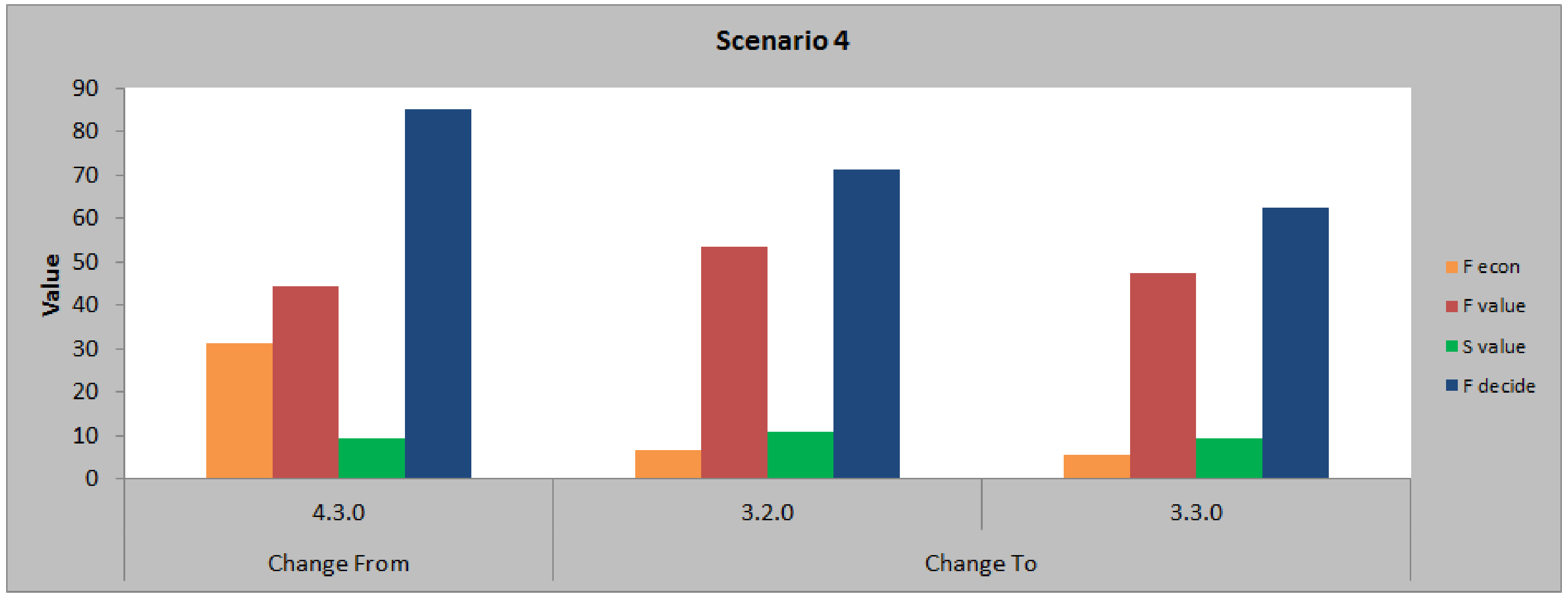
Scenario 4:









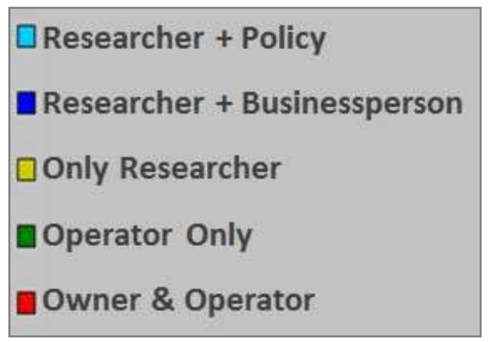


Description of Decision Factors Chart:

The values above zero indicate support for changing the land use to another land use, whereas values below zero indicate support for retaining the land use in its existing state. Zero indicates no preference for changing or not changing the land use. The factors in the chart include; land capability, elevation, slope, aspect, dominant soil, drainage lines, area of land, surrounding land use, and policy.

Contextual Comments:

In the survey conducted, a few participants indicated that if more sustainable or more productive uses can be found, then evidence of the decision factors would persuade them to change, surrounding land uses being grazing. A few indicated that the aspect makes it cold, frost prone country, and the dominant soil being chromosol cannot be abused and needs careful cropping. One participant indicated that drainage lines could cause water pollution and another said that access to irrigation would be good.



9.2.1 Scenario 1:

Scenario 1 deals with a change from land use Grazing Natural Vegetation (2.1.0) to: Residual Native Cover (1.3.3), Grazing Modified Pastures (3.2.0), or Habitat Species Management Area (1.1.5).

A. Detailed Summary

The highest F_{decide} in the scenario is observed for land use Grazing Natural Vegetation (2.1.0) managed for fine wool production, demonstrating that farmers as a whole are inclined towards Grazing Natural Vegetation, followed by for Grazing Modified Pastures (3.2.0) managed for breeding and trading first cross ewes. This is followed by the land uses Habitat Species Management Area (1.1.5), i.e. a protected area and by Residual Native Cover (1.3.3) having no prime use. The results of the detailed summary chart show that F_{econ} is a major contributor to F_{decide}. As the land uses Habitat Species Management Area (1.1.5) and Residual Native Cover (1.3.3) have zero F_{econ}, they contribute less to F_{decide} compared with the land use Grazing Natural Vegetation (2.1.0) and Grazing Modified Pastures (3.2.0). Additionally, F_{econ} for Grazing Natural Vegetation (2.1.0) is higher than Grazing Modified Pastures (3.2.0), thus producing higher values for F_{decide}. The F_{value} for Grazing Natural Vegetation (2.1.0) is a little higher than the F_{value} for Grazing Modified Pastures (3.2.0), but it is not serving as a major contributing factor to F_{decide}. The high influence of F_{econ} on F_{decide} is because it carries the highest weight (calculated using expert's opinion through AHP) among the sub-functions, which plays a major role in increasing or reducing the value of F_{decide} for land uses.

F_{value} is highest for Grazing Natural Vegetation (2.1.0), followed by both Grazing Modified Pastures (3.2.0) and Habitat Species Management Area (1.1.5), which are followed by for Residual Native Cover (1.3.3). This shows that, overall, farmers value Grazing Natural Vegetation the highest. In addition, it is noted that farmers overall value the land use Grazing Modified Pastures and Habitat Species Management Area (1.1.5) equally, whereas, farmers overall value Residual Native Cover (1.3.3) the least. This indicates that farmers would prefer Grazing Natural Vegetation (2.1.0) a more native landscape over Grazing Modified Pastures (3.2.0) that is modified and occupies majority of exotic species. Also, it is noted that the values assigned to Habitat Species Management Area (1.1.5) and Grazing Modified Pastures (3.2.0) by farmers are overall the same, although there is an apparent distinction in the former being a protected area with natural and native vegetation, while the latter having predominantly exotic species. The details on differences within the values can further be obtained in individual summary charts for both of the land uses. The farmers overall valued Residual Native Cover (1.3.3) the least, indicating that the values they assign do not solely depend on land having native vegetation, but also depend on how on how it is managed. Residual Native Cover (1.3.3) does not have any prime use and is also legally not regarded as protected area.

Examining the S_{value} in Scenario 1, it is noted that the highest value by other stakeholders is given to Habitat Species Management Area (1.1.5), followed by to Residual Native Cover (1.3.3), Grazing Natural Vegetation (2.1.0) and the least to Grazing Modified Pastures (3.2.0). This provides an overall view of other stakeholders' values, and demonstrates higher values placed on protected (1.15) or untouched/unmanaged (1.3.3) natural and native landscapes. This is followed by Grazing Natural Vegetation (2.1.0) – a landscape comparatively less influenced by external pressures (humans and animals) and still occupying more than 50% of native vegetation, and then Grazing Modified Pastures (3.2.0).

These results indicate that farmers overall value Grazing Natural Vegetation the highest and Residual Native Cover (1.1.5) the lowest in the scenario, whereas other stakeholders overall regard Habitat Species Management Area (1.1.5) of highest value and Grazing modified Pastures (3.2.0) the least.

B. Individual Summary

i. Values

Comparing the cumulative values across land uses for each value type, it is evident that the highest values are assigned to land use Habitat Species Management Area (1.1.5) for each value type except for economic, which is lower (third priority). Major value distinctions for cumulative values between land uses are observed for Biodiversity (range from 50 to 100), Economic (range from 35 to 90), Future (range from 65 to 90), Historic (range from 65 to 90) and Recreation (40 to 75). Other values shown to be almost steady for value types are Aesthetic (range from 85 to 100), Cultural (range from approx. 65 to 80), Intrinsic (range from 80 to 90), Learning (80 to 95), Spiritual (45 to 65) and Therapeutic (70 to 90).

These findings illustrate that Habitat Species Management Area (1.1.5) is prioritised when compared to other land uses, although lower economic value indicates low economic output. A Habitat Species Management Area (1.1.5) is a protected area, possibly having a covenant agreement to protect the habitat. Findings show that the land use is highly valued, although the economic value lowers its total value. Economic value may indicate the economic costs required to maintain this land use or/and economic returns from the land use. The major changes observed in values across land uses are due to differences in values of biodiversity, economic, future, historic and recreation values.

Examining the land use Grazing Natural Vegetation (2.1.0), it is notable that farmers (Owner & Operator, Operator Only) have put a comparatively lower economic value on this land use compared with other stakeholders (Researcher only, Researcher + Businessperson, Researcher + Policy). In the case of Future value and Recreation value, the stakeholders Researcher + Businessperson allocated it the lowest value among others. Thus, we observe differences in values within stakeholders for single land uses and across land uses.

In the survey for land use Grazing Natural Vegetation (2.1.0), 66% of the participants said that the values assigned to the land use are affected by parcel size and adjacent similar parcels, 42% indicated that transport routes may affect the values, while only 14% indicated that irrigation or farm infrastructure may affect the values assigned to the land use.

ii. Decision Factors

This chart and analysis however are not involved in deriving the F_{decide} values or the decisions, yet they provide complementary information in ascertaining whether the results on decision factors coincide with the results obtained from detailed and individual summary charts. Examination of the decision factors chart shows that retaining the current land use (2.1.0) is the most likely outcome, as is also evident from the detailed summary charts. The factors and stakeholders that are supporting no change in the land use are land capability (Owner & Operator, Researcher + Businessperson), elevation (Owner & Operator, Operator Only, Researcher + Businessperson), slope (Owner & Operator, Researcher + Businessperson), dominant soil (Owner & Operator, Operator Only), area of land (Owner & Operator, Operator Only), and policy (Researcher + Businessperson). The factors and stakeholders that are supporting change are aspect (Operator Only, Only Researcher, Researcher + Policy), drainage lines (Operator Only, Only Researcher, Researcher + Policy) and surrounding land use (Only Researcher, Researcher + Policy).

C. Scenario 1 Conclusion and Findings

The detailed and the individual charts, together with the decision factors chart, indicate that the stakeholders' preference is to retain this land use in its current state rather than changing it into any other land use. In addition, the probability of occurrence calculated for this scenario yield 1 for Grazing Natural Vegetation (2.1.0) meaning the land use is selected, whereas, other land uses within the scenario yield 0 meaning the land uses were not selected for change. If a decision to change was made, the preferred change would be to Grazing Modified Pastures (3.2.0), as evidenced by F_{decide} in the detailed summary, although Grazing Modified Pastures (3.2.0) depicts lowest values (lowest F_{value} and S_{value} among scenario land uses) assigned by farmers and other stakeholders.

Grazing Natural Vegetation (2.1.0) is a native vegetation land use, which requires management for grazing and provides economic returns. Whereas Residual Native Cover (1.3.3) land use is also a native vegetation land use, but does not require any (substantive) management and does not provide the economic returns of Grazing Natural Vegetation. Conversely, Habitat Species Management Area (1.1.5) occupies protected land with native vegetation, may require managing the land, and the farmer may receive payments by the Government under various projects and schemes for entering into a covenant. The farmers value Grazing Modified Pastures (3.2.0) and Habitat Species Management Area (1.1.5) equally and more than Residual Native Cover (1.3.3). This illustrates that farmers do not only consider native vegetation while valuing land uses (as the high value of Grazing Modified Pastures shows), rather they also consider the management of the land use that is linked with the economic return from the land use. Conversely, other stakeholders have valued land uses in view of assigning higher priority to native lands.

In this scenario, the farmer has two major pathways to decide between: one towards changing the native vegetation to exotic vegetation (change to Grazing Modified Pastures), the other towards retaining the native vegetation. The values indicate that preference is mostly towards retaining the native vegetation rather than changing it to exotic, although there is higher economic return from changing the vegetation to exotic species. Within the pathway towards retaining the native vegetation, there exist choices of: 1) retaining it in its current land use (Grazing Natural Vegetation) by allowing grazing on native species, 2) leaving the native vegetation with no prime use and discontinuing grazing (Residual Native Cover), or 3) entering into a covenant and declaring it as protected land (Habitat Species Management Area). The first choice provides some economic return by allowing grazing on the land; the other two choices do not provide economic returns in this scenario although, if a covenant is accompanied by payments for conservation, it would provide some economic return as well.

As evident from the detailed summary that takes into account economic returns (F_{econ}), farmers' value (F_{value}) and other stakeholders' value (S_{value}), the results show that the priority remains to retain the land in its existing land use, i.e. Grazing Natural Vegetation. While only looking at cumulative values, Habitat Species Management Area has attained highest values, whereas Grazing Modified Pastures has been least valued by farmers and other stakeholders alike, although it's high economic return has put it in second priority in detailed charts.

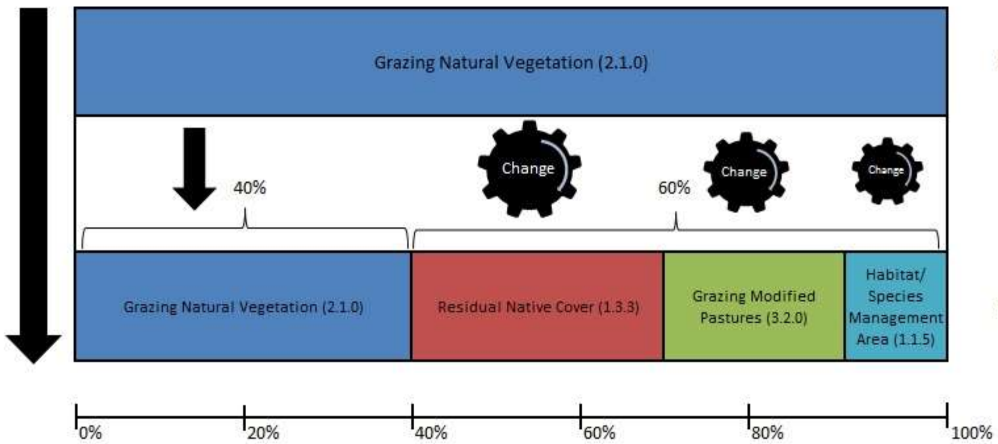


Figure 9.3: Historical change of land use Grazing Natural Vegetation (2.1.0) from 2001 to 2010

Examining the land use change from 2001 to 2010 in the area shown in Figure 9.3, the statistics indicate firstly that 40% of Grazing Natural Vegetation (2.1.0) retained this land use. Where there was change, this was mostly to Residual Native Cover (1.3.3), followed by Grazing Modified Pastures (3.2.0), and thirdly Habitat Species Management Area (1.1.5). Except for economic value, minimum values were assigned to land use Grazing Modified Pastures (3.2.0) and almost highest (second highest) assigned to Residual Native Cover (1.3.3). It is evident that economic value as well as economic return play a significant role in this case. Habitat Species Management Area (1.1.5) does involve entering into a voluntary conservation agreement/covenant. Covenants in the area have been either for set-term or for perpetuity. Set-term were rarely used in Tasmania (Iftekhar *et al.* 2014). Therefore, it may be that farmers have decided to keep land under no prime use (Residual Native Cover) for a certain number of years and then resume grazing over native vegetation (Grazing Natural Vegetation). In addition, the move towards Habitat Species Management Area (1.1.5) evident from historic data, although lower in area than Native Residual Cover (1.3.3) and Grazing Modified Pastures (3.2.0), can be associated with successful policy initiatives towards protecting private lands by providing the monetary payments required to cause this change (absent in the scenario above as $F_{econ} = 0$) in the decade from 2001 to 2010

9.2.2 Scenario 2:

Scenario 2 deals with the change from land use Grazing Modified Pastures (3.2.0) to Residual Native Cover (1.3.3), Cropping (3.3.0), Grazing Natural Vegetation (2.1.0) and Irrigated Cropping (4.3.0).

A. Detailed Summary

In this scenario, the highest F_{decide} is obtained for Irrigated Cropping (4.3.0), followed by Grazing Natural Vegetation (2.1.0) and then for the current land use: Grazing Modified Pastures (3.2.0). The remaining land uses – Cropping (3.3.0) and Residual Native Cover (1.3.3) – have a lower F_{decide} than the original Grazing Modified Pastures (3.2.0). It is evident from this scenario that F_{econ} plays a significant role in determining the F_{decide} values, owing to the high weight assigned to F_{econ}.

Examining the overall farmers' value (F_{value}), it is evident from the detailed chart that land use Grazing Natural Vegetation (2.1.0) has the highest earned values followed by Residual Native Cover (1.3.3), with the least assigned to Irrigated Cropping (4.3.0). This shows that the farmers overall attach the least (non-economic) values to Irrigated Cropping, although it has the highest economic return (F_{econ}). It also indicates that the highest values assigned to Grazing Natural Vegetation (2.1.0) cannot be based entirely on economic returns associated with this land use. The farmers have values other than economic value (Biodiversity, Future, Historic and Recreation – mentioned in Scenario 1) that they regard significant. Therefore, if we only take F_{value} into consideration, the trend would go from Grazing Modified Pastures (3.2.0) to Grazing Natural Vegetation (2.1.0).

Similarly, for other stakeholders' value (S_{value}), the highest values are assigned to land use Residual Native Cover (1.3.3), followed by Grazing Natural Vegetation (2.1.0) and then to Grazing Modified Pastures (3.2.0) itself. The least values by other stakeholders are assigned to the highly modified landscapes of Irrigated Cropping (4.3.0) and lastly Cropping (3.3.0). Thus, it is evident from the values assignment by other stakeholders that they value least disturbed native vegetation landscapes the highest. Residual Native Cover (1.3.3) occupies a second highest place in farmers value overall and first highest place for other stakeholders value among the land uses in the scenario, although owing to the zero economic return (F_{econ}), it has the least F_{decide} among the land uses. The detailed chart suggests that the change trend could be either towards Irrigated Cropping (4.3.0) with highest economic return, or towards Grazing Natural Vegetation (2.1.0) having highest farmers value.

B. Individual Summary

i. Values

The highest individual cumulative values for each value type is obtained by two land uses including land use Residual Native Cover (1.3.3) and Grazing Natural Vegetation

(2.1.0), having some values highest in the former land use and others in latter, except for economic value. Similarly, the lowest cumulative individual values assigned for each value type is for land use Irrigated Cropping (4.3.0) and Cropping (3.3.0), having lowest value types in former land use while others in the latter, except for economic value which is highest for Irrigated Cropping (4.3.0) followed by for Cropping (3.3.0).

Of particular interest are the varied and broad range differences as seen for most of the scenario land use values including Biodiversity, Economic, Future and Life sustaining values. The highest range variation is found in value types Biodiversity, being lowest for Irrigated Cropping (4.3.0) i.e. 23 and highest for Residual Native Cover (1.3.3) i.e. 100.

While investigating the values assigned to Grazing Modified Pastures (3.2.0) by various groups of individuals, we observe that the least values in Biodiversity among other stakeholders are given by Researcher + Businessperson. For Recreation and Spiritual values in Grazing Modified Pastures (3.2.0), the least values are given by Only Researcher and Researcher + Businessperson. These findings inform us that within a particular land use, and a specific value type, i.e. Biodiversity in this case, we can note which participant group has valued least and highest, providing us information about who values it lower than others and so assisting the design of interventions that might increase these values for a target group. Thus, we can observe differences in values within stakeholders for a single land use and across land uses in this way.

In the survey for land use Modified Grazing Pastures (3.2.0), 83% of the responses indicated that the values assigned to the land use are affected by parcel size and 75% said by adjacent similar parcels, 42% indicated that transport routes may affect the values, while 85% indicated that irrigation or farm infrastructure may affect the values assigned to the land use.

ii. Decision Factors

The overall decision factors chart, having an increasingly and prominent upward pattern rather than downwards, suggests change of one land use to another land use. The resultant net factors and stakeholders that are supporting change in the land use are land capability (All Stakeholders), Elevation (All Stakeholders), Slope (All Stakeholders), Dominant Soil (Owner & Operator, Operator Only), Aspect (All Stakeholders except Researcher + Businessperson suggesting no effect), Dominant Soil (All Stakeholders), Drainage Lines (All Stakeholders except Researcher + Businessperson suggesting no effect), Area of land (All Stakeholders), Surrounding Land use (All Stakeholders except Researcher + Businessperson) and Policy (Researcher + Businessperson). The resultant net factors and stakeholders that are not supporting change are Aspect (Operator Only, Only Researcher, Researcher + Policy), Drainage Lines (Operator Only, Only Researcher, Researcher + Policy) and Surrounding Land use (Owner & Operator, Only Researcher).

C. Scenario 2 Conclusion and Findings

The detailed and the individual charts along with decision factors chart indicate that there is a high inclination of stakeholders towards changing this land use to another land use rather than retaining it. The options obtained from the detailed chart suggest the change either to land use Irrigated Cropping (4.3.0), with lowest farmers values and second lowest stakeholder values, or towards land use Grazing Natural Vegetation (2.1.0), having highest farmers and second highest other stakeholders values for the land use. The inclination towards Irrigated Cropping (4.3.0) is based on economic returns, while for Grazing Natural Vegetation, the inclination is based upon some economic return, but also obtaining a native vegetation landscape. . In addition, the probability of occurrence calculated for this scenario yield 1 for Irrigated Cropping (4.3.0), meaning the land use is selected for change, whereas, other land use within the scenario yield 0 denoting the land uses were not selected for change.

In this scenario also, the farmers' value trend has gone towards Grazing Natural Vegetation (2.1.0), which is not the highly profitable land use (highly profitable being Irrigated Cropping with least value). This indicates that for this scenario and land features associated with this scenario, the farmers' values cannot be based solely on economic returns associated with this land use, but some other values (Biodiversity, Future, Historic and Recreation, etc. mentioned in Scenario 1) have precedence over economic value for farmers. It may be attributed to the type of dominant soil, slope, aspect, drainage line and intensive management required that led the farmers to value Grazing Natural Vegetation more over Irrigated Cropping. Looking into the values of other stakeholders: the values are high for land uses with least external disturbance having native vegetation (i.e. Residual Native Cover (1.3.3) and Grazing Natural Vegetation (2.1.0)) and becoming low for land uses with highest external influence (i.e. Irrigated Cropping (4.3.0) and Cropping (3.3.0)). This shows the diverging values among farmers and other stakeholders. These examples illustrates the capacity of the modelling to concisely communicate both overall value trends and detailed component value trends.

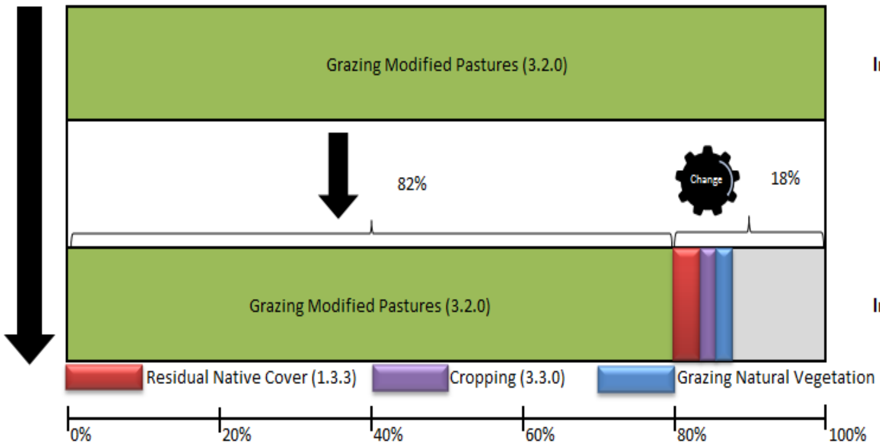


Figure 9.4: Historical change of land use Grazing Modified Pastures (3.2.0) from 2001 to 2010

While examining the land use change from 2001 to 2010 in the area shown in Figure 9.4, the statistics indicate that the trend of change was more towards not changing the land use and retaining it, and if changed, the change was observed to be towards Residual Native Cover (1.3.3), Cropping (3.3.0) and Grazing Natural Vegetation (2.1.0). It is also to be noted here that the change scenario generated by the model depends upon the underlying conditions of the land use parcel in question, as the values and the decision factors depend upon the conditions of the parcel. That is, the high inclination towards changing the land use as indicated by the model may be explained by underlying spatial factors (e.g. parcel size, elevation, soil, slope, adjacency to other land uses). This result from the model does however contradict data observed in the change data analysis from 2001 to 2010, which might be explained by some other change in values, policies or other factors since that time.

9.2.3 Scenario 3:

Scenario 3 deals with the change from land use Residual Native Cover (1.3.3) to Grazing Natural Vegetation (2.1.0) and Habitat Species Management Area (1.1.5).

A. Detailed Summary

In this scenario, the highest F_{decide} is obtained by land use Grazing Natural Vegetation (2.1.0), followed by for land use Habitat Species Management Area (1.1.5). The influence of F_{econ} on increasing and decreasing F_{decide} is also evident in this scenario, as it was in previous scenarios, although the highest values assigned to land use Grazing Natural Vegetation (2.1.0) by farmers (F_{value}), also contribute to the cause. The highest values assigned by other stakeholders goes to Habitat Species Management Area (1.1.5), followed by Residual Native Cover (1.3.3) and the least to Grazing Natural Vegetation (2.1.0) among the lot. This indicates that other stakeholders have highest values for least disturbed land uses with native vegetation intact, although farmers' values depend upon obtaining native vegetation and having an appropriate management practice with economic return. Looking at the overall detailed chart, it suggests that the change may take place towards Grazing Natural Vegetation (2.1.0), but can also result in Habitat Species Management Area (1.1.5) i.e. a protected land, if assisted with economic incentives. The land is currently under no prime use and is not being managed, although is occupied by native vegetation. Therefore, in order to convert the land to Habitat Species Management Area (1.1.5) would only require legalising it, which can be made a first priority with associative incentives in this case.

B. Individual Summary

i. Values

The highest cumulative values for individual value types is obtained for land use Habitat Species Management Area (1.1.5), whereas the lowest values under some value types are for Grazing Natural Vegetation (2.1.0) and others for Residual Native Cover (1.3.3), except for economic value. Economic value is highest for Grazing Natural Vegetation (2.1.0) and lowest for Residual Native Cover (1.3.3). The value ranges for individual value types are not wide, but are rather narrow for the land uses in this scenario except for economic value which ranges from 37 to 62. It is to be noted that the values allocated to land use Grazing Natural Vegetation (2.1.0) by Owner & Operators in Others also plays a significant role in alleviating the overall values of this land use.

While looking at values assigned by various stakeholders to land use Residual Native Cover (1.3.3), it is observed that the least values assigned by Only Researcher, Owner & Operator and Researcher + Businessperson to Economic, to Economic, Recreation and Spiritual values would influence it to change to other land uses, although, other values being mostly between the two other land uses (Habitat Species Management Area (1.1.5) the highest, and Grazing Natural Vegetation (2.10) the lowest). In this way, the model describes the differences in values within stakeholders for a single land use and across land uses.

In the survey for land use Residual Native Cover (1.3.3), 85% of the participants said that the values assigned to the land use are affected by parcel size and 42% said by adjacent similar parcels; 42% indicated that transport routes may affect the values, while only 14% indicated that irrigation or farm infrastructure may affect the values assigned to the land use.

ii. Decision Factors

The overall decision factors chart, having an increasingly and prominent upwards pattern than downwards, suggests change to another land use. The resultant net factors and stakeholders that are supporting change in the land use are Elevation (Only Researcher, Researcher + Businessperson), Slope (Owner & Operator, Only Researcher, Researcher + Businessperson), Aspect (Owner & Operator, Operator Only, Only Researcher, Researcher + Businessperson), Dominant Soil (Only Researcher), Drainage Lines (Owner & Operator, Researcher + Policy), Area of land (Owner & Operator, Researcher + Policy), Surrounding Land use (Owner & Operator, Only Researcher, Researcher + Businessperson), Policy (Researcher + Businessperson) and Others (Only Researcher). The resultant net factor and stakeholders that are not supporting change are Land Capability (Owner & Operator, Operator Only, Researcher + Businessperson, Researcher + Policy).

C. Scenario 3 Conclusion and Findings

The detailed and the individual charts along with decision factors chart indicate that there is high inclination of stakeholders towards changing this land use to another land use rather than retaining it. The charts suggests that the most likely change is towards land use Grazing Natural Vegetation (2.1.0), depending upon highest F_{decide} and farmers values. Although, the change can also be towards Habitat Species Management Area (1.1.5), if assisted with economic incentives. In addition, the probability of occurrence calculated for this scenario yield 1 for Grazing Natural Vegetation (2.1.0) meaning the land use is selected for change, whereas, other land use within the scenario yield 0 denoting the land uses were not selected. Farmers' values in this scenario are also largely based on not only obtaining native vegetation but also on having an appropriate management practice with economic return. On the contrary, other stakeholders have values based on native vegetation prioritisation.

Examining the land use change dataset for years 2001 and 2010 shown in Figure 9.5, analysis suggests change to other land uses, consistent with the results on decision factors obtained from the model. Also, the historical data also suggests a first priority change to land use Grazing Natural Vegetation (2.1.0), followed by a lesser amount of change to Habitat Species Management Area (1.1.5), which also supports the analysis and results obtained from the model. This result of change from Residual Native Cover (1.3.3) largely towards Grazing Natural Vegetation (2.1.0) is also consistent with the findings of Scenario 1, where Grazing Natural Vegetation (2.1.0) was converted to Native Residual Cover (1.3.3), possibly for the same reasons suggested for Scenario 1. In addition, the move towards Habitat Species Management Area (1.1.5) (as shown in Scenario 1) is also evident from historic data, although for a lower total area than Grazing Natural Vegetation (2.1.0). This can be associated with successful policy initiatives aimed at protecting private lands by providing the monetary payments needed to obtain this land use change (absent in the scenario above as $F_{econ} = 0$) in the decade from 2001 to 2010.

The statistical analysis carried out for land use change from 2001 to 2010 in the area indicate that the trend of change was highest towards Grazing Natural Vegetation, supporting our current argument on land use change values derived from the model.

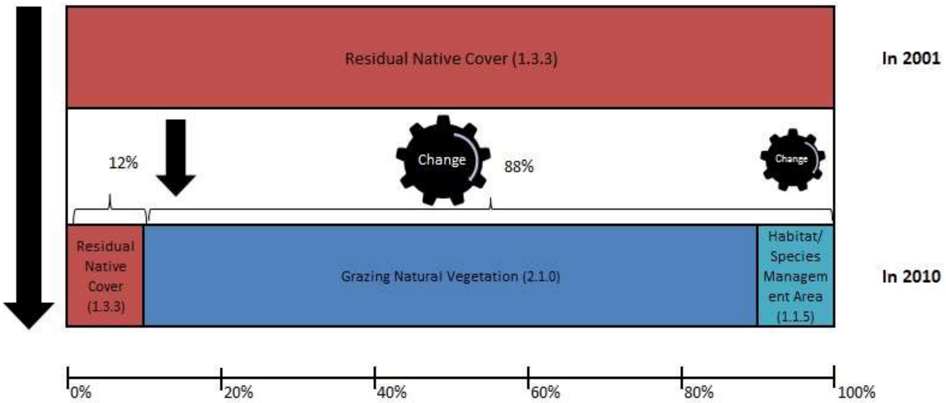


Figure 9.5: Historical change of land use Residual Native Cover (1.3.3) from 2001 to 2010

9.2.4 Scenario 4:

Scenario 4 deals with the change from land use Irrigated Cropping (4.3.0) to Grazing Modified Pastures (3.2.0) and Cropping (3.3.0).

A. Detailed Summary

The highest F_{decide} in the scenario is obtained for the land use Irrigated Cropping (4.3.0) owing to its high F_{econ} values, with the second highest for Grazing Modified Pastures (3.2.0) and lastly for Cropping (3.3.0). In this scenario too, F_{econ} plays the major role in increasing and decreasing F_{decide} in land uses. The highest F_{econ} is for Irrigated Cropping (4.3.0) followed by for Grazing Modified Pastures (3.2.0) and then for Cropping (3.3.0). Thus, the F_{decide} is mainly steered by the high weight assigned to F_{econ} in the model. While looking at the farmers' values i.e. F_{value}, the highest obtained is for Grazing Modified Pastures (3.2.0), followed by for Cropping (3.3.0) and lastly for Irrigated Cropping (4.3.0). Thus, we observe that even though the Irrigated Cropping (4.3.0) has the highest economic return, yet it has been valued least by the farmers. Other stakeholders have valued Grazing Modified Pastures (3.2.0) the highest, followed by Irrigated Cropping (4.3.0) and the least Cropping (3.3.0). Thus farmers as well as other stakeholders both have highest values for Grazing Modified Pastures in this scenario, suggesting that if change occurs, it would be the highest priority for the scenario.

B. Individual Summary

i. Values

The cumulative values across land uses for each value type show that most of the highest values are assigned to land use Grazing Modified Pastures (3.2.0), with some to Irrigated Cropping (4.3.0). Looking at each value type individually, the least values assigned to various value types encompass land use Irrigated Cropping (4.3.0), Cropping (3.3.0) and Grazing Modified Pastures (3.2.0). The value types that are assigned least values to land use Irrigated Cropping (4.3.0) are for biodiversity value, life sustaining value, recreation value, spiritual value and therapeutic value. The least values assigned to land use Cropping (3.3.0) include aesthetic, cultural, economic, future and historic, whereas, some for land use Grazing Modified Pastures (3.2.0) including intrinsic and life sustaining values. Major value distinctions of cumulative values between land uses are observed for biodiversity, cultural and life sustaining, while other value ranges for value types are found to be almost steady. Findings to be noted here are that the modelling concisely communicates that biodiversity, cultural and life sustaining values, play a vital role for decision making in this scenario.

Reviewing the values of stakeholders within the land use Irrigated Cropping (4.3.0), it is evident that the least values are for biodiversity values, followed by recreation and spiritual. For biodiversity, Only Researcher and Researcher + Businessperson have assigned zero values with the least from Owner & Operator, Operator Only and Researcher + Policy. For Life Sustaining value, Only Researcher has assigned zero values, along with minimum value by Owner & Operator. These examples illustrates the capacity of the modelling to concisely communicate both overall value trends within stakeholders for single land use and differences in value trends across land uses.

In the survey for land use Irrigated Cropping (4.3.0), 50% of the participants said that the values assigned to the land use are affected by parcel size and 33% said by adjacent similar parcels; 42% indicated that transport routes may affect the values, while 57% indicated that irrigation or farm infrastructure may affect the values assigned to the land use.

ii. Decision Factors

The overall decision factors chart, having an increasingly upwards pattern rather than downwards, suggests a change of this land use to another land use. The resultant net factors and stakeholders that are supporting change in the land use are Slope (Operator Only, Only Researcher, Researcher + Businessperson), Aspect (Owner & Operator, Operator Only, Only Researcher, Researcher + Policy), Dominant Soil (Owner & Operator, Operator Only, Only Researcher, Researcher + Policy), Drainage Lines (Operator Only, Only Researcher, Researcher + Policy), Area of land (Owner & Operator, Only Researcher), Policy (Only Researcher) and Others (Only Researcher). The resultant net factor and stakeholders that are not supporting change are Land Capability (Owner & Operator, Operator Only, Researcher + Businessperson), Elevation (Owner & Operator, Operator Only, Researcher + Businessperson) and Surrounding land use (Operator Only, Researcher & Business).

C. Scenario 4 Conclusion and Findings

The detailed chart depicts the highest F_{decide} values for land use Irrigated Cropping (4.3.0), thus suggesting no change in the land use based upon the highest values of F_{econ}. If there is to be change, then the most likely change would be to Grazing Modified Pastures (3.2.0) having second highest F_{decide} and subsequent F_{econ} values. Reviewing the chart on decision factors, this also suggests change in an overall view but it also suggests retaining the land use in view of some factors. The net result from the decision factors as a whole suggest change towards another land use. In addition, the probability of occurrence calculated for this scenario yield 1 for Irrigated Cropping (4.3.0), meaning the land use is selected, whereas, other land use within the scenario yield 0 denoting the land uses were not selected for change.

Examining the historical data on land use shown in Figure 9.6, analysis suggests change to other land uses, consistent with the results on decision factors obtained from the model. Also, the historical data suggests a first priority change to land use Grazing Modified Pastures (3.2.0), followed by a lesser amount of change to Cropping (3.3.0), which also supports the analysis and results obtained from the model.

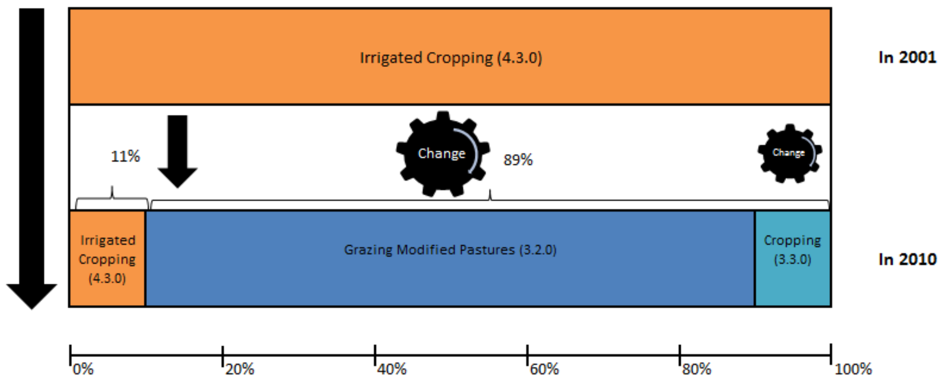


Figure 9.6: Historical change of land use Irrigated Cropping (4.3.0) from 2001 to 2010

9.3 Discussion

The model thus provides information on land use change priorities along with detailed information on values obtained on each value type for each land use. It captures details of different stakeholders and provides information about which values are considered significant from a specific stakeholder's perspective. This provides an opportunity to look at the possible trade-offs among values for specific stakeholders and to formulate policies in line with these values that might focus on enhancing specific values for a specific group of stakeholders to obtain desirable results.

The results obtained are largely consistent with the data analysis carried out on the historical land use data (2001 and 2010), which provides an insight into the reliability of the data values obtained and the success of the model. The decisions obtained in the model are based on the model algorithm having weights assigned to the sub-functions of F_{econ}, F_{value} and S_{value} based on expert opinion. The model can be tested or populated using different weights and is flexible in this regard.

In this research, participants were also questioned about whether spatial factors such as parcel size, adjacency to similar parcels, proximity of transport routes, or proximity of irrigation or farm infrastructure would affect the values they assigned to the land use. The results show that participant values do depend on attributes of the land that are spatially defined (area, proximity, etc.), and so demonstrate the value of incorporating the spatial dimension into these analyses. There is considerable capacity to extend and adapt these spatial factors on a study-by-study basis; this case study demonstrates the capacity to do this.

Each F_{decide} result depends on the values assigned to each land use by each of the participant groups and on the values of the sub-functions (F_{econ}, F_{value} and S_{value}). A Random Forest analysis (Variable Importance) was undertaken in order to better understand these contributing variables (values and sub-functions). This was achieved using "Rattle", which is a data mining graphical user interface for the R statistical programming language.

RF is commonly used to aid in selection of optimal variables for developing models where there is a large number of variables and a need to reduce this to a smaller set of most influential variable (Abdel-Rahman *et al.* 2012). In contrast, in this research, the model has already been developed and the prediction determined; the variable importance feature is here utilised to ascertain which variables are most significantly contributing to that prediction. For a detailed description of Random Forest and its operation, refer to Appendix F.

For this study, random forest and its variable importance feature is utilised to compare relative importance among predictor variables and to ascertain which variable (F_{econ}, F_{value}, S_{value} etc.) contributes more towards already predicted target variable F_{decide}. In this research, percentage increase in mean square error (%IncMSE) is used as a measure for variable importance.

While generating the variable importance through random forest, a training sample of 70% was assigned that is called 'in bag' data, whereas a validation sample of 15% and testing sample of 15% was allocated, and referred to as 'out of bag' data. The OOB sample, which is the set of observations that are not used for building the current tree, is used to estimate the prediction error (OOB error) and then to evaluate variable importance (Williams 2008). As the OOB sample is not used to build trees in the ensemble, therefore, the OOB estimate of error is considered as a reliable form of cross-validation and a good source of prediction accuracy (Breiman 2001, Prasad *et al.* 2006, Siroky 2009). The number of trees 'ntree' and number of variables 'mtry' were optimised based on OOB estimate of error (Liaw and Wiener 2002, Adam *et al.* 2012). Finally, with the least OOB estimate of error, ntree was optimised as 500 and mtry as 7.

Strobl *et al.* (2009) suggest that variable importance can be biased when a dataset contains predictor variables of different types, and that the Gini index measure is more biased than the permutation importance measure. Strobl *et al.* (2009) instead propose a conditional variant (cForest) of the traditional random Forest, as the latter overestimates the results of highly correlated variables (Genuer *et al.* 2010). Strobl *et al.* (2009) also suggest that a 'guaranteed unbiased' variable importance is achieved if conditional cForest is used. Therefore, to avoid biasness in splitting, and overestimation of highly correlated variables, the permutation importance measure (%IncMSE) and conditional variant cForest methods was utilised for this study.

In order to analyse the importance of variables on predictions, the data was set to analyse the target F_{decide} for all the land uses obtained, which serves as the prediction to all the other variables of F_{econ}, F_{value}, S_{value}, and individually assigned values of stakeholders on all value types. Charts 9.1, 9.2, 9.3, 9.4 and 9.5 indicate variables on y-axis and percentage increase in mean square error on x-axis.

The results shown in Chart 9.1 indicate that the most important factor in predicting F_{decide} is F_{econ}, which supports the arguments in the 'Results' section, followed by the Cultural values of stakeholder Only Researcher (SOR Cultural) and F_{value}. A Random Forest analysis was then again carried out but ignoring the F_{econ} variable, with the results shown in Chart 9.2. It was found that among these variables, the highly important variable in predicting F_{decide} was life sustaining value assigned by stakeholder Researcher + Business (SRB Life Sustaining), followed by Other value assigned by Farmer Owner & Operator (FL Other 1), and F_{value}. This also showed that the Other value assigned to land use actually increases the importance of it in predicting F_{decide} for that specific land use, as only a few stakeholders assigned it to a few land uses. Therefore, in the survey questionnaire, it is suggested for future studies, either to remove the 'Other' value option, or to obtain a value in 'Other' from all stakeholders and for all land uses, in order to avoid risk of bias in the result. As the variable 'Other' values did not receive any input by most of the stakeholders, in a subsequent trial of Random Forest all Other values were also ignored, resulting in the output from the Random Forest analysis shown in Chart 9.3. This result finally showed that the most important variable in predicting F_{decide}, after ignoring F_{econ} and Other values, was SRB Life Sustaining (stakeholder Researcher + Business), followed by F_{value}, SOR Cultural (stakeholder Only Researcher), SOR Aesthetic (stakeholder Only Researcher), SRB Historic (stakeholder Researcher + Business).

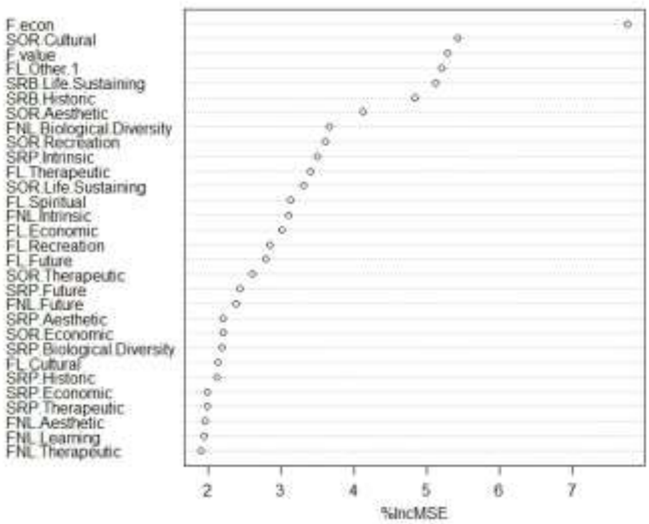


Chart 9.1: Random Forest Variable Importance – All Variables with F_{decide} as target

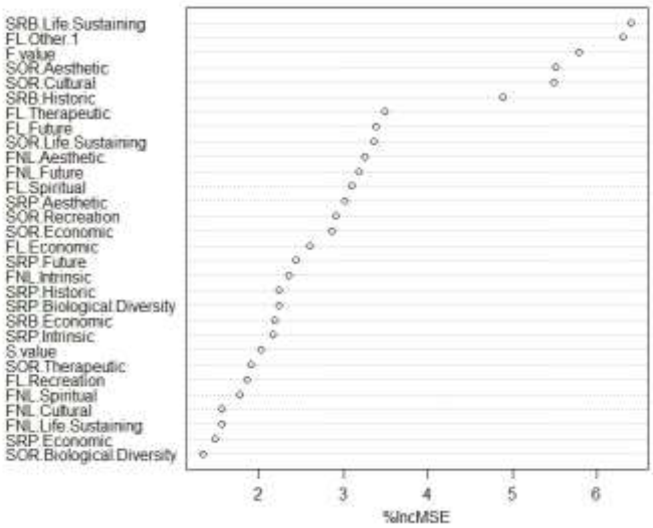


Chart 9.2: Random Forest Variable Importance – All variables except F $_{econ}$

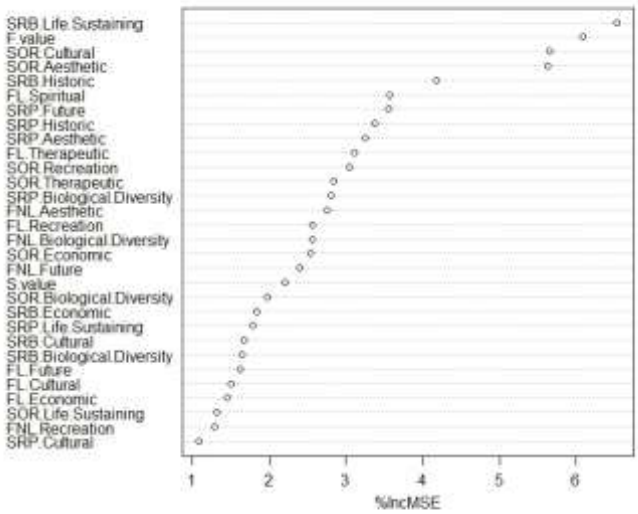


Chart 9.3: Random Forest Variable Importance – All variables except F $_{econ}$ and Other values

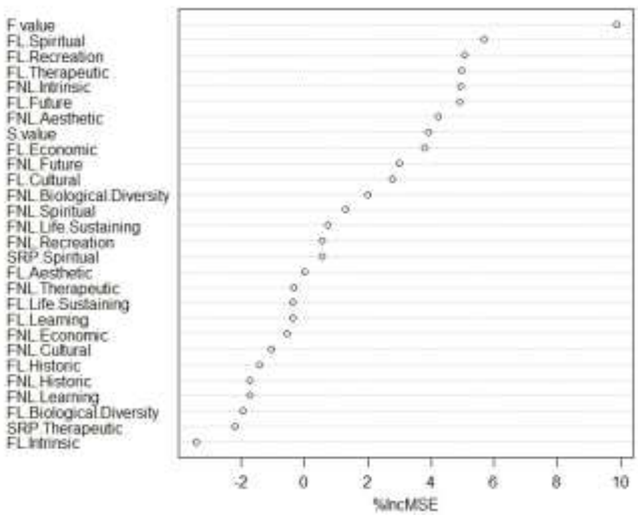


Chart 9.4: Random Forest Variable Importance – Only farmers values as variables

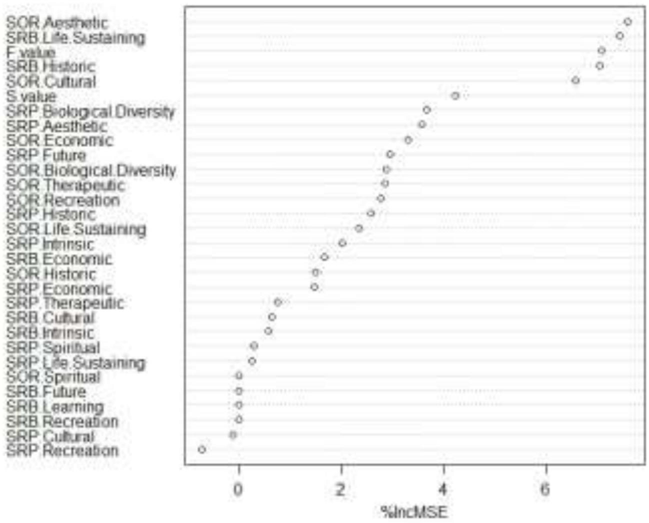


Chart 9.5: Random Forest Variable Importance – Only Other Stakeholders values as variables

Moreover, the relative importance of farmers values as well as for other stakeholders values were analysed individually, to ascertain which values within farmers values and within other stakeholders values have highly contributed in predicting F_{decide}, therefore, Random Forest was generated by ignoring all other variables except for farmers values achieved in Chart 9.4 and by ignoring all other variables except other stakeholders values achieved in Chart 9.5. Chart 9.4 shows that the most important variable was F_{value}, followed by FL Spiritual (farmer Owner & Operator), FL Recreation (farmer Owner and Operator) and FL Therapeutic (farmer Owner & Operator).

This analysis provides an insight into the variables that are influencing the final F_{decide} and thus can help us interpret the results as well as focus on those variables while observing the charts.

9.4 Conclusion

This chapter has described and discussed the outputs and results obtained from population of the SEEM using survey data obtained for Maryland Farm. The model outputs, and the visualisation tools developed to graphically illustrate these outputs, demonstrate how the modelling can be used to understand the complex mix of values held by a diversity of stakeholders and the ways in which these values can drive decision making, and so – in turn – how decision making might most effectively be actively managed through targeted interventions.

The SEEM can be utilised in decision-making setting. For example, a land parcel identified as ideal for various land uses include grazing natural vegetation, grazing modified pastures and habitat species management area. These land use alternatives are based on the values of diverse stakeholders assigned holistically in relation to the spatial location of the parcel. This holistic problem can be viewed by the SEEM, that can provide an alternative approach involving human rationalisation and trade-offs. The trade-offs can also be made in the values by elevating for example, economic incentives of a land use, assisting in introducing such incentives at policy level, thus, economically promoting the land use. The trade-offs can also involve statements such as 'If you let me graze sheep on modified pastures on this parcel, I will let you set aside a habitat species management area on the other one'. Such dialogues are followed by persuasive arguments and group discussions, which are far from mathematical optimisation solutions. The dialogue that can follow may involve a statement as, 'I can let you graze sheep on native pasture on this parcel and set aside the other for habitat species management area' that can lead to an acceptable decision. Such discussion settings use models as the SEEM to focus discussion towards a specific spatial area and its unique distribution of conditions and potential uses. The model assists in developing spatial reasoning skills and reaching an acceptable decision.

10 Discussion and Conclusion – Spatially Enabled EE Modelling

This chapter summarises and discusses the results of this study. Firstly, it highlights the SEEM model characteristics and identifies caveats and limitations of the model. Secondly, it provides recommendations and future directions for the study.

10.1 SEEM Model Characteristics

10.1.1 Flexibility and generalisability

The SEEM model is flexible and generalisable. It can facilitate specific locations, conditions, and stakeholder survey inputs. The user may adopt any value typology or decision factors list appropriate to the study area and research objectives. Weights assigned in the model to the three sub-functions can be modified and redefined. In the case study reported here (Maryland Farm), the Analytical Hierarchy Process (AHP) was selected as the method for calculating weights, however the user may choose any sound method of assigning weights to the three sub-functions. The type and number of farmers and other stakeholders are flexible; the current implementation of the model supports between 1 and 5 types. The categorisation of farmers and other stakeholders can be based on predetermined criteria or through a cluster analysis of surveyed results. The attributes required to populate the model with a shape file are default set to land use, land capability, and dominant soil; however these attributes can be replaced with for example parcel boundaries or any other relevant land characteristics. The model therefore provides flexibility in allocation of attributes from shape files. The model and its implementation are not hardwired, but provide considerable flexibility.

10.1.2 Scalability

The model is capable of functioning at the farm and landscape level, for which it was built, but it could also be applied at larger scales (over larger regions) wherever a unique identification of polygons of interest can be made. Shape files may be assigned unique identifiers and used as an input under land use. The model is readily available for use at the farm and landscape level, but may also be applicable for regional level modelling with appropriate modifications to the dataset.

10.1.3 Public Participation

The model facilitates public participation through a semi-structured questionnaire and interview process, with input from stakeholders objectively catalogued, incorporated into, and communicated by the model and its output visualisation tools.

10.1.4 Integration of Policy Incentives

The model has the capacity to integrate current or proposed policy incentives as a contributing component to F_{econ}.: payments or other financial incentives obtained through adoption of a policy can be easily incorporated when calculating F_{econ}. The SEEM model provides a powerful way to explore the potential impact of policy interventions as well as change in values for different land uses. The model captures other stakeholder values (other than farmers), providing an insight into the values of the community, and thus, allowing future alternative land use pathways to be openly and objectively discussed.

10.1.5 Independent Application - Open Source – Freely Accessible

The SEEM model is a stand-alone application that is built in python. It is built to be made available as open source and will be made freely accessible for research and academic purposes. The current implementation does have one commercial requirement: ArcPy is a Python site package that provides an effective way to perform geographic data analysis and management with Python. ArcPy is a licensed package and requires prior installation of ArcGIS software. Open source alternatives to ArcPy, such as the Geospatial Data Abstraction Library (GDAL/OGR) could be used in place of ArcPy, with appropriate modifications to the SEEM model script.

10.1.6 Time Requirements

The time required to collect and prepare the data prior to running the SEEM model is moderate to high, depending particularly upon the time consumed by the stakeholders' survey. The time for SEEM model execution – that is, the time required to populate the model with the prepared survey data and execute the model – might range from low to moderate. This is mainly because of those components of the data that are manually entered. The time spent on data population depends upon the number and types of stakeholders involved in the survey. Data population could be automated in the future, which would lead to time savings.

10.2 Model Caveats and Limitations

The main limitation of the current implementation is the time taken to populate the model, particularly in terms of the stakeholder survey and the entry of that data; the model currently requires the user to manually enter data on values for each stakeholder and for each land use. However, this limitation could be addressed with some additional coding that allowed the data to be extracted from a spreadsheet or database.

For data entry into the value typology interface and the decision factors interface, there is an acceptable range that the model accepts. For individual values, the minimum and maximum range is from 0 to 20, whereas, for decision factors, the minimum and maximum range is from -10 to +10. Therefore, users wanting to utilise their own value typology and decision factors list and associated surveys need to be compliant with these minimum and maximum values.

10.3 Findings and Discussion

The findings from this study indicate that spatial representation (visualisation) of the model farm and potential change pathways assisted in stakeholders' understanding of the spatial characteristics of the farm and the associated values attached to management decisions. The values obtained from the stakeholder survey demonstrate that spatial aspects of the farm significantly influence the values held by farmers and other stakeholders and thus these values cannot be isolated from their geospatial context. The integration of spatiality into the model and valuation process significantly assisted the collection of values and the views of stakeholders. Moreover, the integration of spatial file formats (shape files) into the SEEM captured the spatial characteristics of the farmland parcels (polygons) and thus assisted in further model computations. Spatiality is an integral part of such problems that should not be ignored. This study has tested the capacity to integrate geospatial context into a framework of ecological economic modelling for land use change decision-making and has practically implemented the model on a model farm, in this case situated in the midlands region of Tasmania, Australia.

The SEEM has unique characteristics when compared to other existing models. The SEEM model provides flexible non-optimised output. Its flexibility is evident by the weights assigned to various parameters in a logistic regression model that can be altered interactively. The results obtained from the SEEM are therefore different (flexible and non-optimised) compared with modelling strategies that provide optimised (one potential) output (e.g. agent based modelling) and models that use such strategies (e.g. Envision (Bolte *et al.* 2007), ARIES (Bagstad *et al.* 2013, Jackson *et al.* 2013, Häyhä and Franzese 2014) and the logistic regression model in Daloğlu *et al.* (2014)).

The SEEM requires and supports engagement with a variety of stakeholders and a variety of values, and integrates non-monetary ecosystem values. Therefore, SEEM is a modelling tool unlike ARIES (Bagstad *et al.* 2013, Jackson *et al.* 2013, Häyhä and Franzese 2014) and MIMES (Andrade *et al.* 2010, Liu *et al.* 2010, Jackson *et al.* 2013, Boumans *et al.* 2015), which are not designed to be used with stakeholder engagement and/or do not integrate non-monetary values.

The purpose of this research was to explore and progress the integration of spatial explicitness into ecological economic modelling of land use decisions. The SEEM integrates the spatial dimension into the ecological and economic values of diverse stakeholders to inform and support land use change decision making. Currently, there are a number of models and strategies prevalent in the literature that either deal with one or another aspect of ecological values, but do not provide an integrated approach to modelling spatially explicit ecological and economic values and then utilising those values as parameters for decision making. The SEEM strategy is different from e.g. the NRM Cradle Coast's strategy that identifies and maps ecosystem services (Williams 2009, 2013), where the economic aspect of land uses is not a significant part of the study. Similarly, Scenario Chooser (Smith *et al.* 2012) is different from SEEM, as the former is a web based tool developed to support public assessment of scenarios and potential future forest landscapes, but does not deal with the diverse values of stakeholders, or the economic aspect of land uses. Similarly, models such as CHIA (Pert *et al.* 2013), REM (Knight and Cullen 2010), MCAS-S (Magierowski *et al.* 2014, The Landscapes and Policy Hub 2015), SRIAS (Mallawaarachchi *et al.* 1996), ASSESS (Hill, Braaten, *et al.* 2005, Chen and Wu 2009), AHP-CA-GIS tool (Chen and Wu 2009), SoLVES (van Riper *et al.* 2012), Landscape Toolkit (Bohnet *et al.* 2011), a hydro-ecological model based on Bayesian Network model (Kragt *et al.* 2011), EcoDynamo (Liacc 2007), LUCI Polyscape (Bagstad *et al.* 2013, Jackson *et al.* 2013), Ecosystem Valuation Toolkit (Earth Economics 2016), EcoServ (Bagstad *et al.* 2012, Bagstad *et al.* 2013) and Co$ting Nature

(Silvestri *et al.* 2010, Policy Support Systems 2016), all serve a purpose, but differ from SEEM as they lack either geospatial integration of ecological and economic values , or do not engage diverse stakeholders and/or integrate their values into the modelling strategy.

The InVest model can be used for constructing spatially explicit scenarios through a rule-based approach (McKenzie *et al.* 2012); however, it does not allow trade-offs of values between and within diverse stakeholders and land uses to be viewed that the SEEM can capture. Similarly Butler *et al.* (2013) use indicators and indices to value ecosystem services and to identify cause-effect relationships, trade-offs and thresholds between services and stakeholders, but do not capture and communicate the trade-offs of values held by diverse stakeholders and assigned to different land uses in the manner that the SEEM allows.

The approach developed by Daloğlu *et al.* (2014) differs from SEEM, although SEEM is adapted and developed from their approach. Daloğlu *et al.* (2014) integrate Agent Based Modelling (ABM) and a Soil and Water Assessment Tool (SWAT) to integrate land management decisions with soil properties, climate information, and land topography, and with a logistic regression model used to define the behavioural model that guided the farmer agent's decision-making, thus producing optimised (one potential) outputs. However, SEEM derives non-optimised and flexible outputs, and does not utilise an agent based modelling approach. SEEM has a distinctive strategy, approach and findings when compared to other approaches described in the current literature.

Numerous research projects addressing landscape management have been carried out in Tasmania and, specifically, in the Midlands region. Many of these were spatially explicit in nature; i.e. they reflected upon the arrangement of these natural resources in space and locality, considering either spatial boundaries, spatial adjacency/proximity, or spatially associated attributes. A small number of these have focussed explicitly on modelling tools in an integrated environment combining the ecological dynamism, or the economic aspects and/or social values (e.g. Cradle Coast NRM's strategy (Williams 2009, 2013), Scenario Chooser (Smith *et al.* 2012), REM (Knight and Cullen 2010), and MCAS-S (Hill, Lesslie, *et al.* 2005, Lesslie *et al.* 2008)), although none have focused on a geospatially integrated approach that combines ecological values and the economic aspects of land use change decisions, and in a manner that can promote dialogue and spatial reasoning in a decision making setting. This research contributes to the development of such a model, that integrates spatially explicit ecological and economic values and with stakeholder engagement, and that has capacity to be utilised in a decision making process, promoting dialogue, spatial reasoning, building consensus, resolving conflicts and arriving at mutually acceptable decisions. Although, SEEM has been developed for a case study in Tasmania, it is readily transferable to other locations.

The SEEM is designed to be employed within land use policy formulation and implementation processes. It allows values associated with alternative trade-offs between different land uses or incentives associated with policy-based interventions to be captured and valued. The SEEM can be used to describe and to communicate complex and varying values held by diverse stakeholders and assigned to different land uses, and so allow insightful discussions about the relationships between values for desired outcomes.

The model includes F_{econ} as a sub-function of the logistic regression model, allowing a wide range of parameters to be included and used to elaborate the economic aspects of changing land use, in turn allowing possible structural and policy interventions to be dynamically modelled as part of a consultative process. In addition, gross margin values utilised in the model can incorporate the currency fluctuations, allowing for accurate calculations of costs and returns.

Effective agro-ecological policies depend in large part on the values of famers, who play a major role in shaping rural and agricultural landscapes. The SEEM provides a platform that has the capacity to integrate farmers' and other stakeholders' values in land use change scenarios, and thus be used to develop better policies that are framed more effectively to support and reflect farmers values and decisions. SEEM is a relatively simple model, based on a logit algorithm, that provides a thinking space for informed and critical questioning and discussion, and that in turn can assist experts to understand complex decision making environments, to develop pathways to land use change that are desirable and sustainable, and to communicate the rationale for land use policy decisions.

This research has pursued an integrative multi-disciplinary approach to fostering human-environmental inquiry. The work further extends the field of GIS beyond traditional spatial data management, modelling and mapping to focus on supporting spatial reasoning and dialogue. Shared understanding, effective communication, and the inclusion and involvement of all interested parties throughout a decision making process helps to lead technically feasible options towards socially acceptable solutions (Parrott 2017). Models such as the SEEM and active engagement from stakeholders can lead to group consensus. Or when consensus cannot be reached, models such as the SEEM help to ensure that differences are at least objectively understood and transparently documented.

This study contributes to the development of models that can be used in complex decision-making settings and that have capacity to be catalysts for, and to inform and support, dialogue between interested parties, supported by spatial reasoning, and providing for socially, environmentally and economically acceptable solutions.

10.4 Conclusion and Future Directions

The overarching research question framed in Chapter 1 was**: How can geospatial capabilities be integrated into ecological economic modelling and applied to land use decisions in agricultural landscapes?** The research objectives have been to:

- review existing methods of integrated ecological economic models, focussing on spatially enabled integrated models and their strategies
- develop a spatially enabled ecological economic model that is capable of modelling land use decisions made by farmers

- implement and assess the model using a case study drawn from the Midlands region of Tasmania, Australia.

The work undertaken to achieve this aim and these objectives has been described in this thesis. A model has been constructed and implemented that allows economic and ecological values to be captured from a variety of stakeholders, in a manner that allows the spatial context of land use management to be explicitly included, geospatial data to be incorporated, and possible land use pathways to be explored.

The following is a short list of possible future development pathways for the SEEM model:

- Existing models deal either with biophysical quantification of an ecosystem to provide decision support (e.g. the InVEST model) or deal with the social values of an ecosystem (e.g. the SolVES model). The SEEM model integrates the values of diverse stakeholders and the economics of the land that plays a vital role in forming decisions and shaping landscapes. The SEEM model does not integrate biophysical measurements of the ecosystem itself to capture the biophysical properties of the ago-ecological lands, although, this could further enhance its capabilities as a decision support tool, and can be regarded as a future research challenge.
- The SEEM model currently is a stand-alone application that requires an ArcPy license. To reduce the dependency on an ArcPy license, the model could also be developed using the R programming language and so converting the model into an entirely open source language through the freely accessible R package. R can provide useful functionality, such as sophisticated graphical plotting and thus can assist in representing value-added graphs. Although the model in this case would lose its graphical user interface, a conversion to R could provide efficiency in data handling and output generation. Alternatively, the SEEM model could be integrated into the ArcGIS environment as an extension, although this would require substantial development and testing. Alternatively the model could be embedded as a custom python script into a toolbox in ArcMap or ArcGIS Pro, and thus by invoking the PyQt script the tool could be setup and run in ArcGIS.
- The model currently involves time intensive data entry of values. This could be resolved by directly capturing the values from an external Microsoft Excel file (or some other e.g. database file). This would increase the efficiency of the model, but would require further modifications and development of the model script. The model could be further equipped with a plug-and-play capability, allowing the user to change the weights assigned to each sub-function of the algorithm, thus for example changing the F_{decide} of all the land uses in any scenarios. This would facilitate the examination of trade-offs between the sub-functions. For example, it would allow alteration of the weights of sub-functions in order to observe consequential impacts on F_{decide} values.
- This research has concentrated on the design, development and calibration of a spatially enabled model and a demonstration of how values can be collected and input to that model, and how the outputs can be visualised and communicated. An extension of this work will be to explore stakeholders' responses to the

collection of data, to the opportunities to interact with the outputs of the model, and to decisions flowing from the modelling process, together with an assessment of the influence of the model on a decision making process and outcome. This is outside the scope of the research undertaken for this thesis, and is likely to benefit from use of a real land management scenario rather than a model farm.

This study and the SEEM model, contribute towards development of geospatially enabled tools that have the capacity to assist in decision-making and policy formulations at farm and landscape scales.

11 References

Abdel-Rahman, E.M., Ahmed, F.B., Ismail, R., Goncalves, J.F., and Correia, K.D., 2012. Random forest regression for sugarcane yield prediction in Umfolozi, South Africa based on Landsat TM and ETM+ derived spectral vegetation indices. *Sugarcane: Production, Cultivation and Uses*, 257–284.

Abila, R., 1998. Utilisation and economic valuation of the Yala swamp wetland. *In*: *Strategies for wise use of wetlands-Best practices in participatory management. Proceedings of the 21st International conference on wetlands and development*.

Aboelela, S.W., Larson, E., Bakken, S., Carrasquillo, O., Formicola, A., Glied, S.A., Haas, J., and Gebbie, K.M., 2007. Defining interdisciplinary research: Conclusions from a critical review of the literature. *Health services research*, 42 (1p1), 329–346.

Achmad, A., Hasyim, S., Dahlan, B., and Aulia, D.N., 2015. Modeling of urban growth in tsunami-prone city using logistic regression: Analysis of Banda Aceh, Indonesia. *Applied Geography*, 62, 237–246.

Adam, E., Mutanga, O., Abdel-Rahman, E.M., and Ismail, R., 2014. Estimating standing biomass in papyrus (Cyperus papyrus L.) swamp: exploratory of in situ hyperspectral indices and random forest regression. *International Journal of Remote Sensing*, 35 (2), 693–714.

Adam, E.M., Mutanga, O., Rugege, D., and Ismail, R., 2012. Discriminating the papyrus vegetation (Cyperus papyrus L.) and its co-existent species using random forest and hyperspectral data resampled to HYMAP. *International Journal of Remote Sensing*, 33 (2), 552–569.

Aerni, P., Rae, A., and Lehmann, B., 2009. Nostalgia versus Pragmatism? How attitudes and interests shape the term sustainable agriculture in Switzerland and New Zealand. *Food Policy*, 34 (2), 227–235.

Akgun, A., 2012. A comparison of landslide susceptibility maps produced by logistic regression, multi-criteria decision, and likelihood ratio methods: a case study at Izmir, Turkey. *Landslides*, 9 (1), 93–106.

Alberti, M., Marzluff, J.M., Shulenberger, E., Bradley, G., Ryan, C., and Zumbrunnen, C., 2003. Integrating Humans into Ecology: Opportunities and Challenges for Studying Urban Ecosystems. *BioScience* , 53 (12), 1169–1179.

Alessa, L. (Naia), Kliskey, A. (Anaru), and Brown, G., 2008a. Social–ecological hotspots mapping: A spatial approach for identifying coupled social–ecological space. *Landscape and Urban Planning*, 85 (1), 27–39.

Alessa, L. (Naia), Kliskey, A. (Anaru), and Brown, G., 2008b. Social–ecological hotspots mapping: A spatial approach for identifying coupled social–ecological space. *Landscape and Urban Planning*, 85 (1), 27–39.

Althuwaynee, O.F., Pradhan, B., Park, H.-J., and Lee, J.H., 2014. A novel ensemble decision tree-based CHi-squared Automatic Interaction Detection (CHAID) and multivariate logistic regression models in landslide susceptibility mapping. *Landslides*, 11 (6), 1063–1078.

Amut, A., Gong, L., Yuan, Z., Crovello, T., and Gao, Z., 2006. Estimation of the ecological degeneration from changes in land use and land covers in the upper reaches of the Tarim River. *In*: W. Gao and S.L. Ustin, eds. *Proceedings of SPIE - Remote Sensing and Modelling of Ecosystems for Sustainability III*.

An, L. and López-Carr, D., 2012. Understanding human decisions in coupled natural and human systems. *Ecological Modelling*, 229, 1–4.

Anderson, D.A., 2010. *Environmental economic and natural resource management*. 3rd Editio. Routledge.

Anderson, N.M., Williams, K.J.H., and Ford, R.M., 2013. Community perceptions of plantation forestry: The association between place meanings and social representations of a contentious rural land use. *Journal of Environmental Psychology*, 34, 121–136.

Anderson, S.T. and West, S.E., 2006. Open space, residential property values, and spatial context. *Regional Science and Urban Economics*, 36 (6), 773–789.

Andrade, D.C., Romeiro, A.R., Boumans, R., Sobrinho, R.P., and Tôsto, S.G., 2010. New methodological perspectives on the valuation of ecosystem services: Toward a dynamic-integrated valuation approach. *In*: *International Society for Ecological Economics Conference: Advancing Sustainability in Time of Crisis, Oldenburg and Bremen, Germany*.

Austin, E.J., Deary, I.J., Gibson, G.J., McGregor, M.J., and Dent, J.B., 1996. Attitudes and values of Scottish farmers: Yeoman and Entrepreneur as factors, not distinct types. *Rural Sociology*, 61 (3), 464–474.

Austin, P.C. and Tu, J. V, 2004. Bootstrap methods for developing predictive models. *The American Statistician*, 58 (2), 131–137.

Australian Bureau of Statistics, 2002a. Tasmanian Pocket Year Book, 2002 [online]. Available from: http://www.abs.gov.au/ausstats/abs@.nsf/cat/1302.6.

Australian Bureau of Statistics, 2002b. Agriculture and Soils in Tasmania [online]. Statistics - Tasmania. Available from: http://www.abs.gov.au/AUSSTATS/abs@.nsf /featurearticlesbytitle/1CDD1718C0BA635DCA256C32002478A?OpenDocument.

Australian Bureau of Statistics, 2011. Tasmania at a Glance [online]. Available from: http://www.abs.gov.au/AUSSTATS/abs@.nsf/DetailsPage/1305.62011?OpenDocument.

Australian Bureau of Statistics, 2016. Australian Demographic Statistics [online]. Available from: http://www.abs.gov.au/ausstats/abs@.nsf/mf/3101.0.

Australian Government, 2010. Land management practices [online]. Department of Agriculture and Water Resources ABARES. Available from:

http://www.agriculture.gov.au/abares/aclump/land-management-practices.

Australian Government, 2012a. Australian Bioregion Framework [online]. *Department of the Environment*. Available from: https://www.environment.gov.au/land/nrs/science/ibra/australias-bioregion-framework [Accessed 20 Jan 2016].

Australian Government, 2012b. The National Reserve System (NRS) - Australia's bioregions (IBRA) [online]. *Department of the Environment*. Available from: https://www.environment.gov.au/land/nrs/science/ibra [Accessed 29 Jan 2016].

Australian Government, 2016. Area of Australia - States and Territories [online]. Geoscience Australia. Available from: http://www.ga.gov.au/scientific-topics/national-location-information/dimensions/area-of-australia-states-and-territories.

Au-Yeung, B., Yigitcanlar, T., and Mayere, S., 2009. Brisbane Urban Growth Model: Integrated Sustainable Urban and Infrastructure Management in Brisbane. *In*: *Infrastructure Research Theme Postgraduate Student Conference*. Queensland University of Technology, Brisbane, 1 – 12.

Aznar, J., Guijarro, F., and Moreno-Jiménez, J., 2011. Mixed valuation methods: a combined AHP-GP procedure for individual and group multicriteria agricultural valuation. *Annals of Operations Research*, 190 (1), 221–238.

Bagstad, K., Semmens, D., and Winthrop, R., 2013. Comparing approaches to spatially explicit ecosystem service modeling: A case study from the San Pedro River, Arizona. *Ecosystem Services*, 5, 40–50.

Bagstad, K.J., Semmens, D., Winthrop, R., Jaworski, D., and Larson, J., 2012. Ecosystem services valuation to support decision-making on public lands: A case study for the San Pedro River, Arizona. *USGS Scientific Investigations Report*, 5251.

Bagstad, K.J., Semmens, D.J., Waage, S., and Winthrop, R., 2013. A comparative assessment of decision-support tools for ecosystem services quantification and valuation. *Ecosystem Services*, 5, 27–39.

Bagstad, K.J., Semmens, D.J., Winthrop, R., Jaworksi, D., and Larson, J., 2012. *Ecosystem Services Valuation to Support Decision Making on Public Lands - A Case Study of the San Pedro River Watershed, Arizona*. Scientific Investigation Reports. US Geological Survey.

Bao, Y., Wu, W., Wang, M., and Liu, W., 2007. Disadvantages and future research directions in valuation of ecosystem services in China. *International Journal of Sustainable Development and World Ecology*.

Baral, H., Keenan, R.J., Sharma, S.K., Stork, N.E., and Kasel, S., 2014. Spatial assessment and mapping of biodiversity and conservation priorities in a heavily modified and fragmented production landscape in north-central Victoria, Australia. *Ecological Indicators*, 36, 552–562.

Baral, H., Keenan, R.J., Stork, N.E., and Kasel, S., 2013. Measuring and managing ecosystem goods and services in changing landscapes: a south-east Australian perspective. *Journal of Environmental Planning and Management*, 57 (7), 961–983.

Barbier, E.B., Adams, W.M., Kimmage, K., and others, 1991. Economic valuation of wetland benefits: the Hadejia-Jama'are floodplain, Nigeria. *LEEC Paper-IIED/UCL London Environmental Economics Centre*, (91-02).

Barry, J. and Proops, J., 1999. Seeking sustainability discourses with Q methodology. *Ecological economics*, 28 (3), 337–345.

Barton, D.N., 1999. The quick, the cheap and the dirty: Benefit transfer approaches to the non-market valuation of coastal water quality in Costa Rica. Citeseer.

Bastian, C.T., McLeod, D.M., Germino, M.J., Reiners, W.A., and Blasko, B.J., 2002. Environmental amenities and agricultural land values: a hedonic model using geographic information systems data. *Ecological Economics*, 40 (3), 337–349.

Bastian, O., Grunewald, K., and Khoroshev, A. V, 2015. The significance of geosystem and landscape concepts for the assessment of ecosystem services: exemplified in a case study in Russia. *Landscape Ecology*, 30 (7), 1145–1164.

Bateman, I.J., Harwood, A.R., Abson, D.J., Andrews, B., Crowe, A., Dugdale, S., Fezzi, C., Foden, J., Hadley, D., Haines-Young, R., Hulme, M., Kontoleon, A., Munday, P., Pascual, U., Paterson, J., Perino, G., Sen, A., Siriwardena, G., and Termansen, M., 2014. Economic Analysis for the UK National Ecosystem Assessment: Synthesis and Scenario Valuation of Changes in Ecosystem Services. *Environmental and Resource Economics*, 57, 273–297.

Bateman, I.J., Jones, A.P., Lovett, A.A., Lake, I.R., and Day, B.H., 2002. Applying geographical information systems (GIS) to environmental and resource economics. *Environmental and Resource Economics*, 22 (1-2), 219–269.

Bateman, I.J., Mace, G.M., Fezzi, C., Atkinson, G., and Turner, K., 2011. Economic analysis for ecosystem service assessments. *Environmental and Resource Economics*, 48 (2), 177–218.

Baumgartner, S., Becker, C., Frank, K., Muller, B., and Quaas, M., 2008. Relating the philosophy and practice of ecological economics: The role of concepts, models, and case studies in inter- and transdisciplinary sustainability research. *Ecological Economics*, 67 (3), 384–393.

Baumgärtner, S., Becker, C., Frank, K., Müller, B., and Quaas, M., 2008. Relating the philosophy and practice of ecological economics: The role of concepts, models, and case studies in inter-and transdisciplinary sustainability research. *Ecological Economics*, 67 (3), 384–393.

Baveye, P.C., Baveye, J., and Gowdy, J., 2013. Monetary valuation of ecosystem services: It matters to get the timeline right. *Ecological Economics*, 95, 231–235.

Beder, S., 2011. Environmental economics and ecological economics: the contribution of interdisciplinarity to understanding, influence and effectiveness. *Environmental conservation*, 38 (02), 140–150.

Beggs, S., Cardell, S., and Hausman, J., 1981. Assessing the potential demand for electric cars. *Journal of econometrics*, 17 (1), 1–19.

Beratan, K.K., 2007. A cognition-based view of decision processes in complex social--ecological systems. *Ecology and Society*, 12 (1), 27.

Bergez, J.-E., Colbach, N., Crespo, O., Garcia, F., Jeuffroy, M.-H., Justes, E., Loyce, C., Munier-Jolain, N., and Sadok, W., 2010. Designing crop management systems by simulation. *European Journal of Agronomy*, 32 (1), 3–9.

van den Bergh, J.C., 2001. Ecological economics: themes, approaches, and differences with environmental economics. *Regional Environmental Change*, 2 (1), 13–23.

Bergh, J.C.J.M. van den, 1996. *Ecological Economics and Sustainable Development - Theory, methods ad applications*. Edward Elgar Publishing Limited.

Bergh, J.C.J.M. van den and Nijkamp, P., 1991. Operationalizing sustainable development: dynamic ecological economic models. *Ecological Economics*, 4 (1), 11–33.

Bergstrom, J.C., Stoll, J.R., Titre, J.P., and Wright, V.L., 1990. Economic value of wetlands-based recreation. *Ecological economics*, 2 (2), 129–147.

Blamey, R., Rolfe, J., Bennett, J., and Morrison, M., 2000. Valuing Remnant Vegetation in Central Queensland Using Choice Modelling. *Australian Journal of Agricultural and Resource Economics*, 44 3, 439–456.

Blamey, R.K. and James, R., 1999. Citizens' juries: An alternative or an input to environmental cost-benefit analysis.

Bockstael, N., Costanza, R., Strand, I., Boynton, W., Bell, K., and Wainger, L., 1995. Ecological economic modeling and valuation of ecosystems. *Ecological Economics*, 14 (2), 143–159.

Bockstael, N.E., Freeman, A.M., Kopp, R.J., Portney, P.R., and Smith, V.K., 2000. On measuring economic values for nature. *Environmental Science & Technology*, 34 (8), 1384–1389.

Bohnet, I.C., Roebeling, P.C., Williams, K.J., Holzworth, D., Grieken, M.E., Pert, P.L., Kroon, F.J., Westcott, D. a., and Brodie, J., 2011. Landscapes Toolkit: an integrated modelling framework to assist stakeholders in exploring options for sustainable landscape development. *Landscape Ecology*, 26 (8), 1179–1198.

Boithias, L., Acuña, V., Vergoñós, L., Ziv, G., Marcé, R., and Sabater, S., 2014. Assessment of the water supply:demand ratios in a Mediterranean basin under different global change scenarios and mitigation alternatives. *The Science of the total environment*, 470-471, 567–77.

Bolte, J.P., Hulse, D.W., Gregory, S. V, and Smith, C., 2007. Modeling biocomplexity--actors, landscapes and alternative futures. *Environmental Modelling \& Software*, 22 (5), 570–579.

Boumans, R., Roman, J., Altman, I., and Kaufman, L., 2015. The Multiscale Integrated

Model of Ecosystem Services (MIMES): Simulating the interactions of coupled human and natural systems. *Ecosystem Services*, 12, 30–41.

Boyd, J. and Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics*, 63 (2–3), 616–626.

Boyle, K.J., Kuminoff, N. V, Parmeter, C.F., and Pope, J.C., 2010. The benefit-transfer challenges. *Annu. Rev. Resour. Econ.*, 2 (1), 161–182.

Breiman, L., 2001. Random forests. *Machine learning*, 45 (1), 5–32.

Brisaboa, N.R., Luaces, M.R., Andrea Rodríguez, M., and Seco, D., 2014. An inconsistency measure of spatial data sets with respect to topological constraints. *International Journal of Geographical Information Science*, 28 (1), 56–82.

Brodnig, G. and Mayer-Schonberger, V., 2000. Bridging the gap: the role of spatial information technologies in the integration of traditional environmental knowledge and western science. *The Electronic Journal of Information Systems in Developing Countries*, 1.

Brondízio, E.S., Gatzweiler, F.W., Zografos, C., Kumar, M., Jianchu, X., McNeely, J., Kadekodi, G.K., and Martinez-Alier, J., 2010. *Socio-cultural context of ecosystem and biodiversity valuation*. The Economics of Ecosystems and Biodiversity. The Economics of Ecosystems and Biodiversity.

Brown, G. and Kyttä, M., 2014. Key issues and research priorities for public participation GIS (PPGIS): A synthesis based on empirical research. *Applied Geography*, 46, 122–136.

Brown, G.M. and Hammack, J., 1973. Dynamic economic management of migratory waterfowl. *The Review of Economics and Statistics*, 73–82.

Brown, M.J., 2010. *Landscape scale conservation planning in Tasmania the Spatial identification of contemporary refugia*. NRM South, Tasmania.

BSR, 2011. New Business Decision-Making Aids in an Era of Complexity, Scrutiny, and Uncertainty. *Business*, (May), 1–40.

Buenemann, M., Martius, C., Jones, J.W., Herrmann, S.M., Klein, D., Mulligan, M., Reed, M.S., Winslow, M., Washington-Allen, R.A., Lal, R., and Ojima, D., 2011. Integrative geospatial approaches for the comprehensive monitoring and assessment of land management sustainability: Rationale, Potentials, and Characteristics. *Land Degradation Development*, 22 (2), 226–239.

Bureau of Meteorology, 2013. *Guide to environmental accounting in Australia*. Series 3. Canberra, Australia: Bureau of Meteorology.

Burrough, P.A. and Frank, A.U., 1995. Concepts and paradigms in spatial information: are current geographical information systems truly generic? *International Journal of Geographical Information Systems*, 9 (2), 101–116.

Bush Heritage, 2008. *Midlands of Tasmania anchor region*. Annual Conservation Report

Bush Heritage in Action.

Butler, J.R.A., Wong, G.Y., Metcalfe, D.J., Honzák, M., Pert, P.L., Rao, N., van Grieken, M.E., Lawson, T., Bruce, C., Kroon, F.J., and Brodie, J.E., 2013. An analysis of trade-offs between multiple ecosystem services and stakeholders linked to land use and water quality management in the Great Barrier Reef, Australia. *Agriculture, Ecosystems & Environment*, 180, 176 – 191.

Campiglio, E., 2011. Ecological vs. Environmental Economics [online]. *The NEF blog*. Available from: http://www.neweconomics.org/blog/entry/ecological-vs.-environmental-economics [Accessed 11 Apr 2016].

Carlsson, F., Frykblom, P., and Liljenstolpe, C., 2003. Valuing wetland attributes: an application of choice experiments. *Ecological Economics*, 47 (1), 95–103.

Celio, E., Flint, C.G., Schoch, P., and Grêt-Regamey, A., 2014. Farmers' perception of their decision-making in relation to policy schemes: A comparison of case studies from Switzerland and the United States. *Land Use Policy*, 41, 163–171.

Cernusca, A., Bahn, M., Chemini, C., Graber, W., Siegwolf, R., Tappeiner, U., and Tenhunen, J., 1998. ECOMONT: a combined approach of field measurements and process-based modelling for assessing effects of land-use changes in mountain landscapes. *Ecological Modelling*, 113 (1-3), 167–178.

Chan, K.M.A., Shaw, M.R., Cameron, D.R., Underwood, E.C., and Daily, G.C., 2006. Conservation planning for ecosystem services. *PLoS biology*, 4 (11), e379.

Chen, F., Kusaka, H., Bornstein, R., Ching, J., Grimmond, C.S.B., Grossman-Clarke, S., Loridan, T., Manning, K.W., Martilli, A., Miao, S., Sailor, D., Salamanca, F.P., Taha, H., Tewari, M., Wang, X., Wyszogrodzki, A. a., and Zhang, C., 2011. The integrated WRF/urban modelling system: development, evaluation, and applications to urban environmental problems. *International Journal of Climatology*, 31 (2), 273–288.

Chen, N., Li, H., and Wang, L., 2009. A GIS-based approach for mapping direct use value of ecosystem services at a county scale: Management implications. *Ecological Economics*, 68 (11), 2768–2776.

Chen, Y. and Wu, J.P., 2009. Cellular automata and GIS based landuse suitability simulation for irrigated agriculture. *In*: *18th World IMACS / MODSIM Congress 13 - 17 July*. 3584–3590.

Clement, J.M. and Cheng, A.S., 2006. Public values and preferences regarding forest uses and management on the Pike and San Isabel National Forests, Colorado. *Survey results. Department of Forest, Rangeland and Watershed Stewardship, Colorado State University*.

Common, M. and Stagl, S., 2005. *Ecological Economics An Introduction*. Cambridge University Press.

Constantin, C., 2015. Using the Logistic Regression model in supporting decisions of establishing marketing strategies. *Bulletin of the Transilvania University of Brasov.*

Economic Sciences. Series V, 8 (2), 4.

Coote, A. and Lenaghan, J., 1997. *Citizens' juries: theory into practice*. Institute for Public Policy Research.

Cork, S., Gorrie, G., Ampt, P., Maynard, S., Rowland, P., Oliphant, R., Reeder, L., and Stephens, L., 2012. *Discussion paper on ecosystem services for the Department of Agriculture, Fisheries and Forestry*. Final Report. Australia 21 Limited.

Costanza, R., 1989. What is ecological economics? *Ecological Economics*, 1 (1), 1–7.

Costanza, R., 1991. *Ecological economics: the science and management of sustainability*. Seminars in Plastic Surgery. Columbia University Press.

Costanza, R., 1992. *Ecological economics: the science and management of sustainability*. Columbia University Press.

Costanza, R., 2001. Visions, values, valuation, and the need for an ecological economics. *BioScience*, 51 (6), 459–468.

Costanza, R., 2003. Ecological economics is post-autistic. *Post-Autistic Economics Review*, 20 (2), 1–3.

Costanza, R., 2006. Nature: ecosystems without commodifying them. *Nature*, 443 (7113), 749.

Costanza, R., 2008. Ecosystem services: multiple classification systems are needed. *Biological conservation*, 141 (2), 350–352.

Costanza, R., Cumberland, J.H., Daly, H., Goodland, R., Norgaard, R.B., Kubiszewski, I., and Franco, C., 2014. *An introduction to ecological economics*. CRC Press.

Costanza, R., D'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R. V, Paruelo, J., and others, 1997. The value of the world's ecosystem services and natural capital. *Nature*, 387 (1), 253–260.

Costanza, R., Daly, H.E., and Bartholomew, J.A., 1991. Goals, Agenda and Policy Recommendations for Ecological Economics. *In*: R. Costanza, ed. *Ecological economics The science and management of sustainability*. Columbia University Press, 525.

Costanza, R., Duplisea, D., and Kautsky, U., 1998. Ecological Modelling on modelling ecological and economic systems with STELLA. *Ecological Modelling*, 110 (1), 1–4.

Costanza, R., Norgaard, R., Daly, H., Goodland, R., and Cumberland, J.H., 2014. An introduction to ecological economics - Chapter 2. *In*: *An introduction to ecological economics*. CRC Press.

Costanza, R., Voinov, A., Boumans, R., Maxwell, T., Villa, F., Wainger, L., and Voinov, H., 2002. Integrated ecological economic modeling of the Patuxent River watershed, Maryland. *Ecological Monographs*, 72 (2), 203–231.

Couclelis, H., 1991. Requirements for planning-relevant GIS: a spatial perspective. *Papers in Regional Science*, 70 (1), 9–19.

Cox, D.R., 1958. The regression analysis of binary sequences. *Journal of the Royal Statistical Society. Series B (Methodological)*, 215–242.

Dai, E., Wu, S., Shi, W., Cheung, C., and Shaker, A., 2005. Modeling Change-Pattern-Value Dynamics on Land Use: An Integrated GIS and Artificial Neural Networks Approach. *Environmental Management*, 36 (4), 576–591.

Daily, G.C., 1997. *Nature's services: societal dependence on natural ecosystems*. Ecology. Island Press.

Daily, G.C., Alexander, S., Ehrlich, P.R., Goulder, L., Lubchenco, J., Matson, P.A., Mooney, H.A., Postel, S., Schneider, S.H., Tilman, D., and Woodwell, G.M., 1997. Ecosystem Services: Benefits Supplied to Human Societies by Natural Ecosystems. *Issues in Ecology*, (2).

Daily, G.C., Polasky, S., Goldstein, J., Kareiva, P.M., Mooney, H.A., Pejchar, L., Ricketts, T.H., Salzman, J., and Shallenberger, R., 2009. Ecosystem services in decision making: time to deliver. *Frontiers in Ecology and the Environment*, 7 (1), 21–28.

Dalheimer, M., 2002. *Programming with QT: Writing portable GUI applications on Unix and Win32*. ' O'Reilly Media, Inc.'

Daloğlu, I., Nassauer, J.I., Riolo, R.L., and Scavia, D., 2014. An integrated social and ecological modeling framework - Impacts of agricultural conservation practices on water quality. *Ecology and Society*, 19 (12).

Daly, H.E. and Farley, J., 2011. *Ecological Economics Principles and Applications*. 2nd ed. Island Press, Washington.

DEFRA, 2007. *An introductory guide to valuing ecosystem services*. Department for Environment, Food and Rural Affairs. Department for Environment, Food and Rural Affairs.

Defrancesco, E., Gatto, P., Runge, F., and Trestini, S., 2008. Factors affecting farmers participation in agri-environmental measures: A Northern Italian perspective. *Journal of agricultural economics*, 59 (1), 114–131.

Dhakal, A., Ooba, M., and Hayashi, K., 2014. Assessing impacts of forest conversion on terrestrial vertebrates combining forestry cost with HSI and InVEST: case of Toyota city, Japan. *International Journal of Biodiversity Science, Ecosystem Services & Management*, 10 (3), 198–215.

Diaz-Uriarte, R. and De Andres, S.A., 2006. Gene selection and classification of microarray data using random forest. *BMC bioinformatics*, 7 (1), 1.

Dixon, J.A., Sherman, P.B., and others, 1990. *Economics of protected areas: a new look at benefits and costs.* Island Press.

Dow, S.C., 2007. Variety of methodological approach in economics. *Journal of Economic Surveys*, 21 (3), 447–465.

DPIPWE, 1999. *Land Capability Classification Handbook*. Department of Primary

Industries, Water and Environment. Tasmania.

DPIPWE, 2015. Gross Margin Analysis Tool [online]. *Farm Business Planning Tool*. Available from: http://dpipwe.tas.gov.au/agriculture/investing-in-irrigation/farm-business-planning-tools#GrossMarginAnalysisTools.

Drechsler, M., Wätzold, F., Johst, K., Bergmann, H., and Settele, J., 2007. A model-based approach for designing cost-effective compensation payments for conservation of endangered species in real landscapes. *Biological Conservation*, 140 (1–2), 174–186.

Dugdale, S., 2010. *Habitat association modelling for farmland birds*. Report to the Economics Team of the UK National Ecosystem Assessment, CSERGE, University of East Anglia. Citeseer.

Dury, J., Garcia, F., Reynaud, A., and Bergez, J.-E., 2013. Cropping-plan decision-making on irrigated crop farms: A spatio-temporal analysis. *European Journal of Agronomy*, 50, 1–10.

Earth Economics, 2016. Ecoystem Valuation Toolkit [online]. *Earth Economics*. Available from: http://esvaluation.org/.

Edwards-Jones, G., 2006. Modelling farmer decision-making: concepts, progress and challenges. *Animal science*, 82 (06), 783–790.

Egoh, B., Dunbar, M.B., Maes, J., Willemen, L., and Drakou, E.G., 2012. *Indicators for mapping ecosystem services: a review*. European Commission, Joint Research Centre, Institute for Environment and Sustainability.

Egoh, B., Reyers, B., Rouget, M., Richardson, D.M., Le Maitre, D.C., and van Jaarsveld, A.S., 2008. Mapping ecosystem services for planning and management. *Agriculture, Ecosystems & Environment*, 127 (1), 135–140.

Ehrlich, P.R. and Ehrlich, A.H., 1981. *Extinction: the causes and consequences of the disappearance of species*. Gollancz London.

Eichner, T. and Pethig, R., 2006. Economic land use, ecosystem services and microfounded species dynamics. *Journal of environmental economics and management*, 52 (3), 707–720.

Elsawah, S., Guillaume, J.H.A., Filatova, T., Rook, J., and Jakeman, A.J., 2015. A methodology for eliciting, representing, and analysing stakeholder knowledge for decision making on complex socio-ecological systems: From cognitive maps to agent-based models. *Journal of environmental management*, 151, 500–516.

Emerton, L., 1996. Valuing the subsistence use of forest products in Oldonyo Orok forest, Kenya. *Rural Development Forestry Network Paper*, 19, 21–30.

Emerton, L., 2005. *Values and rewards: counting and capturing ecosystem water services for sustainable development*. IUCN.

English, M.R., 1999. Environmental decision making by organizations: Choosing the right tools. *In*: K. Sexton, A.A. Marcus, W. Easter, and T.D. Burkhardt, eds. *Better environmental decisions: strategies for governments, businesses, and communities*. Island Press, 57 – 67.

ESRI, 2002. *COTS GIS: The value of a commercial geographic information system*. White Paper. ESRI.

ESRI, 2015. *Deprecation Plan for ArcGIS 10.3 and ArcGIS 10.2.x*. ESRI.

Everingham, Y.L., Smyth, C.W., and Inman-Bamber, N.G., 2009. Ensemble data mining approaches to forecast regional sugarcane crop production. *Agricultural and forest meteorology*, 149 (3), 689–696.

Faber, M., 2008. How to be an ecological economist. *Ecological Economics*, 66 (1), 1–7.

Farber, S.C., Costanza, R., and Wilson, M.A., 2002. Economic and ecological concepts for valuing ecosystem services. *Ecological Economics*, 41 (3), 375–392.

Fisher, B., Christie, M., de Groot, R., Aronson, J., Braat, L., Gowdy, J., Haines-Young, R., Maltby, E., Neuville, A., and Others, 2010. *Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation*. The Economics of Ecosystems and Biodiversity. The Economics of Ecosystems and Biodiversity.

Fisher, B. and Turner, R.K., 2008. Ecosystem services: Classification for valuation. *Biological Conservation*, 141 (5), 1167–1169.

Foley, N.S., van Rensburg, T.M., and Armstrong, C.W., 2010. The ecological and economic value of cold-water coral ecosystems. *Ocean \& Coastal Management*, 53 (7), 313–326.

Fowler, K.R., Jenkins, E.W., Ostrove, C., Chrispell, J.C., Farthing, M.W., and Parno, M., 2015. A decision making framework with MODFLOW-FMP2 via optimization: Determining trade-offs in crop selection. *Environmental Modelling \& Software*, 69, 280–291.

Franek, J. and Kresta, A., 2014. Judgment Scales and Consistency Measure in AHP. *Procedia Economics and Finance*, 12, 164–173.

Freudenburg, W.R., 1999. Tools for understanding the socioeconomic and political settings for environmental decision making. *In*: V.H. Dale and M.R. English, eds. *Tools to aid environmental decision making*. Springer Verlag New York, 94 – 129.

Fu, B., Xu, P., Wang, Y., Peng, Y., and Ren, J., 2013. Spatial Pattern of Water Retetnion in Dujiangyan County. *Acta Ecologica Sinica*, 33 (3), 789–797.

Fu, B.-J., Su, C.-H., Wei, Y.-P., Willett, I., Lü, Y.-H., and Liu, G.-H., 2011. Double counting in ecosystem services valuation: causes and countermeasures. *Ecological Research*, 26 (1), 1–14.

Furlanello, C., Neteler, M., Merler, S., Menegon, S., Fontanari, S., Donini, A., Rizzoli, A., and Chemini, C., 2003. GIS and the random forest predictor: Integration in R for tick-borne disease risk assessment. *In*: *Proceedings of Distributed Statistical Computing*.

2.

Fürst, C., Volk, M., Pietzsch, K., Makeschin, F., and Furst, C., 2010. Pimp your landscape: a tool for qualitative evaluation of the effects of regional planning measures on ecosystem services. *Environ Manage*, 46 (6), 953–968.

Gadsby, S., Lockwood, M., Moore, S., and Curtis, A., 2013. *Tasmanian Midlands Socio-Economic Profile*. Landscapes and Policy Hub.

Gascoigne, W.R., Hoag, D., Koontz, L., Tangen, B.A., Shaffer, T.L., and Gleason, R.A., 2011. Valuing ecosystem and economic services across land-use scenarios in the Prairie Pothole Region of the Dakotas, USA. *Ecological Economics*, 70 (10), 1715–1725.

Genuer, R., Poggi, J.-M., and Tuleau-Malot, C., 2010. Variable selection using random forests. *Pattern Recognition Letters*, 31 (14), 2225–2236.

German, L., Mazengia, W., Nyangas, S., Meliyo, J., Adimassu, Z., Bekele, B., and Tirwomwe, W., 2012. Participatory Integrated Watershed Management. *In*: L. German, J. Mowo, T. Amede, and K. Masuki, eds. *Integrated Natural Resource Management in the Highlands of Eastern Africa - from concept to practice*. Earthscan and International Development Research Centre, 83–158.

Gibbons, J.M. and Ramsden, S.J., 2008. Integrated modelling of farm adaptation to climate change in East Anglia, UK: scaling and farmer decision making. *Agriculture, ecosystems \& environment*, 127 (1), 126–134.

Giri, S. and Nejadhashemi, A.P., 2014. Application of analytical hierarchy process for effective selection of agricultural best management practices. *Journal of Environmental Management*, 132, 165–177.

Gislason, P.O., Benediktsson, J.A., and Sveinsson, J.R., 2006. Random forests for land cover classification. *Pattern Recognition Letters*, 27 (4), 294–300.

Goel, A.K., Rugaber, S., and Vattam, S., 2009. Structure, behavior, and function of complex systems: The structure, behavior, and function modeling language. *Artificial Intelligence for Engineering Design, Analysis and Manufacturing*, 23 (01), 23–35.

Goetz, S.J., Jantz, C.A., Prince, S.D., Smith, A.J., Varlyguin, D., and Wright, R.K., 2004. Integrated Analysis of Ecosystem Interactions With Land Use Change: The Chesapeake Bay Watershed. *Ecosystems and Land Use Change*, 153, 263–275.

Gómez-Baggethun, E., de Groot, R., Lomas, P.L., and Montes, C., 2010. The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes. *Ecological Economics*, 69 (6), 1209–1218.

Granitto, P.M., Furlanello, C., Biasioli, F., and Gasperi, F., 2006. Recursive feature elimination with random forest for PTR-MS analysis of agroindustrial products. *Chemometrics and Intelligent Laboratory Systems*, 83 (2), 83–90.

Gray, A., 1931. *Development of Economic Doctrine: An Introductory Survey*. Longmans Greens and Co.

Green, G. and Knight, R., 2015. *Linking revegetation, remnant restoration and ecosystem management priorities in Tasmania's lowland grazing districts*. Bushlinks 500.

Greening Australia, 2003. *Bushcare Support 2003*. Native Vegetation Management - A needs analysis of regional service delivery in Tasmania.

de Groot, R., Alkemade, R., Braat, L., Hein, L., and Willemen, L., 2010a. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecological Complexity*, 7 (3), 260–272.

de Groot, R., Fisher, B., Christie, M., Aronson, J., Braat, L., Gowdy, J., Haines-Young, R., Maltby, E., Neuville, A., Polasky, S., Portela, R., and Ring, I., 2010. *Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation*. The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations. Earthscan, London and Washington: TEEB.

De Groot, R., Kumar, P., Ploeg, S. van der, and Sukhdev, P., 2010. *Estimates of monetary values of ecosystem services*. Appendix 3 in: The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations. Earthscan, London, UK: TEEB.

de Groot, R.S., Alkemade, R., Braat, L., Hein, L., and Willemen, L., 2010b. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecological Complexity*, 7 (3), 260–272.

de Groot, R.S., Wilson, M.A., and Boumans, R.M.. J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics*, 41 (3), 393–408.

Guerry, A.D., Ruckelshaus, M.H., Arkema, K.K., Bernhardt, J.R., Guannel, G., Kim, C.-K., Marsik, M., Papenfus, M., Toft, J.E., Verutes, G., Wood, S.A., Beck, M., Chan, F., Chan, K.M.A., Gelfenbaum, G., Gold, B.D., Halpern, B.S., Labiosa, W.B., Lester, S.E., Levin, P.S., McField, M., Pinsky, M.L., Plummer, M., Polasky, S., Ruggiero, P., Sutherland, D.A., Tallis, H., Day, A., and Spencer, J., 2012. Modeling benefits from nature: using ecosystem services to inform coastal and marine spatial planning. *International Journal of Biodiversity Science, Ecosystem Services & Management*, 8 (1-2), 107–121.

Guillaume, S., Bruzeau, C., Justes, E., Lacroix, B., and Bergez, J.-E., 2016. A conceptual model of farmers' decision-making process for nitrogen fertilization and irrigation of durum wheat. *European Journal of Agronomy*, 73, 133–143.

Guillem, E.E., Murray-Rust, D., Robinson, D.T., Barnes, A., and Rounsevell, M.D.A., 2015. Modelling farmer decision-making to anticipate tradeoffs between provisioning ecosystem services and biodiversity. *Agricultural Systems*, 137, 12–23.

Gunawardena, M. and Rowan, J.S., 2005. Economic valuation of a mangrove ecosystem threatened by shrimp aquaculture in Sri Lanka. *Environmental Management*, 36 (4), 535–550.

Guswa, A.J., Brauman, K.A., Brown, C., Hamel, P., Keeler, B.L., and Sayre, S.S., 2014. Ecosystem services: Challenges and opportunities for hydrologic modeling to support decision making. *Water Resources Research*, 50 (5), 4535–4544.

Guzy, M.R., Smith, C.L., Bolte, J.P., Hulse, D.W., and Gregory, S. V, 2008. Policy research using agent-based modeling to assess future impacts of urban expansion into farmlands and forests. *Ecology and Society*, 13 (1), 37.

Von Haaren, C., 2002. Landscape planning facing the challenge of the development of cultural landscapes. *Landscape and Urban Planning*, 60 (2), 73–80.

Hadzilacos, T. and Tryfona, N., 1992. A model for expressing topological integrity constraints in geographic databases. *In*: *Theories and methods of spatio-temporal reasoning in geographic space*. Springer, 252–268.

Han, P., Zhang, X., Norton, R.S., and Feng, Z., 2007. Reducing overfitting in predicting intrinsically unstructured proteins. *In*: *Pacific-Asia Conference on Knowledge Discovery and Data Mining*. 515–522.

Hanley, N.D. and Ruffell, R.J., 1993. The contingent valuation of forest characteristics: two experiments. *Journal of Agricultural Economics*, 44 (2), 218–229.

Hansen, B.G. and Greve, A., 2014. Dairy farmers values and how their values affect their decision making. *Agricultural and Food Science*, 23, 278 – 290.

Hao, F., Lai, X., Ouyang, W., Xu, Y., Wei, X., and Song, K., 2012. Effects of Land Use Changes on the Ecosystem Service Values of a Reclamation Farm in Northeast China. *Environmental Management*, 50 (5), 888–899.

Haslett, J.R., Berry, P.M., Bela, G., Jongman, R.H.G., Pataki, G., Samways, M.J., and Zobel, M., 2010. Changing conservation strategies in Europe: a framework integrating ecosystem services and dynamics. *Biodiversity and Conservation*, 19 (10), 2963–2977.

Häyhä, T. and Franzese, P.P., 2014. Ecosystem services assessment: A review under an ecological-economic and systems perspective. *Ecological Modelling*, 289, 124–132.

Hill, M.J., Braaten, R., Veitch, S.M., Lees, B.G., and Sharma, S., 2005. Multi-criteria decision analysis in spatial decision support: the ASSESS analytic hierarchy process and the role of quantitative methods and spatially explicit analysis. *Environmental Modelling & Software*, 20 (7), 955–976.

Hill, M.J., Lesslie, R., Barry, A., and Barry, S., 2005. A simple, portable, spatial multi-criteria analysis shell--MCAS-S. *In*: *MODSIM 2005 International Congress on Modelling and Simulation. Modelling and Simulation Society of Australia and New Zealand*. 12–15.

Holling, C.S., 1978. *Adaptive environmental assessment and management.* Wiley-Interscience.

Holtz, G. and Nebel, M., 2014. Testing Model Robustness--Variation of Farmers' Decision-Making in an Agricultural Land-Use Model. *In*: *Advances in Social Simulation*. Springer, 37–48.

Holzman C., D., 2012. Accounting for Nature's Benefits: The Dollar Value of Ecosystem Services. *Environmental Health Perspectives*, 120 (4), A153.

Howarth, R.B. and Farber, S., 2002. Accounting for the value of ecosystem services. *Ecological Economics*, 41 (3), 421–429.

Hoyer, R. and Chang, H., 2014. Assessment of freshwater ecosystem services in the Tualatin and Yamhill basins under climate change and urbanization. *Applied Geography*, 53, 402–416.

Iftekhar, M.S., Tisdell, J.G., and Gilfedder, L., 2014. Private lands for biodiversity conservation: Review of conservation covenanting programs in Tasmania, Australia. *Biological Conservation*, 169, 176–184.

Illge, L. and Schwarze, R., 2009. A matter of opinion - How ecological and neoclassical environmental economists and think about sustainability and economics. *Ecological Economics*, 68 (3), 594–604.

van Ingrid, E.P., Jennings, S.M., Louviere, J.J., and Burgess, L.B., 2011. Tasmanian landowner preferences for conservation incentive programs: A latent class approach. *Journal of environmental management*, 92 (10), 2647–2656.

Institute for Digital Research and Education, 2016. How do I interpret odds ratios in logistic regression? [online]. *UCLA: Statistical Consulting Group*. Available from: http://www.ats.ucla.edu/stat/mult_pkg/faq/general/citingats.htm [Accessed 19 Aug 2016].

Iovanna, R., 2005. Benefits Transfer and Valuation Databases : Are We Heading in the Right Direction ? *In*: R. Iovanna and A. Associates, eds. *Benefits Transfer and Valuation Databases : Are We Heading in the Right Direction ?* Environment Canada, 1–272.

Isbell, R.F., 2002. *The Australian Soil Classification*. Revised Ed. CSIRO Publishing Collingwood, Vic.

Isely, E.S., Isely, P., Seedang, S., Mulder, K., Thompson, K., and Steinman, A.D., 2010. Addressing the information gaps associated with valuing green infrastructure in west Michigan: INtegrated Valuation of Ecosystem Services Tool (INVEST). *Journal of Great Lakes Research*, 36 (3), 448–457.

Ishizaka, A. and Labib, A., 2009. Analytic hierarchy process and expert choice: Benefits and limitations. *OR Insight*, 22 (4), 201–220.

Ismail, R. and Mutanga, O., 2010. A comparison of regression tree ensembles: Predicting Sirex noctilio induced water stress in Pinus patula forests of KwaZulu-Natal, South Africa. *International Journal of Applied Earth Observation and Geoinformation*, 12, S45–S51.

Jackson, B., Pagella, T., Sinclair, F., Orellana, B., Henshaw, A., Reynolds, B., Mcintyre, N., Wheater, H., and Eycott, A., 2013. Polyscape: A GIS mapping framework providing efficient and spatially explicit landscape-scale valuation of multiple ecosystem services. *Landscape and Urban Planning*, 112 (1), 74–88.

Jacobs, M., 1997. Environmental valuation, deliberative democracy and public decision-making institutions. *Valuing nature*, 211–231.

James, R.F. and Blamey, R.K., 1999. Citizen participation - some recent Australian developments. *In*: *Pacific Science Conference, Sydney, Australia*.

Jones, Z. and Linder, F., 2015. Exploratory data analysis using random forests. *In*: *Prepared for the 73rd annual Midwest Political Science Association conference*.

Jopp, F., Reuter, H., and Breckling, B., 2012. *Modelling complex ecological dynamics: An introduction into ecological modelling for students, teachers & scientists*. Springer.

Kahneman, D., 2003. Maps of bounded rationality: Psychology for behavioral economics. *The American economic review*, 93 (5), 1449–1475.

Kahneman, D. and Sugden, R., 2005. Experienced utility as a standard of policy evaluation. *Environmental and resource economics*, 32 (1), 161–181.

Kallis, G. and Norgaard, R.B., 2010. Coevolutionary ecological economics. *Ecological Economics*, 69 (4), 690–699.

Karali, E., Rounsevell, M.D. a., and Doherty, R., 2011. Integrating the diversity of farmers' decisions into studies of rural land-use change. *Procedia Environmental Sciences*, 6, 136–145.

Kavzoglu, T., Sahin, E.K., and Colkesen, I., 2014. Landslide susceptibility mapping using GIS-based multi-criteria decision analysis, support vector machines, and logistic regression. *Landslides*, 11 (3), 425–439.

Kelly, R.A., Jakeman, A.J., Barreteau, O., Borsuk, M.E., ElSawah, S., Hamilton, S.H., Henriksen, H.J., Kuikka, S., Maier, H.R., Rizzoli, A.E., van Delden, H., and Voinov, A.A., 2013. Selecting among five common modelling approaches for integrated environmental assessment and management. *Environmental Modelling & Software*, 47, 159–181.

Kettunen, M., Vihervaara, P., Kinnunen, S., D'Amato, D., Badura, T., Argimon, M., and Ten Brink, P., 2012. Understanding and assessing the value of nature - Chapter 5. *In*: *Socio-economic importance of ecosystem services in the Nordic Countries - Synthesis in the context of TEEB*. Nordic Council of Ministers, 55.

Kleindorfer, P.R., 1999. Understanding individual's environmental decisions: A decision science approach. *In*: K. Sexton, A.A. Marcus, W. Easter, and T.D. Burkhardt, eds. *Better environmental decisions: strategies for governments, businesses, and communities*. Island Press.

Kliskey, A.D., 1995. The role and functionality of GIS as a planning tool in natural-resource management. *Computers, environment and urban systems*, 19 (1), 15–22.

Knight, R.I., 2010. *User guide to the spreadsheet version of the Regional Ecosystem Model for biodiversity in the Tasmanian Midlands, version 1.0*. 'Using landscape ecology to prioritise property management actions in Tasmania'. Caring for Our Country.

Knight, R.I. and Cullen, P.J., 2010. *Specifications for a Regional Ecosystem Model of natural* *resources in the Tasmanian Midlands - A report of the Caring for Our Country project 'Using landscape ecology to prioritise property management actions in Tasmania'*. Caring for our Country.

Van Kooten, G.C., Sohngen, B., and others, 2007. Economics of forest ecosystem carbon sinks: a review.

Kragt, M.E., Newham, L.T.H., Bennett, J., and Jakeman, A.J., 2011. An integrated approach to linking economic valuation and catchment modelling. *Environmental Modelling Software*, 26 (1), 92–102.

Kralisch, S., Zander, F., and Krause, P., 2009. Coupling the RBIS Environmental Information System and the JAMS Modelling Framework. *In*: *18th World IMACS / MODSIM Congress 13 - 17 July*. Cairns, Australia.

Kreuter, U.P., Harris, H.G., Matlock, M.D., and Lacey, R.E., 2001. Change in ecosystem service values in the San Antonio area, Texas. *Ecological Economics*, 39 (3), 333–346.

Kula, E., 1998. *History of Environmental Economic Thought*. Routledge, London.

Kułakowski, K., 2015. Notes on order preservation and consistency in AHP. *European Journal of Operational Research*, 245 (1), 333–337.

Lamarque, P., Artaux, A., Barnaud, C., Dobremez, L., Nettier, B., and Lavorel, S., 2013. Taking into account farmers' decision making to map fine-scale land management adaptation to climate and socio-economic scenarios. *Landscape and Urban Planning*, 119, 147–157.

Lamarque, P., Meyfroidt, P., Nettier, B., and Lavorel, S., 2014. How Ecosystem Services Knowledge and Values Influence Farmers' Decision-Making. *PLoS ONE*, 9 (9), e107572.

Lambin, E.F., Geist, H.J., and Lepers, E., 2003. Dynamics of land-use and land-cover change in tropical regions. *Annual review of environment and resources*, 28 (1), 205–241.

Lambin, E.F., Rounsevell, M.D.A., and Geist, H.J., 2000. Are agricultural land-use models able to predict changes in land-use intensity? *Agriculture, Ecosystems \& Environment*, 82 (1), 321–331.

Lange, W.J. De, Stafford, W.H.L., Forsyth, G.G., and Maitre, D.C. Le, 2012. Incorporating stakeholder preferences in the selection of technologies for using invasive alien plants as a bio-energy feedstock: Applying the analytical hierarchy process. *Journal of Environmental Management*, 99, 76–83.

Lant, C.L., Kraft, S.E., Beaulieu, J., Bennett, D., Loftus, T., and Nicklow, J., 2005. Using GIS-based ecological–economic modeling to evaluate policies affecting agricultural watersheds. *Ecological Economics*, 55 (4), 467–484.

LeBlanc, M. and Fitzgerald, S., 2000. Logistic regression for school psychologists. *School Psychology Quarterly*, 15 (3), 344.

Lee, S., 2005. Application of logistic regression model and its validation for landslide susceptibility mapping using GIS and remote sensing data. *International Journal of Remote Sensing*, 26 (7), 1477–1491.

Leh, M.D.K., Matlock, M.D., Cummings, E.C., and Nalley, L.L., 2013. Quantifying and mapping multiple ecosystem services change in West Africa. *Agriculture, Ecosystems & Environment*, 165, 6–18.

Lehtonen, M., 2004. The environmental–social interface of sustainable development: capabilities, social capital, institutions. *Ecological Economics*, 49 (2), 199–214.

Leopold, A. and Schwartz, C.W., 1989. *A Sand County almanac, and sketches here and there*. Oxford University Press.

Lesslie, R., Hill, M., Hill, P., Cresswell, H., and Dawson, S., 2008. The Application of a Simple Spatial Multi-Criteria Analysis Shell to Natural Resource Management Decision Making. *In*: C. Pettit, W. Cartwright, I. Bishop, K. Lowell, D. Pullar, and D. Duncan, eds. *Landscape Analysis and Visualisation SE - 5*. Springer Berlin Heidelberg, 73–95.

Lesslie, R., Mewett, J., and Walcott, J., 2011. *Landscapes in transition: tracking land use change in Australia*. Science and Economics Insights. Australian Government.

Lewandrowski, J., Darwin, R.F., Tsigas, M., and Raneses, A., 1999. Estimating costs of protecting global ecosystem diversity. *Ecological economics*, 29 (1), 111–125.

Li, X., Gao, Q., Lei, T., and Yang, X., 2010. Application of an integrative hydro-ecological model to study water resources management in the upper and middle parts of the Yellow River basin. *Frontiers of Earth Science*, 5 (1), 45–55.

Liacc, R., 2007. An Integrated Economic Modelling and Decision Support Methodology. *Simulation*, 4 (Cd), 497–502.

Liaw, A. and Wiener, M., 2002. Classification and regression by randomForest. *R news*, 2 (3), 18–22.

Lisson, S.N. and Cotching, W.E., 2011. Modelling the fate of water and nitrogen in the mixed vegetable farming systems of northern Tasmania, Australia. *Agricultural Systems*, 104 (8), 600–608.

Liu, S., 2011. *Environmental Benefit Transfers of Ecosystem Service Valuation*. Treatise on Estuarine and Coastal Science. Elsevier Inc.

Liu, S., Costanza, R., Farber, S., and Troy, A., 2010. Valuing ecosystem services. *Annals of the New York Academy of Sciences*, 1185 (1), 54–78.

Liu, S., Costanza, R., Troy, A., D'Aagostino, J., and Mates, W., 2010. Valuing New Jersey's ecosystem services and natural capital: a spatially explicit benefit transfer approach. *Environmental management*, 45 (6), 1271–85.

Liu, Y., Gupta, H., Springer, E., and Wagener, T., 2008. Linking science with environmental decision making: Experiences from an integrated modeling approach to supporting sustainable water resources management. *Environmental Modelling & Software*, 23 (7), 846–858.

Liu, Y., Li, J., and Zhang, H., 2012. An ecosystem service valuation of land use change in Taiyuan City, China. *Ecological Modelling*, 225, 127–132.

Lockwood, M., 1997. Integrated value theory for natural areas. *Ecological Economics*, 20 (1), 83–93.

Lockwood, M., 1998. Integrated value assessment using paired comparisons. *Ecological Economics*, 25 (1), 73–87.

Lubowski, R.N., Plantinga, A.J., and Stavins, R.N., 2006. Land-use change and carbon sinks: Econometric estimation of the carbon sequestration supply function. *Journal of Environmental Economics and Management*, 51 (2), 135–152.

Lubowski, R.N., Plantinga, A.J., and Stavins, R.N., 2008. What Drives Land-Use Change in the United States ? A National Analysis of Landowner Decisions. *Land Economics*, 84 (4), 529–550.

Madin, J.S., Bowers, S., Schildhauer, M.P., and Jones, M.B., 2008. Advancing ecological research with ontologies. *Trends in ecology & evolution*, 23 (3), 159–68.

Magierowski, R., Carter, O., Gilfedder, L., Gaynor, S., Lefroy, T., and Davies, P.E., 2014. *Aquatic refuge MCAS-S datapack for the Tasmanian Midlands - A worked example of using the MCAS-S tool for addressing aquatic questions in the Tasmanian Midlands*. University of Tasmania.

Mahan, B.L., 1997. *Valuing Urban Wetlands: A Property Pricing Approach*. US Army Corps of Engineers Institute for Water Resources, Evaluation of Environmental IWR Report.

Malawska, A. and Topping, C.J., 2016. Evaluating the role of behavioral factors and practical constraints in the performance of an agent-based model of farmer decision making. *Agricultural Systems*, 143, 136–146.

Mallawaarachchi, T., Walker, P.A., Young, M.D., Smyth, R.E., Lynch, H.S., and Dudgeon, G., 1996. GIS-based integrated modelling systems for natural resource management. *Agricultural Systems*, 50 (2), 169–189.

Marinoni, O., 2004. Implementation of the analytical hierarchy process with VBA in ArcGIS. *Computers \& Geosciences*, 30 (6), 637–646.

Marsh, D. and Smith, M., 2000. Understanding policy networks: towards a dialectical approach. *Political Studies*, 48 (1), 4–21.

Martínez-Harms, M.J. and Balvanera, P., 2012. Methods for mapping ecosystem service supply: a review. *International Journal of Biodiversity Science, Ecosystem Services \& Management*, 8 (1-2), 17–25.

Mauser, W., Klepper, G., Rice, M., Schmalzbauer, B.S., Hackmann, H., Leemans, R., and

Moore, H., 2013. Transdisciplinary global change research: the co-creation of knowledge for sustainability. *Current Opinion in Environmental Sustainability*, 5 (3–4), 420–431.

Maynard, S., James, D., and Davidson, A., 2010. The development of an ecosystem services framework for South East Queensland. *Environmental Management*, 45 (5), 881–895.

McComb, G., Lantz, V., Nash, K., and Rittmaster, R., 2006. International valuation databases: Overview, methods and operational issues. *Ecological Economics*, 60 (2), 461–472.

McCune, J., 2012. ArcGIS for Desktop 10.1 Add-In with Custom Toolbox [online]. Available from: http://joelmccune.com/arcgis-for-desktop-10-1-add-in-with-custom-toolbox/.

McDonald, J.H., 2014. *Handbook of Biological Statistics*. 3rd ed. Sparky House Publishing, Baltimore, Maryland.

McKenzie, E., Rosenthal, A., Bernhardt, J., Girvetz, E., Kovacs, K., Olwero, N., and Toft, J., 2012. *Developing Scenarios to Assess Ecosystem Service Tradeoffs: Guidance and Case Studies for InVEST Users*. Natural Capital Project. Washington, DC: World Wildlife Fund.

Merwade, V., 2012. Watershed and Stream Network Delineation using ArcHydro Tools.

Meyer, B.C., Lescot, J.-M., and Laplana, R., 2009. Comparison of two spatial optimization techniques: a framework to solve multiobjective land use distribution problems. *Environmental management*, 43 (2), 264–81.

Michaels, K., Lacey, M., Norton, T., and Williams, J., 2008. Vegetation Futures for Tasmania. *In*: *Vegetation Future - Australia's National Vegetation Conference*. 2–15.

Midlands Initiatives for Local Enterprise Inc, 2003. Gateway to online information on all aspects of life including local government, in Tasmania's heartland [online]. *A joint initiative of the Northern Midlands, the Southern Midlands, and the Central Highlands Councils*. Available from: https://stors.tas.gov.au/au-7-0050-00001721.

Millennium Ecosystem Assessment, 2005. *Ecosystems and Human Well-Being: Our Human Planet: Summary for Decision Makers*. Island Press.

Miller, A.J., 1984. Selection of subsets of regression variables. *Journal of the Royal Statistical Society. Series A (General)*, 389–425.

Molkentin, D., 2007. *The Book of Qt 4: The art of building Qt applications*. No Starch Press.

Morse-Jones, S., Luisetti, T., Turner, R.K., and Fisher, B., 2011. Ecosystem valuation: some principles and a partial application. *Environmetrics*, 22 (5), 675–685.

Muller, F., Breckling, B., Jopp, F., and Reuter, H., 2012. What are the general conditions under which ecological models can be applied? *In*: *Modelling complex ecological dynamics: An introduction into ecological modelling for students, teachers &*

scientists. American Library Association.

Münier, B., 2004. Combined ecological and economic modelling in agricultural land use scenarios. *Ecological Modelling*, 174 (1-2), 5–18.

Nadeau, R., 2011. Environmental and ecological economics [online]. *The Encyclopedia of Earth*. Available from: http://www.eoearth.org/view/article/152604 [Accessed 11 Apr 2016].

Naidoo, R. and Ricketts, T.H., 2006. Mapping the economic costs and benefits of conservation. *PLoS biology*, 4 (11), e360.

National Research Council, 2004. *Valuing Ecosystem Services: Toward Better Environmental Decision-Making*. Washington, DC: The National Academies Press.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, Dr., Chan, K.M.A., Daily, G.C., Goldstein, J., Kareiva, P.M., and others, 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment*, 7 (1), 4–11.

Nelson, E.J. and Daily, G.C., 2010. Modelling ecosystem services in terrestrial systems. *F1000 biology reports*, 2.

Niraj, S.K., Dayal, V., and Krausman, P.R., 2010. Applying methodological pluralism to wildlife and the economy. *Ecological Economics*, 69 (8), 1610–1616.

Norgaard, R.B., 1989. The case for methodological pluralism. *Ecological Economics*, 1 (1), 37–57.

Norton, B.G. and Noonan, D., 2007. Ecology and valuation: Big changes needed. *Ecological Economics*, 63 (4), 664–675.

NRM North, 2010. *Northern Tasmania Natural Resource Management Strategy 2010 - 2015*. Northern Tasmania Natural Resource Management Committee. Northern Tasmania Natural Resource Management Committee.

Opdam, P., Steingrover, E., and Rooij, S., 2006. Ecological networks: A spatial concept for multi-actor planning of sustainable landscapes. *Landscape and Urban Planning*, 75 (3-4), 322–332.

Osborn, F., 1948. Our Plundered Planet.: 68--85. *Little, Brown. Boston, Mass*.

Pannell, D.J. and Roberts, A.M., 2009. Conducting and delivering integrated research to influence land-use policy: salinity policy in Australia. *Environmental Science & Policy*, 12 (8), 1088–1098.

Parks and Wildlife Service Tasmania, 2015. Complete listing of National Parks and Reserves [online]. *Department of Primary Industries, Parks, Water and Environment*. Available from: http://www.parks.tas.gov.au/index.aspx?base=5710.

Park, H., 2013. An introduction to logistic regression: from basic concepts to interpretation with particular attention to nursing domain. *Journal of Korean Academy of Nursing*, 43 (2), 154–164.

Parrott, L., 2017. The modelling spiral for solving wicked environmental problems: guidance for stakeholder involvement and collaborative model development. *Methods in Ecology and Evolution*.

Pascual, U., Muradian, R., Brander, L., Gómez-Baggethun, E., Martín-López, B., Verma, M., Armsworth, P., Christie, M., Cornelissen, H., Eppink, F., and others, 2010. *The economics of valuing ecosystem services and biodiversity - Chapter 5*. The Economics of Ecosystems and Biodiversity. TEEB.

Pattanayak, S.K. and Kramer, R.A., 2001. Worth of watersheds: a producer surplus approach for valuing drought mitigation in Eastern Indonesia. *Environment and Development Economics*, 6 (01), 123–146.

Patterson, M.G., 2006. Development of ecological economics in Australia and New Zealand. *Ecological Economics*, 56 (3), 312–331.

Peach, D., Hughes, A., Kessler, H., Mathers, S., and Giles, J., 2011. *Integrated Environmental Modeling - The New Dream for Geological Surveys*. British Geological Survey.

Perrings, C., 1997. *Biodiversity loss: economic and ecological issues*. Cambridge University Press.

Pert, P.L., Lieske, S.N., and Hill, R., 2013. Participatory development of a new interactive tool for capturing social and ecological dynamism in conservation prioritization. *Landscape and Urban Planning*, 114, 80–91.

Petit, O. and Vivien, F.-D., 2015. When economists and ecologists meet on Ecological Economics: two science paths around two interdisciplinary concepts. *In*: *11th biennal Conference of the European Society for Ecological Economics*.

Piorr, A., Ungaro, F., Ciancaglini, A., Happe, K., Sahrbacher, A., Sattler, C., Uthes, S., and Zander, P., 2009. Integrated assessment of future CAP policies: land use changes, spatial patterns and targeting. *Environmental Science & Policy*, 12 (8), 1122–1136.

Plant, R. and Ryan, P., 2013. Ecosystem services as a practicable concept for natural resource management: some lessons from Australia. *International Journal of Biodiversity Science, Ecosystem Services & Management*, 9 (1), 44–53.

Van der Ploeg, S. and De Groot, R., 2010. *A searchable database of 1310 estimates of monetary values of ecosystem services*. The TEEB Valuation Database. Wageningen, the Netherlands: Foundation for Sustainable Development.

Van der Ploeg, S., De Groot, R., and Wang, Y., 2010. *Overview of structure, data and results*. The TEEB Valuation Database. Wageningen, the Netherlands: Foundation for Sustainable Development.

Policy Support Systems, 2016. Co$ting Nature [online]. *King's College London and AmbioTEK*. Available from: http://www.policysupport.org/costingnature.

Poppenborg, P. and Koellner, T., 2013. Do attitudes toward ecosystem services determine agricultural land use practices? An analysis of farmers' decision-making in a South Korean watershed. *Land Use Policy*, 31, 422–429.

Potschin, M.B. and Haines-Young, R.H., 2011. Ecosystem services: Exploring a geographical perspective. *Progress in Physical Geography*, 35 (5), 575–594.

Power, A.G., 2010. Ecosystem services and agriculture: tradeoffs and synergies. *Philosophical transactions of the Royal Society of London. Series B, Biological sciences*, 365 (1554), 2959–2971.

Power, D., 2016. Are computerized decision aids decision support systems? *Decision Support News*, 17 (10).

Prasad, A.M., Iverson, L.R., and Liaw, A., 2006. Newer classification and regression tree techniques: bagging and random forests for ecological prediction. *Ecosystems*, 9 (2), 181–199.

Radeloff, V.C., Nelson, E., Plantinga, A.J., Lewis, D.J., Helmers, D., Lawler, J.J., Withey, J.C., Beaudry, F., Martinuzzi, S., Butsic, V., Lonsdorf, E., White, D., and Polasky, S., 2012. Economic-based projections of future land use in the conterminous United States under alternative policy scenarios. *Ecological Applications*, 22 (3), 1036–1049.

Rao, E., Xiao, Y., Ouyang, Z., and Zheng, H., 2013. Spatial characteristics of soil conservation service and its impact factors in Hainan Island. *Acta Ecologica Sinica*, 33 (3), 746–755.

Rao, M.N., Waits, D. a, and Neilsen, M.L., 2000. A GIS-based modeling approach for implementation of sustainable farm management practices. *Environmental Modelling & Software*, 15 (8), 745–753.

Raudsepp-Hearne, C., Peterson, G.D., and Bennett, E.M., 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. *Proceedings of the National Academy of Sciences*, 107 (11), 5242–5247.

Read, D., 2004. Utility theory from Jeremy Bentham to Daniel Kahneman.

Reagan, D.P., 2006. An Ecological Basis for Integrated Environmental Management. *Human and Ecological Risk Assessment*, 12 (5), 819–833.

Ren, J., Wang, Y., Fu, B., and Xu, P., 2011. Soil conservation assessment in the Upper Yangtze River Basin based on invest model. *In*: *2011 International Symposium on Water Resource and Environmental Protection*. IEEE, 1833–1836.

van Riper, C.J., Kyle, G.T., Sutton, S.G., Barnes, M., and Sherrouse, B.C., 2012. Mapping outdoor recreationists' perceived social values for ecosystem services at Hinchinbrook Island National Park, Australia. *Applied Geography*, 35 (1-2), 164–173.

Roberts, M.S., 2015. Understanding Farmer Decision Making in Northern Lao PDR. *Culture, Agriculture, Food and Environment*, 37 (1), 14–27.

Rolfe, J., Bennett, J., and Louviere, J., 2002. Stated values and reminders of substitute goods: Testing for framing effects with choice modelling. *Australian Journal of ...*.

Røpke, I., 2005a. Trends in the development of ecological economics from the late 1980s to the early 2000s. *Ecological Economics*, 55 (2), 262–290.

Røpke, I., 2005b. Trends in the development of ecological economics from the late 1980s to the early 2000s. *Ecological economics*, 55 (2), 262–290.

Rudner, M., Biedermann, R., Schroder, B., and Kleyer, M., 2007. Integrated Grid Based Ecological and Economic (INGRID) landscape model - A tool to support landscape management decisions. *Environmental Modelling Software*, 22 (2), 177–187.

Saaty, T.L., 1977. A scaling method for priorities in hierarchical structures. *Journal of mathematical psychology*, 15 (3), 234–281.

Saaty, T.L., 1988. What is the Analytic Hierarchy Process? *In*: G. Mitra, H.J. Greenberg, F.A. Lootsma, M.J. Rijkaert, and H.J. Zimmermann, eds. *Mathematical Models for Decision Support*. Berlin, Heidelberg: Springer Berlin Heidelberg, 109–121.

Saaty, T.L., 1994a. How to make a decision: the analytic hierarchy process. *Interfaces*, 24 (6), 19–43.

Saaty, T.L., 1994b. Highlights and critical points in the theory and application of the analytic hierarchy process. *European Journal of operational research*, 74 (3), 426–447.

Saaty, T.L., 2000. *Fundamentals of decision making and priority theory with the analytic hierarchy process*. Rws Publications.

Saaty, T.L., 2008. Decision making with the analytic hierarchy process. *International journal of services sciences*, 1 (1), 83–98.

Saaty, T.L. and Vargas, L.G., 2012. *Models, methods, concepts \& applications of the analytic hierarchy process*. Springer Science \& Business Media.

Sánchez-Canales, M., López Benito, A., Passuello, A., Terrado, M., Ziv, G., Acuña, V., Schuhmacher, M., and Elorza, F.J., 2012. Sensitivity analysis of ecosystem service valuation in a Mediterranean watershed. *The Science of the total environment*, 440, 140–53.

Sandri, M. and Zuccolotto, P., 2006. Variable selection using random forests. *In*: *Data analysis, classification and the forward search*. Springer, 263–270.

Sarewitz, D., 2010. Against Holism. *In*: R. Frodeman, J.T. Klein, and C. Mitcham, eds. *The Oxford handbook of interdisciplinarity*. Oxford University Press.

Satija, K., 2009. *Textbook on Economics for Law Students*. Universal Law Publishing.

Scheirer, J., 2011. Developing GUI in Python for ArcGIS geoprocessing using PyQT? [online]. *GIS Stack Exchange*. Available from: http://gis.stackexchange.com/questions/6848/developing-gui-in-python-for-arcgis-geoprocessing-using-pyqt/6850#6850.

Schirmer, J. and Bull, L., 2011. *Technical Report 218 Planting trees for carbon sequestration: what do landholders think?* Cooperative Research Centre for

Forestry.

Schou, J.S., Skop, E., and Jensen, J.D., 2000. Integrated agri-environmental modelling: A cost-effectiveness analysis of two nitrogen tax instruments in the Vejle Fjord watershed, Denmark. *Journal of Environmental Management*, 58 (3), 199–212.

Schultz, C., Alegría, A.C., Cornelis, J., and Sahli, H., 2016. Comparison of spatial and aspatial logistic regression models for landmine risk mapping. *Applied Geography*, 66, 52–63.

Schutzberg, A., 2001. Update: ESRI has 40+% of GIS MArketshare. *Energy and Infrastructure, ESRI Technology*.

Seitz, W. and La Torre, D., 2014. Modelling Investment Optimization on Smallholder Farms through Multiple Criteria Decision Making and Goal Programming: A Case Study from Ethiopia. *INFOR: Information Systems and Operational Research*, 52 (3), 97–107.

Selten, R., 1999. What is bounded rationality. *Bounded Rationality: The Adaptive Toolbox (Cambridge, MA: MIT Press, 2001, pp. 13-36)*.

Sen, A., 1979. Utilitarianism and welfarism. *The Journal of Philosophy*, 76 (9), 463–489.

Sen, A., Harwood, A.R., Bateman, I.J., Munday, P., Crowe, A., Brander, L., Raychaudhuri, J., Lovett, A.A., Foden, J., and Provins, A., 2014. Economic assessment of the recreational value of ecosystems: methodological development and national and local application. *Environmental and Resource Economics*, 57 (2), 233–249.

Seppelt, R., Dormann, C.F., Eppink, F. V, Lautenbach, S., and Schmidt, S., 2011. A quantitative review of ecosystem service studies: approaches, shortcomings and the road ahead. *Journal of Applied Ecology*, 48 (3), 630–636.

Serafy, S. El, 1998. Pricing the invaluable:: the value of the world's ecosystem services and natural capital. *Ecological Economics*, 25 (1), 25–27.

Serrao, A., Coelho, L., and others, 2005. Analysing Farmers' Decision-Making Process Face to the Mid-Term Review of the Common Agricultural Policy in the Alentejo region of Portugal. *In*: *2005 Annual meeting, July 24-27, Providence, RI*.

Settle, C., Crocker, T.D., and Shogren, J.F., 2002. On the joint determination of biological and economic systems. *Ecological Economics*, 42 (1-2), 301–311.

Sherrouse, B.C., Clement, J.M., and Semmens, D.J., 2011. A GIS application for assessing, mapping, and quantifying the social values of ecosystem services. *Applied Geography*, 31 (2), 748–760.

Sherrouse, B.C. and Semmens, D.J., 2014. Validating a method for transferring social values of ecosystem services between public lands in the Rocky Mountain region. *Ecosystem Services*, 8, 166–177.

Sherrouse, B.C., Semmens, D.J., and Clement, J.M., 2014. An application of Social Values for Ecosystem Services (SolVES) to three national forests in Colorado and

Wyoming. *Ecological Indicators*, 36, 68–79.

Siew, T.-F. and Döll, P., 2012. Transdisciplinary research for supporting the inte-gration of ecosystem services into land and water management in the Tarim River Basin, Xinjiang, China. *Journal of Arid Land*, 4 (2), 196–210.

Silva, M.C. e and Teixeira, A.A.C., 2011. A bibliometric account of the evolution of EE in the last two decades: Is ecological economics (becoming) a post-normal science? *Ecological Economics*, 70 (5), 849–862.

Silvestri, S., Kershaw, F., and others, 2010. *Framing the flow: innovative approaches to understand, protect and value ecosystem services across linked habitats.* United Nations Environment Programme (UNEP).

Simon, H.A., 1955. A behavioral model of rational choice. *The quarterly journal of economics*, 99–118.

Simon, H.A., 1972. Theories of bounded rationality. *Decision and organization*, 1 (1), 161–176.

Simon, H.A., 2000. Bounded rationality in social science: Today and tomorrow. *Mind \& Society*, 1 (1), 25–39.

Siray, G.Ü., Toker, S., and Kaçiranlar, S., 2015. On the restricted Liu estimator in the logistic regression model. *Communications in Statistics-Simulation and Computation*, 44 (1), 217–232.

Siroky, D.S., 2009. Navigating random forests and related advances in algorithmic modeling. *Statistics Surveys*, 3, 147–163.

Smith, E.L., Bishop, I.D., Williams, K.J.H., and Ford, R.M., 2012. Scenario Chooser: An interactive approach to eliciting public landscape preferences. *Landscape and Urban Planning*, 106 (3), 230–243.

Söderbaum, P., 2000. *Ecological Economics A Political Economics Approach to Environment and Development*. Earthscan Publications Ltd.

Söderbaum, P., 2006. Democracy and sustainable development--what is the alternative to cost-benefit analysis? *Integrated Environmental Assessment and Management*, 2 (2), 182–190.

Soofi, A., 1995. Economics of Ibn Khaldun Revisited. *History of Political Economy*, 27 (2), 387.

Spash, C.L., 2008. *Ecosystems services valuation*. CSIRO Sustainable Ecosystems.

Spash, C.L., 2012. New foundations for ecological economics. *Ecological Economics*, 77 (0), 36–47.

Stacey, D., Légaré, F., Col, N.F., Bennett, C.L., Barry, M.J., Eden, K.B., Holmes-Rovner, M., Llewellyn-Thomas, H., Lyddiatt, A., Thomson, R., and others, 2014. Decision aids for people facing health treatment or screening decisions. *The Cochrane Library*.

Stephenson, W., 1953. The study of behavior; Q-technique and its methodology.

Steward, D.R. and Bernard, E.A., 2006. Integrated Engineering and Landscape Architecture Approaches To Address Groundwater Declines in the High Plains Aquifer. *In*: *Association of Environmental Engineering and Science Professors Case Studies Compilation*. 122 –134.

Stewart, R.N. and Purucker, S.T., 2006. SADA : A Freeware Decision Support Tool Integrating GIS , Sample design , Spatial Modeling , and Risk Assessment. *In*: A. Voinov, A.J. Jakeman, and A.E. Rizzoli, eds. *Proceedings of the iEMSs Third Biennial Meeting: 'Summit on Environmental Modelling and Software'. International Environmental Modelling and Software Society*. Burlington, USA.

Stock, P. and Burton, R.J.F., 2011. Defining terms for integrated (multi-inter-trans-disciplinary) sustainability research. *Sustainability*, 3 (8), 1090–1113.

Strobl, C., Malley, J., and Tutz, G., 2009. An introduction to recursive partitioning: rationale, application, and characteristics of classification and regression trees, bagging, and random forests. *Psychological methods*, 14 (4), 323.

Strother H. Walker, D.B.D., 1967. Estimation of the Probability of an Event as a Function of Several Independent Variables. *Biometrika*, 54 (1/2), 167–179.

Stuip, M.A.M., Baker, C.J., and Oosterberg, W., 2002. The socio-economics of wetlands. *Wetlands International and RIZA, The Netherlands*.

Su, C. and Fu, B., 2013. Evolution of ecosystem services in the Chinese Loess Plateau under climatic and land use changes. *Global and Planetary Change*, 101, 119–128.

Sullivan, S., 2014. *The natural capital myth; or will accounting save the world?*

Swetnam, R.D., Fisher, B., Mbilinyi, B.P., Munishi, P.K.T., Willcock, S., Ricketts, T., Mwakalila, S., Balmford, A., Burgess, N.D., Marshall, A.R., and others, 2011. Mapping socio-economic scenarios of land cover change: A GIS method to enable ecosystem service modelling. *Journal of environmental management*, 92 (3), 563–574.

Tasmanian Government's Wealth from Water, 2012. *An agricultural profile of the Midlands region, Tasmania*. Department of Economic Development, Tourism and the Arts.

Tasmanian Natural Heritage Trust, 2002. *Annual Report Tasmania 2001/2002*.

Tasmanian Natural Heritage Trust, 2003. *Annual Report Tasmania 2002/2003*.

Tereshenkov, A., 2015. ArcGIS-based solution (script tools + Python add-ins) [online]. *GIS Stack Exchange*. Available from: http://gis.stackexchange.com/questions/133354/recomendations-about-graphic-interface-for-arcpy-python-script.

The Landscapes and Policy Hub, 2015. *Aquatic life in the Tasmanian Midlands: escaping the summer heat*. National Environmental Research Program.

Tian, W., Bai, J., Sun, H., and Zhao, Y., 2013. Application of the analytic hierarchy process to a sustainability assessment of coastal beach exploitation: A case study of the wind power projects on the coastal beaches of Yancheng, China. *Journal of*

Environmental Management, 115, 251–256.

Tietenberg, T. and Lewis, L., 2015a. Environmental & Natural Resource Economics - Chapter 1. *In*: *Environmental & Natural Resource Economics*. Pearson Education Limited, 27 – 40.

Tietenberg, T. and Lewis, L., 2015b. Environmental & Natural Resource Economics - Chapter 13. *In*: *Environmental & Natural Resource Economics*. Pearson Education Limited, 342 – 368.

Tietenberg, T.H. and Lewis, L., 2015c. *Environmental and natural resource economics*. 10th Editi. Pearson Education.

Tilman, D., Polasky, S., and Lehman, C., 2005. Diversity, productivity and temporal stability in the economies of humans and nature. *Journal of Environmental Economics and Management*, 49 (3), 405–426.

Toman, M., 1998. Why not to calculate the value of the world's ecosystem services and natural capital. *Ecological Economics*, 25 (1), 57–60.

Trenouth, A.L., Harte, C., de Heer, C.P., Dewan, K., Grage, A., Primo, C., and Campbell, M.L., 2012. Public perception of marine and coastal protected areas in Tasmania, Australia: Importance, management and hazards. *Ocean \& Coastal Management*, 67, 19–29.

Treu, M.C., Magoni, M., Steiner, F., and Palazzo, D., 2000. Sustainable landscape planning for Cremona, Italy. *Landscape and Urban Planning*, 47 (1-2), 79–98.

Troy, A. and Wilson, M.A., 2006. Mapping ecosystem services: Practical challenges and opportunities in linking GIS and value transfer. *Ecological Economics*, 60 (2), 435–449.

Tschirhart, J., 2009. Integrated Ecological Economic Models. *Annual Review of Resource Economics*, 1 (1), 381–407.

Tucker, G.M. and Braat, L.C., 2009. *Scenarios and models for exploring future trends of biodiversity and ecosystem services changes*. Institute for European Environmental Policy (IEEP).

Turner, K.G., Anderson, S., Chang, M.G., Costanza, R., Courville, S., Dominati, E., Kubiszewski, I., Ogilvy, S., Porfirio, L., Ratna, N., and others, 2014. *Toward an integrated ecology and economics of land degradation and restoration: Methods, data, and models*. The Economics of Land Degradation (ELD).

Turner, R.K., Bergh, J.C.J.M. Van Den, and So, T., 2000. The values of wetlands: landscape and institutional ecological-economic analysis of wetlands: scientific integration for management and policy. *Ecological Economics*, 35 (1), 7–23.

US Government, 2015. Multiscale Integrated Earth Systems Model (MIMES) [online]. *US Climate Resilience Toolkit*. Available from: https://toolkit.climate.gov/tool/multiscale-integrated-earth-systems-model-mimes-0.

Vigerstol, K.L. and Aukema, J.E., 2011. A comparison of tools for modeling freshwater ecosystem services. *Journal of environmental management*, 92 (10), 2403–2409.

Vogt, W., 1948. Road to Survival. *Road to Survival.*

Voinov, A., Costanza, R., Wainger, L., Boumans, R., Villa, F., Maxwell, T., and Voinov, H., 1999. Patuxent landscape model: integrated ecological economic modeling of a watershed. *Environmental Modelling & Software*, 14 (5), 473–491.

Voinov, A., Kolagani, N., McCall, M.K., Glynn, P.D., Kragt, M.E., Ostermann, F.O., Pierce, S.A., and Ramu, P., 2016. Modelling with stakeholders--Next generation. *Environmental Modelling & Software*, 77, 196–220.

Volk, M., Hirschfeld, J., Dehnhardt, A., Schmidt, G., Bohn, C., Liersch, S., and Gassman, P.W., 2008. Integrated ecological-economic modelling of water pollution abatement management options in the Upper Ems River Basin. *Ecological Economics*, 66 (1), 66–76.

Volk, M., Hirschfeld, J., Schmidt, G., Bohn, C., Dehnhardt, A., Liersch, S., and Lymburner, L., 2007. A SDSS-based Ecological-economic Modelling Approach for Integrated River Basin Management on Different Scale Levels – The Project FLUMAGIS. *Water Resources Management*, 21 (12), 2049–2061.

Wainger, L. and Mazzotta, M., 2011. Realizing the Potential of Ecosystem Services: A Framework for Relating Ecological Changes to Economic Benefits. *Environmental Management*, 48 (4), 710–733.

Wallace, K.J., 2007. Classification of ecosystem services: Problems and solutions. *Biological Conservation*, 139 (3–4), 235–246.

Walls, M. and Riddle, A., 2012. *Biodiversity, Ecosystem Services and Land Use - Comparing Three Federal Policies*. Washington, DC.

Wang, D., Li, Y., Zheng, H., and Ouyang, Z., 2014. Ecosystem services' spatial characteristics and their relationships with residents' well-being in Miyun reservoir watershed. *Acta Ecologica Sinica*, 34 (1), 70–81.

Wang, J., Chen, J., Ju, W., and Li, M., 2010. IA-SDSS: A GIS-based land use decision support system with consideration of carbon sequestration. *Environmental Modelling & Software*, 25 (4), 539–553.

Wätzold, F., Drechsler, M., Armstrong, C.W., BaumgÄRtner, S., Grimm, V., Huth, A., Perrings, C., Possingham, H.P., Shogren, J.F., Skonhoft, A., and others, 2006. Ecological-Economic Modeling for Biodiversity Management: Potential, Pitfalls, and Prospects. *Conservation Biology*, 20 (4), 1034–1041.

Westman, W.E., 1977. How much are nature's services worth? *Science*, 197 (4307), 960–964.

White, S.S. and Selfa, T., 2013. Shifting lands: exploring kansas farmer decision-making in an era of climate change and biofuels production. *Environmental management*, 51 (2), 379–391.

Whitten, S.M. and Bennett, J.W., 2002. A travel cost study of duck hunting in the Upper South East of South Australia. *Australian Geographer*, 33 (2), 207–221.

Williams, G., 2008. *Data Mining Desktop Survival*.

Williams, J., 2009. *Ecosystem services mapping Stage 1 - Project scoping and data reconnaissance - Final Report*. Cradle Coast NRM.

Williams, J., 2013. *Mapping & Valuing Ecosystem Services of the Cradle Coast region, Tasmania - Final Report*. Cradle Coast NRM. Caring for Our Country.

Williams, K.J.H., 2014. Public acceptance of plantation forestry: Implications for policy and practice in Australian rural landscape. *Land Use Policy*, 38, 346–354.

Willis, K.J., Jeffers, E.S., Tovar, C., Long, P.R., Caithness, N., Smit, M.G.D., Hagemann, R., Collin-Hansen, C., and Weissenberger, J., 2012. Determining the ecological value of landscapes beyond protected areas. *Biological Conservation*, 147 (1), 3–12.

Wilson, G.A., 1997. Factors influencing farmer participation in the environmentally sensitive areas scheme. *Journal of environmental management*, 50 (1), 67–93.

Wilson, G.A. and Hart, K., 2000. Financial imperative or conservation concern? EU farmers' motivations for participation in voluntary agri-environmental schemes. *Environment and planning A*, 32 (12), 2161–2185.

Windle, J., Rolfe, J., and others, 2013. The limitations of applying benefit transfer to assess the value of ecosystem services in a generic peri-urban, coastal town in Australia. *In*: *th Annual conference of the Australian Agricultural and Resource Economics Society, February*. 5–8.

Woodward, R.T. and Wui, Y.-S., 2001. The economic value of wetland services: a meta-analysis. *Ecological economics*, 37 (2), 257–270.

Wright, D.J., Goodchild, M.F., and Proctor, J.D., 1997. Demystifying the persistent ambiguity of GIS as tool versus science. *Annals of the Association of American Geographers*, 87 (2), 346–362.

Wunder, S., 2007. The efficiency of payments for environmental services in tropical conservation. *Conservation biology*, 21 (1), 48–58.

Wunderlich, A.L., 2012. Automation in ArcGIS 10: Understanding Changes in Methods of Customization and Options for Migration of Legacy Code. *In*: *Digital Mapping Techniques '10 - Workshop Proceedings'*. US Geological Survey.

Xepapadeas, A., 2008. *Ecological economics - The New Palgrave Dictionary of Economics*. 2nd ed. Palgrave MacMillan.

Xia, O.Y., Xie, G., and Ka, A.N., 2003. Economic value of ecosystem services in Mangcuo Lake drainage basin. *Chinese Journal of Applied Ecology*, 14 (5), 676–680.

Xie, G.D., Lu, C.X., Leng, Y.F., Zheng, D., and Li, S.C., 2003. Ecological assets valuation of the Tibetan Plateau. *Journal of Natural Resources*, 18, 189–198.

Zhang, B., Li, W., and Xie, G., 2010. Ecosystem services research in China: Progress and perspective. *Ecological Economics*, 69 (7), 1389–1395.

Zhang, Z. and Liu, J., 2011. Progress in the valuation of ecosystem services. *Huanjing Kexue Xuebao/Acta Scientiae Circumstantiae*, 31 (9), 1835–1842.

Zhao, B., Kreuter, U., Li, B., Ma, Z., Chen, J., and Nakagoshi, N., 2004. An ecosystem service value assessment of land-use change on Chongming Island, China. *Land Use Policy*, 21 (2), 139–148.

Appendix A: Detailed charts presenting analysis of all the studies (65) in different categories

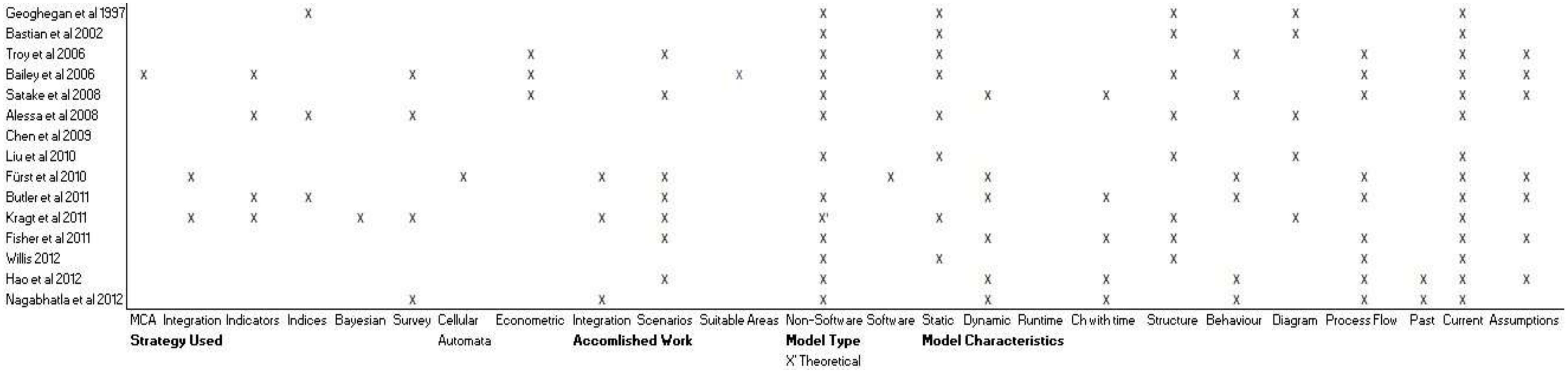


Figure 1a: Presentation of studies under ESV category.[24][25]

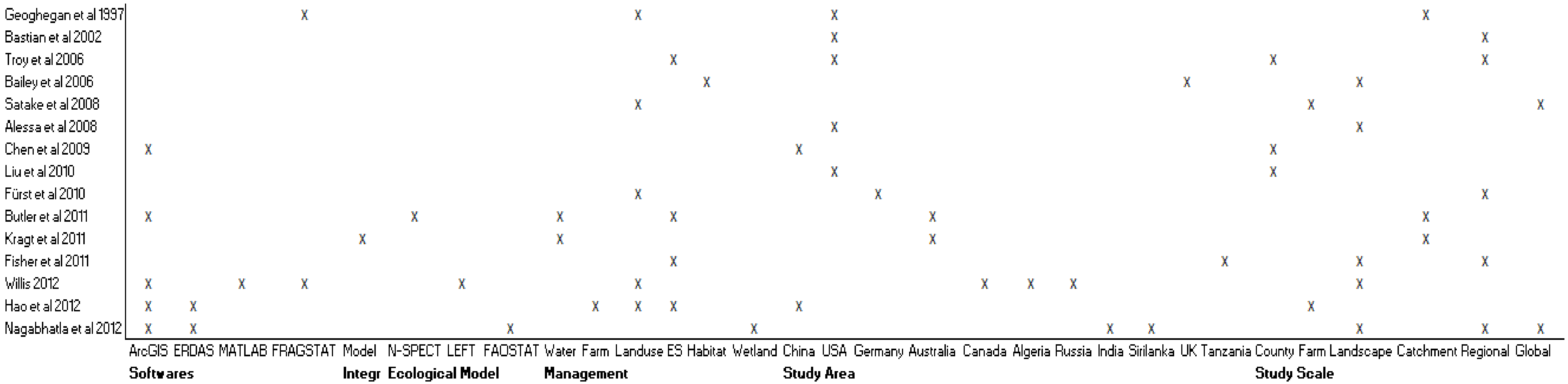


Figure 1b: Presentation of studies under ESV category.

[24] **Structure** of a model system is represented in terms of its components, the substances contained in the components (attributes) and connection (relationships) among the components. System structure is represented as diagrams (Goel *et al.* 2009).

[25] **Behaviour** of a system is what a system will do in response to its external environment. It is represented as a process flow or sequence of states/operations of the behaviour (Goel *et al.* 2009).

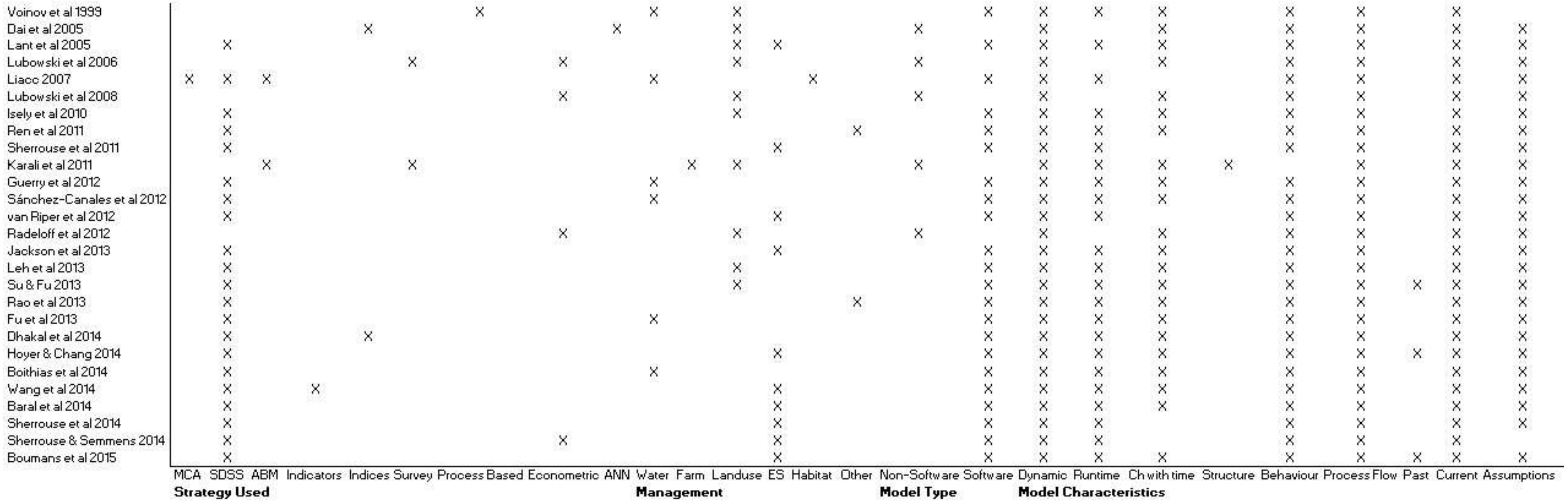


Figure 2a: Presentation of studies under Scenarios category.

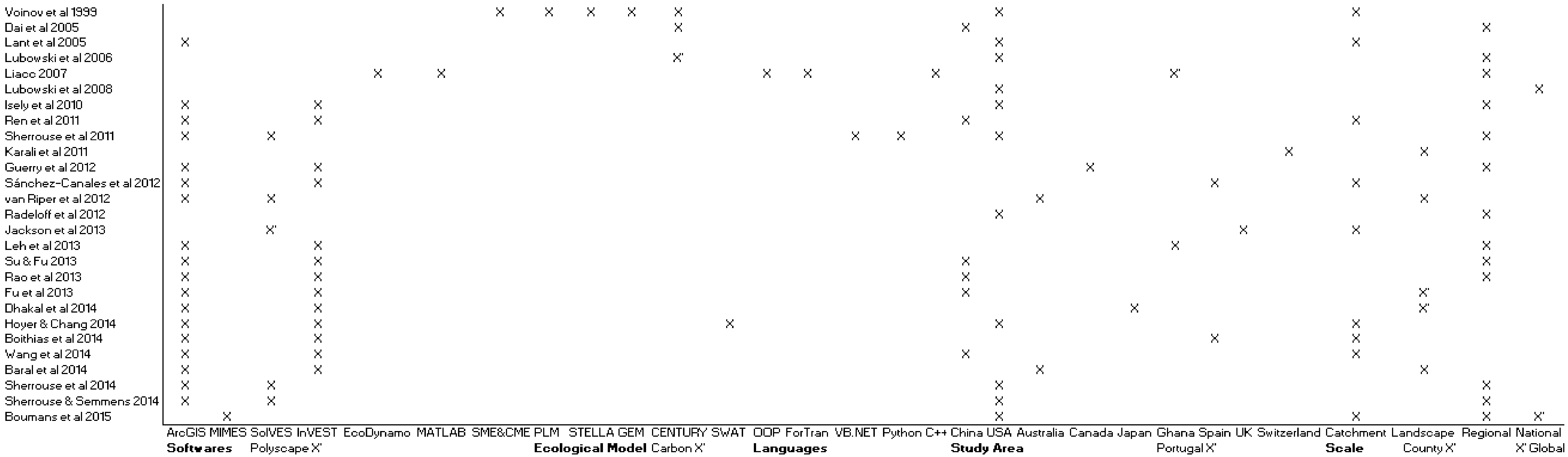


Figure 2b: Presentation of studies under Scenarios category.

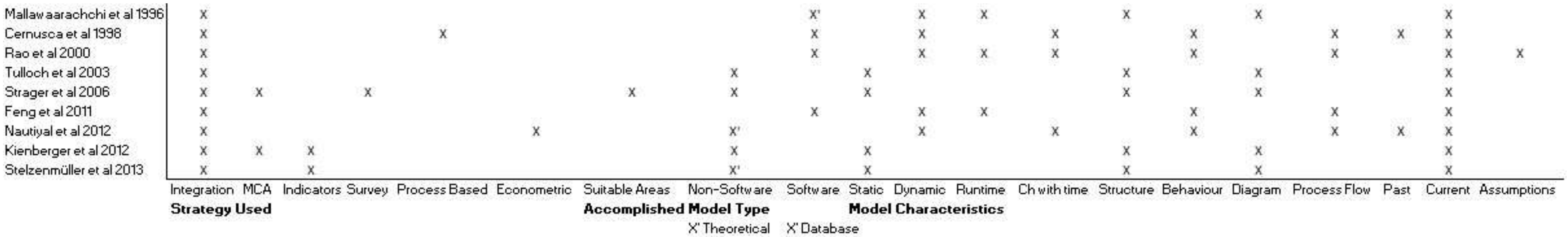


Figure 3a: Presentation of studies under Integration category.

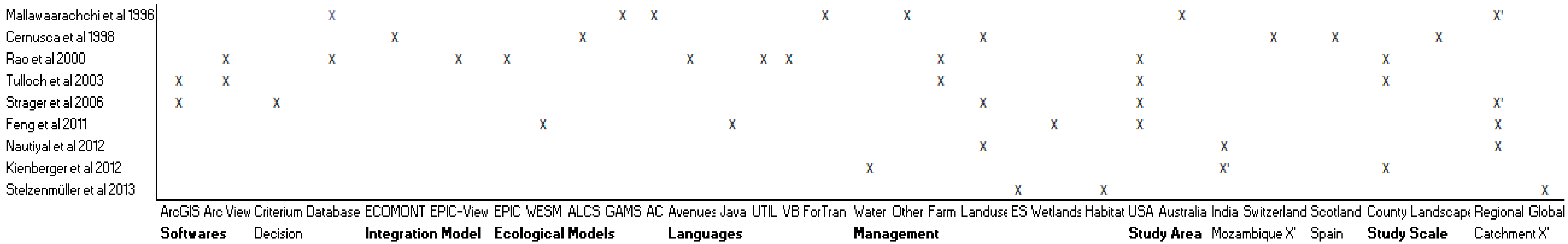


Figure 3b: Presentation of studies under Integration category.

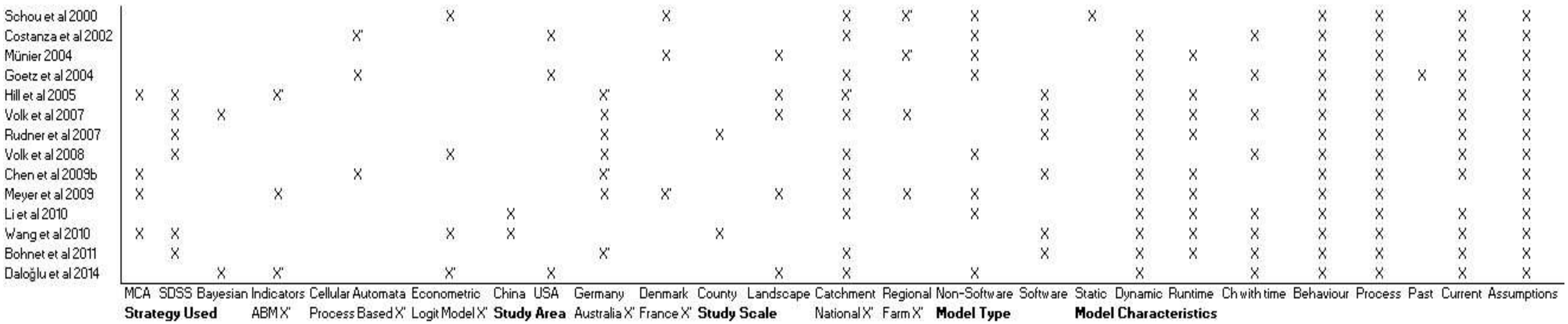


Figure 4a: Presentation of studies under Integration & Scenarios category.

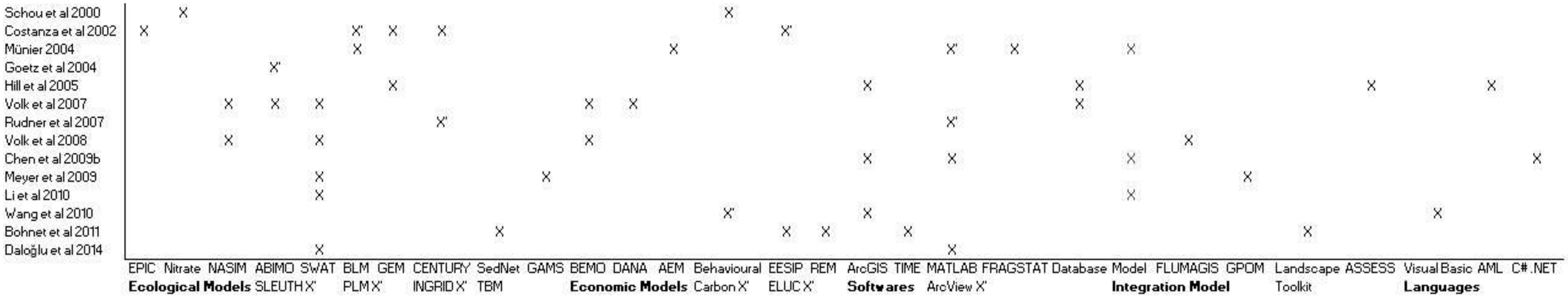


Figure 4b: Presentation of studies under Integration & Scenarios category.

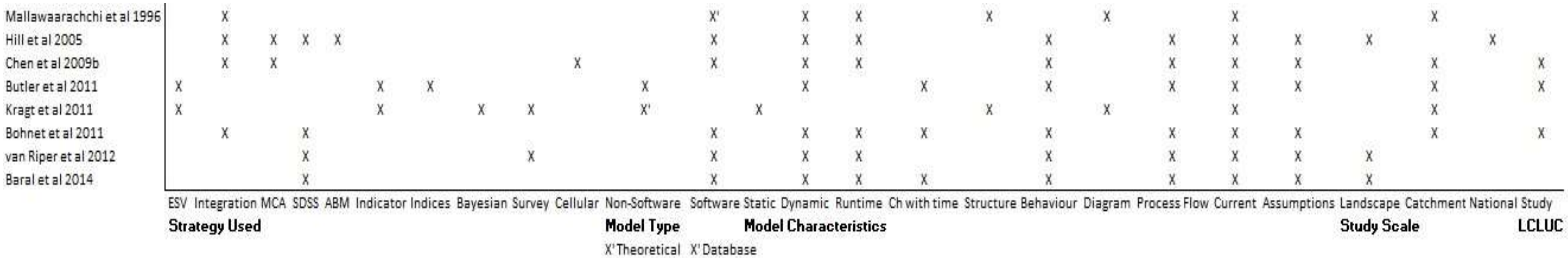


Figure 5a: Presentation of studies having Australia as study area.

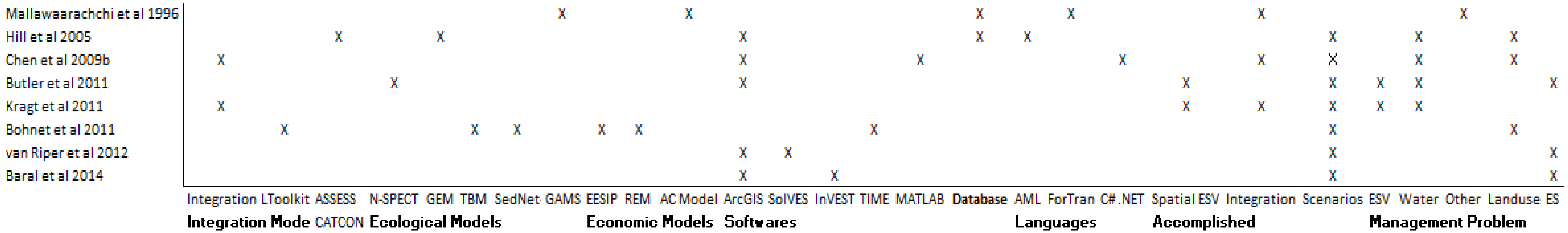


Figure 5b: Presentation of studies having Australia as study area.

Appendix B: Detailed Description of Models

The following is a detailed description and discussion of the 65 studies identified and reviewed in Chapter 4.

Spatial Models

In view of the analysis and results summarising the findings for the literature reviewed, a number of spatial models were identified that have been used, or are currently being implemented, world-wide. For convenience of presentation, these models have been divided into three categories:

1) independent models
2) models that can be used as an extension in ArcGIS or ArcView, and
3) web-based models.

Independent models identified in the literature are Envision, LUCI Polyscape, Landscape Toolkit, Eco-Aim, Eco-Value, ECOMONT, FLUMAGIS, Eco-Dynamo, ABM-SWAT model and the Hydro-ecological model. Models that can be used within ArcGIS or ArcView include ASSESS, SolVES, InVEST, EPIC-View, Biotope Landscape Model, and the AHP-CA-GIS model. Web-based models include Ecosystem Valuation Toolkit, ARIES, MIMES, EcoServ, Co$ting Nature, EPM and InFOREST.

Independent Models

Among the independent models, Envision (formerly known as EvoLand: for Evolving Landscapes) (Bolte *et al.* 2007) is intended to employ a holistic approach to multiple ecosystem services and to be flexible in regards to the types and number of ecosystem services considered. It has been designed to be flexible in spatial and temporal scales. It uses the empirical approaches of principle component analysis (PCA) and agent based modelling (ABM) to quantify ecosystem services using biophysical models. The model is location-specific and data intensive and requires rich data (Bagstad *et al.* 2013). Envision has the capacity to support and explore how development policies affect land use agent behaviour, leading to changes in development patterns and landscape metrics (Bolte *et al.* 2007, Guzy *et al.* 2008). It is an open-source modular model capable of integration with models such as InVEST. It has largely been applied in the U.S Pacific Northwest because it was initially intended for policy research for the State of Oregon (Guzy *et al.* 2008). Bagstad *et al.* (2013) describe the model as place specific and it cannot currently be applied in locations for which it has not been developed, although Bagstad *et al.* (2013) report that implementation studies have commenced in Colombia and New Zealand.

LUCI, formerly known as Polyscape (Bagstad *et al.* 2013, Jackson *et al.* 2013) has been designed to use simple algorithms, and to be robust in situations where there are knowledge and data limitations, and to communicate ecosystem trade-offs in settings with stakeholders and decision makers, assisted by spatially explicit maps designed to be intuitive to use and interpret. Bagstad *et al.* (2013) describe LUCI is an open source GIS toolbox that maps area, portraying gain or loss of services under specified management scenarios, and suggest that it is potentially suitable for widespread use (i.e. generalisable) given the feasibility of conducting a full stakeholder engagement process intended to aid in localising the data and models. The model captures and presents trade-offs between ecosystem services, and is primarily designed for application at local scales. The application of LUCI has been tested at various locations including UK, New Zealand, Ghana and Greece (Bagstad *et al.* 2013).

Landscape Toolkit is another such model (Bohnet *et al.* 2011), and has been used on a case study in the Tully–Murray catchment in the Great Barrier Reef region of Australia. Bohnet *et al.* (2011) claim that Landscape Toolkit links spatially explicit disciplinary models, to enable integrated assessment of water quality, biodiversity and the economic outcomes of stakeholder-defined land use and management change scenarios. The model framework presented by Bohnet *et al.* (2011) explains that the model has capacity to flexibly integrate additional models into its design, and is configured for three existing models: SedNET/ANNEX, EESIP and the TBM.

Eco-AIM and Eco-Value are two of the few proprietary tools that are consultant-driven for mapping ecosystem services and stakeholder preferences at landscape level. They require a trained analyst to operate (Bagstad *et al.* 2013), and are relatively time consuming to run. EcoAIM uses a series of publicly available spatial datasets combined with a weighting or aggregation function to derive spatially explicit scores for ecosystem services of interest (Bagstad *et al.* 2013). EcoAIM can also integrate stakeholder preferences in considering ecosystem service impacts, using a modified risk-analysis approach. Eco-Value combines expert- and literature-derived data to develop ecosystem service production functions and involves a stakeholder engagement process for ecosystem services using focus groups. ESValue specifies the relative values that society, managers, and stakeholders place on ecosystem services, as developed during a stakeholder-engagement process. It carries out non-monetary valuation through ranked analysis of trade-offs by comparing what can be produced (i.e. the production function), with what the stakeholders want to produce (i.e. the valuation function). EcoAIM and EcoValue can be used in Australia for specified purposes, although, it is a proprietary tool and may incur additional costs and conditions.

Other strategies that integrate various disciplinary models include ECOMONT (Cernusca *et al.* 1998), a spatial decision support system developed by the FLUMAGIS project (Volk *et al.* 2008), and a hydro-ecological model developed by (Li *et al.* 2010). ECOMONT uses field measurements and process-based modelling to assess the effects of land-use changes in European terrestrial mountain ecosystems (Cernusca *et al.* 1998); the aim was to investigate the ecological effects of agriculture and forestry induced land use changes. The project was carried out by nine European partners and was headed by the University of Innsbruck. The model facilitated analysis of influencing factors in the ecosystem and the application of landscape scenarios and land use changes scaling from 'the leaf to the landscape level' (Cernusca *et al.* 1998). It also allowed integration of submodules, thus facilitating ecological, economic and socio-political decision making processes. The model is reported to have been data intensive, requiring data rich sites for the model parameterisation in order to support pixel by pixel integration (Cernusca *et al.* 1998).

FLUMAGIS is a spatial decision support system developed to support implementation of the European Water Framework Directive (WFD) for river basin management with GIS (Volk *et al.* 2008). The approach involved integration of ecological and socio-economic assessment methods, GIS-based data and a knowledge base (KB) representing the knowledge and experience of experts and was implemented in Germany (Volk *et al.* 2008). The FLUMAGIS approach integrates simulation models from different disciplines to evaluate river basin management options and forecast their effects on water quality, habitat conditions, and socio-economic aspects.

An ABM-SWAT model reported by Daloğlu *et al.* (2014) was constructed to investigate the influence of policy and farmer characteristics on conservation practice selection and, in turn, their effects on water quality. Daloğlu *et al.* (2014) claim that the model synthesizes social, economic, and ecological aspects of landscape change to evaluate how different agricultural policy and land tenure scenarios and land management preferences affect landscape pattern and downstream water quality. For this purpose, they construct a social ecological system (SES) based on a logistic regression model, which informs an agent based model (ABM) of farmers' adoption of conservation practices. ABM is implemented in Java using Repast J agent-based libraries within the Eclipse integrated environment and is linked to Soil and Water Assessment Tool (SWAT) using MatLab. This framework, thus, integrates land management decisions with soil properties, climate information, and land topography to estimate water quality metrics (Daloğlu *et al.* 2014). The model is run for an agricultural watershed of Lake Erie (Daloğlu *et al.* 2014).

Another model, reported by Li *et al.* (2010) is a hydro-ecological model, with capacity to integrate disciplinary models to support water resource management for the Yellow River Basin, China. The model was developed to simulate hydrologic, ecologic, and economic variables and was used to model four water management scenarios over a 10-year period to reflect alternative future water management pathways. The model provides various outputs including, but not limited to, stream flow, soil water, ground water, crop yield and the monetary value of crop productivity.

ECOMONT was specifically developed for European mountain ecosystems, with the collaboration of nine European partners in six composite landscapes i.e. Eastern Alps, Swiss Alps, Spanish Pyrenees and the Scottish Highlands (Cernusca *et al.* 1998). FLUMAGIS is applicable for river basin management and uses knowledge base (obtained from experts).

EcoDynamo (Liacc 2007) was designed to simulate different physical and biogeochemical processes of aquatic ecosystems. The model integrates existing models and also uses multi-criteria Analytical Hierarchy Process (AHP) and pairwise comparison method for user preferences so that a priority ranking of the pre-processes scenarios can be obtained. EcoDynamo was used with the decision support system to simulate different management scenarios in a coastal lagoon at the South of Portugal and may be widely used for coastal management elsewhere.

Models with ArcView or ArcGIS Integration

Among the models that can be used within ArcGIS/ArcView, four have already been used and implemented in Australia including ASSESS, SolVES, InVEST and AHP-CA-GIS models. ASSESS and AHP-CA-GIS (Hill, Braaten, *et al.* 2005, Chen and Wu 2009) are described in thesis, Chapter 4 'Australian Research'. The following section addresses SolVES, InVEST, EPIC-View and the Biotope Landscape Model.

SolVES (Social Values for Ecosystem Services) is an ArcGIS toolbar that maps quantitative social values for ecosystem services based on survey data or value transfer. It is a spatially explicit and a generalisable tool (Bagstad *et al.* 2013). It is intended to quantify and map the perceived social values for ecosystem services calculated from a combination of spatial and non-spatial responses to public attitude and preference surveys using value typology. The model has been upgraded by integrating the Maxent maximum entropy modelling software into SolVES 2.0, which allows SolVES to produce more complete and robust social value maps. Public value and attribute survey are time consuming, but this model been been successfully implemented in different studies (Sherrouse *et al.* 2011, 2014, Sherrouse and Semmens 2014), including a study conducted in Australia (van Riper *et al.* 2012).

InVEST (Integrated Valuation of Ecosystem Services) is one of the best known and widely used open source, public domain and generalisable software (Vigerstol and Aukema 2011) that can be accessed through ArcGIS (Nelson *et al.* 2009, Bagstad *et al.* 2013, Baral *et al.* 2013, Guswa *et al.* 2014, Boumans *et al.* 2015). InVEST integrates disciplinary models and currently includes nine marine and seven freshwater and terrestrial ecosystem service models and has been used widely at different locations and for a diverse range of purposes (Isely *et al.* 2010, Power 2010, Ren *et al.* 2011, Sánchez-Canales *et al.* 2012, Guerry *et al.* 2012, Bagstad *et al.* 2013, Rao *et al.* 2013, Su and Fu 2013, Fu *et al.* 2013, Leh *et al.* 2013, Baral *et al.* 2014, Wang *et al.* 2014, Boithias *et al.* 2014, Dhakal *et al.* 2014, Hoyer and Chang 2014). The tool is the outcome of the Natural capital Project formed by three key authors of Millennium Ecosystem Assessment (MEA) (Baral *et al.* 2013). InVEST can be used to analyse the effect of different land use scenarios on the provision of diverse ecosystem services.

InVEST provides functionality to map and value ecosystem services, The approaches used for making spatially explicit scenarios within InVEST include; 1) drawing maps with stakeholders; 2) statistical techniques based on past experience and trend analysis and; 3) rule-based approach (McKenzie *et al.* 2012). In the first approach, maps are drawn by working with stakeholders showing where different land and marine uses and development activities would occur for each scenario. A GIS expert then helps to translate the paper scenario maps into digital land use/land cover GIS maps to facilitate the InVEST analysis (McKenzie *et al.* 2012). In the second approach, the most likely change is statistically deduced from past experience. The change is analysed from one time point to another and causal factors are identified to model the probability of change into future (McKenzie *et al.* 2012). In the third approach, rules based on socioeconomic or biophysical principles are defined and areas most suitable for particular activity are highlighted (McKenzie *et al.* 2012). The rules can be defined using focus groups of experts or decision makers (Swetnam *et al.* 2011). InVEST has been employed in north-central Victoria (Baral *et al.* 2014) to highlight areas of high biodiversity conservation value at less modified land cover sites.

Other ecological economic models that require ArcView (a previous version of ArcGIS) to operate include EPIC-View (Rao *et al.* 2000) and Biotope Landscape Model, (Münier

2004). EPIC-View is an integrated system developed to integrate a hydrologic-crop management model called EPIC with the desktop GIS, ArcView, for implementing sustainable farm management practices. The tool has been implemented in the USA (Rao *et al.* 2000) as a planning tool for sustainable farm management. Biotope Landscape model (Münier 2004) was a result of a multidisciplinary research project, addressing the consequences of changes in agricultural production with respect to ecology, environment and economy. The Biotope Landscape Model has been built as an ArcView extension, requiring the ArcView Spatial Analyst extension. An agro-economic model as well as an economic model based on information on land use (crop types), livestock husbandry, and the main soil type of the farm, have been linked to the Biotope Landscape Model, allowing scenario definition and integrated evaluation of results. The model has also been linked to FRAGSTATS that works directly on the vegetation maps generated by the Biotope Landscape Model. Biotope Landscape Model has been implemented in a case study in Denmark (Münier 2004).

Web-based Models

Among the web-based models (Ecosystem Valuation Toolkit, ARIES, EcoServ, Co$ting Nature, EPM, MIMES and InFOREST) the best known generalisable open source and public domain tool is ARIES (Artificial Intelligence for Ecosystem Services) (Vigerstol and Aukema 2011). It biophysically models ecosystem service flows and analyses trade-offs among multiple ecosystem services (Bagstad *et al.* 2013, Baral *et al.* 2013, Guswa *et al.* 2014, Boumans *et al.* 2015). Data and models available in ARIES are for the western US states only but its global model and online interface are under development in order to enable more widespread future use (Bagstad *et al.* 2013). ARIES uses artificial intelligence techniques to pair locally appropriate agent based models with spatial data, enabling quantification of actual ecosystem service provision and use (Bagstad *et al.* 2013, Jackson *et al.* 2013, Häyhä and Franzese 2014). ARIES is extremely flexible in terms of spatial scales, temporal scales, and types and number of ecosystem services in consideration. ARIES uses neural networks, bayesian networks and agent based modelling as modelling approaches. It is highly data intensive, requiring data rich sites for study (Jackson *et al.* 2013), although has been applied in case studies in Arizona (Bagstad, Semmens, and Winthrop 2013).

Multiscale Integrated Models of Ecosystem Services (MIMES) is a spatially explicit, public domain tool that biophysically models ecosystem services and performs monetary economic valuation through input output analysis. It can be used at multiple scales and deals simultaneously with multiple ecosystem services production and demands (Jackson *et al.* 2013, Boumans *et al.* 2015). MIMES is a suite of applications, delivered to end users through a standard web browser. MIMES builds on GUMBO (Global Unified Meta-model of the Biosphere) that integrates earth systems while also modelling social and economic dynamics. MIMES was developed using SIMILE, a commercial coding and simulation software package and thus, can be independently applied if users have access to SIMILE (Andrade *et al.* 2010, Liu, Costanza, Troy, *et al.* 2010, Jackson *et al.* 2013, Boumans *et al.* 2015). MIMES integrates existing biophysical models, thus making it possible to explore potential futures in economic, social and ecological terms over time and space. A recent publication (Boumans *et al.* 2015) reports the application of MIMES at three different scales: at a global scale, at a watershed scale for the Albemarle-Pamlico watershed in North Carolina i.e. a place based model, and for a marine application in the Eastern part of USA in Massachusetts coastal waters as a case study, the last two in Eastern United States. MIMES models are developed as case studies to be applicable for further locations, as has been applied in New Zealand (Manuwatu MIMES) and is under development for the Ministry of Fisheries in Cambodia to help manage fisheries policies on the Tonle Sap Lake (US Government 2015). Although, MIMES is developing further case studies to be applicable in other locations, yet it requires a long lead time to develop it, except where it has previously been developed (e.g. Eastern USA, (Boumans *et al.* 2015)). MIMES is a public domain and place-specific tool, that is customised for application in a particular geographic context as a case study (Bagstad, .

The Ecosystem Valuation Toolkit is a subscription based valuation toolkit that includes a Researcher's Library, SERVES and resources. Researcher's library is a growing searchable database of ecosystem services that can be used for benefit transfer (Earth Economics 2016). SERVES (Simple and Effective Resource for Valuing Ecosystem Services) is a web-based proprietary tool for calculating ecosystem service value and performing natural capital appraisal, whereas, resources are general material and links to further resources around the web (Bagstad *et al.* 2013). The Ecosystem Valuation Toolkit can be used in Australia for specified purposes, although, it will require SERVES, which is a proprietary tool and may incur additional costs and conditions.

Similarly, EcoServ is a web-based tool for ecosystem services modelling and mapping under development by scientists at the USGS and Chinese Academy of Sciences (Bagstad *et al.* 2012) that uses production functions as well as external models to substitute a service of interest. It links external ecosystem process models and spatial data and makes it accessible to public, but does not involve economic valuation of ecosystem services (Bagstad *et al.* 2013). Case studies of EcoServ are still under development in the US and Canada and is still not generalizable until future development of global models (Bagstad *et al.* 2012).

Co$ting Nature is a another web-based, spatially explicit, public domain tool that biophysically models ecosystem services and jointly maps ecosystem services and conservation priorities (Bagstad *et al.* 2013). It can identify potential ecosystem services hotspots, but cannot disaggregate services for trade-off analysis and valuation. It works for global and chosen site analysis. It is designed to help test policies for land use and other interventions by simulating their impact on the distribution of service provision (Silvestri *et al.* 2010, Policy Support Systems 2016). Bagstad *et al.* (2013) suggest that Co$ting Nature can rapidly be applied to terrestrial environments globally.

The Ecosystem Portfolio Model (EPM) and InForest models are both place-specific, web based, public domain tools. Bagstad *et al.* (2013) in their review, proclaim that EPM models land use change dynamics and has been used in South Florida, Washington and Arizona. InForest as described by Bagstad *et al.* (2013) quantifies carbon, watershed nutrients, sediment loads and biodiversity and is designed as an ecosystem credit calculator and is only applicable for the state of Virginia.

Summary and Conclusions

The following observations, drawn from this review, informed the choice of model for the research reported here.

The main purpose of the proposed research is to integrate spatial explicitness into ecological economic modelling of land use decisions, and therefore models that lack or unlikely to facilitate either integration of the spatial dimension, or ecological values or economic values, are not suitable for the current research. The models and strategies that fall in this category include NRM Cradle Coast's strategy to model ecosystem services values (Williams 2009, 2013), Scenario Chooser (Smith *et al.* 2012), CHIA (Pert *et al.* 2013), REM (Knight and Cullen 2010), MCAS-S (Magierowski *et al.* 2014, The Landscapes and Policy Hub 2015), SRIAS (Mallawaarachchi *et al.* 1996), ASSESS (Hill, Braaten, *et al.* 2005, Chen and Wu 2009), AHP-CA-GIS tool (Chen and Wu 2009), SoLVES (van Riper *et al.* 2012), Landscape Toolkit (Bohnet *et al.* 2011), a hydro-ecological model based on Bayesian Network model (Kragt *et al.* 2011), EcoDynamo (Liacc 2007), LUCI Polyscape (Bagstad *et al.* 2013, Jackson *et al.* 2013), Ecosystem Valuation Toolkit (Earth Economics 2016), EcoServ (Bagstad *et al.* 2012, Bagstad *et al.* 2013) and Co$ting Nature (Silvestri *et al.* 2010, Policy Support Systems 2016).

The model adopted for this study must be one that can provide a flexible non-optimised output. Therefore modelling strategies that provide optimised (one potential) output (e.g. agent based modelling) are excluded from consideration, along with the models that use such strategies (e.g. Envision (Bolte *et al.* 2007) and ARIES (Bagstad *et al.* 2013, Jackson *et al.* 2013, Häyhä and Franzese 2014)).

The model adopted for this study is required to engage with a variety of stakeholders and a variety of values, and must be able to integrate non-monetary ecosystem values. Therefore, modelling tools that do not undertake stakeholder engagement and/or do not integrate non-monetary values (e.g. MIMES (Andrade *et al.* 2010, Liu, Costanza, Troy, *et al.* 2010, Jackson *et al.* 2013, Boumans *et al.* 2015) and ARIES (Bagstad *et al.* 2013, Jackson *et al.* 2013, Häyhä and Franzese 2014)) are also excluded from consideration.

InVest is a potentially suitable model and tool for the proposed research, in which case the third of the approaches described above for constructing spatially explicit scenarios would best fit the intentions of the research (i.e. the rule-based approach of McKenzie *et al.* (2012)). However, this would not allow trade-offs of values between and within diverse stakeholders and land uses to be viewed. Consequently, it was not selected as the preferred approach for the work reported here.

The requirements for the current study was to model farmers' decision making and to capture and communicate the trade-offs between and within diverse stakeholders. A method that approaches this purpose was one that integrates the ecological values of ecosystem services and analyses trade-offs between multiple ecosystem services (Butler *et al.* 2013), described in Chapter 4. Butler *et al.* (2013) used indicators and indices to value various ecosystem services using the Millennium Ecosystem Assessment. The study was carried out to identify cause-effect relationships, trade-offs and thresholds between services and stakeholders; however it did not capture the trade-offs of values between and within diverse stakeholders and land uses. Therefore, the ecosystems services strategy adopted by Butler *et al.* (2013) was also not implemented in this research and so adapting that methodology was not considered a likely optimal approach.

The approach developed Daloğlu *et al.* (2014) is intended to synthesize social, economic, and ecological aspects of landscape change, in their case in order to evaluate how different agricultural policy and land tenure scenarios and land management preferences affect landscape pattern and downstream water quality. Their study integrated Agent Based Modelling (ABM) and a Soil and Water Assessment Tool (SWAT) to integrate land management decisions with soil properties, climate information, and land topography. In each ABM simulation, the farmer agent is informed by a social ecological system (SES) based on a behavioural model and attributes of the farmers. The farmer agents decide whether to adopt or not to adopt a specific conservation practice. A logistic regression model was used to define the behavioural model that guided the farmer agent's decision-making. The farmer agent's decision updates the landscape management (Daloğlu *et al.* 2014). The logistic regression model in Daloğlu *et al.* (2014), incorporates agricultural profit dynamics ($F_{econ\ (i,\ j)}$), intrinsic attributes of the farmers ($F_{profile\ (i,\ j)}$), the influence of the social network of the farmer ($F_{profile\ (i,\ j)}$) and the influence of a spatial network)(the farmer's neighbours), on the farmer agent decision.

The methods of Daloğlu *et al.* (2014), particularly the logistic regression model, has the capacity to be spatially enabled, can integrate ecological and economic values, and has the capacity to generate scenarios and allow stakeholder engagement, and is thus, regarded as an appropriate starting point for model development for this research.

Appendix C: Modelling Strategies undertaken in Tasmania

Numerous research projects addressing natural resource management have been carried out in Tasmania and, specifically, in the Midlands region. Many of these were spatially explicit in nature; i.e. they reflected upon the arrangement of these natural resources in space and locality, considering either, spatial boundaries, spatial adjacency/proximity; or spatially associated attributes. A small number of these have focussed explicitly on modelling tools in an integrated environment combining the ecological dynamism, economic aspects and social values.

In Tasmania, some studies have focussed on the ecological and economic value of marine ecosystems (Foley *et al.* 2010), others have contributed towards acquiring public opinion on natural resources within the region, including perception and values on the coastal areas (Trenouth *et al.* 2012) and plantation forestry (Anderson *et al.* 2013, Williams 2014), while others focussed on landowner's preferences and behaviour towards decisions about engagement with conservation programs (van Ingrid *et al.* 2011).

The first project to use an ecosystem services concept at a regional level in Tasmania was conducted by Cradle Coast NRM, in northwest Tasmania, concentrating on the Leven and Tarkine catchments. In this project, ecosystem services in the Cradle Coast were first identified and then mapped (Williams 2009, 2013). Given the lack of an appropriate and easily deployed tool, the project developed its own framework and approach to map ecosystem services and values and the benefits they provide. A range of variables were used to consider eight ecosystem service themes chosen for examination and mapping (Williams 2009, 2013).

In order to elicit public opinion on forested landscapes, a tool known as Scenario Chooser was developed (Smith *et al.* 2012), that has the capacity to model preferred management scenarios in a spatial manner. Scenario Chooser is a web based tool developed to support public assessment of scenarios and potential future forest landscapes. It was developed to present a range of hypothetical future forested landscapes to the public with the aim of eliciting the most or the least preferred forest management scenarios. The goal of the research was to develop an interactive interface to present multiple scenarios for comparison and selection and to evaluate the effectiveness of the interface by analysing the participant interaction logs and the evaluation questionnaire. The tool ascertained people's landscape preferences by presenting information in an easily understandable manner through 3D panoramas and quantifiable outputs.

Another such tool for capturing social and ecological dynamism in conservation prioritisation has been applied in Queensland, Australia, and can be generalised to other areas in future (Pert *et al.* 2013). The participatory tool for conservation prioritization is called The Collaborative Habitat Investment Atlas (CHIA). Pert *et al.* (2013) argue that the model has robust dynamic capability for the stakeholder's interaction. It allows participants to modify variable weights to investigate the varied biodiversity protection requirements and the results updated immediately depicts parcel based maps of three models i.e. biodiversity importance, level of protection and level of threat. Also, suitability maps are generated that aid in conservation decision making.

Another tool known as Regional Ecosystem Model (REM) is a spatial system for storing data on biodiversity, assessing relationships in ecosystems, and identifying priorities for management. It is built around a spatial data model that stores all base and derived data in a single GIS layer for each of the three Asset Classes i.e. land and soils, freshwater and biodiversity in the Tasmanian Midlands. The REM inherits a list of Natural Resource Management (NRM) Issues, Assets and Asset Classes, and the relationships between them, from the project's Strategy Review (Knight and Cullen 2010). The Issues identified in the Review are used as inputs to the REM to be addressed in management. The structure is reflected in a series of matrices that integrate the prioritisation of each Issue with other Issues in the same Asset or Asset Class to determine Level of Concern. REM generates Level of Concern and identifies management priorities through input matrices/indices on biological significance and landscape function (Knight and Cullen 2010). REM also has a spreadsheet version that assists in stand-alone application of the model and allows a user to produce outputs from the REM for a number of sites or areas, and to test the impacts of the relative priority for further management actions (Knight 2010). The REM has been adapted for use in a wide variety of situations, including urban development and planning, property management and planning, prioritising funding for on ground works, in forest management and forest certification (Knight 2010, Knight and Cullen 2010, Green and Knight 2015).

Another spatially explicit model that has been applied in Tasmania is the Criteria Analysis Shell for Spatial Support (MCAS-S) model. MCAS-S is a decision support tool designed for non–GIS users to integrate spatial data, developed by Australian Bureau of Agricultural and Resource Economics and Sciences. It has the ability to cater for and analyse large amounts of environmental, social and economic information, thus overlaying and combining maps of management scenarios to inform policy decisions through multi-criteria analysis (Hill, Lesslie, *et al.* 2005, Lesslie *et al.* 2008). Another multi-criteria interface called ASSESS, that works in ArcInfo has been widely used, (ASSESS has been discussed in detail in section on ‘Australian Research’). Thus, MCAS-S builds on ASSESS by providing an interface that is designed for a non-GIS user (Hill, Lesslie, *et al.* 2005). Among other projects using MCAS-S in Australia, a project called ‘The Midlands Aquatic Refuge’ was carried out in Tasmanian Midlands. The project used MCAS-S to identify potential refuges or sanctuaries from predation, climate change and other threatening processes by overlaying maps of suitable habitat and likely threats (Magierowski *et al.* 2014, The Landscapes and Policy Hub 2015). (Hill, Lesslie, *et al.* 2005, Lesslie *et al.* 2008)

Appendix D: Normalisation of algorithm outputs

$New\ Value = \frac{(max' - min')}{(max - \mathrm{min})} * (\mathrm{Value} - \mathrm{max}) + \mathrm{max}'$ ……………………………….(Eq. 1)

We have a value V, which lies between a minimum A and a maximum B range of values. We wish to rescale it to a new value v so it is between a and b instead. The above formula can be written as:

$v\ = \frac{(\mathrm{b-a})}{(\mathrm{B-A})} * (\mathrm{V} - \mathrm{B}) + \mathrm{b}$ ………………………………………. (Eq. 2)

In the rescaling, we preserve the ratio (V-B)/(B-A), so that it is the same as the ratio (v-b)/(b-a):

i.e. $\frac{(\mathrm{V-B})}{(\mathrm{B-A})} = \frac{(v - \mathrm{b})}{(\mathrm{b-a})}$

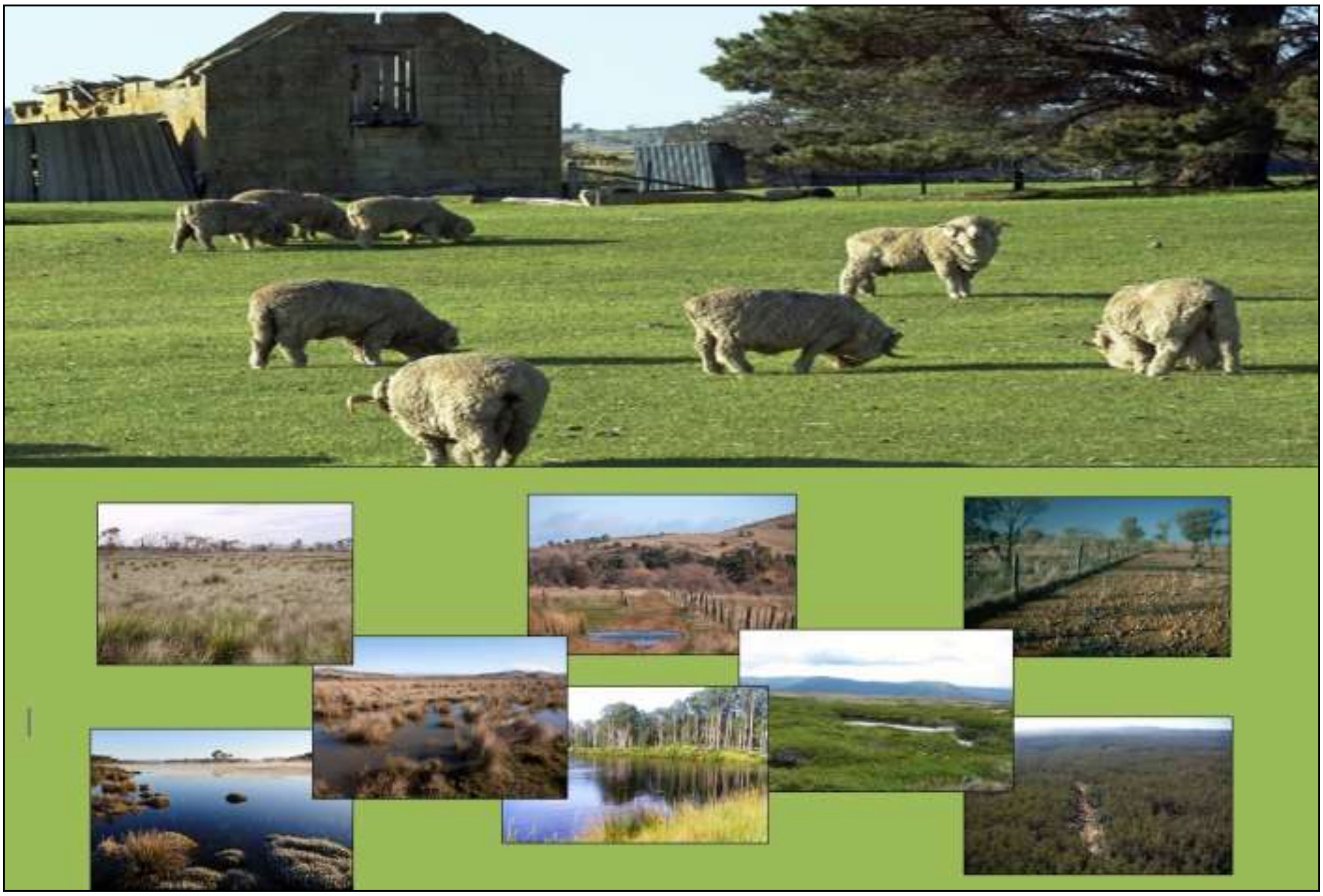
So $v = \frac{(\mathrm{V-B})}{(\mathrm{B-A})} * (\mathrm{b} - \mathrm{a}) + \mathrm{b}$

i.e. $v = \frac{(\mathrm{b-a})}{(\mathrm{B-A})} * (\mathrm{V} - \mathrm{B}) + \mathrm{b}$ ………………………………………… (Eq. 3)

Therefore, the Eq. 1, 2 and 3 are similar and contributing to the same concept, therefore, Eq. 1 is used for normalisation of values in the model.

Appendix E: Questionnaire

How do landscape values affect land use?



Researcher: Juwairia Mahboob
Supervisory Team: Dr Jon Osborn, Dr Jagannath Aryal and Assoc Prof Rohan Nelson
School of Land and Food
Department of Geography and Spatial Sciences
University of Tasmania

UNIVERSITY of TASMANIA

Why value agricultural landscapes?

In this project, we would like to work with you to understand how you value agricultural landscapes, and how these values affect your preferences for the way agricultural land is used now and into the future. The information you provide will help us to map the values that land managers and others associate with different types of land use. This will enable us to provide you with a better understanding of why current patterns of land use have evolved, and how land use may evolve into the future under alternative scenarios. For example, what will Tasmania's midlands look like in 20 years' time as a result of expanded irrigation infrastructure?

We need your insights

Your local knowledge will help us to create models that are useful for informing decisions about future land use. Geographic information systems (GIS) enable us to associate the ecological attributes of land, with the economic values that you tell us are associated with these attributes. We can already map some of the more important attributes of land such as its productivity class, current use, area, soil type, vegetation, elevation slope, as well as proximity to roads and population centres. You can help us to understand the relative importance of these attributes in driving land use change, and how land use change is perceived from different government, industry, and community perspectives.

Who is doing this?

This interview process is part of PhD research by Juwairia Mahboob at the University of Tasmania. Juwairia is being supervised by Drs Jon Osborn, Jagannath Aryal and Rohan Nelson from the School of Land and Food.

The survey process

To help us to understand how you value the attributes of land, Juwairia has constructed a *typical* agricultural property. This farm, Maryland, in not a real farm but has a variety of land uses typical of the midlands region of Tasmania. In the survey that follows, Juwairia will ask you to place a relative value on various attributes of land. When you answer these questions, we want you to answer as best you can based on the scenario presented to you and your knowledge of midlands agriculture. For example, if you are a land manager, we would like you to answer the questions as though you managed this farm.

This survey starts by introducing you to Maryland farm.

Confidentiality

This project is subject to UTAS's ethical standards. This means that we will use the information you provide us carefully, and respect your privacy. We will not use information you provide us in ways that could be used to identify you.

Welcome to Maryland Farm

We'd like to start by introducing Maryland Farm to you.

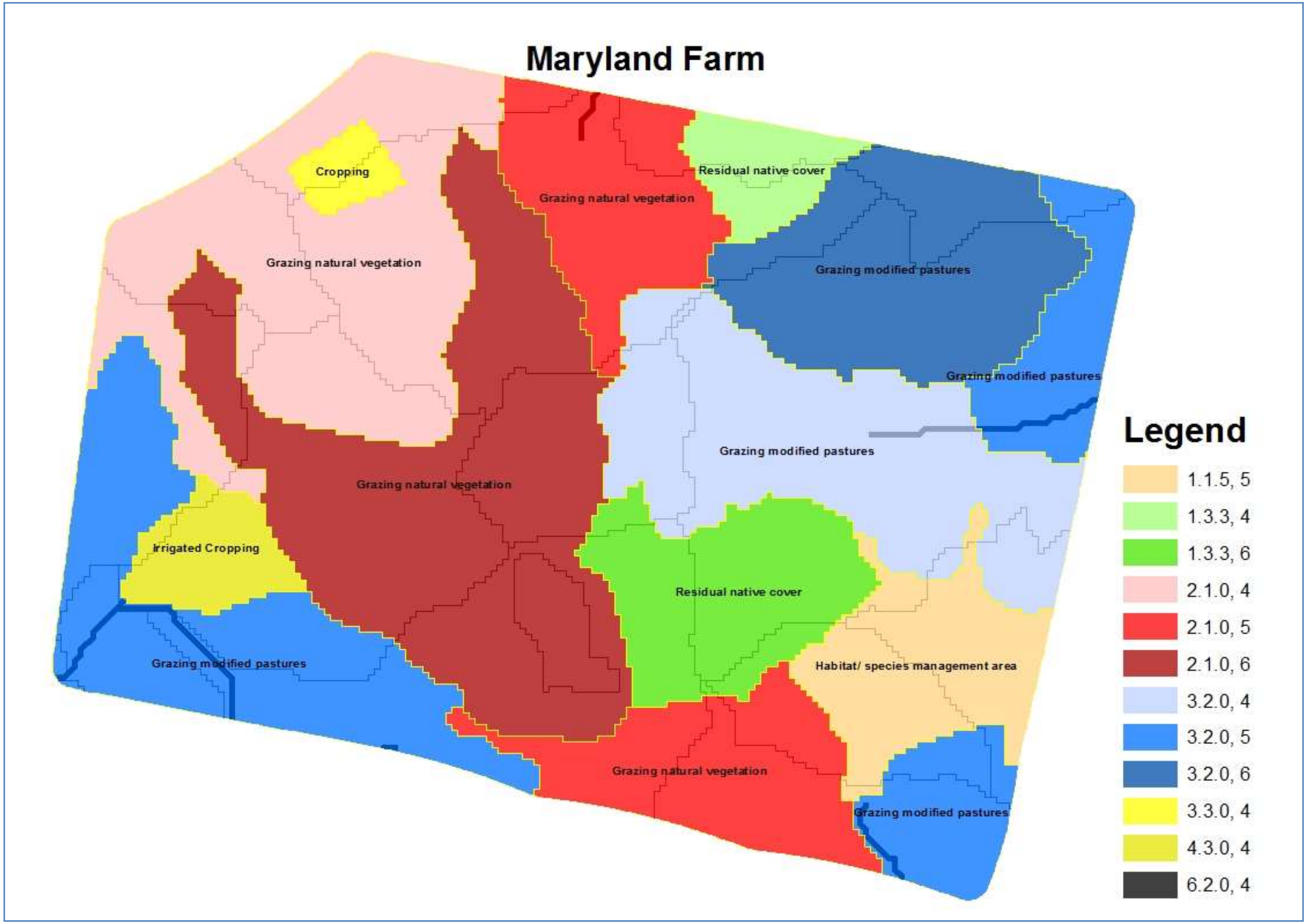
The maps and images that follow are probably the best way to learn about Maryland Farm. The maps show topography (elevation, slope, aspect), hydrology (drainage lines), and current land use. Catchment boundaries are shown with black lines. Drainage lines are shown as blue or white lines. The land use in combination with land capability boundaries are shown with yellow lines.

The dominant soil type on this farm is Chromosol. The farm includes the following *land uses* and *land capabilities* with their respective area (hectares) also shown in the Table below. Detailed information about the *land use classifications* and *land capability classifications* is provided in a separate document, in case you want to refer to it.

Land use Code	Land use	Land Capability	Area in Hectares
1.1.5	Habitat/ species management area	5	68
1.3.3	Residual Native Cover	4	25
1.3.3	Residual Native Cover	6	77
2.1.0	Grazing Natural Vegetation	4	174
2.1.0	Grazing Natural Vegetation	5	146
2.1.0	Grazing Natural Vegetation	6	230
3.2.0	Grazing Modified Pasture	4	150
3.2.0	Grazing Modified Pasture	5	204
3.2.0	Grazing Modified Pasture	6	115
3.3.0	Cropping	4	11
4.3.0	Irrigated Cropping	4	28
6.2.0	Reservoir	4	
TOTAL			1231

Land Capability Class	Short Description – see separate document for full description
Class 4	Land primarily suitable for grazing but which may be used for occasional cropping. Severe limitations restrict the length of cropping phase and/or severely restrict the range of crops that could be grown. Major conservation treatments and/or careful management is required to minimise degradation.
Class 5	This land is unsuitable for cropping, although some areas on easier slopes may be cultivated for pasture establishment or renewal and occasional fodder crops may be possible. The land may have slight to moderate limitations for pastoral use.
Class 6	Land marginally suitable for grazing because of severe limitations. This land has low productivity, high risk of erosion, low natural fertility or other limitations that severely restrict agricultural use. This land should be retained under its natural vegetation cover.

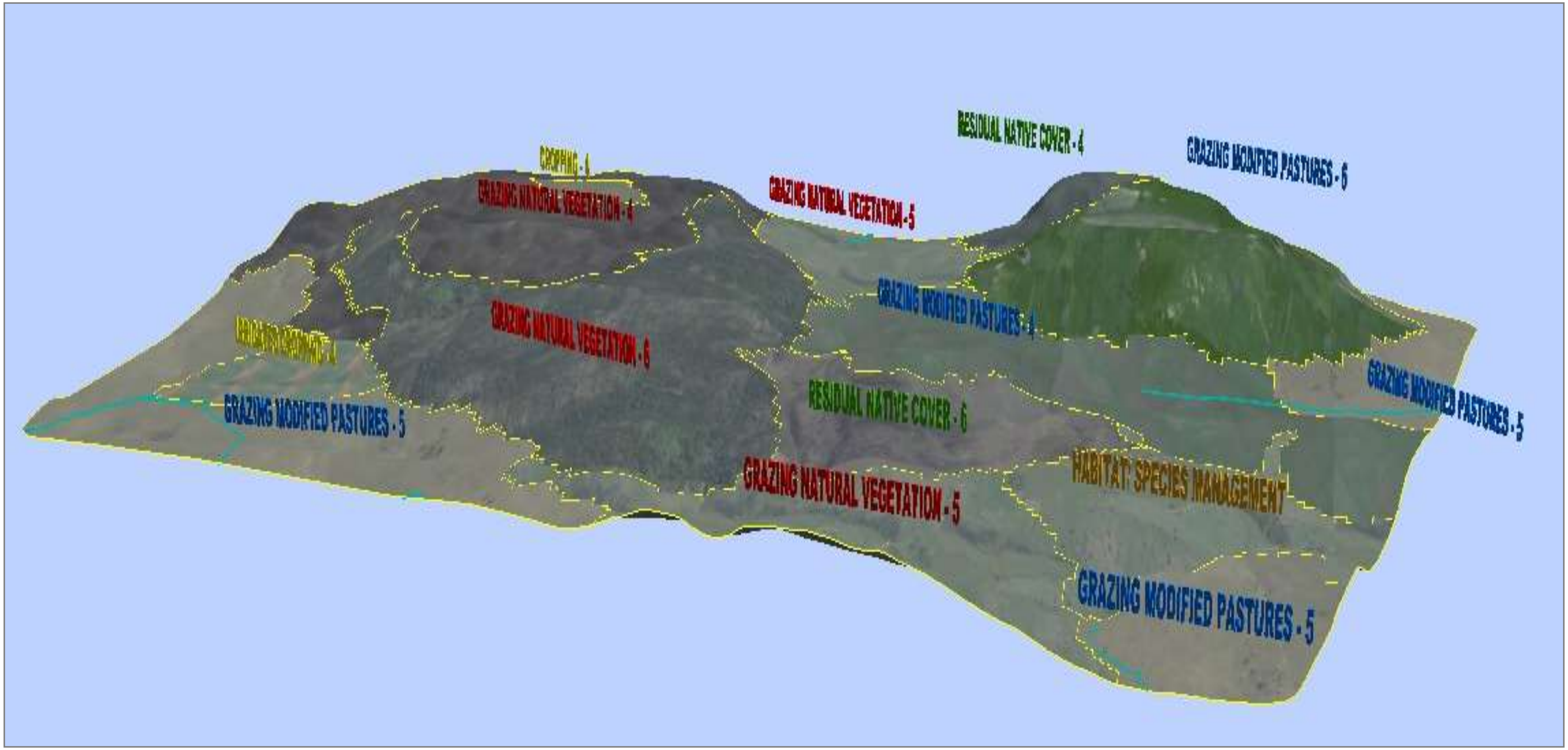
Land Uses on Maryland Farm



Land Uses on Maryland Farm – Plan View

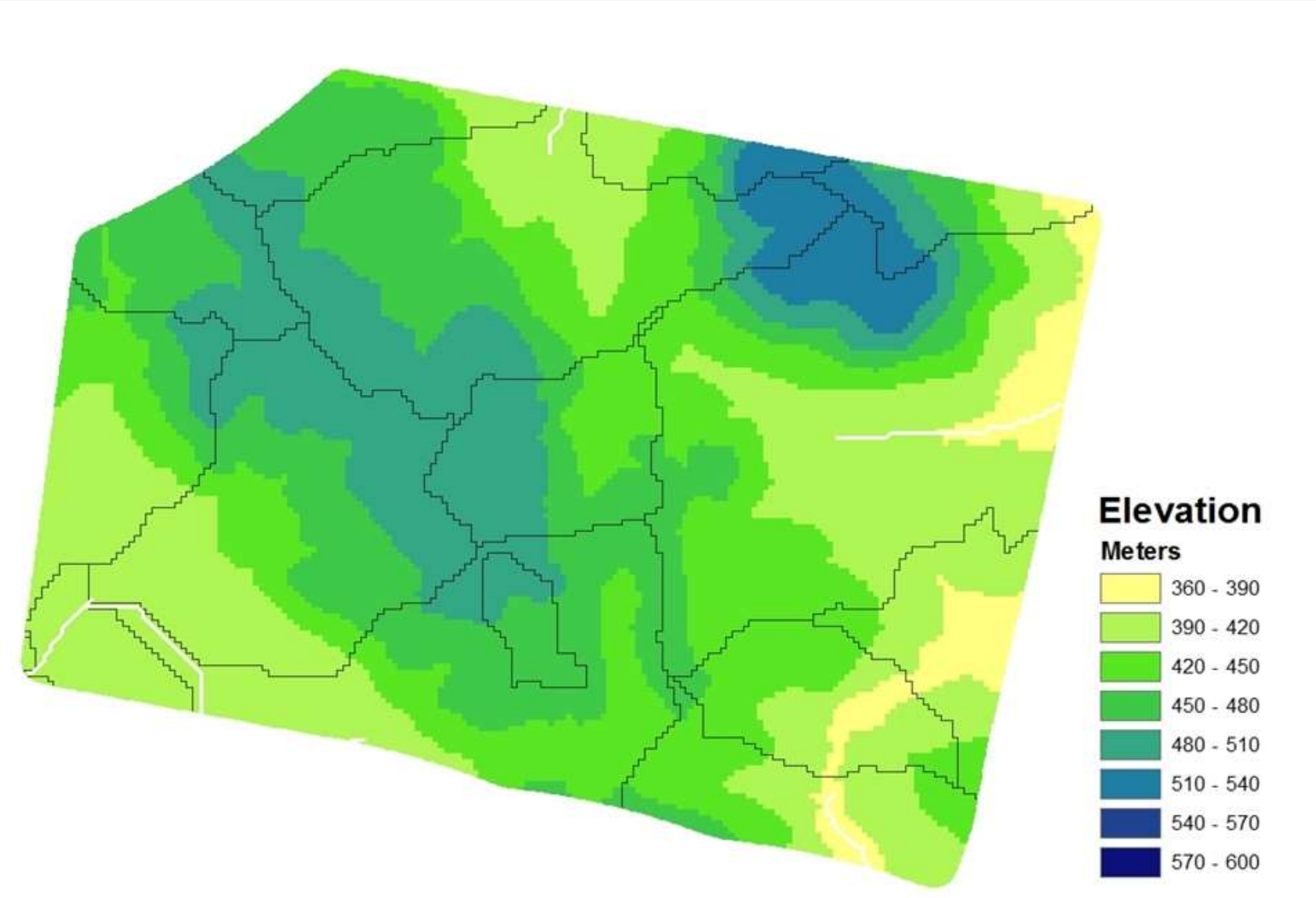
Land Uses on Maryland Farm – Birds-eye View, looking to the Northeast

Land Uses on Maryland Farm – Birds-eye View, looking to the Northwest

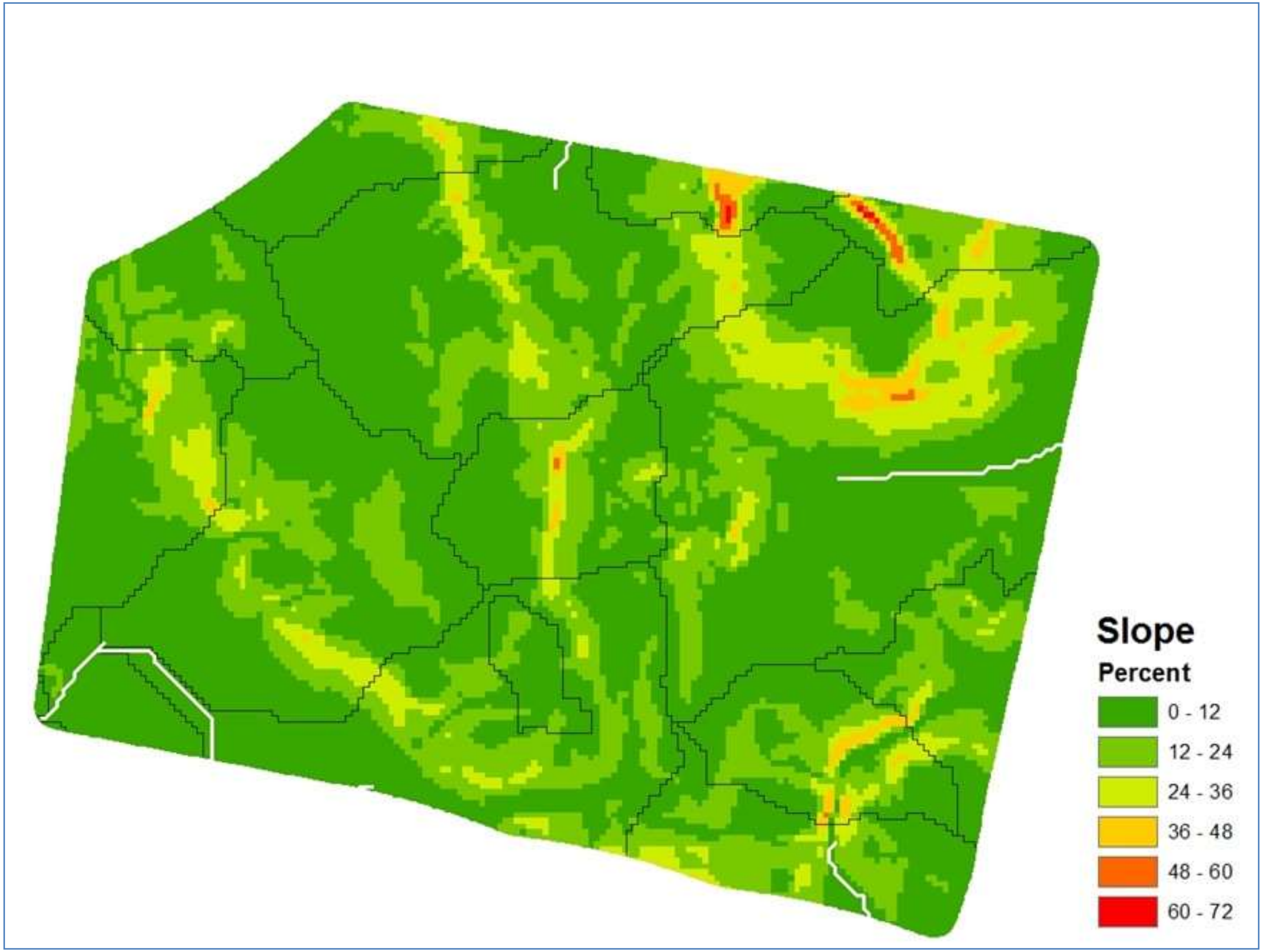


Land Uses on Maryland Farm – Birds-eye View, looking to the North

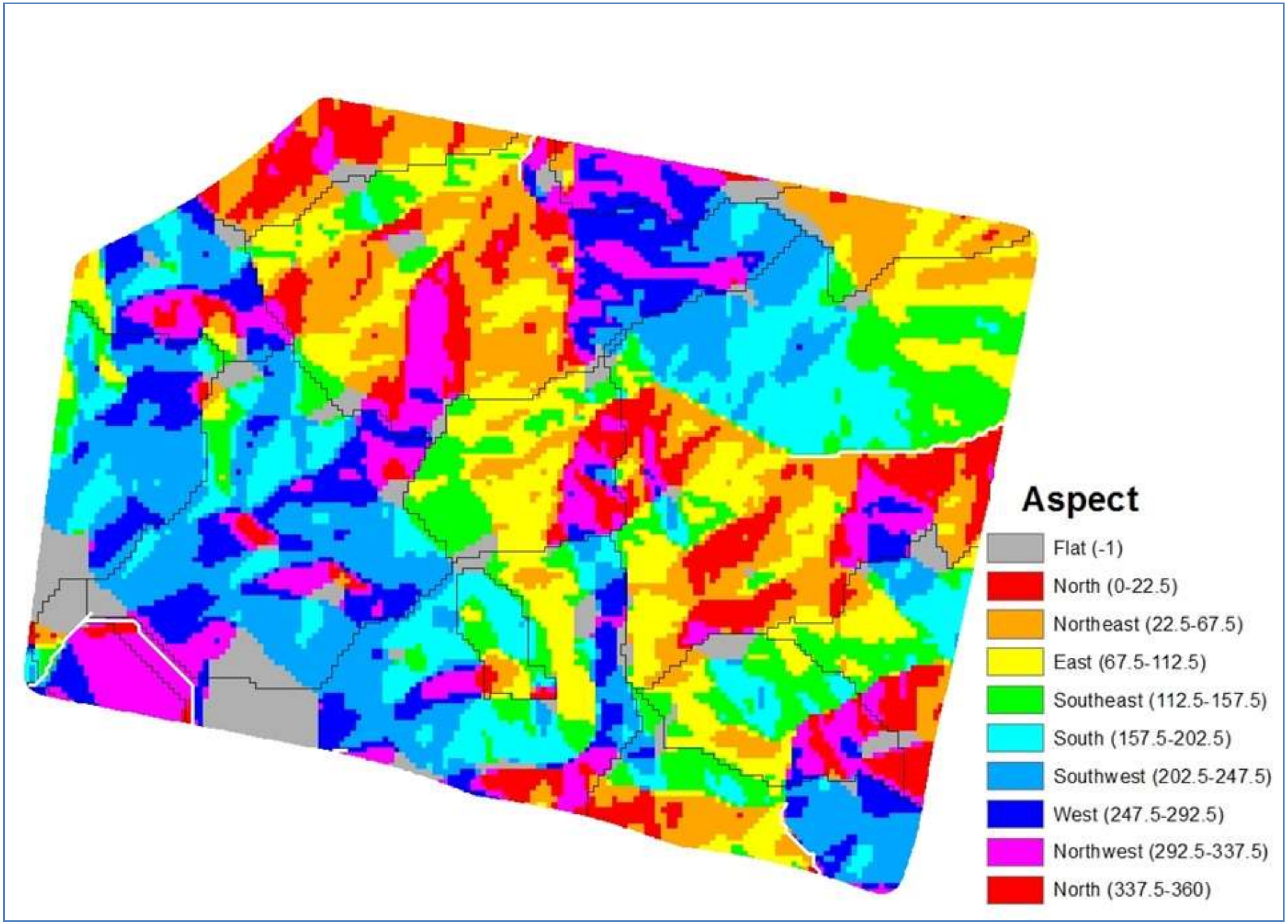
Maryland Farm – Elevation



Maryland Farm – Slope



Maryland Farm – Aspect



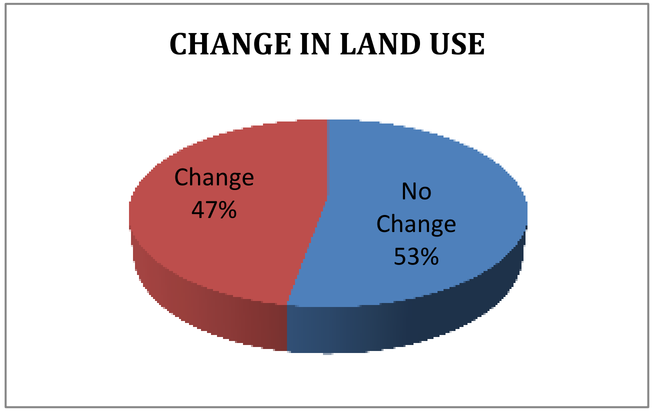
Questionnaire

As noted before, we understand that you won't have the experiences and knowledge of Maryland Farm that you would have with any real farm that you actually own, manage, or have an interest in. But that doesn't matter too much for our survey. Your participation will help us to learn how best to gather information about the ways in which key stakeholders value agricultural land, and we hope it will also provide some sensible data that Juwairia can use to populate her prototype model.

We now ask you to respond to some questions about the values you associate with the different land uses on Maryland Farm.

Question A

Records show that land use on about 165,000 hectares of land in the South East Midlands region has changed during the last decade. That's nearly half of the total area.



- Please give one example of a land use change that you have been involved with, or observed, in the Midlands region. Please give just one example.
    - What was the land originally used for?
    - What has the use changed to?

When you describe the initial and changed land use, you might choose the land uses listed on page 2, but it is fine to use your own words.

Initial land use:

____

Changed land use: ____

Reason for the change to land use:

____

____

____

____

____

Question B

We'd now like to know something about your interests and experiences.

1. What activities are you involved in Midlands region? Please tick all responses that are correct for you.
   a. Farm business
      Owner/operator or Operator or Absentee owner
      i. Mostly cropping
      ii. Mostly grazing
      iii. Mixed cropping/grazing
   b. Forest
      Owner/operator or Operator or Absentee owner
      i. Mostly plantation
      ii. Mostly natural forest
      iii. Mixed plantation/natural
   c. Recreation
      i. Camping
      ii. Walking
      iii. Fishing
      iv. Hunting
      v. Other recreation ………………………….
   d. Other

2. Do you live in the midlands region?
   a. No
   b. Yes: on a farming property
   c. Yes: on a non-farming property (but not in a township)
   d. Yes: in a town/city

3. Do you use land in the Midlands for recreational purpose?
   a. No
   b. Yes

   If Yes, please tell us what types of recreation you engage in?____

4. Do you attend or engage in meetings conducted by any of the following organisations? Please tick any or all that apply to you.
   a. Tasmanian Farmers and Graziers Association (TFGA)
   b. Landcare
   c. Tasmanian Land Conservancy (TLC)
   d. Natural Resource Management (NRM)
   e. Other ………………………….
   f. None

5. How often do you attend or engage in State or Local Government meetings that might influence land use policies in the Midlands region?
   a. Occasionally
   b. Often

c. Not at all
d. Please specify what meetings you attend:…………………………………

6. Are you currently, or have you ever, contributed to research on any aspect of farming, grazing, forested, or natural land in Midlands region?
    a. Yes
    b. No

7. Have you ever applied for, or received, stewardship payments associated with the natural values of your land?
    a. Yes
    b. No
    c. Please specify……………………………..

Question C

We would now like to learn about the values you hold for different land uses in the Midlands region. We will only ask about land uses that occur on Maryland Farm, or land use changes that we might expect for this farm. Again, we understand that you will not have the experiences and knowledge of Maryland Farm that you would have with any real farm that you actually own, manage or have an interest in, but please try to respond in a way that reflects the values you believe you would hold if Maryland Farm was a property that you did know well.

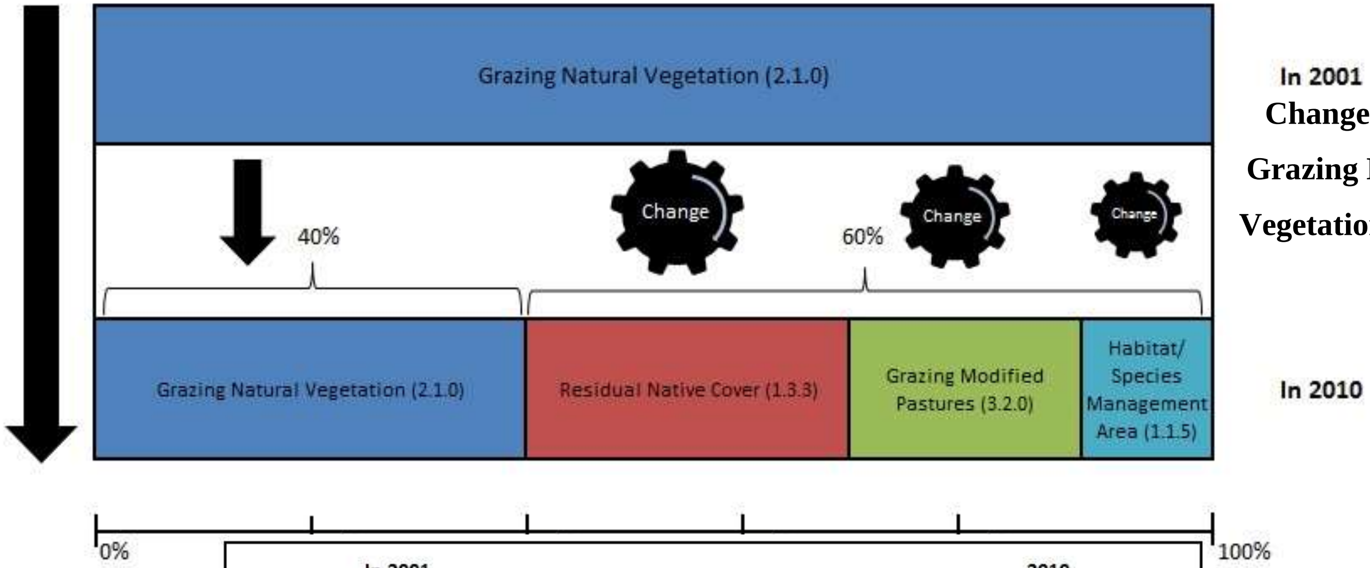
GRAZING NATURAL VEGETATION (LAND USE 2.1.0)

On Maryland Farm, there is native vegetation being grazed (known by its land use as ***Grazing Natural Vegetation***). The primary use of grazing sheep on this land is fine wool production. The land has land capability level 6. The picture below shows an example of Grazing Natural Vegetation being grazed by sheep and the table explains what Grazing Natural Vegetation really means according to Australian Land Use Management (ALUM) classification.

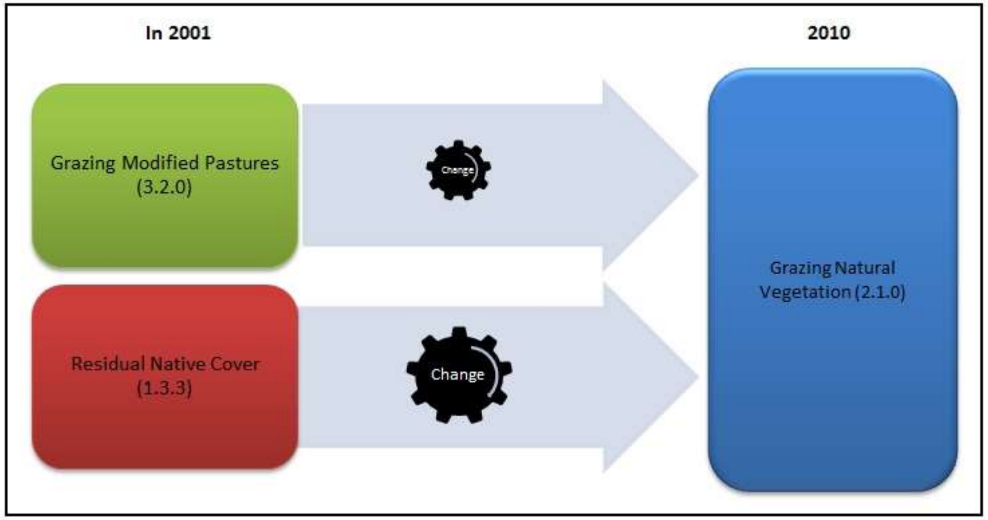
Grazing Natural Vegetation (2.1.0)


Land use ALUM Code	Land Use Level 1	Land Use Level 2	Land Use Level 3
2.1.0	Production from Relatively Natural Environments	Grazing natural vegetation	Grazing natural vegetation
2.1.0 **Grazing Natural Vegetation** **Low level of intervention – Grazing of domestic stock on native vegetation**	**2 Production from relatively natural environments** This class includes land that is subject to relatively low levels of intervention. The land may not be used more intensively because of its limited capability. The structure of the native vegetation generally remains intact despite deliberate use, although the floristics of the vegetation may have changed markedly. Where the native vegetation structure is, for example, open woodland or grassland, the land may be grazed. Where native grasses have been deliberately and extensively replaced with improved species, the use should be treated under class 3, 'Production from dryland agriculture and plantations'. **2.1 Grazing Natural Vegetation** Land uses based on grazing by domestic stock on native vegetation where there has been limited or no deliberate attempt at pasture modification. Some change in species composition may have occurred. For ALUM purposes, this class is used when there is greater than 50 per cent dominant native species.		

Records show that about 60% of Grazing Natural Vegetation in the south-eastern midlands has been subject to change over the last decade. There have been two-way changes in the past decade in relation to Grazing Natural Vegetation i.e. ***From*** Grazing Natural Vegetation and ***To*** Grazing Natural Vegetation. ***From Grazing Natural Vegetation***, the change has been to Residual Native Cover (1.3.3), Grazing Modified Pastures (3.2.0) and Habitat/ Species Management Area (1.1.5). The change has also taken place ***To Grazing Natural Vegetation*** from Residual Native Cover (1.3.3) and Grazing Modified Pastures (3.2.0).



Change From Grazing Natural Vegetation (2.1.0)



Change To Grazing Natural Vegetation (2.1.0)

We would like to learn about the ***values*** you have for the land use: <u>**Grazing Natural Vegetation.**</u>

Question 1.1:

Please indicate the values you hold for the land use: Grazing Natural Vegetation.

Values	Strongly agree	Slightly agree	Neither agree nor disagree	Slightly disagree	Strongly disagree
Aesthetic value – I value this land use because I enjoy the scenery, sights, sounds, smells, etc.					
Biological diversity value – I value this land use because it provides a variety of fish, wildlife, plant life, etc.					
Cultural value - I value this land use because it is a place for me to continue and pass down the wisdom and knowledge, traditions, and way of life of my ancestors.					
Economic value – I value this land use because it provides an economic return.					
Future value – I value this land use because it allows future generations to know and experience the land as it is now.					
Historic value – I value this land use because it has places and things of natural and human history that matter to me, others, or the nation.					
Intrinsic value – I value this land use in and of itself, whether people are present or not.					
Learning value – I value this land use because we can learn about environment through scientific observation and experimentation.					
Life Sustaining value – I value this land use because it helps produce, preserve, clean and renew air, soil and water.					
Recreation value – I value this land use because it provides a place for my favourite outdoor recreation activities.					
Spiritual value – I value this land use because it is a sacred, religious, or spiritually special place to me or because I feel reverence and respect for nature here.					
Therapeutic value – I value this land because it makes me feel better physically and/or mentally.					
Other reason/s – Please mention the reason here:					

Question 1.2:

Please consider the area of Grazing Natural Vegetation (2.1.0) on Maryland Farm with land capability 6. Knowing the associated information about land capability, elevation, slope, aspect, dominant soil, hydrology, area of land, surrounding land uses and any relevant policies in the

Midlands area, we would like to know what factors would influence your decision to support (or to not support) retaining land use Grazing Natural Vegetation rather than change this area to another land use.

Decision Factors	Strongly influence me to **support a change** to any other land use	Slightly influence me to **support a change** to any other land use	No influence	Slightly influence me to **support NO change** to any other land use	Strongly influence me to **support NO change** to any other land use	Please add any comments to explain your decision.
Land Capability						
Elevation						
Slope						
Aspect						
Dominant Soil						
Drainage Lines						
Area of land						
Surrounding land use						
Policy						
Other 1:						
Other 2:						
Other 3:						

Question 1.3:

Would any of the values you have placed on land use Grazing Natural Vegetation (e.g Aesthetic, Cultural, Economic etc.) be affected, if the size of the land use parcel considered was larger or smaller? If Yes, explain how and why

____

____

____

Question 1.4:

Would any of the values you have placed on land use Grazing Natural Vegetation (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to other similar parcels having similar land use? If Yes, explain how and why

____

____

____

Question 1.5:

Would any of the values you have placed on land use Grazing Natural Vegetation (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to transport routes? If Yes, explain how and why

____

____

____

Question 1.6:

Would any of the values you have placed on land use Grazing Natural Vegetation (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to irrigation and/or farm infrastructure? If Yes, explain how and why

____

____

____

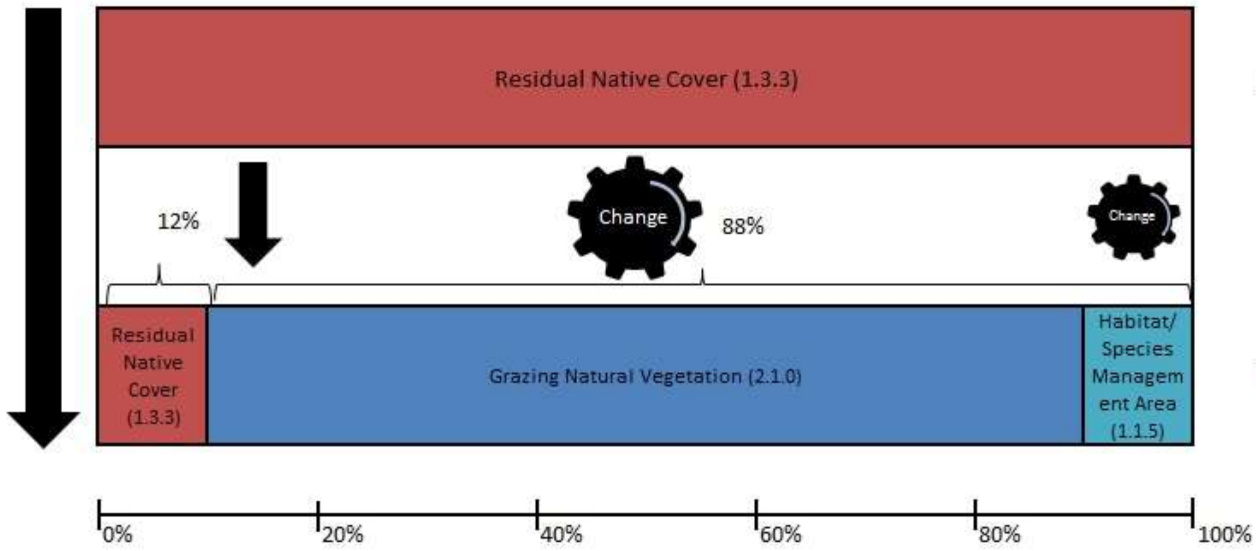
RESIDUAL NATIVE COVER (LAND USE 1.3.3)

On Maryland Farm, there is native vegetation cover having no prime use (known by its land use as ***Residual Native Cover***). The native cover is sometimes conserved for environmental purposes. The land has land capability level 6. The pictures below show examples of Residual Native Cover and the table explains what Native Residual Cover really means according to Australian Land Use Management (ALUM) classification.

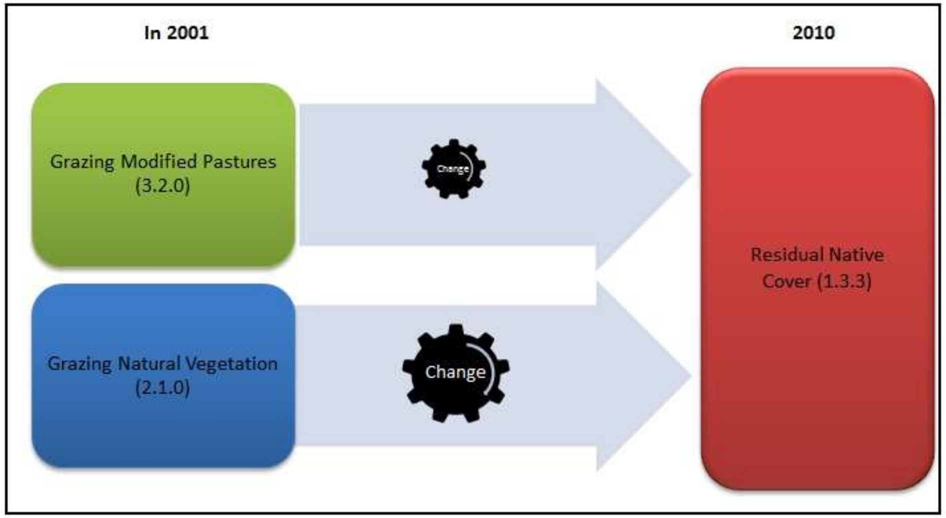
Residual Native Cover (1.3.3)

Land use ALUM Code	**Land Use Level 1**	**Land Use Level 2**	**Land Use Level 3**
1.3.3	Conservation and Natural Environments	Other minimal use	Residual native cover
1.3.3 **Residual Native Cover** **No prime use native cover land used for any environmental purpose**	**1 Conservation and Natural Environments** This class includes land that has a relatively low level of human intervention. The land may be formally reserved by government for conservation purposes, or conserved through other legal or administrative arrangements. Areas may have multiple uses, but nature conservation is the prime use. Some land may be unused as a result of a deliberate decision of the government or landowner, or due to circumstance. **1.3 Other Minimal Use** Areas of land that are largely unused (in the context of the prime use) but may have ancillary uses. This may be a deliberate decision by the land manager or the result of other circumstances. The land may be available for use but remain ‘unused’ for various reasons. **1.3.3 Residual Native Cover** Land under native cover, mainly unused (no prime use) or used for non-production or environmental purposes (e.g. to conserve native vegetation and wildlife, or for natural resources protection).		

Records show that about 88% of Residual Native Cover in the south-eastern midlands has been subject to change over the last decade. There have been two-way changes in the past decade in relation to Residual Native Cover i.e. ***From*** Residual Native Cover and ***To*** Residual Native Cover. ***From Residual Native Cover***, the change has been to Grazing Natural Vegetation (2.1.0) and Habitat/ Species Management Area (1.1.5). The change has also taken place ***To Residual Native Cover*** from Grazing Natural Vegetation (2.1.0) and Grazing Modified Pastures (3.2.0).



Change From Residual Native Cover (1.3.3)



Change To Residual Native Cover (1.3.3)

We would like to learn about the ***values*** you have for the land use: **<u>Residual Native Cover.</u>**

Question 2.1:

Please indicate the values you hold for the land use: Residual Native Cover.

Values	Strongly agree	Slightly agree	Neither agree nor disagree	Slightly disagree	Strongly disagree
Aesthetic value – I value this land use because I enjoy the scenery, sights, sounds, smells, etc.					
Biological diversity value – I value this land use because it provides a variety of fish, wildlife, plant life, etc.					
Cultural value - I value this land use because it is a place for me to continue and pass down the wisdom and knowledge, traditions, and way of life of my ancestors.					
Economic value – I value this land use because it provides an economic return.					
Future value – I value this land use because it allows future generations to know and experience the land as it is now.					
Historic value – I value this land use because it has places and things of natural and human history that matter to me, others, or the nation.					
Intrinsic value – I value this land use in and of itself, whether people are present or not.					
Learning value – I value this land use because we can learn about environment through scientific observation and experimentation.					
Life Sustaining value – I value this land use because it helps produce, preserve, clean and renew air, soil and water.					
Recreation value – I value this land use because it provides a place for my favourite outdoor recreation activities.					
Spiritual value – I value this land use because it is a sacred, religious, or spiritually special place to me or because I feel reverence and respect for nature here.					
Therapeutic value – I value this land because it makes me feel better physically and/or mentally.					
Other reason/s – Please mention the reason here:					

Question 2.2:

Please consider the area of Residual Native Cover (1.3.3) on Maryland Farm with land capability 6. Knowing the associated information about land capability, elevation, slope, aspect, dominant soil, hydrology, area of land, surrounding land uses and any relevant policies in the Midlands area, we would like to know what factors would influence your decision to support (or to not support) retaining land use Residual Native Cover rather than change this area to another land use.

Decision Factors	Strongly influence me to **support a change** to any other land use	Slightly influence me to **support a change** to any other land use	No influence	Slightly influence me to **support NO change** to any other land use	Strongly influence me to **support NO change** to any other land use	Please add any comments to explain your decision.
Land Capability						
Elevation						
Slope						
Aspect						
Dominant Soil						
Drainage Lines						
Area of land						
Surrounding land use						
Policy						
Other 1:						
Other 2:						
Other 3:						

Question 2.3:

Would any of the values you have placed on land use Residual Native Cover (e.g Aesthetic, Cultural, Economic etc.) be affected, if the size of the land use parcel considered was larger or smaller? If Yes, explain how and why

____

____

____

Question 2.4:

Would any of the values you have placed on land use Residual Native Cover (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to other similar parcels having similar land use? If Yes, explain how and why

____

____

____

Question 2.5:

Would any of the values you have placed on land use Residual Native Cover (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to transport routes? If Yes, explain how and why

____

____

____

Question 2.6:

Would any of the values you have placed on land use Residual Native Cover (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to irrigation and/or farm infrastructure? If Yes, explain how and why

____

____

____

GRAZING MODIFIED PASTURES (LAND USE 3.2.0)

On Maryland Farm, there exists modified pasture being grazed by sheep (known by its land use as ***Grazing Modified Pasture***) for breeding and trading first cross ewes. The land has land capability level 4. The picture below shows an example of Grazing Modified Pastures and the table explains what Grazing Modified Pastures really means according to Australian Land Use Management (ALUM) classification.

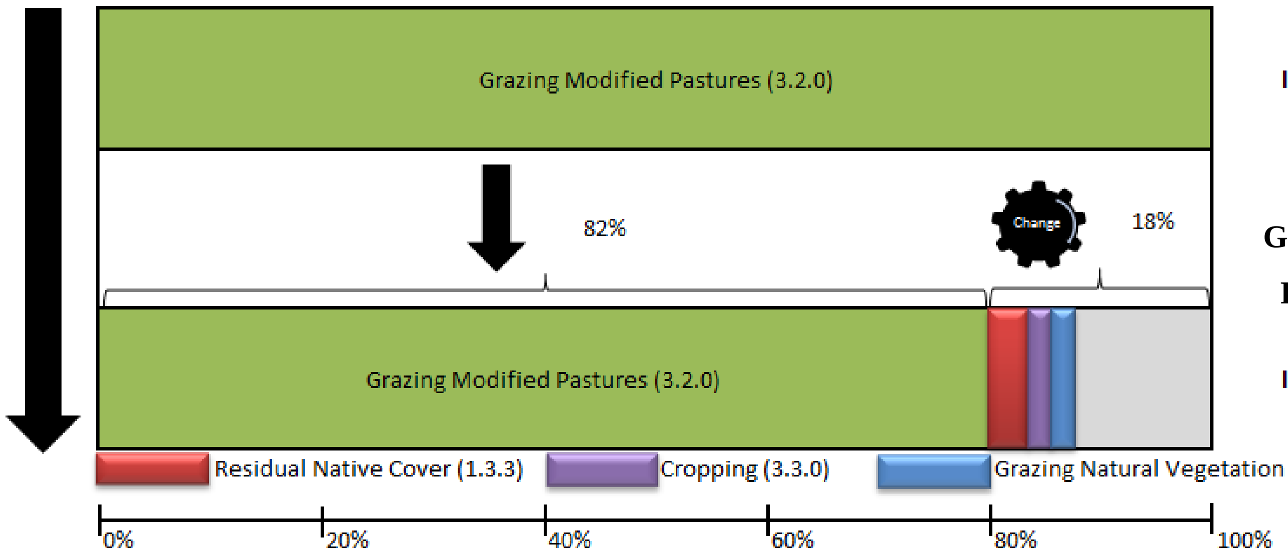
Grazing Modified Pastures (3.2.0)



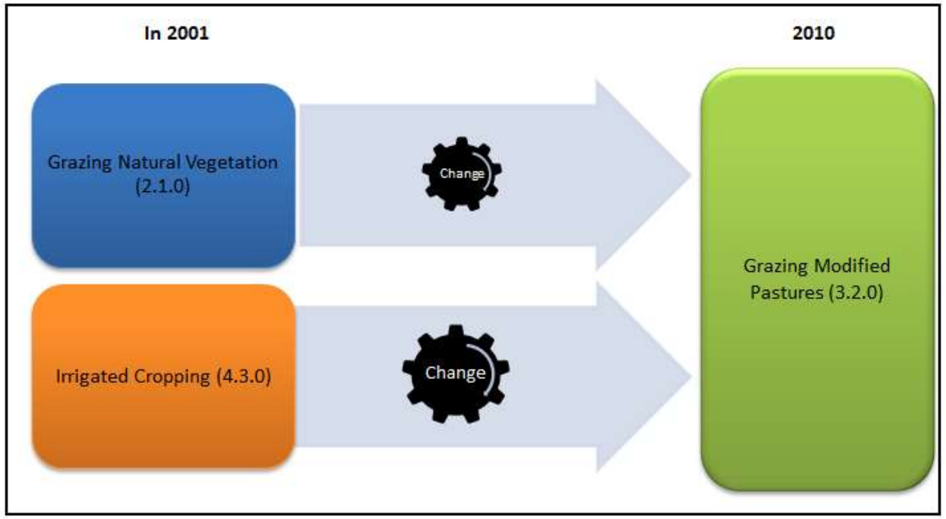
Land use ALUM Code	**Land Use Level 1**	**Land Use Level 2**	**Land Use Level 3**
3.2.0	Production from Dryland Agriculture and Plantations	Grazing modified pastures	N.A or Grazing modified pastures

3.2.0 **Grazing modified pastures**	**3 Production from dryland agriculture and plantations** This class includes land that is used principally for primary production, based on dryland farming systems. Native vegetation has largely been replaced by introduced species through clearing, the sowing of new species, the application of fertilisers or the dominance of volunteer species. The range of activities in this category includes plantation forestry, pasture production for stock, cropping and fodder production, and a wide range of horticultural production. **3.2 Grazing modified pastures** Pasture and forage production, both annual and perennial, based on significant active modification or replacement of the initial vegetation. For ALUM purposes, this class is used when there is greater than 50 per cent dominant exotic species.
Production of pasture and forage from dryland through modification - For grazing	

Records show that about 18% of Grazing Modified Pastures in the south-eastern midlands has been subject to change over the last decade. There have been two-way changes in the past decade in relation to Grazing Modified Pastures i.e. ***From*** Grazing Modified Pastures and ***To*** Grazing Modified Pastures. ***From Grazing Modified Pastures***, the change has been to Residual native Cover (1.3.3), Cropping (3.3.0), and Grazing Natural Vegetation (2.1.0). The change has also taken place ***To Grazing Modified Pastures*** from Grazing Natural Vegetation (2.1.0) and Irrigated Cropping (4.3.0).



Change From Grazing Modified Pastures (3.2.0)



Change To Grazing Modified Pastures (3.2.0)

We would like to learn about the ***values*** you have for the land use: **<u>Grazing Modified Pastures.</u>**

Question 3.1:

Please indicate the values you hold for the land use: Grazing Modified Pastures.

Values	Strongly agree	Slightly agree	Neither agree nor disagree	Slightly disagree	Strongly disagree
Aesthetic value – I value this land use because I enjoy the scenery, sights, sounds, smells, etc.					
Biological diversity value – I value this land use because it provides a variety of fish, wildlife, plant life, etc.					
Cultural value - I value this land use because it is a place for me to continue and pass down the wisdom and knowledge, traditions, and way of life of my ancestors.					
Economic value – I value this land use because it provides an economic return.					
Future value – I value this land use because it allows future generations to know and experience the land as it is now.					
Historic value – I value this land use because it has places and things of natural and human history that matter to me, others, or the nation.					
Intrinsic value – I value this land use in and of itself, whether people are present or not.					
Learning value – I value this land use because we can learn about environment through scientific observation and experimentation.					
Life Sustaining value – I value this land use because it helps produce, preserve, clean and renew air, soil and water.					
Recreation value – I value this land use because it provides a place for my favourite outdoor recreation activities.					
Spiritual value – I value this land use because it is a sacred, religious, or spiritually special place to me or because I feel reverence and respect for nature here.					
Therapeutic value – I value this land because it makes me feel better physically and/or mentally.					
Other reason/s – Please mention the reason here:					

Question 3.2:

Please consider the area of Grazing Modified Pastures (3.2.0) on Maryland Farm with land capability 4. Knowing the associated information about land capability, elevation, slope, aspect, dominant soil, hydrology, area of land, surrounding land uses and any relevant policies in the Midlands area, we would like to know what factors would influence your decision to support (or to not support) retaining land use Grazing Modified Pastures rather than change this area to another land use.

Decision Factors	Strongly influence me to **support a change** to any other land use	Slightly influence me to **support a change** to any other land use	No influence	Slightly influence me to **support NO change** to any other land use	Strongly influence me to **support NO change** to any other land use	Please add any comments to explain your decision.
Land Capability						
Elevation						
Slope						
Aspect						
Dominant Soil						
Drainage Lines						
Area of land						
Surrounding land use						
Policy						
Other 1:						
Other 2:						
Other 3:						

Question 3.3:

Would any of the values you have placed on land use Grazing Modified Pastures (e.g Aesthetic, Cultural, Economic etc.) be affected, if the size of the land use parcel considered was larger or smaller? If Yes, explain how and why

____

____

____

Question 3.4:

Would any of the values you have placed on land use Grazing Modified Pastures (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to other similar parcels having similar land use? If Yes, explain how and why

____

____

____

Question 3.5:

Would any of the values you have placed on land use Grazing Modified Pastures (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to transport routes? If Yes, explain how and why

____

____

____

Question 3.6:

Would any of the values you have placed on land use Grazing Modified Pastures (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to irrigation and/or farm infrastructure? If Yes, explain how and why

____

____

____

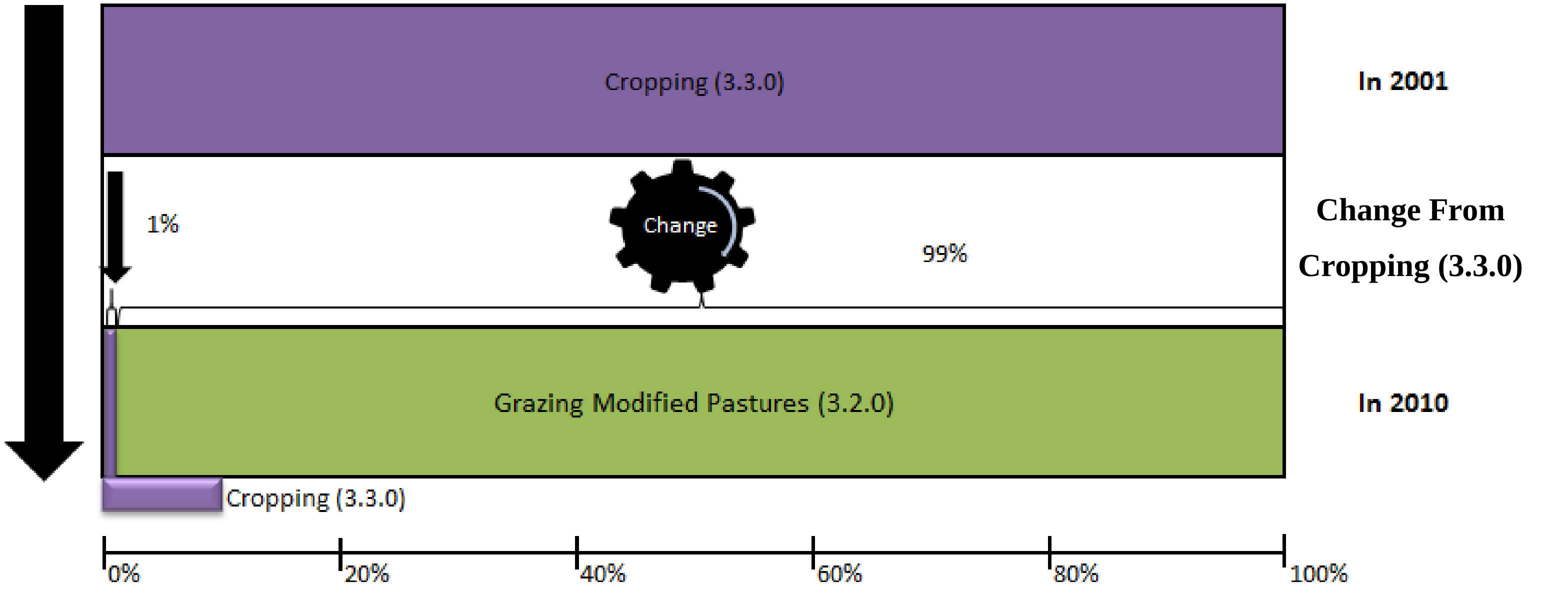
CROPPING (LAND USE 3.3.0)

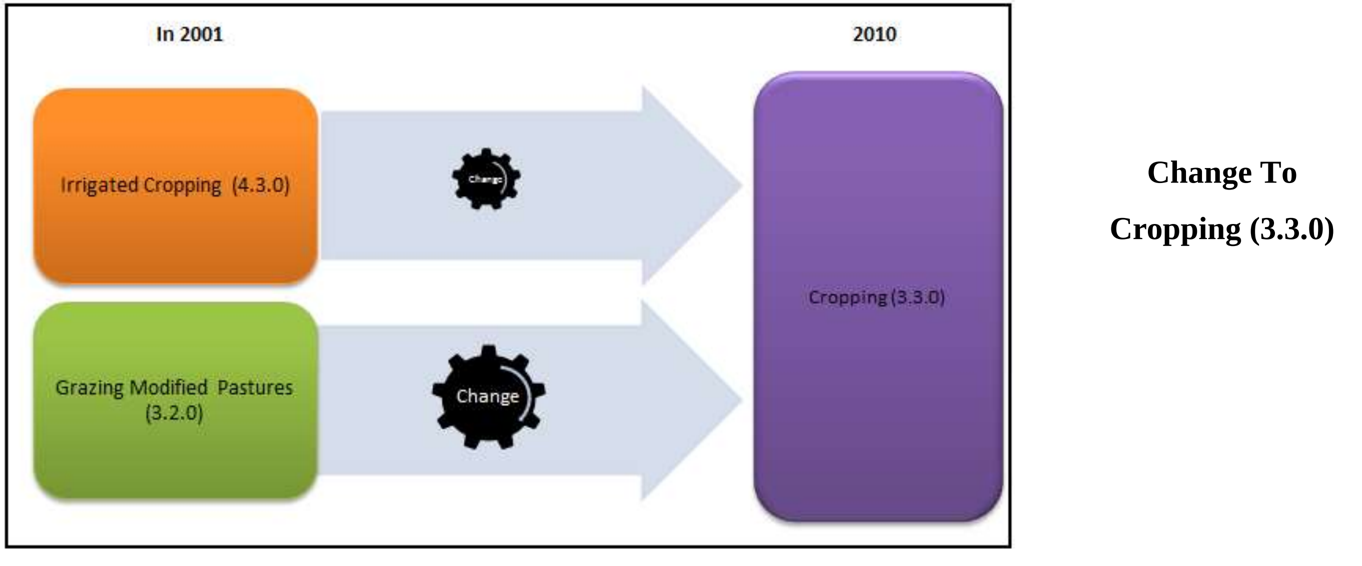
On Maryland Farm, there is a dryland farm being cultivated with crops (known by its land use as ***Cropping***). The land has land capability level 4. The picture below shows an example of Cropping and the table explains what Cropping really means according to Australian Land Use Management (ALUM) classification.

Cropping (3.3.0)

Land use ALUM Code	**Land Use Level 1**	**Land Use Level 2**	**Land Use Level 3**
3.3.0	Production from Dryland Agriculture and Plantations	Cropping	Cropping
3.3.0 **Cropping**	**3 Production from dryland agriculture and plantations** This class includes land that is used principally for primary production, based on dryland farming systems. Native vegetation has largely been replaced by introduced species through clearing, the sowing of new species, the application of fertilisers or the dominance of volunteer species. The range of activities in this category includes plantation forestry, pasture production for stock, cropping and fodder production, and a wide range of horticultural production.		
Production from dryland through cropping	**3.3 Cropping** Land that is under cropping. Land under cropping at the time of mapping may be in a rotation system, so that at another time the same area may be, for example, under pasture. Land in a rotation system should be classified according to the land use at the time of mapping. Cropping can vary markedly over relatively short distances in response to changes in the nature of the land and the preferences of the land manager. It may also change over time in response to market conditions. Production of fodder, such as lucerne hay, is considered cropping as there is no harvesting by stock.		

Records show that about 99% of Cropping in the south-eastern midlands has been subject to change over the last decade. There have been two-way changes in the past decade in relation to Cropping i.e. ***From*** Cropping and ***To*** Cropping. ***From Cropping***, the change has been to Grazing Modified Pastures (3.2.0). The change has also taken place ***To Cropping*** from Grazing Modified Pastures (3.2.0) and Irrigated Cropping (4.3.0).





We would like to learn about the ***values*** you have for the land use: <u>**Cropping.**</u>

Question 4.1:

Please indicate the values you hold for the land use: Cropping.

Values	Strongly agree	Slightly agree	Neither agree nor disagree	Slightly disagree	Strongly disagree
Aesthetic value – I value this land use because I enjoy the scenery, sights, sounds, smells, etc.					
Biological diversity value – I value this land use because it provides a variety of fish, wildlife, plant life, etc.					
Cultural value - I value this land use because it is a place for me to continue and pass down the wisdom and knowledge, traditions, and way of life of my ancestors.					
Economic value – I value this land use because it provides an economic return.					
Future value – I value this land use because it allows future generations to know and experience the land as it is now.					
Historic value – I value this land use because it has places and things of natural and human history that matter to me, others, or the nation.					
Intrinsic value – I value this land use in and of itself, whether people are present or not.					
Learning value – I value this land use because we can learn about environment through scientific observation and experimentation.					
Life Sustaining value – I value this land use because it helps produce, preserve, clean and renew air, soil and water.					
Recreation value – I value this land use because it provides a place for my favourite outdoor recreation activities.					
Spiritual value – I value this land use because it is a sacred, religious, or spiritually special place to me or because I feel reverence and respect for nature here.					
Therapeutic value – I value this land because it makes me feel better physically and/or mentally.					
Other reason/s – Please mention the reason here:					

Question 4.2:

Please consider the area of Cropping (3.3.0) on Maryland Farm with land capability 4. Knowing the associated information about land capability, elevation, slope, aspect, dominant soil, hydrology, area of land, surrounding land uses and any relevant policies in the Midlands area, we would like to know what factors would influence your decision to support (or to not support) retaining land use Cropping rather than change this area to another land use.

Decision Factors	Strongly influence me to **support a change** to any other land use	Slightly influence me to **support a change** to any other land use	No influence	Slightly influence me to **support NO change** to any other land use	Strongly influence me to **support NO change** to any other land use	Please add any comments to explain your decision.
Land Capability						
Elevation						
Slope						
Aspect						
Dominant Soil						
Drainage Lines						
Area of land						
Surrounding land use						
Policy						
Other 1:						
Other 2:						
Other 3:						

Question 4.3:

Would any of the values you have placed on land use Cropping (e.g Aesthetic, Cultural, Economic etc.) be affected, if the size of the land use parcel considered was larger or smaller? If Yes, explain how and why

____

____

____

Question 4.4:

Would any of the values you have placed on land use Cropping (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to other similar parcels having similar land use? If Yes, explain how and why

____

____

____

Question 4.5:

Would any of the values you have placed on land use Cropping (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to transport routes? If Yes, explain how and why

____

____

____

Question 4.6:

Would any of the values you have placed on land use Cropping (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to irrigation and/or farm infrastructure? If Yes, explain how and why

____

____

____

HABITAT/ SPECIES MANAGEMENT AREA (LAND USE 1.1.5)

On Maryland Farm, there is a land for conserving nature and regarded as a protected area (known by its land use as ***Habitat/ Species Management Area***). The land is mainly conserved through management interventions to ensure maintenance of habitats. The land has land capability level 5. The picture below shows an example of Habitat/ Species Management Area and the table explains what Habitat/ Species Management Area really means according to Australian Land Use Management (ALUM) classification.

Habitat / Species Management Area (1.1.5)



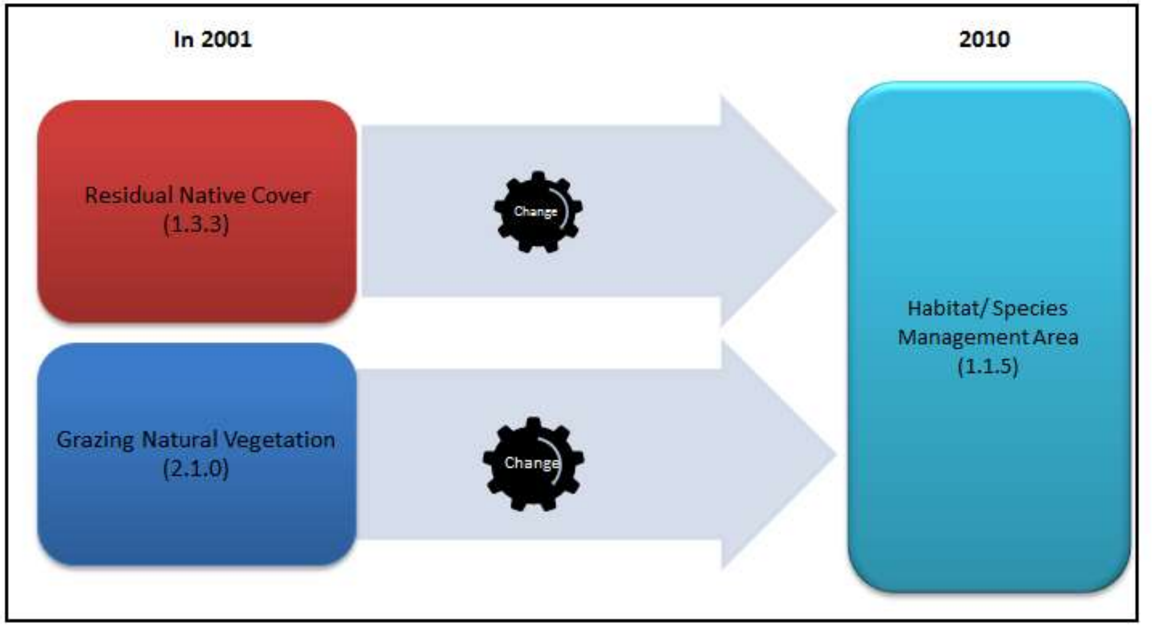
Land use ALUM Code	Land Use Level 1	Land Use Level 2	Land Use Level 3
1.1.5	Conservation and Natural Environments	Nature conservation	Habitat/species management area

1.1.5 **Habitat/species management area** **Maintenance of habitat or species through management interventions Limited human interventions**	**1 Conservation and Natural Environments** This class includes land that has a relatively low level of human intervention. The land may be formally reserved by government for conservation purposes, or conserved through other legal or administrative arrangements. Areas may have multiple uses, but nature conservation is the prime use. Some land may be unused as a result of a deliberate decision of the government or landowner, or due to circumstance. **1.1 Nature conservation** Tertiary classes 1.1.1–1.1.6 are based on the Collaborative Australian Protected Areas Database (CAPAD) classification (Cresswell & Thomas 1997). **1.1.5 Habitat/species management area** Protected area managed mainly for conservation through management intervention. An area of land or sea subject to active intervention for management purposes to ensure the maintenance of habitats or to meet the requirements of specific species. This may include areas on private land.

Records show that about 1% of Habitat/ Species Management Area in the south-eastern midlands has been subject to change over the last decade. There have been two-way changes in the past decade in relation to Habitat/ Species Management Area i.e. ***From*** Habitat/ Species Management Area and ***To*** Habitat/ Species Management Area. ***From Habitat/ Species Management Area***, the change is insignificant. Although, the change has also taken place ***To Habitat/ Species Management Area*** from Grazing Natural Vegetation (2.1.0) and Residual Native Cover (1.3.3).

NO SIGNIFICANT CHANGE OBSERVED FROM LAND USE HABITAT/ SPECIES MANAGEMENT AREA (1.1.5) TO OTHER LAND USES

Change From Habitat/ Species Management Area (1.1.5)



Change To Habitat/ Species Management Area (1.1.5)

We would like to learn about the ***values*** you have for the land use: **<u>Habitat/ Species Management Area.</u>**

Question 5.1:

Please indicate the values you hold for the land use: Habitat/ Species Management Area.

Values	Strongly agree	Slightly agree	Neither agree nor disagree	Slightly disagree	Strongly disagree
Aesthetic value – I value this land use because I enjoy the scenery, sights, sounds, smells, etc.					
Biological diversity value – I value this land use because it provides a variety of fish, wildlife, plant life, etc.					
Cultural value - I value this land use because it is a place for me to continue and pass down the wisdom and knowledge, traditions, and way of life of my ancestors.					
Economic value – I value this land use because it provides an economic return.					
Future value – I value this land use because it allows future generations to know and experience the land as it is now.					
Historic value – I value this land use because it has places and things of natural and human history that matter to me, others, or the nation.					
Intrinsic value – I value this land use in and of itself, whether people are present or not.					
Learning value – I value this land use because we can learn about environment through scientific observation and experimentation.					
Life Sustaining value – I value this land use because it helps produce, preserve, clean and renew air, soil and water.					
Recreation value – I value this land use because it provides a place for my favourite outdoor recreation activities.					
Spiritual value – I value this land use because it is a sacred, religious, or spiritually special place to me or because I feel reverence and respect for nature here.					
Therapeutic value – I value this land because it makes me feel better physically and/or mentally.					
Other reason/s – Please mention the reason here:					

Question 5.2:

Please consider the area of Habitat/ Species Management Area (1.1.5) on Maryland Farm with land capability 5. Knowing the associated information about land capability, elevation, slope, aspect, dominant soil, hydrology, area of land, surrounding land uses and any relevant policies in the Midlands area, we would like to know what factors would influence your decision to support (or to not support) retaining land use Cropping rather than change this area to another land use.

Decision Factors	Strongly influence me to **support a change** to any other land use	Slightly influence me to **support a change** to any other land use	No influence	Slightly influence me to **support NO change** to any other land use	Strongly influence me to **support NO change** to any other land use	Please add any comments to explain your decision.
Land Capability						
Elevation						
Slope						
Aspect						
Dominant Soil						
Drainage Lines						
Area of land						
Surrounding land use						
Policy						
Other 1:						
Other 2:						
Other 3:						

Question 5.3:

Would any of the values you have placed on land use Habitat/ Species Management Area (e.g Aesthetic, Cultural, Economic etc.) be affected, if the size of the land use parcel considered was larger or smaller? If Yes, explain how and why

____

____

____

Question 5.4:

Would any of the values you have placed on land use Habitat/ Species Management Area (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to other similar parcels having similar land use? If Yes, explain how and why

____

____

____

Question 5.5:

Would any of the values you have placed on land use Habitat/ Species Management Area (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to transport routes? If Yes, explain how and why

____

____

____

Question 5.6:

Would any of the values you have placed on land use Habitat/ Species Management Area (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to irrigation and/or farm infrastructure? If Yes, explain how and why

____

____

____

IRRIGATED CROPPING (LAND USE 4.3.0)

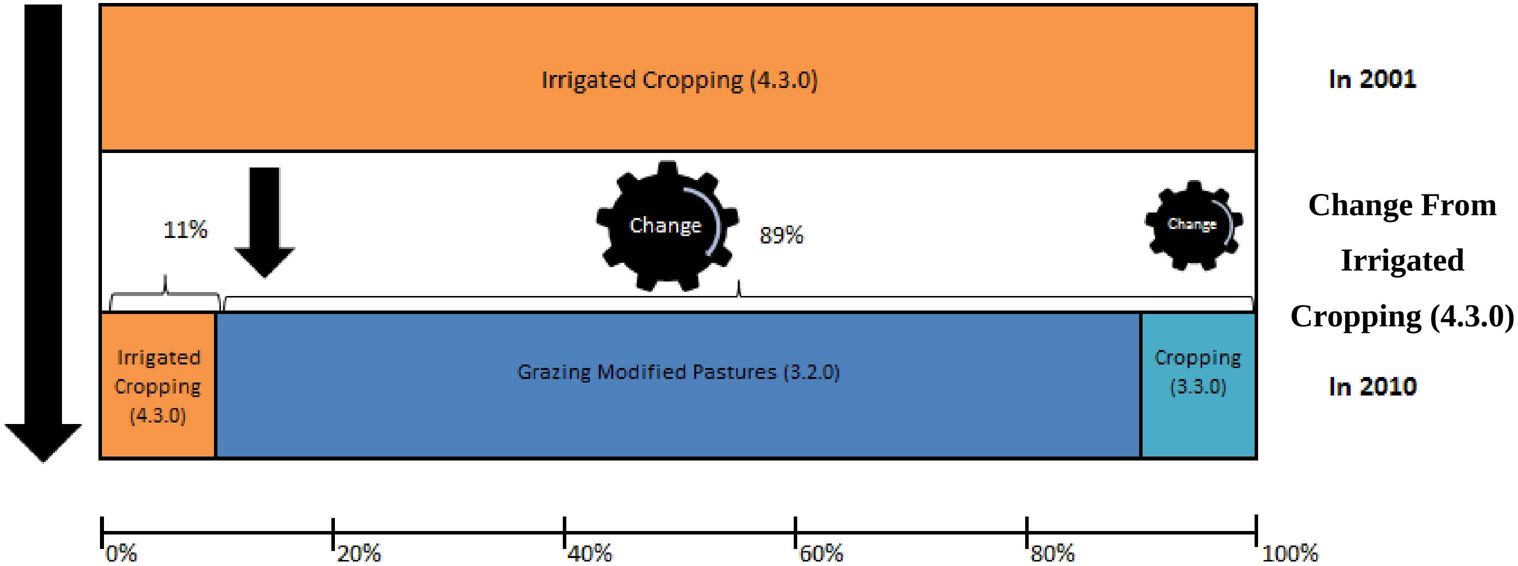
On Maryland Farm, there exists agricultural land where water is applied for cropping (known by its land use as ***Irrigated Cropping***). The main purpose of the land is cropping by application of water to promote growth. The land has land capability level 4. The picture below shows an example of Irrigated Cropping and the table explains what Irrigated Cropping really means according to Australian Land Use Management (ALUM) classification.

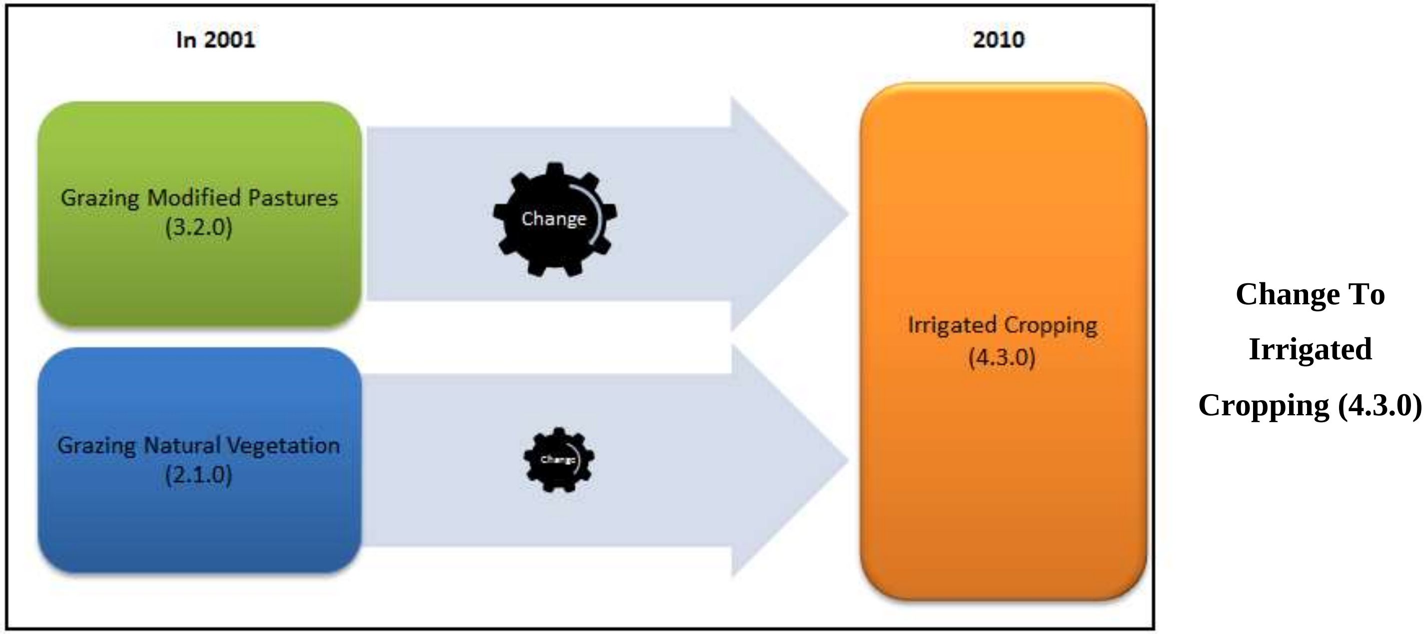
Irrigated Cropping (4.3.0)



Land use ALUM Code	Land Use Level 1	Land Use Level 2	Land Use Level 3
4.3.0	Production from Irrigated Agriculture and Plantations	Irrigated Cropping	N.A or Irrigated Cropping
4.3.0 **Irrigated Cropping** **Production from irrigated land through cropping**	**4 Production from dryland agriculture and plantations** This class includes agricultural land uses where water is applied to promote additional growth over normally dry periods, depending on the season, water availability and commodity prices. This includes land uses that receive only one or two irrigations per year, through to those uses that rely on irrigation for much of the growing season. **4.3 Cropping** Land that is under irrigated cropping. This class may include land in a rotation system that at other times may be under pasture.		

Records show that about 89% of Irrigated Cropping in the south-eastern midlands has been subject to change over the last decade. There have been two-way changes in the past decade in relation to Irrigated Cropping i.e. ***From*** Irrigated Cropping and ***To*** Irrigated Cropping. ***From Irrigated Cropping***, the change has been to Grazing Modified Pastures (3.2.0) and Cropping (3.3.0). The change has also taken place ***To Irrigated Cropping*** from Grazing Natural Vegetation (2.1.0) and Grazing Modified Pastures (3.2.0).





We would like to learn about the ***values*** you have for the land use: **<u>Irrigated Cropping.</u>**

Question 6.1:

Please indicate the values you hold for the land use: Irrigated Cropping.

Values	Strongly agree	Slightly agree	Neither agree nor disagree	Slightly disagree	Strongly disagree
Aesthetic value – I value this land use because I enjoy the scenery, sights, sounds, smells, etc.					
Biological diversity value – I value this land use because it provides a variety of fish, wildlife, plant life, etc.					
Cultural value - I value this land use because it is a place for me to continue and pass down the wisdom and knowledge, traditions, and way of life of my ancestors.					
Economic value – I value this land use because it provides an economic return.					
Future value – I value this land use because it allows future generations to know and experience the land as it is now.					
Historic value – I value this land use because it has places and things of natural and human history that matter to me, others, or the nation.					
Intrinsic value – I value this land use in and of itself, whether people are present or not.					
Learning value – I value this land use because we can learn about environment through scientific observation and experimentation.					
Life Sustaining value – I value this land use because it helps produce, preserve, clean and renew air, soil and water.					
Recreation value – I value this land use because it provides a place for my favourite outdoor recreation activities.					
Spiritual value – I value this land use because it is a sacred, religious, or spiritually special place to me or because I feel reverence and respect for nature here.					
Therapeutic value – I value this land because it makes me feel better physically and/or mentally.					
Other reason/s – Please mention the reason here:					

Question 6.2:

Please consider the area of Irrigated Cropping (4.3.0) on Maryland Farm with land capability 4. Knowing the associated information about land capability, elevation, slope, aspect, dominant soil, hydrology, area of land, surrounding land uses and any relevant policies in the Midlands area, we would like to know what factors would influence your decision to support (or to not support) retaining land use Cropping rather than change this area to another land use.

Decision Factors	Strongly influence me to **support a change** to any other land use	Slightly influence me to **support a change** to any other land use	No influence	Slightly influence me to **support NO change** to any other land use	Strongly influence me to **support NO change** to any other land use	Please add any comments to explain your decision.
Land Capability						
Elevation						
Slope						
Aspect						
Dominant Soil						
Drainage Lines						
Area of land						
Surrounding land use						
Policy						
Other 1:						
Other 2:						
Other 3:						

Question 6.3:

Would any of the values you have placed on land use Irrigated Cropping (e.g Aesthetic, Cultural, Economic etc.) be affected, if the size of the land use parcel considered was larger or smaller? If Yes, explain how and why

____

____

____

Question 6.4:

Would any of the values you have placed on land use Irrigated Cropping (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to other similar parcels having similar land use? If Yes, explain how and why

____

____

____

Question 6.5:

Would any of the values you have placed on land use Irrigated Cropping (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to transport routes? If Yes, explain how and why

____

____

____

Question 6.6:

Would any of the values you have placed on land use Irrigated Cropping (e.g Aesthetic, Cultural, Economic etc.) be affected, if the land use parcel was closer to irrigation and/or farm infrastructure? If Yes, explain how and why

____

____

____

Appendix F: Random Forest

Random Forest (RF) methods can provide useful algorithms for exploratory data analysis and variable selection. Many alternative automated variable importance measures exist in literature (Miller 1984, Austin and Tu 2004) but they involve assumptions about the functional form of models or the distribution of residuals (Sandri and Zuccolotto 2006). In contrast, the RF algorithm chosen for this study does not require these assumptions and can be applied in situations where a large number of observed variables are present, and the sample size is small (Sandri and Zuccolotto 2006).

The RF algorithm generates multiple similar data sets (called bootstrap samples) by resampling with replacement from the original training data set to create multiple regression trees (ntree). The algorithm allows these regression trees to grow to maximum size without pruning. Each tree is grown with a randomised subset of predictors (mtry) to determine the best split at each node of the tree (Breiman 2001, Prasad *et al.* 2006, Adam *et al.* 2014). The results from each aggregation are then averaged to get the overall prediction accuracy.

Random Forests can be used for a number of purposes, such as describing the relationship between predictors and responses i.e. regression, forecasting, variable selection, missing data imputation and classification. It can be used with mixed discrete and continuous predictors and responses (Siroky 2009). Random forest has been widely used in various disciplines including landscape epidemiology (Furlanello *et al.* 2003) medicine (Sandri and Zuccolotto 2006), genetics (Diaz-Uriarte and De Andres 2006) micro-biology (Han *et al.* 2007), agro-industrial research (Granitto *et al.* 2006) and remote sensing (Abdel-Rahman *et al.* 2012). In remote sensing, RF has been widely used for classification as well as for prediction (Gislason *et al.* 2006, Everingham *et al.* 2009, Ismail and Mutanga 2010, Abdel-Rahman *et al.* 2012, Adam *et al.* 2014).

Variable importance is evaluated based on how much worse the prediction would be if the data for that predictor were permuted randomly (Prasad *et al.* 2006). The tables generated can be used to compare relative importance among predictor variables. The importance of variables can be assessed by their impact on the accuracy of predictions, which allows for a quick assessment of the relevance of a predictor for the outcome of interest (Jones and Linder 2015). There are two means by which variable importance is measured. The first is the scaled average of the prediction accuracy of each variable, known as percentage increase in mean square error (%IncMSE); the second is the total decrease in node impurities splitting on the variable over all trees, using the Gini index (Williams 2008).

The importance measures show how much MSE or impurity increases when that variable is randomly permuted or, in other words, how much the model accuracy decreases if we drop that variable. If a variable is randomly permuted and no gain is achieved in the prediction, then there is evidence that removing that measure will substantially degrade predictions; with the converse also true: a high change in

%IncMSE indicates an important variable (Prasad *et al.* 2006, Williams 2008, Siroky 2009). The difference between the accuracy of the prediction before and after permutation provides the importance of the i^{th} variable for one tree, and the importance for the forest is calculated by averaging over all trees (Breiman 2001, Sandri and Zuccolotto 2006, Strobl *et al.* 2009).