The Optimal Design of Conservation Investments

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Abstract

A substantial proportion of biodiversity occurs on private land. In attempting to halt the decline of this biodiversity, voluntary contracts with private landowners have become a cornerstone of conservation investments. Taking a theoretical modelling approach, I investigate several issues in the design of voluntary conservation investments. First, I consider the implications for conservation of the informational advantage that landowners have in contract negotiations. I find that landowners have the potential to use their private information on conservation costs to make conservation considerably more expensive than suggested by estimates of opportunity costs relied on in previous studies. Next, I consider how cost structures affect the optimal distribution of nature reserves, showing that common cost structures used in conservation studies could lead to ineffective and inefficient reserves designs. Following on from this, by mapping the trade-off between conservation improvements and the increasing costs of achieving those improvements over various species distributions, I find that conservation will be most cost-effective when species are highly nested and conservation targets include widespread species. Finally, I analyse how socio-economic and ecological uncertainty affects the choice between short or long conservation contracts and reveal that socio-economic factors have a greater impact on contract choice than those of an ecological nature.

Several important conclusions follow from these results. First, conservation science must deal with the complexities of conservation planning problems. Simplifications, such as assuming conservation costs equal opportunity costs, can result in distorted conservation priorities and unrealistic expectations. Second, incorporating in analyses the uncertainties inherent in conservation planning, such as uncertainties in cost and the future ecological condition of sites, is crucial for developing effective conservation strategies. Finally, the results reinforce the conclusion of other recent studies that variation in socio-economic aspects of conservation can be more important for effective conservation planning than variation in indicators of biodiversity.

Publications

Publications and manuscripts following from this thesis:

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In Review:

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[6] **Chapter 6 - Gareth D. Lennox**, Kevin J. Gaston, Martin Dallimer and Paul R. Armsworth. How does the distribution of biodiversity affect the trade-off between conservation costs and species coverage? Submitted to *Biodiversity and Conservation*.

In Preparation:

[7] **Chapter 5 - Gareth D. Lennox** and Paul R. Armsworth. Cost structures in conservation planning.

This is planned to be a synthetic review of cost structures in conservation planning. I present initial ideas and results in Chapter 5.

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“Don’t worry, don’t be afraid, ever, because this is just a ride...”

– Bill Hicks

Chapter 1

General introduction

1.1 Biodiversity crisis

It is a widely held consensus that the Earth currently faces a biodiversity crisis. Estimates suggest that the contemporary rate of species extinction is 100 to 1000 times the average rate seen in the evolutionary time scale of life on the planet (Pimm et al. 1995). This rate of species extinction has led some conservation biologists to contend that we are in the midst of life's sixth mass extinction, an event in which more than 75% of all species will be lost (Chapin III et al. 2000; Dirzo and Raven 2003; Wake and Vrendenburg 2008; but see Barnosky et al. 2011).

This extinction crisis affects all kingdoms of life. For example, Schipper et al. (2008) calculate, as a conservative estimate, that approximately 25% of all mammalian species are at risk of extinction. For marine mammals the situation is even bleaker: one-third of all such species are currently endangered. Similarly, it is estimated that 14% of all avian species, 29% of all reptilian species and 31% of all amphibian species are currently threatened with extinction (Barnosky et al. 2011). However, these figures pale in comparison with the future likely status of the planet's species if current practices go unabated. Wilson (2002) estimates that by the end of this century more than half of all species could face serious risk of extinction.

The most important factors contributing to this erosion of biodiversity are: habitat destruction, degradation and fragmentation; over-exploitation; the introduction of alien and invasive species; and, increasingly, climate change (Pimm et al. 2001; Thomas et al. 2004; Hoffman et al. 2010; Bellard et al. 2012). Among these, habitat destruction, degradation and fragmentation is considered to pose the greatest threat to biodiversity (Dirzo and Raven 2003; Millennium Ecosystem Assessment 2005), yet habitats continue to be altered and degraded by human actions. For example, remote-sensing studies during the period 2002-

2009 indicate that the rate of selective logging in the Brazilian rainforest varied from between approximately 8,000 to 28,000 km^2/year (Sodhi et al. 2011); global mapping studies have shown that approximately 50% of all temperate grasslands, tropical dry forests, and temperate broadleaf forests have been converted to human dominated land uses while only between 4-10% has been protected (Hoekstra et al. 2005); and throughout Europe less than one-sixth of all land area is estimated to remain unaltered by human activities (Primark 2006).

Consideration of each of the major threats to biodiversity shows that they share a commonality: they are, in most part, anthropogenic in nature. Humanity's inability to curb its deleterious impacts on nature has resulted in the continuing and accelerating collapse of global biodiversity (Butchart et al. 2011). Besides failing our ethical imperative to protect the species with which we share the planet, these loses impose a significant cost on society given that biodiversity underpins many of the ecosystem services on which we rely (Chapin III et al. 2000; Hooper et al. 2005; Balvanera et al. 2006; Bennett et al. 2009).

1.2 Conservation investments

In response to the continuing decline in biodiversity throughout the world, the conservation movement has developed into a wide-ranging social, political and scientific endeavour. The question therefore arises: why have conservation interventions failed to arrest rates of species extinctions? The first explanation is simply the funding available for conservation. James et al. (1999) estimate that to purchase and manage a broadly representative system of nature reserves would cost US$22.6 billion per year. This figure is in marked contrast to the $6 billion the authors calculate as being spent. The authors also estimate that a comprehensive global conservation programme would cost approximately $300 billion per year. Comparing this figure to government subsidies that are used to promote ecologically damaging activities, which total in the region of $1-1.5 trillion per year (Myers 1998; Van Beers and de Moor 1999), demonstrates how little of an investment is required to protect the planet's vitally important biodiversity. The second component of the explanation as to why, despite conservation interventions, species continue to be lost at an accelerating rate relates to how conservation funding has been used. Ferraro and Pattanayak (2006) state: "For far too long, conservation scientists and practitioners have depended on intuition and anecdote to guide the design of conservation investments." If we are to have any hope of reversing the loss of global biodiversity this prevailing paradigm must be changed so that all aspects of conservation investments are based on rigorously evaluated scientific principles. With the work in this thesis I hope to contribute to this effort.

An important sphere of conservation investments has become voluntary agreements between private landowners and conservation organisations (Ferraro and Kiss 2002). With these investments, conservation groups attempt to alter private land use decisions for the benefit of biodiversity. The analyses presented in this thesis are all related to aspects of the design of voluntary conservation investments. In the following sections: I discuss the general principles behind schemes that are designed to protect/enhance biodiversity on private land along with the practical limitations of such schemes; I then outline a formal optimisation framework that encapsulates many of components of the design of voluntary conservation investments and discuss this framework in the context of prominent conservation approaches; next, I outline the analyses presented in this thesis and discuss how they deal with unresolved issues in the design of voluntary conservation agreements; finally, I conclude by briefly outlining some important areas of conservation that I do not cover in this thesis.

1.3 Conservation on private land

At a global scale, a significant proportion of biodiversity occurs on privately owned land. For example, in the United States of America (USA) two-thirds of federally listed endangered species occur on private land (Grooves et al. 2000) and in the European Union (EU) farmland creates multiple ecological niches that provide habitat for a variety of species (Kleijn et al. 2009; Cooper et al. 2009). The way in which private land is managed is therefore critical to biodiversity. However, biodiversity has the characteristics of a public good: it is non-excludable, meaning that if one person benefits from biodiversity no one else is excluded from similarly benefiting, and it is non-rival, meaning that if one person benefits from biodiversity the amount of biodiversity available to others is not reduced (Cooper et al. 2009). These public good characteristics of biodiversity mean that private landowners receive no financial benefit for protecting/enhancing biodiversity in traditional markets and, in fact, can incur considerable costs to do so (Hanley et al. 1998; 2012). Biodiversity and other public goods are therefore not provided sufficiently by traditional markets (Hanley et al. 2012).

Conservation organisations have made a variety of different types of investment in private land to overcome the insufficient supply of biodiversity. Outright acquisitions of land are an important component of conservation measures and have attracted large sums of funding. For example, protected areas now cover approximately 12% of the Earth's terrestrial surface (United Nations 2011), with investment levels in the region of $6.5 billion per annum (Emerton et al. 2006); and by 2010 over 1,700 land trust organisations in the USA had conserved 47 million acres of private land (Land Trust Alliance 2011). With such investments, conservation organisations potentially hold all land rights in perpetuity.

In contrast, increasing attention has been focused on temporary conservation contracts, which have been collectively termed “payments for environmental services” (PES). PES have been defined as a voluntary transaction where a well-defined environmental service (or a land use likely to secure that service) is being “bought” by a service buyer from a service provider if and only if the service provider secures service provision (Wunder 2005) (an alternative definition that recognises ambiguities in the voluntary nature of PES is offered by Sommerville et al. (2009)). This definition of PES therefore covers widely applied conservation investments such as agri-environment schemes (which are discussed in detail in Section 1.4.3) and shares many of the characteristics of conservation easements (which are discussed in Section 1.4.2) to the extent that some authors have argued that conservation easements are PES (Engel et al. 2008). Unlike land acquisitions, PES are conditional on the delivery of environmental services with the discontinuation of funding should delivery cease and thus may be a more efficient means of protecting/enhancing biodiversity and other environmental services.

Underlying PES is a simple theoretical basis: the undersupply of environmental services is corrected by performance-based financial transfers from those who use the service to those who provide it to secure increased provision (Ferraro 2011). PES therefore act to internalise the external costs borne by society by private land use decisions that diminish biodiversity and other environmental services. While attractive in theory, PES schemes suffer from several general problems that impair their effectiveness and efficiency. PES can lack additionality, a situation in which payments are made to landowners for actions that would have been undertaken in the absence of the payment. This has been identified as a problem in the forest conservation scheme in Costa Rica, for example, where it has been calculated that 71% of payments were directed to lands with limited or no agricultural potential (Hartshorn et al. 2005). As a result, a significant proportion of payments may have gone to lands that were under no threat of conversion. Also, those implementing PES schemes often have incomplete information regarding the service providers and poor targeting methods (Ferraro 2011). This can lead to the problem of adverse selection in which landowners are paid to deliver environmental services without the ability to do so (Franks 2011). Moreover, the net benefits of PES schemes can be limited or even negative if payments to particular landowners simply displace the environmental pressures to other areas of environmental importance, a phenomenon known as leakage. For example, Wu (2000) found that payments to US farmers to remove land from production produce an unintended incentive for additional plantings, such that 20 acres of land are added to production for every 100 removed. Finally, in contrast to land acquisitions, PES can produce benefits that are only temporary, with environmentally damaging land uses recommencing when payments end.

Both land acquisitions and PES are generalised forms of contracting with private landowners to secure biodiversity. Both of these methods encompass a large variety of contract designs and allocation mechanisms. Nonetheless, as has been described, both can suffer from ineffectiveness and inefficiency if designed injudiciously. Consequently, a focus on improving the design of these voluntary conservation investments is needed.

1.4 Designing voluntary conservation schemes

The objective of a conservation group which wishes to conserve biodiversity on private land is to contract with a set of landowners so as to return the highest biodiversity benefit subject to a budget constraint. The principal components in the design of conservation investments relate to the rights that will be purchased and the period of time for which those rights will be held by the conservation group. These decisions will depend on the cost of the investments and these costs can take various forms, as described in Box 1.1. This optimisation problem can be formally stated as:

$$\max_{\theta_i,\boldsymbol{\omega}_i} \{f(r_1(\theta_1,\boldsymbol{\omega}_1),\ldots,r_N(\theta_N,\boldsymbol{\omega}_N))\} \tag{1.1}$$

$$\text{subject to} \quad g(c_1(\theta_1,\boldsymbol{\omega}_1),\ldots,c_N(\theta_N,\boldsymbol{\omega}_N)) \leq B \tag{1.2}$$

where, $i = 1,\ldots,N$ is the set of sites of conservation interest and r_i is the function that describes the reward that the conservation group obtains by procuring the rights $\boldsymbol{\omega}_i$ on site i for $\theta_i \geq 0$ years. $\boldsymbol{\omega}_i$ is a binary vector in which an element equals 1 if the corresponding rights are purchased and is zero otherwise. f is the function that describes the combined reward from the investments. This combined reward depends on the objective of the conservation group. Consequently, the functional form has a range of complexity. For example, when seeking to cover as much of the abundance of a rare species as possible with areas under conservation contracts, biodiversity values could simply sum over conserved sites. Maximising the richness of species represented in a reserve network, on the other hand, results in a complex, non-additive function in which the biodiversity value of one site depends on the biodiversity value of other conserved sites.

There is a cost associated with securing a site for conservation and this cost depends on the rights purchased and the length of time for which those rights are held, $c_i(\theta_i,\boldsymbol{\omega}_i)$. g is the function that describes the total cost of investing in each of the sites, which may or may not be additive (see Chapter 5), and this total cost must not exceed the budget, B.

Additionally, the conservation group will be uncertain regarding r_i and c_i, the benefit that will follow from investing in site i and the cost of those investments. In general, landowners

Box 1.1. Economic costs of conservation

The economic costs of conservation can take various forms. The upfront cost of securing the rights to a parcel of land are known as acquisition costs. Primary data regarding conservation acquisition costs is not widely available (Naidoo et al. 2006). Consequently, researchers in conservation planning have developed a host of proxies that are thought to estimate site acquisition costs and developed regression techniques that allow acquisition costs to be modelled against related site characteristics (e.g.,Bastian et al. 2002; Goodwin et al. 2003; Naidoo and Adamowicz 2006).

Recurrent costs in conservation come in the form of continuing payments for the delivery of environmental services and management costs of acquired sites. Globally, terrestrial conservation management costs vary by seven orders of magnitude, positively correlate with human population density and negatively correlate with reserve area (Balmford et al. 2003). As with conservation acquisition costs, primary data regarding management costs is limited, especially at fine scales needed for conservation prioritisation exercises (Balmford et al. 2003).

Conservation also leads to opportunity costs, the costs to society of opportunities forgone by precluding economic activities on a site. For example, if farmland is used for conservation, the opportunity costs would be the net present value of revenues that could have been gained from undertaking agriculture on the land instead of conservation. In the context of site acquisitions, opportunity costs would equal acquisition costs if markets were perfectly competitive. However, markets in conservation do not generally exist and even in cases where it can be argued that they do, they are certainly not perfectly competitive (Hanley et al. 2012). The relationship between conservation acquisition costs and opportunity costs remains largely unresolved and I explore this issue in Chapters 2-4.

Finally, the costs associated with negotiating an economic exchange, such as the cost of identifying sites of conservation interest, negotiating with landowners and gaining approval for the transfer of the land titles, are collectively known as transaction costs (Naidoo et al. 2006). Transaction costs can often be substantial, depending on the complexity of the allocation mechanism (Falconer 2002; Naidoo et al. 2006).

will have greater information regarding these functions, the consequences of which I investigate in detail in Chapters 2-4.

Many voluntary conservation schemes are essentially special cases of the above formulation. In the following subsections, I discuss several important examples of voluntary conservation investments and outline how they are related to the optimisation formulation given in Eqns. (1.1) & (1.2).

1.4.1 Formulation 1 - Classic site selection

Conservation reserves serve as one of the most important examples of voluntary conservation investments and much of the science of conservation concerns how to locate and design reserves. To illustrate how such a site selection exercise could be undertaken, below is a general formulation following from Eqns. (1.1) & (1.2) that captures many of the features of the classic site selection problem,

$$\max_{x_i}\{f(r_1x_i,\ldots,r_Nx_N)\} \tag{1.3}$$

subject to

$$\sum_{i=1}^{N} c_ix_i \leq B \tag{1.4}$$

where x_i is the binary choice variable that takes the value 1 if site i is selected for investment and is 0 otherwise. In this formulation there is no uncertainty, sites are bought in totality (and thus there are no intra-site differences in land rights to consider) and in perpetuity (thus no consideration to contract length is necessary). The cost structure (Eqn. (1.4)) is simple and linear in which the cost to secure multiple sites for conservation is the sum of the costs of the individual sites.

In the classic site selection problem, the objective is to procure a network of sites that contain a representative sample of biodiversity and that is designed to ensure that species are able to persist in the long-term (Margules and Pressey 2000). This objective would be described by the function f in Eqn. (1.3) with the form dependent on the conservation context. Approaching site selection in this manner is known as systematic conservation planning (Margules and Pressey 2000). Unfortunately, conservation planning has often lacked such a systematic approach. An important contributing factor to the ad-hoc development of reserve networks is that levels of biodiversity are correlated with the value of land (Ferraro 2003; Moore et al. 2004). Therefore, to represent the biota of a region, land that is of high value has to be protected from anthropogenic disturbance. This has rarely occurred, with conservation reserves being located only on the least valuable and unproductive land, leading to a bias in the species protected (Pressey et al. 1996).

Moreover, systematic conservation planning has often been blighted by a lack of sophistication/realism. For example, to determine the most efficient and effective site prioritisations, spatially explicit estimates of the total cost of conservation should be incorporated in reserve design mechanisms. This has rarely occurred for two primary reasons. First, conservation planning research has often neglected the economics of conservation, focusing mainly on the biological aspects of the problem (Naidoo et al. 2006). In the earliest attempts at systematic conservation planning, costs, if included at all, were incorporated in a simplistic manner in which reserve selections were limited by the number or total area of reserves (Dobson et al. 1997; Araújo et al. 2002). The implicit assumption in such formulations is that the cost of land is homogeneous. Yet conservation costs vary by many orders of magnitude (Balmford et al. 2003; Davies et al. 2010) and often this variation is considerably greater than variation in indicators of biodiversity (Naidoo et al. 2006). Failing to account for variation in conservation costs has consistently and unsurprisingly been shown to be highly inefficient (Ando et al. 1998; Polasky et al. 2001; Possingham and Wilson 2005; Carwardine et al. 2008) and, in fact, a growing number of studies show that economic aspects of conservation may more influential in setting priorities than biological issues (Ando et al. 1998; Naidoo and Iwamura 2007; Bode et al. 2008; Perhans et al. 2008). For example, Murdoch et al. (2010) show that a conservation strategy that disregards biology completely is more effective than one that disregards costs completely.

Second, despite the growing recognition of the need to include cost information accurately in site prioritisation exercises, seldom has the total cost of conservation been considered with researchers focusing on the component(s) of cost thought to be the largest (Adams et al. 2009) (see Box 1.1 for the different components of the cost of conservation). For example, in developing a global strategy for the conservation of mammals, Wilson et al. (2011) calculate the cost of conservation as the sum of the opportunity costs of forgone agriculture and predicted management costs. Whilst unknown, it is arguable that the transaction costs of such a global conservation strategy would be vast (McDonald 2009; Armsworth et al. 2012). Whether neglecting to include particular aspects of the cost of conservation would distort conservation prioritisations depends on the relative magnitude of accounted to unaccounted for costs and their spatial correlation. If one unaccounted for component of cost was a substantial fraction of the total cost and negatively covaried with the component of cost used in the analyses, the result could be grossly inefficient site selections.

The design of reserves has similarly been lacking. For example, reserve design mechanisms that have had widespread use in conservation planning studies, such as the set covering problem (Ando et al. 1998; Rodrigues et al. 2000; Moore et al. 2003) and the maximum covering problem (Church and ReVelle 1974; Polasky et al. 2001; Gaston and Rodrigues 2003), produce reserve networks that lack spatial coherence. While such reserve designs may represent the biodiversity of a region adequately, species persistence may be jeopardised due to the lack of spatial considerations (Cabeza and Moilanen 2001).

Despite the many failures to apply systematic conservation planning effectively, the methodology has been responsible for notable successes. For example, several software platforms have been designed, such as C-Plan (Pressey et al. 1999), Marxan (Ball et al. 2009), and Zonation (Moilanen et al. 2005). Among other features, these allow conservation practitioners to design new reserve networks, test the performance of existing reserves, and develop plans for natural resource management that combine multiple land uses alongside conservation. As examples of the widespread applicability of these programmes, they have been used to: design transnational reserve networks in Southern Africa (Smith et al. 2008); prioritise areas for avian species conservation in a 3,200 km corridor from Yellowstone National Park in the USA to the Yukon in Canada (Pearce et al. 2008); and been used to integrate functional ecological interactions in the spatial design of a reserve network in the boreal forest in Quebec, Canada (Rayfield et al. 2009).

1.4.2 Formulation 2 - Conservation easements

Conservation easements are a prevalent form of voluntary conservation investment on private land (Fishburn et al. 2009a; Pocewicz et al. 2011). Easements are conservation contracts in which the landowner receives a direct payment or a tax rebate in recompense for ceding particular land rights to a conservation group while retaining the titles to the land (Hanley et al. 2012). These acquired rights are held by the conservation group in perpetuity. Therefore, in addition to considering which sites to select, the conservation group and landowner must agree which land rights will be purchased by the conservation group. Maximising the benefit from conservation easements can therefore be formulated as a simple extension of the site selection problem:

$$\max_{\boldsymbol{\omega}_i} \{f(r_1(\boldsymbol{\omega}_1),\ldots,r_N(\boldsymbol{\omega}_N))\}$$

$$\text{subject to} \qquad \sum_{i=1}^{N} c_i(\boldsymbol{\omega}_i) \leq B$$

In recent decades, there has been a rapid rise in the use of easements as a conservation tool in the USA. In 2010, the acreage protected through fee-simple (or full-rights) purchases by local and state land trusts was 2.1 million, a 91% increase on the 2000 level. In contrast, the acreage protected through the use of conservation easement had increased in the same time period by 283% to 8.8 million (Land Trust Alliance 2011). The rapid increase in conservation easements has also been witnessed in Latin America, Australia and South Africa (Armsworth and Sanchirico 2008).

There are many advantages to the use of easements as opposed to fee-simple purchases. Most obviously, easements can be significantly cheaper as only a subset of rights is purchased by the conservation group (Davies et al. 2010). Consequently, all else being equal, more land can be placed under conservation management. Moreover, it is often the case that conservation objectives and some forms of productive activity are compatible (Polasky et al. 2005). To set land aside completely in such scenarios would not only be unnecessarily expensive but would diminish social welfare. Finally, given that landowners maintain ownership of the land titles with a conservation easement, it may be easier to persuade them to agree to conservation than if they had to cede the land completely.

Despite the rapid advance and huge investment in conservation easements, little analysis has been undertaken into their efficacy. For example, little is known about which species and habitats conservation easements aim to protect, how much structural development they allow, or what types of activity they commonly permit (Rissman et al. 2007). Moreover, under current practices, easements are considered primarily to be private transactions and thus receive little public scrutiny (Morris 2008). However, despite being considered in this light, it has been argued that they are in fact only partially private (Raymond and Fairfax 2002; Morris 2008). Not only must the state define the property rights of easements but many conservation easements receive substantial public funding and, in fact, many easements are actually held by public agencies (Morris 2008). In reference to the privateness of publicly funded easement programmes, Fairfax et al. (2005) state: “Perhaps the only thing worse than private control of public resources and authority, in terms of public accountability, is reducing the scrutiny of substantially public programs by declaring them private.”

1.4.3 Formulation 3 - Agri-environment schemes

In contrast to the USA, in the EU easements are rarely used as a conservation tool. Here, voluntary conservation agreements on private land are commonly made in agri-environment schemes (AES). Generally, AES consist of finite length contracts offered by the state which recompense landowners for undertaking a subset of available management actions on their land. This increases the complexity of the formulation in comparison to conservation easements as now consideration must also be given to the contract duration. The formulation that captures this type of scheme is:

$$\begin{aligned} & \max_{\theta_i, \boldsymbol{\omega}_i} \{f(r_1(\theta_1, \boldsymbol{\omega}_1), \ldots, r_N(\theta_N, \boldsymbol{\omega}_N))\} \\ \text{subject to} \quad & \sum_{i=1}^{N} c_i(\theta_{1i}, \boldsymbol{\omega}_i) \leq B \end{aligned}$$

AES have a relatively long history of use within the EU. The first such scheme was the Environmentally Sensitive Areas (ESA) programme developed in the United Kingdom (UK) (Dobbs and Pretty 2008). Implemented in 1986, this first of many agri-environment schemes grew to contain 43 designated ESAs (Hanley et al. 1999) and was complemented by a variety of different programmes that had differing objectives. Implemented in 1991, the Countryside Steward Scheme (CSS) became most prominent among these. By 2003, over

10% of all agricultural land in England was enrolled in either the ESA or CSS programmes (Dobbs and Pretty 2008). More widely within the EU, Regulation 1257/1997 compels each member state to instigate AES to protect the environment and maintain the countryside (Donald and Evans 2006). In total, AES cover approximately 20% of EU farmland, resulting in annual payments to landowners of $7.2 billion (Cooper et al. 2009).

Each state in the EU is responsible for the design and implementation of AES within its borders, which has resulted in a large variety of different schemes. However, an important distinction between different AES is exemplified by the Environmental Stewardship Scheme, the flagship agri-environment scheme in England that replaced the ESA and CSS programmes. This programme has two contract types: the Entry Level Scheme (ELS) and the Higher Level Scheme (HLS). The ELS is “broad and shallow”, meaning a scheme that is not highly targeted and offers simple, low-cost management options to all landowners (Vickery et al. 2005). In contrast, the HLS is “narrow and deep”, meaning a scheme that may involve significant alterations to land management and which is targeted to landowners with sites that contain the highest ecological potential and/or face the greatest threat (Vickery et al. 2005).

As with conservation easements, despite the large scale implementation of AES, little assessment of their efficacy has been undertaken (Kleijn et al. 2006; Wilson et al. 2007). In a meta-analysis of assessments of AES, Kleijn and Sutherland (2003) found that only 5 out of the 22 EU countries that employ such programmes had undertaken any research into their impacts. Moreover, simple evaluation techniques that are often employed in assessments, such as comparing species richness or abundances on sites with agri-environment measures and control sites, fail to account for complex dynamics that can arise. For example, source-sink dynamics can result in sites of high ecological quality producing populations that can disperse to low quality sites thus obscuring the effects of the implemented measures (Foppen et al. 2000; Kleijn et al. 2011). Methodological considerations aside, a mixed picture is painted by those studies that have evaluated the effectiveness of agri-environment schemes. For example, the meta-analysis by Kleijn and Sutherland (2003) showed that 54% of the examined species increased in abundance while 6% decreased and 23% showed no response to the implemented measures. Kleijn et al. (2006) found that threatened species in five EU countries rarely benefited from agri-environment measures. Finally, the results of Wilson et al. (2007) indicate that habitat enhancement measures in the ESA programme resulted in increasing populations trends for three avian species. Lower cost measures aimed at habitat maintenance produced negligible benefits, however.

1.5 Thesis structure

As outlined above, all of the major forms of voluntary conservation investments suffer from criticism. Some of this relates to a lack of assessment while other criticisms relate to the achieved outcomes. Comparing the formulations of each of the investment types described above to the full optimisation problem described by Eqns. (1.1) & (1.2) indicates that many factors that may be of critical importance are not recognised and/or incorporated. In this thesis, I address several of these outstanding issues to investigate how they affect the optimal design of conservation investments.

1.5.1 Uncertainty regarding the cost of investments

I begin in Chapters 2-4 by investigating the economic and biological consequences of the conservation group's uncertainty regarding the cost of investments. As noted, when deciding with which landowners to negotiate to secure conservation agreements, the conservation group is unaware of the landowner's cost type. With respect to the formulation of the optimisation problem, there is uncertainty regarding c_i for all landowners. To overcome this, researchers seeking to identify the most cost-effective site selections have relied on proxies for the cost of conservation, most usually using the proxy of the agricultural value of land. However, this is an estimate of the landowner's opportunity costs, the minimum the landowner should be willing to accept when agreeing to the conservation investment. Given that the conservation group cannot observe the landowner's opportunity costs, landowners may be able to use the conservation group's uncertainty regarding c_i to gain payments higher than the minimum required level. By failing to account for the uncertainty in c_i and simply assuming that the cost of conservation is equal to a landowner's opportunity costs, conservation planning exercises may result in distorted site prioritisations and unrealistic expectations regarding conservation outcomes.

To overcome this shortcoming, I determine the upper bound on conservation costs, investigate where negotiated settlements between conservation groups and landowners would likely conclude, and examine the consequences for conservation of failing to account for cost uncertainties. These analyses begin in Chapter 2 in which I model the situation where a conservation group identifies one site for protection. After having identified this site, I am able to calculate the maximum that the conservation group would be willing to pay to secure it for conservation. In Chapter 3, I generalise the situation to one in which the conservation group identifies multiple sites for investment. In this chapter, I also simulate discriminatory-price conservation auctions so as to determine where in the range of possible conservation costs a negotiation between a conservation group and landowner would conclude. These two chapters deal with binary optimisations: a site is either enrolled for conservation or not.

However, many conservation investments, such as agri-environment schemes or conservation easements, can be continuous in nature with specific amounts of land enrolled in the conservation programme, for example. In Chapter 4, I examine the range of conservation costs in conservation investments of this type.

1.5.2 Cost structures in conservation planning

In describing the optimisation formulation of various conservation schemes, all the investments were limited by a cost structure that is used almost ubiquitously in conservation planning studies. With this cost structure, the cost to conserve multiple sites is simply the sum of the costs of the individual sites. This is the simplest cost structure that could be applied while recognising heterogeneous costs and is likely a gross oversimplification of the economics of conservation. In Chapter 5, I model two conservation planning problems to investigate how conservation reserves are spatially distributed when alternative cost structures are incorporated into analyses.

1.5.3 Trade-offs in conservation investments

When planning conservation investments, conservation groups must trade-off conservation improvements against the increasing costs of achieving those improvements. The nature of the trade-off curve faced depends on the particular distribution pattern of biodiversity targets across sites. Focusing on species richness as an indicator of biodiversity, in Chapter 6 I analyse how the trade-off between conservation costs and species coverage levels is manifest over a range of biodiversity distributions by varying the nestedness and the scarcity of species over the landscape. To undertake the analyses, I develop a linear programme that is specifically designed to map trade-off curves. Alongside the trade-off analyses, I investigate the potential improvement in cost-effectiveness that this new linear programme offers in comparison to the maximum coverage problem, a problem formulation that has been employed in previous trade-off analyses.

1.5.4 The optimal length of conservation contracts

Variable length conservation contracts are the mainstay of many conservation programmes. As noted, finite length contracts in agri-environment schemes in the EU receive \$7.2 billion per annum and cover 20% of European farmland (Cooper et al. 2009). Moreover, within the EU and globally there is large variability in the length of conservation contract employed in different schemes, ranging from those that last a few years to those that last a few decades (see Table 7.1 for examples). Despite such levels of investment, widespread application, and variation in duration, very little research has been undertaken to analyse the situations in which contracts of differing lengths are optimal. In Chapter 7, I address this question by investigating how uncertainty regarding the future availability of sites and their future ecological condition influences the choice between either short or long conservation contracts.

1.6 Areas not covered in this thesis

The investigations in this thesis cover several diverse areas in the design of voluntary conservation investments. Nonetheless, there are many important areas of conservation planning that I do not consider. To conclude this general introduction, I highlight and exemplify areas of conservation outwith the scope of this thesis but which nonetheless warrant discussion and/or offer promising avenues for future research.

1.6.1 Other forms of conservation instrument - Regulations

While conservation can be pursued by voluntary measures, states may compel citizens to change behaviours to achieve conservation objectives through command and control regulations. Regulations, such as limiting access to or use of land, were the policy choice of favour throughout the developed world in the 1970s and 1980s (Ekins 1999). However, command and control regulations suffer from several drawbacks, the recognition of which has led to the growing development and implementation of voluntary schemes. Optimally designed conservation measures are those that target landowners who offer the highest conservation benefit at the lowest cost. However, targeting such landowners in a regulatory setting places a very substantial information burden upon the regulator. As a result, regulations are often structured with limited flexibility, applying the same restrictions uniformly, leading to inefficiencies (Engel et al. 2008). The use of regulations also requires strong government institutions to ensure monitoring and enforcement, a situation that is lacking in many developing countries of conservation importance (Engel et al. 2008; Clements et al. 2010). Also, even in developed countries, many people depend on the services provided by ecosystems for their livelihood. The inflexibility of regulations and hence their insensitivity to the needs of different communities can lead to economic hardships and social conflict (Bulte and Engel 2006). Finally, voluntary conservation measures can be more politically palatable than regulations as they provide positive incentives to engage in conservation activities (e.g., financial payments) compared to the negative incentives of regulations (e.g., fines for non-compliance).

Voluntary approaches to conservation can alleviate some of these problems. For example, PES schemes are often run and/or funded by non-governmental organisations (NGOs), such as the World Bank or Conservation International (Pagiola 2008; Mittermeier et al. 2009). These organisations provide an institutional framework that can administer conservation projects even in countries with weak or dysfunctional governments. Nonetheless, many design issues are shared across voluntary and regulatory approaches. For example, the majority of voluntary conservation schemes offer uniform payments to all landowners irrespective of spatial variation in conservation opportunity costs (Hanley et al. 2012). This also leads to inefficiencies with all but marginal landowners receiving surplus from conservation contracts (Armsworth et al. 2012). The question therefore arises: what are the relative merits and demerits of different conservation instruments across planning scenarios and is there is some optimal mix of differing approaches to securing the provision of biodiversity?

1.6.2 Other issues in contract design - Paying for actions or results

As outlined in this introduction, the problem of optimally designing conservation contracts spans a wide range of disparate issues. As such, I am only able to consider a limited subset of design factors. Whether PES schemes should pay for actions undertaken or for results achieved provides an example of an interesting and developing area of research in contract design that the models presented in this thesis do not address.

For several reasons, most PES schemes pay for actions rather than outcomes (Gibbons et al. 2011; Hanley et al. 2012). First, biodiversity responds to a host of factors, operating at varying scales. For example, while avian species diversity depends on farm-scale factors, such as labour inputs, it also depends on regional characteristics and climate change (Dallimer et al. 2009). Therefore, biodiversity levels on an individual landowner's site can only be partly influenced by his actions. As a result, outcomes-based contracts come with greater risk to landowners than do actions-based contracts, meaning that risk premiums may have to be paid to ensure participation rates (Whitten et al. 2007). Second, actions are generally thought easier to observe than outcomes. For example, counting bird populations on a farm will be significantly more expensive than noting whether or not a landowner drained a wetland (Hanley et al. 2012).

Despite the disproportionate reliance on payments for actions, payments for outcomes can be advantageous in several respects. First, payments for outcomes naturally incentivise the participation of those landowners for whom it is relatively inexpensive to secure improvements to biodiversity (Gibbons et al. 2011). As a consequence, this can reduce the requirement for conservation groups to actively target low cost landowners, a task that can be costly and complex. Second, conservation groups often have limited information on how biodiversity will respond to particular management actions on individual sites and how this response will vary over the landscape (Hanley et al. 2012). Payments for outcomes encourage landowners to use their private information on site characteristics to optimise outcomes (Gibbons et al. 2011). Finally, if particular management actions are very expensive for the buyers of environmental services to observe, paying for outcomes can be more efficient than paying for actions (Hanley et al. 2012).

A small number of theoretical papers have addressed the question of whether PES should pay for actions or results (Whitten et al. 2007; Gibbons et al. 2011). These investigations indicate that the disproportionate use of conservation contracts that pay for actions may, in some contexts, come at the expensive of efficiency and cost-effectiveness. A greater focus on uncovering when and where payments for outcomes are the more appropriate tool is needed.

1.6.3 Factors besides biology and economics - Human perceptions and attitudes

In this thesis, I consider only the biological and economic aspects of conservation planning and assume that landowners are focused solely on revenue maximisation. However, evidence suggests that individuals are rarely motivated only by their levels of income (Franks 1987) and that attitudes towards and willingness to engage in conservation activities are determined by a complex web of social, economic, cultural and religious factors (Waylen et al. 2009; Guerrero et al. 2010; Hanley et al. 2012). For example, as will be discussed in Chapters 2-4, it is commonly assumed in conservation planning studies that all land in a given area is available for conservation and can be purchased at landowners' opportunity costs (e.g., Ando et al. 1998; Polasky et al. 2001; Naidoo and Iwamura 2007; Carwardine et al. 2008; Stoms et al. 2011). Testing this contention, Knight et al. (2011) mapped the willingness of 48 land managers in the Eastern Cape province of South Africa to sell their land for conservation. In stark contrast to the assumption of full availability of land, the authors found that less than one-quarter of land managers were willing to sell their land, severely limiting the potential for conservation. As well as economic factors, land managers were unwilling to sell their land for conservation, even at a price premium, because of historical and cultural ties to the area. Other attitudinal factors have also been shown to limit the willingness of landowners to engage with conservation. From the same survey of 48 land managers in South Africa, Knight et al. (2010) found that negative past experiences and concerns about motives resulted in approximately 90% of surveyed landowners being unwilling to collaborate with conservation agencies or NGOs. While In Italy, Defrancesco et al. (2008) showed that AES participation is determined not only by levels of financial compensation but also by local attitudes towards environmental conservation.

These results offer the first glimmerings of a rich new vein in conservation research. Notwithstanding the limitations of spatial prioritisations outlined earlier in this chapter, methods that incorporate socio-economics and biology continue to grow in sophistication, offering new insights into the problems of conservation biology (Cooke et al. 2009). Mapping human perceptions of and attitudes towards conservation and understanding how these affect conservation opportunities has been lacking, however (Knight et al. 2011). The most successful conservation plans will be those that are able to navigate the multidimensional landscape created by humanity's interactions with nature. Prioritisations that consider factors besides biology and economics are a necessary step in creating such plans.

Chapter 2

Landowners' ability to leverage in negotiations over habitat conservation

2.1 Introduction

Habitat destruction, degradation and fragmentation are the primary drivers of losses to terrestrial biodiversity (Millennium Ecosystem Assessment 2005). One of the key mechanisms used to combat this loss has been habitat protection through land acquisitions, either in full or in part. Many such agreements are reached through a negotiation process between the landowner and the conservation group.

Tools have been developed to assist conservation planners to prioritise the locations and sites in which to invest (Possingham et al. 1993; Costello and Polasky 2004; Meir et al. 2004; McDonald-Madden et al. 2008). In recent studies, research has focused on the relative impact of biodiversity and cost variance and on the role of their spatial covariation in determining the most effective outcome for conservation. Naidoo et al. (2006) illustrate that one can achieve the greatest coverage of biodiversity at the lowest overall cost when biodiversity and site cost are strongly negatively correlated across the candidate land parcels.

With the exception of Carwardine et al. (2010) who investigate the sensitivity of conservation priorities when sites have uncertain acquisition costs, previous studies assume conservation costs are predetermined and can be estimated using proxies for the value of alternative land uses on a site. For example, Ando et al. (1998) studied the effect of conservation costs on conservation priorities across the USA using average county-level agricultural land values to estimate costs. Polasky et al. (2001) studied the same problem using finer grain data, which partitioned the western two-thirds of the state of Oregon into 289 individual land parcels, taking the average land value as the cost of providing conservation. On a global scale, Naidoo and Iwamura (2007) estimated conservation costs using the gross value of yields from livestock and crop production based on average prices.

There are few empirically derived estimates of actual conservation costs. In the first large scale study of actual acquisition costs, Davies et al. (2010) found that the upfront cost of land acquisition ranged over six to eight orders of magnitude. This variation greatly exceeds that found in estimates of conservation costs relied on in previous studies, which ranged between two to four orders of magnitude (Ando et al. 1998; Naidoo et al. 2006). This begs the question as to why the disparity between variation in estimated and actual conservation costs exists and what it implies for conservation.

While past studies have assumed that the cost of conservation can be predetermined, in a voluntary negotiation process, the range of cost outcomes is bounded below by the landowner's willingness-to-accept (WTA) and above by the conservation group's willingness-to-pay (WTP). The landowner's WTA is a measure of the cost to the individual of providing conservation benefits and is a measure of several components, including the value of the best alternative land use as well as a site owner's individual preferences (Guerrero et al. 2010). Crucially, a landowner's WTA is private information that a conservation group cannot observe, resulting in landowners having an informational advantage over conservation groups. The use of proxy data is an attempt to estimate the WTA solely through often coarse estimates of the value of the best alternative land uses. Moreover, assuming that negotiated settlements can be reached at the landowner's WTA presumes that those who "produce" biodiversity (landowners) receive none of the surplus available from trade and that those who "consume" biodiversity (conservation groups) receive all of the surplus. This situation is the best case-scenario for conservation. The informational advantage that landowners hold means that this scenario may be impossible to achieve.

In this chapter, I analyse the situation where the cost of conservation is not fixed a priori and the site owner can exploit the conservation group's uncertainty regarding their WTA to hold-out for a higher payment, thereby leveraging some of the conservation group's excess WTP. The conservation group's WTP is the maximum value that they would be willing to pay in a negotiation on a particular site. Should agreement be reached at the WTP, the landowner would obtain the entire surplus available from trade, resulting in the worst-case scenario for conservation. While the WTA is unknown to the conservation group, they base their WTP in part on the a priori estimates of the cost of securing conservation on other sites and their budget constraint, should one apply. I assume that the objective of the conservation group is simply to arrive at the highest biodiversity coverage possible. Therefore, WTP is bounded by the conservation group's overall budget and is set by the availability of substitute sites that are as important to their overall conservation strategy.

Provided WTP>WTA, a negotiation that arrived at any value between these two would allow conservation to proceed. Therefore, the site owner can hold-out for greater amounts of surplus so long as WTP−WTA ≥ 0. The leverage potential is thus WTP−WTA, which I denote by Δc_i for site i. As leverage potential represents the worst-case scenario for conservation it exists independently of any negotiation strategy. Different negotiation strategies will, however, result in different cost outcomes within the range of WTA to WTP. The final figure that will be arrived at in a negotiation is dependent upon the chosen negotiation structure and the information available to each party. However, leverage potential bounds the set of possible outcomes.

Landowners' leverage potential should be greater on parcels for which there are few substitutes (because they contain particularly unique species or a disproportionate amount of some conservation target). All else being equal, by definition leverage potential will also be larger on sites where landowners' WTA is lower. Since $\Delta c_i = \text{WTP}_i - \text{WTA}_i$, a low WTA means that there is more scope for a landowner to leverage before the WTP is reached. Much of the analysis, therefore, hangs on the distribution of the biodiversity benefit to WTA ratio for a particular site.

To illustrate these ideas, in Section 2.1.1 I simulate leveraging by landowners in the context of an optimisation which maximises the coverage of species subject to a budget constraint. Then, in Section 2.3, I examine how leverage potential on particular parcels is affected by variation in the biodiversity value of properties, variation in the value of alternative land uses (landowners' WTA) and covariation between these two factors. Using both simplified analytical and numerical approximations that do not yet account for community complementarity, I focus on the prioritisation of one site. Returning to the maximisation of species richness, in Section 2.4 I use multiple conservation datasets and simulated data to explore the magnitude of leverage potential in real data. Finally, in Section 2.5, I incorporate community complementarity and determine leverage potential in the multi-site selection context as well as investigating the factors that influence an individual's ability to leverage.

2.1.1 Illustration of leveraging

To illustrate these ideas, I run a maximum coverage optimisation (Church and ReVelle 1974). With this optimisation a conservation group seeks to maximise species richness across conserved properties through the acquisition of land rights subject to a budget constraint. The optimisation is run with site costs equal to landowners' WTA.

Data for the optimisation comes from biodiversity and socio-economic surveys of 44 farms in the Peak District of the UK. The biodiversity data documents the presence or absence of 81 bird species over the farms (Dallimer et al. 2009) while the socio-economic data records the estimated annual, per hectare, rental price of the properties (Acs et al. 2010). I use these rental prices to approximate spatial variation in landowner's WTA, which

Table 2.1: *Top section:* Optimal reserve set for illustrative maximum coverage optimisation using the Peak District data. *Bottom three sections:* Leveraging by each of the landowners from the initial optimal set. Costs were increased iteratively until there was a change in the optimal set and the relevant change is noted. The process was then repeated until the site of the landowner who was applying leverage was no longer included in the new optimal set. At this point the landowner's leverage potential is reached. Δc_i is the leverage applied by landowner i given as the absolute value and as the percentage of the site WTA. All costs are in US$1000s

Optimal Set	Costs	Total Cost	Richness
[1 5 19]	[24.8 65.0 43.7]	133.5	58
Δc_1	New Set	Total Cost	Richness
32.1(129%)	[1 5 12]	162.2	57
35.4(143%)	[1 5 31]	165.2	56
35.8(144%)	[1 5 9]	143.9	55
55.5(223%)	[1 5 32]	156.7	54
64.4(259%)	[5 7 19]	161.2	53
Δc_5	New Set	Total Cost	Richness
32.1(48%)	[1 5 12]	162.3	57
35.4(54%)	[1 5 31]	165.2	56
35.8(55%)	[1 5 9]	143.9	55
26.8(85%)	[1 5 32]	156.7	54
64.4(99%)	[1 2 12]	150.7	51
Δc_{19}	New Set	Total Cost	Richness
32.1(73%)	[1 5 12]	130.2	57

is unobservable, and transform them to represent whole farm rental for a decade. All figures are given in USD at 2006 values. I use a budget constraint of $165,600 (three times the mean site cost). For the purposes of this illustration, I also assume rights can only be purchased on at most three of the 44 parcels. This constraint is used for expository purposes in this section only given the need to limit the number of possible combinations in order to document the sequence of changes in the reserve sets brought about by landowner leveraging (Table 2.1). The presented figures for leverage potential will therefore be slightly inflated. In Section 2.5, I return to multisite selections without this additional constraint.

The top section of Table 2.1 shows the result of the initial optimisation when assuming rights over conservation activities can be purchased at the landowners' WTA. The bottom three sections of Table 2.1 show leveraging by each of the landowners from the initial optimal set. The amount of leveraging applied that leads to the documented changes is determined by performing the same optimisation but this time one of the landowners in the initial optimal set demands iteratively higher payments.

When the conservation group can buy land rights at the landowners' WTA, the initial optimal set contains sites labelled 1, 5 and 19. Of the possible 81 species, 58 occur on these three properties and the overall cost of securing conservation agreements on them is \$133,500. Now suppose that the landowner of site 1 holds-out for more money than that offered initially by the conservation group. Agreement can still be reached even at an increase of 129% of the initial cost for site 1. At this point the conservation group's budget constraint forces it to consider a different set of sites. Even with the increase in cost demanded by the landowner of site 1, the next best alternative still includes this site (Table 2.1). Instead, site 19 drops out of the optimal set and there is a decrease in the richness of species protected by the conservation agreements. This process can continue with the landowner of site 1 holding-out for higher payments until the cost of securing a conservation agreement on that site has increased by 259%, at which point the conservation group is forced to choose a set of sites that does not include site 1. The same processes are evident with leveraging on the other sites in the initial optimal set.

For the landowners of sites 1 and 5, the initial effects of their demand for higher payment levels are felt elsewhere in the optimal set, with sites other than their own dropping out. On the other hand, site 19 is the first to drop out if the landowner of that site demands an increase that forces the conservation group to the limit of its budget. These different dynamics among sites in the optimal set suggests an important role for site irreplaceability (Pressey et al. 1994) in setting an individual landowner's leverage potential, an issue I return to in Section 2.5.

Overall, leveraging by landowners reduces the conservation group's ability to achieve its objective, with fewer species conserved for a given budget.

2.2 Methodological approach to single site problem

Before returning to multisite selections and the effects of complementarity, I first considered simpler consequences for landowners' leverage potential based on variation in and co-variation between the biodiversity value of parcels and landowners' willingness-to-accept conservation agreements upon them.

2.2.1 Step 1 - analytical approximation

To develop an analytical approximation that examines the consequences of the distributions of biodiversity and landowners' WTA, I consider the situation where a conservation group seeks to identify one site in which to invest as part of its broader conservation strategy.

Let the biodiversity value of candidate sites and landowner's WTA each be identically and independently distributed normal random variables that may be cross-correlated with one another. I denote by b_i the value of biodiversity on site i and by c_i the WTA on site i ($i = 1, 2, \ldots, n$). To identify priority sites for investment, one strategy would be to pick the site that offered the greatest improvement in biodiversity per dollar spent, based on the ranking of sites

$$\frac{b_n}{c_n} > \frac{b_{n-1}}{c_{n-1}} > \ldots > \frac{b_1}{c_1} \tag{2.1}$$

where b_n/c_n denotes the highest benefit to WTA ratio offered by any site, b_{n-1}/c_{n-1} denotes the next highest, and so on. Leverage potential is then related to the difference between the highest and next highest ratios. I will refer to this as the ratio difference. Estimating this ratio difference proves easier than estimating leverage potential itself.

I rely on an approximation of the Hinkley-Fieller (HF) distribution (Fieller 1932; Hinkley 1969) to develop the analytical approach. The HF distribution is the ratio distribution of two correlated normal variables. The probability density function (pdf), $f(x)$, and cumulative density function (cdf), $F(x)$, of the approximation are

$$f(x) = \frac{b(x)d(x)}{\sqrt{2\pi}\sigma_1\sigma_2 a^3(x)} \tag{2.2}$$

$$F(x) = \Phi\left\{\frac{\theta_2 x - \theta_1}{\sigma_1\sigma_2 a(x)}\right\} \tag{2.3}$$

where θ_1 and θ_2 are the biodiversity mean and the WTA mean, respectively, and σ_1^2 and σ_2^2 are the corresponding variances. Expressions for a, b, d and Φ are given in the appendix (Section 2.7). I use the above approximation of the full HF distribution for two reasons. First, it is significantly simpler than the full HF distribution and, second, the approximation is such that the denominator distribution takes only non-negative values. In the context of the problem in hand, this distribution means that biodiversity is measured as the change in biodiversity from the mean improvement possible on all sites and costs are non-negative. The assumption underlying this approximation is that the variance in the denominator distribution (landowners' WTA) is smaller than the mean. Indeed, as $\theta_2/\sigma_2^2 \rightarrow \infty$ the approximation becomes exact. (See Hinkley (1969) for a discussion of the difference between the exact distribution and the approximation and the appendix for the mathematical description of the full HF distribution and the approximation).

2.2.2 Step 2 - Distributions of b_n/c_n and b_{n-1}/c_{n-1} calculated through the order statistics of the HF distribution

To calculate the relevant expected values for the ratio distribution corresponding to the largest values in Eqn. (2.1) I rely on the pdf for the n-th and $(n-1)$-th order statistics. Using the standard form of the pdf for the k-th order statistic from any distribution it can be shown that the pdf of the n-th and $(n-1)$-th order statistic have the following respective forms, where $f(x)$ and $F(x)$ are the parent distribution's pdf and cdf respectively (Arnold et al. 2007),

$$f_{X_{(n)}}(x) = nF(x)^{n-1}f(x) \tag{2.4}$$

$$f_{X_{(n-1)}}(x) = n(n-1)F(x)^{n-2}(1-F(x))f(x) \tag{2.5}$$

2.2.3 Step 3 - Calculation of the expected value of $\frac{b_n}{c_n} - \frac{b_{n-1}}{c_{n-1}}$

I am now in a position to combine the definition of the HF distribution from step 1 with the order statistics from step 2 to calculate the expected value of the ratio difference. Using Eqns. (2.2) - (2.5), this is

$$\begin{aligned} E(X_{(n)} - X_{(n-1)}) =& n\int_{-\infty}^{\infty} xF(x)^{n-1}f(x)\ \ dx \\ & -n(n-1)\int_{-\infty}^{\infty} xF(x)^{n-2}(1-F(x))f(x)\ \ dx \end{aligned} \tag{2.6}$$

Eqn. (2.6) allows me to determine how the ratio difference is affected by variation and covariation in biodiversity and landowners' WTA, and sheds light on how leverage potential is affected. I solve Eqn. (2.6) using the adaptive Gauss-Kronrod quadrature method.

2.2.4 Simulation methods

I also conducted numerical simulations to approach the same questions as with the analytical approximations. These simulations served two purposes. The first was to evaluate the analytical approximation. The second was that they allowed me to transform the ratio difference into the leverage potential. The analytical approximations calculate the ratio difference because it was infeasible to determine the expected value of the leverage potential itself given the complexity of Eqn. (2.6). However, this can be approximated in the numerical simulations. The ratio difference is given by

$$\frac{b_n}{c_n} - \frac{b_{n-1}}{c_{n-1}}$$

Δc_n is the increase in WTA from the anticipated level for the site owner with highest benefit to cost ratio. Thus, the WTP is exhausted when

$$\frac{b_n}{c_n + \Delta c_n} = \frac{b_{n-1}}{c_{n-1}}$$

From which it can be easily seen that the leverage potential is

$$\Delta c_n = \frac{b_n c_{n-1}}{b_{n-1}} - c_n \tag{2.7}$$

This value can be extracted in the numerical simulations. While this quantity is itself informative, of greater importance is its relation to the WTA. Therefore, in the analysis, I document the maximum percentage increase in cost to the conservation group of securing a conservation agreement on a given site that could result from leveraging relative to the landowner's WTA.

2.2.5 Organisation of the results

The analysis in Section 2.3 is comprised of two parts. In the first part, I quantify the role of the variance in biodiversity values and in landowners' WTA in influencing leverage potential. In the second part, I do the same for covariation between biodiversity and landowners' WTA. I investigate the role of variance and covariation in determining the ratio difference with the analytical approximation and both the ratio difference and leverage potential in the numerical simulations. To explore the role of variance in biodiversity values and WTA, in the analytics I changed a variance parameter for WTA while keeping that for biodiversity value across sites constant and vice versa. In the simulations, I randomly drew values from a normal distribution for the biodiversity values and values from a normal distribution that was truncated below by zero for the landowner's WTA. The parametric variance in one distribution was held constant and I determined the leverage potential and ratio difference 1000 times for iteratively higher levels of variance in the other distribution. I report the mean ratio difference and leverage potential as well as 95% confidence intervals. To examine the role of biodiversity to WTA covariance in setting the ratio difference and leverage potential, I varied the covariance parameter from -1 to 1 in the analytical approximation and in the numerics I draw values from the required distributions 1000 times for each value of correlation coefficient again reporting the mean value along with 95% confidence intervals.

2.2.6 Parameter values

To parameterise the model, I use values that approximate those from the Dallimer et al. (2009) and Acs et al. (2010) data. Biodiversity mean is set at zero with variance of 100. I have used biodiversity mean of zero because as I am using a normal distribution it seems more appropriate to measure biodiversity as the change in mean improvement possible on all sites, thus allowing for negative values. For WTA, the mean value is set to 1000 with variance of 5×10^4.

2.3 The role of variance and covariance in setting leverage potential

2.3.1 Ratio difference and leverage potential for WTA and biodiversity variance

Fig. 2.1 shows how the ratio difference and leverage potential are affected by increasing variance in the value of biodiversity across sites, while keeping the variance in landowners' WTA constant and vice versa. Both the ratio difference and leverage potential respond to a greater degree to increasing WTA variance. While the ratio difference increases asymptotically for biodiversity variance, the leverage potential remains stable at around 30% of the WTA. For WTA variance, the ratio difference continues to increase over the range of variance and this is replicated in the leverage potential, which ranges from approximately 22% to over 55% of landowners' WTA. Also evident is that the ratio difference and leverage potential have the same shape for increasing WTA variance. This is not true for the biodiversity value variance but inspection of the figure highlights that there is a correspondence between the ratio difference and leverage potential in this case, with peaks and troughs at the same levels of variance.

2.3.2 Ratio difference and leverage potential for WTA and biodiversity covariance

Turning to the role of spatial covariation of biodiversity and landowners' WTA, I solve Eqn. (2.6) for varying correlation coefficient. Fig. 2.2 shows that the correlation coefficient of landowners' WTA and the biodiversity value across properties plays an important role in determining both the ratio difference and the leverage potential. Both are highest for strong negative correlation of biodiversity value and landowners' WTA across properties, the situation thought to be most favourable for conservation outcomes (Naidoo et al. 2006), and lowest at strong positive correlation of these distributions. When biodiversity value and landowners' WTA across properties are strongly negatively correlated, the cost of securing

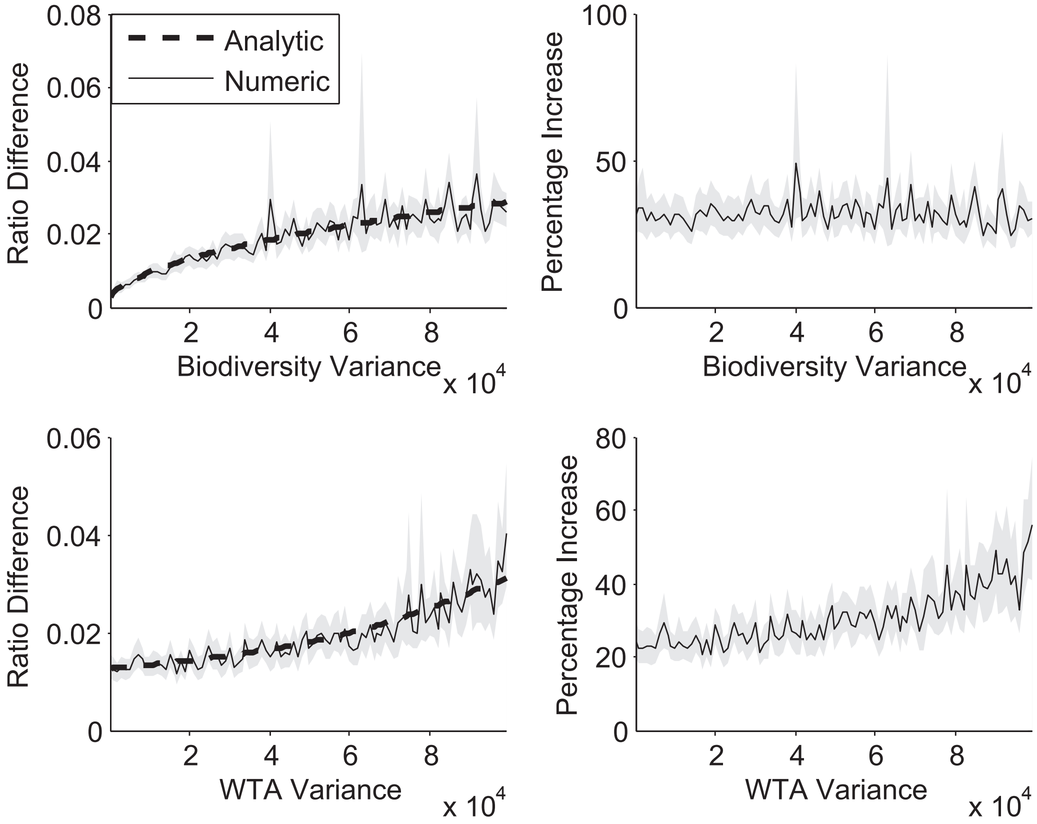


Figure 2.1: Effect of variance in biodiversity value and variance in landowners' WTA across properties on one landowner's ability to leverage. *Left Column*: Analytically and numerically derived ratio difference. *Right Column*: Numerical estimate of leverage potential expressed as a percentage of the landowner's WTA. *Top Row*: Ratio difference and leverage potential when biodiversity variance is increased and variance in WTA is held constant. *Bottom Row*: Ratio difference and leverage potential when variance in WTA is increased and variance in biodiversity value is held constant. Parameters are $\theta_1 = 0$, $\theta_2 = 1000$ and the number of site is $n = 50$. The grey region is the 95% bootstrap confidence interval.

conservation agreements on a property can be increased by approximately 50% over the landowner's WTA. When these factors are strongly positively correlated across properties, the possible increase in cost of securing a conservation agreement is below 20%. In this case there is also a strong correspondence between the shape of the ratio difference (estimated analytically and numerically) and leverage potential (estimated numerically only).

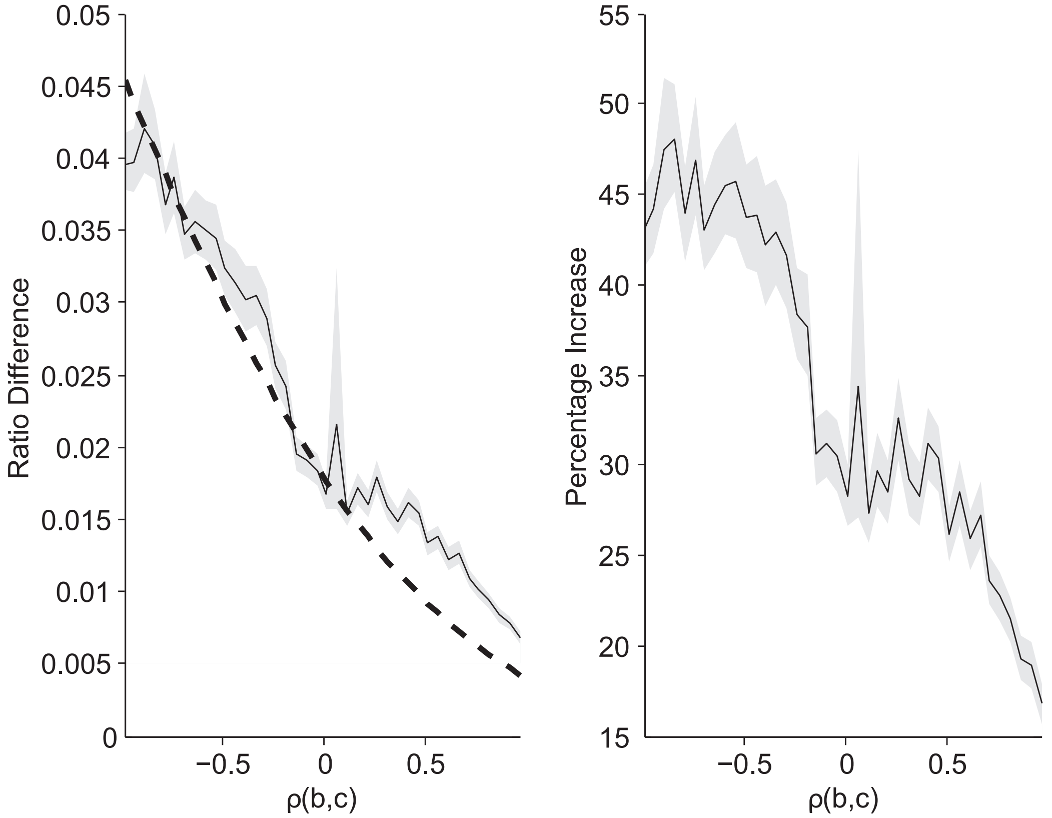


Figure 2.2: Effect of covariance between biodiversity values and landowners' WTA across properties on one landowner's ability to leverage. *Left Panel*: Analytically and numerically derived ratio difference. *Right Panel*: Numerically estimated leverage potential. The parameters are $\theta_1 = 0$, $\theta_2 = 1000$, $\sigma_1^2 = 100$, $\sigma_2^2 = 5 \times 10^4$ and $n = 50$. The grey area is the 95% bootstrap confidence interval.

2.4 Estimate of leverage potential for single site problem from empirical data

Dallimer et al. (2009) and Acs et al. (2010) provide data on biodiversity and land rents on individual properties allowing an empirical estimate of leverage potential in the single site problem to be obtained (Table 2.1). Other published sources presenting biodiversity and cost data tend to report these at much coarser scales (e.g., ecoregions in Murdoch et al. (2007) or states in Fishburn et al. (2009b)). While these other sources are clearly not reporting data at an adequate resolution to depict the process of individual landowners leveraging the biodiversity value on their properties, I include them here for illustrative purposes and for want of published studies presenting biodiversity and cost data at the property scale.

For each dataset listed in Table 2.2, I solved Eqn. (2.7). First, I ranked locations by their

Table 2.2: Leverage potential in empirical data. Leverage potential is given as the percentage increase that the owner of the site having highest biodiversity value to WTA ratio can demand before the site is no longer more desirable than the second best choice and is represented by Δc_1. Leverage potential is calculated from 5 conservation datasets. Also reported is the biodiversity variance (σ_1^2), WTA variance (σ_2^2), and their covariation ($\rho(b,c)$). The Underwood et al. (2008) paper looked at different regions and I used their data from Australia and the Mediterranean separately.

Source	Δc_1	σ_1^2	σ_2^2	$\rho(b,c)$
Fishburn et al. (2009b)	30%	2.1×10^3	2.2×10^6	-0.22
Underwood et al. (2008) Aus	24%	2×10^5	4.7×10^{18}	0.14
Murdoch et al. (2007)	14%	584	7.3×10^4	0.21
Dallimer et al. (2009)	9%	73.4	4.3×10^4	-0.33
Underwood et al. (2008) Med	8%	1×10^6	6.1×10^{11}	0.08

biodiversity to WTA ratio. Then, I determined the extent to which a “landowner” (likely a regional government for the coarser datasets) representing the location attributed the highest priority in this ranking could leverage in negotiations with a conservation group looking to secure a conservation agreement. Leverage potential was again determined by the point at which the biodiversity benefit to WTA ratio becomes equal to that of the second most beneficial location. The values for the leverage potential in Table 2.2 are in the same range (8-30% of the landowner’s WTA) as those found in Section 2.3.

2.5 Multisite selections

In Sections 2.3 and 2.4, I calculated leverage potential for the simplified case where a conservation group is seeking to secure a conservation agreement on a single site. Now I return to the multisite case illustrated in Section 2.1.1. As with that illustration, I still focus on estimating leverage potential when a single landowner holds-out for a higher payment. However, now I examine the situation where the relevant land parcel is part of a network of several sites on which the conservation group are seeking to secure habitat conservation agreements.

The leverage potentials from Section 2.1.1 (73-259% of WTA) were much larger than those found in Sections 2.3 and 2.4 (approximately 8-55% of WTA) that focused on securing a conservation agreement on a single site. Several reasons might account for this disparity. (I) The increased leverage potential in Section 2.1.1 is an artefact derived from only a single landowner holding-out for a higher payment. This individual is then able to capture the overall leverage potential across the set of sites prioritised for conservation that would likely be shared across different landowners. (II) If possible reserve networks were ranked instead of individual sites, the distributions of biodiversity value and landowners' summed WTA across reserve networks might look very different to those for individual sites. (III) The result may have been influenced by the constraint that a maximum of three sites could be selected. (IV) Finally, the disparity may arise because the substitutability of sites diminishes when recognising community complementarity.

2.5.1 Leverage potential in multisite problem for summed occurrences / without community complementarity

I first set out to explore the simpler role of factors (I) and (II) in setting leverage potential and then incorporate community complementarity to test the role of factors (III) and (IV). To achieve this, I first consider prioritising sets of sites based simply on the sum of species richness found on each site. This is equivalent to setting the conservation objective to maximising the occurrences of species.

When maximising the sum of species occurrences, rare and common species are valued equally. With the incorporation of community complementarity, the value of rare species increases given that the objective is to include as many species as possible in the network. Thus, sites that contain rare species are more desirable when considering complementarity as opposed to simply the sum of species occurrences. In the single site selection context the two metrics are equivalent. However, in the multisite selection situation the change in conservation objective could affect leverage potential. When first seeking to consider only factors (I) (one landowner is capturing everyone's leverage potential) and (II) (changed benefit-cost distributions) in setting leverage potential, I used both simulated data and the datasets used in the Section 2.4.

As noted earlier, only one available data source represents the true spatial scale of the problem, the Dallimer et al. (2009) biodiversity data and the Acs et al. (2010) cost data. Thus, in generating the simulated data throughout Section 2.5, I produced random binary matrices for species representation with the same dimensions as the Dallimer et al. (2009) data and bootstrapped the Acs et al. (2010) cost data. In the summed occurrences analyses, the rows of the binary matrix were summed to give the occurrences of species on each site.

To identify the optimal reserve network, I used a simple greedy heuristic. This algorithm selects the best site to be added to the network iteratively until the budget is exhausted. After having determined the reserve set, I calculate the leverage potential as in Section 2.1.1: for each of the sites in the reserve set, I iteratively increased its WTA until it was no longer a member of the set. I used a budget constraint of five times the mean site WTA. The use of the heuristic algorithm as opposed to the exhaustive search algorithm used in Section 2.1.1 removes the necessity of the three site constraint included there for illustrative purposes.

Table 2.3: Leverage potential of a single landowner when selecting multiple sites for protection. *Top Section:* Leverage potential from simulated data and conservation datasets when the objective of the optimisation (Opt) is to maximise the total number of species occurrences (SO). *Bottom Section:* Leverage potential from simulated data and the Dallimer et al. (2009) data when the objective is to maximise species richness (SR). Recorded are the mean figures along with an interval of leverage potential values. For the simulated data this interval is the 95% confidence interval and for the datasets the interval is the range of leverage potential values.

Source	Opt	Mean	Interval
Simulated Data	SO	70.2	[61.9 79.1]
Fishburn et al. (2009b)	SO	97.7	[17.8 172.7]
Underwood et al. (2008) Aus	SO	119.4	[0.1 208.1]
Murdoch et al. (2007)	SO	19.6	[2.9 93.6]
Dallimer et al. (2009)	SO	104.2	[0 233]
Underwood et al. (2008) Med	SO	90.7	[25.6 180.2]
Simulated Data	SR	122.9	[103.7 142.6]
Dallimer et al. (2009)	SR	132.6	[52 237.3]

The top section of Table 2.3 illustrates that even when not yet accounting for community complementarity in the multisite case, leverage potential can be significantly higher than in the single-site case. The mean value for the leverage potential in the simulated data was 70.2% of WTA. The conservation datasets gave similar values. Although for some sites there was little scope for leveraging, large increases were also possible, with some sites having leverage potential above 150% of the WTA.

2.5.2 Leverage potential in multisite problem with community complementarity

Finally, I account for the role of community complementarity when the objective is to cover as many species as possible in the reserve set and redundancy in species occurrences across sites is not rewarded. Because in this situation rare species are valued much more highly, the value of sites has the potential to be higher than when simply maximising species occurrences. This greater desirability of particular sites may allow landowners to hold-out for higher payments. Evidence of this can be seen both in Table 2.3 and the initial illustration of leverage potential in Section 2.1.1. Comparing the leverage potential results for the simulated data for summed occurrences and species richness shows that focusing on community complementarity considerably increases the leverage potential of the individual landowner.

Table 2.4: Leverage potential and irreplaceability for each of the sites in the initial optimal reserve from Section 2.1.1.

Site	Leverage Potential	Irreplaceability Value
1	31.1	0.30
5	31.1	0.57
19	15.5	0.09

The Dallimer et al. (2009) and Acs et al. (2010) data also allows for a direct comparison between the two and again leverage potential can be higher when valuing community complementarity.

Also evident is that the three-site constraint artificially inflated leverage potential. Comparison between the Dallimer et al. (2009) leverage potential for the community complementarity analysis and that calculated in Section 2.1.1 shows that the ability to hold out is around 20% lower when this constraint is removed.

In the illustrative example of Section 2.1.1, Table 2.1 highlights the fact that even when the sites labelled 1 and 5 demand higher payment levels, their inclusion remains necessary to meet the conservation objective through several changes in the optimal reserve set. In contrast, site 19 drops out of the optimal set when leveraging causes the budget constraint to be met for the first time. This difference among sites in the initial optimal reserve network may reflect site "irreplaceability", a metric that has been commonly used in the literature to capture site uniqueness, and which has been defined as the likelihood that a site will be required to meet a given set of targets (Pressey et al. 1994; Ferrier et al. 2000). To investigate the possibility that irreplaceability was an important factor in determining variation in leverage potential across sites in the multi-site selection scenario, I calculated the irreplaceability value of each of the three sites in the initial optimal set from Section 2.1.1. I determined all the combinations of sites that would lead to within 5 species of the maximum possible species richness. I measured site i's irreplaceability to the conservation strategy as the proportion of all possible solutions within this near optimal set that contained site i. The results in Table 2.4 suggest there may be a relationship between leverage potential and irreplaceability: the sites with higher leverage potential have a higher irreplaceability value.

In order to explore the possible relationship between site irreplaceability and leverage potential further I calculated leverage potential and irreplaceability in 1000 similar situations. To test the null hypothesis that there was no significant relationship between irreplaceability and leverage potential, I constructed a quasi-Poisson generalised linear model. The full model had as explanatory variables: irreplaceability, biodiversity to cost correlation and site cost.

The model rejected the null hypothesis of no relationship between irreplaceability (coefficient = 1.82, p-value < 0.001) and the biodiversity to cost correlation (coefficient = −0.39, p-value < 0.01). However, the model showed no evidence against the null hypothesis with respect to site cost (p-value = 0.46). Thus, irreplaceability and biodiversity to cost correlation are important variables in determining leverage potential in the multisite selection scenario.

These regression coefficients should be interpreted with caution because the response variable (leverage potential) does not necessarily satisfy the assumption of independence from the latter two explanatory variables (biodiversity to cost correlation and site cost). That being said, the issue of potential non-independence is more obvious when dealing with site cost, which offered little explanatory power anyway.

2.6 Discussion

Conservation planning studies increasingly emphasise the importance of accounting for the costs of securing conservation benefits. To estimate conservation costs, these studies rely on estimates of the market value of alternative land uses, which, at best, provide partial estimates of landowners' WTA conservation agreements (Dutton et al. 2008; Guerrero et al. 2010; Knight et al. 2010). In principle, a voluntary transaction between a conservation group and landowner could go ahead as long as the conservation group's WTP for conservation on the site exceeds the landowner's WTA. However, due to the fact that conservation groups cannot observe a landowner's true WTA, the possibility exists in a negotiation that landowners may be able to hold-out for higher payments and capture some of the surplus available from the trade. The maximum amount of surplus that landowners could capture is WTP-WTA, a quantity that I termed the landowner's leverage potential. In this chapter I examined how ecological and economic factors combine to determine this leverage potential across sites and the possible consequences of landowners leveraging for conservation strategies and their effectiveness.

Leverage potential is a function of landowners' WTA conservation agreements, the conservation group's overall budget and the potential substitutability of different sites when trying to achieve particular conservation objectives. While specific to this context, leverage potential is therefore related to ideas such as irreplaceability (Pressey et al. 1994; Ferrier et al. 2000), asset specificity (Huusom and Strange 2008) and fungibility (Salzman and Ruhl 2000) that arise elsewhere in the conservation planning literature.

Leveraging by landowners has the potential to considerably increase the cost of securing conservation on a site. When prioritising one site for protection, I found that the upper bound on the increase in cost that the conservation group could face due to leveraging was approximately 55% of the WTA. When moving to the multisite selection situation, leverage potential had the capacity to be higher still. For simulated data and a range of conservation datasets, the upper bounds on the cost of securing conservation agreements was often over 100% and in several cases over 200% of the landowner's WTA. Leveraging by landowners will reduce the overall effectiveness of conservation investments by increasing the costs of securing conservation agreements on a site. Interestingly, the illustrative example in Section 2.1.1 indicated that leveraging also determines the particular properties that should be priorities for investment and that leveraging by one landowner can change a conservation group's prioritisation of other available sites.

Evidence for this type of phenomenon was found in the recent study by Carwardine et al. (2010). In scenarios that sought to expand the protected area network in Queensland, Australia, the authors investigated the effect of uncertain costs on conservation priorities. Specifically, cost was estimated through a valuation of the unimproved land value and using this as the cost baseline a site prioritisation was then carried out. The authors then varied site costs and determined how the prioritisations were affected. One of the most striking results was that sites that were essential remained so throughout the range of cost change. This result complements that found in Section 2.1.1 of this chapter and highlights that landowners of high priority sites have the scope to demand much higher than anticipated costs. This is something that needs to be accounted for in conservation plans.

To unpick how different factors combine to influence leverage potential, I first considered the implications of trying to secure a conservation agreement on a single site. In these analyses, I examined the role of variance in biodiversity and landowners' WTA conservation agreements across sites. I found that WTA variance is more important in determining leverage potential than the variation in biodiversity values. Leverage potential was stable at around 30% of the landowner's WTA for increasing biodiversity variance. However, for increasing WTA variance, the increase that the landowner could demand became substantially higher. At the lowest level of variance the leverage potential was around 17% of the WTA. At the highest level of variance considered, the leverage potential rose to approximately 55% of the WTA. This result reinforces those from other recent studies that have shown that socio-economic factors can be more important in determining conservation priorities than variation in biodiversity levels (Ando et al. 1998; Naidoo and Iwamura 2007; Bode et al. 2008; Perhans et al. 2008).

The role of covariation between biodiversity values and landowners' WTA over sites was shown to have significant influence on the landowners' ability to leverage. Leverage potential was highest when there was a strong negative correlation between these distributions and lowest when this correlation was strongly positive. The change in leverage potential when moving from a strong positive to strong negative correlation was approximately 35% of the landowner's WTA. In the context of fixed predetermined costs, Naidoo et al. (2006) show that conservation will achieve most when there is a strong negative correlation between biodiversity value and costs across sites. However, it is in this situation that the ability for the site owner to hold-out for a higher payment level is greatest, potentially dissipating some of this advantage. More generally, this finding suggests that regional prioritisations for conservation investments that one might arrive at based on a priori estimates of WTA may ultimately prove less efficient than anticipated. Moreover, the disparity between the estimates of WTA and the actual cost arrived at through a negotiation process could produce an unrealistic picture of what conservation investments can achieve.

The results from Section 2.5 demonstrate that leverage potential is determined in part by the conservation group's objective. When moving beyond the special case of securing a conservation agreement on a single site to looking at networks of sites, I contrasted leverage potential when the conservation objective was to maximise species occurrences and when it was to maximise species richness. In general, flexibility on how conservation outcomes can be achieved reduces the potential for landowners to hold-out and leverage. Leverage potential was highest when the objective was to maximise species richness across a reserve network. When seeking to protect as many species as possible, the maximum increase available to landowners from the conservation datasets was 237% of WTA and for the simulated data the mean leverage potential was 123% of WTA. This value is in the region of 50% greater than the corresponding figure when seeking to maximise species occurrences. Evidence that this effect is due to the reduced substitutability of sites when seeking to maximise richness as opposed to occurrences was seen in the fact that site irreplaceability was positively correlated with leverage potential.

Past studies that have focused only on landowners' WTA to estimate the costs of delivering a conservation strategy present the most optimistic outcome for conservation groups. The estimates from landowners' leverage potential illustrate the most pessimistic situation for conservation. The actual cost of securing a conservation agreement on a property could in principle fall anywhere in between. The negotiated settlement that will be agreed in a willing-buyer, willing-seller transaction depends in part on the negotiation process that is followed. The nature of the negotiation process will depend on the particular conservation programme, which can range from governments contracting on large numbers of sites at one time to a conservation group seeking to purchase rights on a particular property in a fee-simple or easement transaction. In the next chapter, I analyse how different contract allocation mechanisms can affect negotiation outcomes between conservation groups and landowners.

Alongside the design of mechanisms for the allocation of conservation contracts, other factors will determine the figure in the range of WTA to WTP at which a negotiated settlement arrives. For instance, the role of timing will be influential in setting the negotiating power of either party. If a conservation group must act quickly to secure biodiversity on a site then the bargaining power of the landowner will be higher. As evidenced in Section 2.5 of this chapter, the substitutability of the sellers influences negotiation power. However, this is true also of the buyers, in this case the conservation groups. If the landowner has multiple conservation groups vying for his land then his negotiating power is strengthened and his ability to hold-out is increased.

2.6.1 Assumptions

Throughout the analyses, I made a number of assumptions. In Sections 2.2 and 2.3, for example, I used the Hinkley-Fieller distribution. This distribution is the ratio of two correlated normal random variables. I used a simplified approximation of this distribution that led to the appealing consequence of non-negative costs. The approximation becomes exact when the ratio of WTA mean to variance tends to infinity. Therefore, it would appear that a necessary assumption for the use of the approximate distribution was that WTA mean be significantly higher than its variance. However, throughout the analyses this assumption was violated: the smallest ratio of WTA mean to variance used in the investigations was $1/100$. Yet comparison between the theoretical and numerical results in Figs. 2.1 and 2.2 shows that they are in almost exact correspondence. Therefore, violation of the assumption that WTA mean should be much higher than the variance had no bearing on the results. Consequently, it appears that the use of the HF distribution to questions that require a ratio distribution need not be limited to the rather specialised cases where the mean of the denominator distribution is significantly larger than the associated variance.

In a second example, I focused on the leverage potential that one site owner could demand when analysing leverage potential in the multisite prioritisation problem, assuming that the other sites could be protected at the anticipated WTA. This gave the upper bound for the increase that an individual landowner could demand. Presumably, if more than one landowner demanded increased payment levels, the ability of other landowners to place leverage on the conservation group would be influenced. Leverage potential when multiple landowners hold-out for higher payments is the topic of Chapter 3.

Finally, the assumed level of the budget constraint needs to be carefully considered in the formulation that maximises species richness within a budget constraint (Section 2.5). In the absence of a valuation function that sets WTP for different levels of biodiversity coverage there exists the possibility that leverage potential could scale with the budget. Should a global optimal solution be found for a site selection problem within a particular budget, raising the budget from this level will increase the leverage potential by the same amount. I have attempted to avoided this particular pitfall by removing the artificial three site constraint used in Section 2.1.1 for illustrative purposes only and by assuming that budgets are always relatively small (5 times the mean site cost). Nonetheless, unrealistic and inflated values for leverage potential could in principle be arrived at with this formulation should the budget be made arbitrarily large (i.e. large enough that all species are protected). This factor though does not affect the analyses of leverage potential based solely on site substitutability (Section 2.3).

2.6.2 Conclusions

The results presented in this chapter reveal that there is potentially a large gap between landowners' WTA conservation agreements and a conservation group's WTP that bounds the actual cost at which a willing-buyer, willing-seller negotiation could arrive. Moreover, I have begun to analyse factors that cause variation in just how large this gap will be in different circumstances. Taken together, these results suggest that the current practice in conservation planning studies of estimating conservation costs based on partial estimates of landowners' WTA an agreement will overestimate just how much a conservation initiative can achieve. Equally importantly, this practice may also lead to inefficient distortions in funding priorities. In light of this, further studies that focus on the actual costs of securing conservation agreements (see for example, Stoneham et al. (2003); Latacz-Lohmann and Schilizzi (2005); Davies et al. (2010)) may offer the most promise for obtaining a more accurate empirical picture of what conservation can achieve. Moreover, greater attention is needed in conservation planning studies to how the negotiation process governing the allocation of conservation contracts can be designed to minimise landowner leveraging and drive the actual cost of securing conservation agreements as close to landowners' WTA as possible.

2.7 Appendix

Here, I present the mathematical formulation for the distribution of two correlated normal random variables. Let X_1 and X_2 be randomly distributed normal variables with means θ_i, variances σ_i^2 $(i = 1,2)$, correlation coefficient ρ and let $W = X_1/X_2$. Hinkley (1969) derived the exact distribution of W and also the distribution on the basis of the assumption that $X_2 > 0$.

If the joint density of (X_1, X_2) is $g(x,y)$ and the probability density function of W is $f(w)$ then

$$f(w) = \int_{-\infty}^{\infty} |y| g(wy, y)\, dy$$

Substituting the bivariate normal density for $g(x,y)$ and integrating gives

$$f(w) = \frac{b(w)d(w)}{\sqrt{2\pi}\sigma_1\sigma_2 a^3(w)}\left[\Phi\left\{\frac{b(w)}{a(w)\sqrt{1-\rho^2}}\right\} - \Phi\left\{-\frac{b(w)}{a(w)\sqrt{1-\rho^2}}\right\}\right] + \frac{\sqrt{1-\rho^2}}{\pi\sigma_1\sigma_2 a^2(w)}\exp\left(-\frac{c}{2(1-\rho^2)}\right)$$

where

$$a(w) = \left(\frac{w^2}{\sigma_1^2} - \frac{2\rho w}{\sigma_1\sigma_2} + \frac{1}{\sigma_2^2}\right)^{1/2}$$

$$b(w) = \frac{\theta_1 w}{\sigma_1^2} + \frac{\rho(\theta_1 + \theta_2 w)}{\sigma_1\sigma_2} + \frac{\theta_2}{\sigma_2^2}$$

$$c = \frac{\theta_1^2}{\sigma_1^2} + \frac{2\rho\theta_1\theta_2}{\sigma_1\sigma_2} + \frac{\theta_2^2}{\sigma_2^2}$$

$$d(w) = \exp\left\{\frac{b^2(w) - ca^2(w)}{2(1-\rho^2)a^2(w)}\right\}$$

Also

$$\Phi(y) = \int_{-\infty}^{y} \phi(u)\, du, \text{ where } \phi(u) = \frac{1}{\sqrt{2\pi}}e^{-\frac{1}{2}u^2}$$

The cumulative distribution function, F, of W is given to be

$$F(w) = L\left\{\frac{\theta_1 - \theta_2 w}{\sigma_1\sigma_2 a(w)}, -\frac{\theta_2}{\sigma_2}; \frac{\sigma_2 w - \rho\sigma_1}{\sigma_1\sigma_2 a(w)}\right\} + L\left\{\frac{\theta_2 w - \theta_1}{\sigma_1\sigma_2 a(w)}, \frac{\theta_2}{\sigma_2}; \frac{\sigma_2 w - \rho\sigma_1}{\sigma_1\sigma_2 a(w)}\right\}$$

where

$$L(h,\kappa,\gamma) = \frac{1}{2\pi\sqrt{1-\gamma^2}}\int_h^\infty\int_\kappa^\infty \exp\left\{-\frac{x^2 - 2\gamma xy + y^2}{2(1-\gamma^2)}\right\} dx\, dy$$

From the cumulative distribution function, it can be seen that as $\theta_2/\sigma_2 \to \infty$, or in other words as $\mathrm{pr}(X_2 > 0) \to 1$ then

$$F(w) \to \Phi\left\{\frac{\theta_2 w - \theta_1}{\sigma_1\sigma_2 a(w)}\right\}$$

Thus, assuming that the denominator distribution is bounded below by zero, the cumulative density function is as above with corresponding density function

$$f(w) = \frac{b(w)d(w)}{\sqrt{2\pi}\sigma_1\sigma_2 a^3(w)}$$

Chapter 3

Landowner surplus from conservation contracts: cooperatives, auctions, and conservation objectives

3.1 Introduction

As discussed in Chapter 2, in negotiations over habitat conservation, landowners have an informational advantage over conservation groups because they have more knowledge regarding the opportunity costs of conservation measures. The voluntary nature of many conservation schemes means that landowners may be able to use this informational advantage to gain some of the surplus available from trade. Consequently, conservation costs may be significantly greater than suggested by estimates of the landowner's willingness-to-accept (WTA), the cost lower bound that has been assumed in many conservation planning studies (e.g., Ando et al. 1998; Polasky et al. 2001; Naidoo and Iwamura 2007; Carwardine et al. 2008; Stoms et al. 2011). To ensure that conservation planning exercises accurately estimate conservation outcomes, it is necessary to know the range of possible conservation costs as well as how much surplus landowners could obtain in any negotiations.

In beginning to investigate these ideas in Chapter 2, I modelled the situation where a conservation group undertakes a prioritisation exercise to identify one or many potential sites in which to invest. I then determined the maximum amount that a single prioritised landowner could demand above his WTA, a value I termed the landowner's leverage potential, assuming that all other landowners agreed to sell at their WTA. Both the theoretical and numerical results from that chapter showed that, in this simplified situation for conservation investments, there is a large gap between the landowner's WTA and the conservation group's willingness-to-pay (WTP).

However, negotiation dynamics distinct from those considered in Chapter 2 can arise when multiple sites are identified for protection. Most obviously, attempting to secure multiple agreements permits more than a single landowner to engage in hold-out behaviour. This could not only influence the overall surplus that the conservation group would cede but could also influence the leverage potential of individual landowners. Negotiations with multiple landowners also create the conditions required for cooperation. Allowing for the collective provision of biodiversity by cooperating land managers is an idea that has a relatively long history in the academic literature (Franks and Russell 1996; MacFarlane 1998; Falconer 2000; Hodge and McNally 2000; Falconer 2002; Franks 2011). Implementation of such ideas has been epitomised by the Netherlands. Policy makers there have developed agri-environment schemes in which farmers work in collaboration with each other and with local, regional and national agencies (Franks 2011). By 2004, cooperative agreements existed between 10 per cent of all farmers in the Netherlands, covering 40 per cent of all agricultural land (Cooper et al. 2009). On a much smaller scale, the UK's Higher Level Stewardship Scheme offers a financial incentive for group applications for a single management option (Franks 2011). In contrast to this encouraged and overt form of cooperation, landowners can covertly cooperate through collusion (Clarke 1983; Laffont and Martimort 1997) by, for example, sharing information with one another on bids.

In this chapter, I first determine the maximum producer surplus that varying numbers of landowners can achieve when the conservation group wishes to secure rights on multiple sites. Given that the conservation group's WTP for any site is dependent upon the site's biodiversity value, central to the determination of this worst-case scenario for conservation is the assumption that landowners form agricultural cooperatives. In this way, those landowners who hold-out for payments larger than their WTA are able to leverage against the total biodiversity over their sites.

The second objective of the analyses in this chapter is to investigate where in the range of WTA to WTP a negotiated settlement is likely to arrive. This outcome is dependent, among other factors, upon the chosen contract allocation mechanism. Awareness of the problem of asymmetric information in conservation contracting has spurred research into the design of tendering schemes that minimise the ability of landowners to hold-out and thus push the price of conservation as close to the minimum payment level as possible. One allocation mechanism that has received substantial attention in conservation research and practice is discriminatory-price procurement auctions (Stoneham et al. 2003; White and Burton 2005). These have been shown to offer theoretical advantages over fixed-price schemes given that their competitive nature creates a price revelatory mechanism that can compensate for the informational advantage that the landowner holds (Latacz-Lohmann and Van der Hamsvoort

1997). Conservation schemes that have chosen to allocate contracts using such auctions include the Conservation Reserve Program (CRP). The CRP is the largest land retirement programme in the USA and has the structure of a multi-round, discriminatory-price auction with a bid cap (Khanna and Ando 2009).

Therefore, to gauge where in the range of cost between WTA and WTP that negotiations would conclude, and to investigate how different negotiation structures affect the outcome, I simulate a range of multi-round auctions with bid caps, a structure similar to the CRP. Auctions with three common conservation objectives are simulated. In the first, the biodiversity value of sites is assumed to be homogeneous and the conservation group protects the largest amount of land subject to their budget (Latacz-Lohmann and Van der Hamsvoort 1997; Schilizzi and Latacz-Lohmann 2007). In the second, biodiversity values are heterogeneous over the landscape and benefits accrue additively over sites (Stoneham et al. 2003; White and Burton 2005). Such a target is consistent with the maximisation of species' occurrences, for example. With these objectives, sites are considered to be substitutes, in that if the cost of one site increases, the demand for other sites would increase. The objective of the third conservation auction is to maximise species richness, in which biodiversity values are heterogeneous and benefits accrue non-additively. Specifically, the biodiversity value gained by protecting one site is dependent upon the biodiversity value of other protected sites. In contrast to the other objectives, maximising the number of species covered by the protected sites results in sites being partial complements rather than substitutes (Margules et al. 1988; Church et al. 1996; Justus and Sarkar 2002). Finally, given the growing prevalence of agricultural cooperatives in conservation projects, I analyse how landowner cooperation affects both the economic and biodiversity outcomes of the different auctions.

3.2 Materials and methods

3.2.1 Data generation

To undertake the analyses, I simulated biodiversity and WTA data by fitting the best-fit distribution to the biodiversity and economic data from the Dallimer et al. (2009) and Acs et al. (2010) studies that were introduced in Chapter 2. As described, this biodiversity data records the presence or absence of 81 avian species over 44 farms and the economic data records the annual, per hectare, rental prices of the properties. In all cases, I used the Kolmogorov-Smirnov test to determine the best-fit distributions. For both species richness and incidences, the best fit distribution was the three-parameter Weibull distribution. For richness on sites, the distribution parameters were $\alpha = 4.3, \beta = 28.8, \theta = -3.6$. For species incidences over sites, the distribution parameters were $\alpha = 0.9, \beta = 11.3, \theta = 0$. For the site WTAs, the best-fit distribution was the generalised β-distribution on the interval [5.5 124], with distribution parameters $\alpha = 0.66, \beta = 0.91$. As in Chapter 2, I scale these per hectare values by the farm area and assume that contracts last for 10 years. Figures are given in USD at 2006 values.

I constructed random presence or absence matrices that accounted for the nestedness of natural populations (Wright et al. 1998). To achieve this, I drew 44 random variables from the best-fit distribution for species richness and ranked them in descending order and then drew random numbers from the best-fit distribution for species incidences until the required number of species had been allocated to each site.

3.2.2 Calculation of leverage potential

To calculate the leverage potential as varying numbers of landowners engaged in hold-out behaviour, I first prioritised a set of sites for investment by applying a linear programme (the full mathematical formulation of the programme is given the appendix, Section 3.5). This programme determines the lowest cost set of sites that contains the maximum species richness within the budget, which was set at 10 times the mean site WTA. After having identified the prioritised sites, I randomly selected a group among them that was equal in size to the number of landowners engaged in hold-out behaviour. Since cooperation sets leverage potential, the biodiversity over the selected sites was combined along with the sites' WTAs. This "new" site (actually the amalgamation of the individual sites whose landowners are cooperating) was then added to the collection of properties and the single sites within the cooperative were removed. The optimisation was then run iteratively with increasing cost for the cooperative until it was no longer prioritised for investment. At this point, the conservation group's WTP for the cooperative had been reached and the difference between the summed WTAs and the WTP gives the leverage potential of the number of landowners in question.

3.2.3 Simulation of conservation auctions

The conservation auctions were simulated using the technique of agent-based modelling (ABM). ABM approximates economic systems by specifying the behavioural characteristics of the individuals and institutions that comprise the system under consideration (Hailu and Schilizzi 2004; Tesfatsion 2002). The aspect of landowner behaviour that was modelled was learning. Biodiversity and WTA data were generated as described above. In the initial round of the auction, it was assumed that landowners bid their WTA whilst also seeking to extract a small, randomly chosen profit. Individual bids were: bid = WTA + ϵ, where $\epsilon \sim$ Log-N(10,100) with support $\epsilon \in [5\ 10]$. Using this data, I first eliminated the sites whose landowners bid above the bid cap, which was assumed to be unknown to the landowners.

Then, from the remaining sites, I determined the lowest cost set that maximised the conservation target within the budget, which was again set at 10 times the mean site WTA. In the subsequent rounds of the auctions, landowners were able to learn from the results of the previous round and augment their bids according to the following rules: landowners who did not receive conservation investment either maintained their bid in the following round or decreased it by 10%; landowners who received conservation investment either maintained their bid in the following round or increased it by 10%. Whether a bid was maintained or augmented was dependent on the landowner's risk profile (see the appendix for full details). Each auction ran for 10 rounds.

3.3 Results

3.3.1 Illustrative example

Before coming to the main analyses, I present a simplified, illustrative example of landowner leveraging. The example allows me to outline how leverage potential is calculated, how landowner cooperatives alter negotiation outcomes and how leveraging generally affects biodiversity coverage and the cost of investments. The reduced scale of the example also highlights some important factors that set the leverage potential of landowners. Simplification of the example is achieved by assuming that the conservation group prioritises at most three sites for protection. The budget constraint is set at three times the mean WTA of the 44 properties, which for this example is $169,000.

The salient site combinations from the example are given in the top section of Table 3.1. The sets of sites are ranked first by the decreasing number of species covered in protected sites and then by the increasing summed WTAs. The set of sites initially prioritised for protection in this example is [3 13 15], which contains 50 different species at a total cost of $146,300. In the bottom section of Table 3.1, the leverage potential of these landowners when each leverages individually and when combinations act cooperatively and then apply leverage is shown.

Given the structure of the model, there are two main factors that set leverage potential. These are the substitutability of the sites whose landowners are demanding higher payments and the budget available to the conservation group. Should the conservation group have a substitute set of sites of similar worth that does not contain the sites whose landowners are holding-out for higher payments then the ability of these landowners to gain surplus would be limited. This is evident when the landowners of sites 3 and 13 act individually. When these landowners demand increased payments, the conservation group can move to negotiate with the landowners from the first best alternative set of sites, set [1 2 15]. This set of sites also contains the maximum species richness. Therefore, the leverage potential

Table 3.1: Illustrative example of landowner leveraging. The top section of the table gives the combinations of sites on which the conservation group wishes to secure land rights to protect the extant species. The bottom section gives the leverage potential, Δc, of the individual landowners from the first set of prioritised sites and the leverage potential when combinations from this set act cooperatively. All costs are in $1000s

Sites	Costs	Total Cost	Richness
[3 13 15]	[42.3 48.5 55.5]	146.3	50
[1 2 15]	[9.3 92.3 55.5]	157.1	50
[3 7 8]	[42.3 8.8 11.6]	62.7	49
[1 3 13]	[9.3 42.3 48.5]	100.1	49
[3 8 13]	[42.3 11.6 48.5]	102.4	49
[1 3 15]	[9.3 42.3 55.5]	107.1	49
[3 9 13]	[42.3 23.7 48.5]	114.5	49
[1 2 3]	[9.3 92.3 42.3]	143.9	49
[1 7 15]	[9.3 8.8 55.5]	73.6	48
[1 3 9]	[9.3 42.3 23.7]	75.3	48
[3 7 13]	[42.3 8.8 48.5]	99.6	48
[7 8 14]	[8.8 11.6 84.1]	104.5	48
Leveraging Site(s)	**Δc**	**Alternative Set**	**Total Cost**
3	10.8	[1 2 15]	157.1
13	10.8	[1 2 15]	157.1
15	22.7	[3 7 8]	62.7
[3 13]	10.8	[1 2 15]	157.1
[3 15]	61.9	[7 8 14]	104.5
[13 15]	22.7	[3 7 8]	62.7
[3 13 15]	22.7	[7 8 14]	104.5

of the landowners of sites 3 and 13 is limited to the difference in cost between the first and second set of prioritised sites, which is $10,800. However, should the landowner of site 15 demand higher payments, the conservation group does not have a substitute set of sites that contains the maximum species richness but that does not include site 15. Given that the dominant priority is to maximise species richness, the conservation group would be willing to pay their entire budget to obtain a set of sites with the highest richness. Thus, should the landowner of site 15 demand payments beyond the level of the budget, the conservation group must consider sets of sites with lower richness. This means that the leverage potential of the landowner of site 15 is at least the difference between the cost of the first prioritised set of sites and the budget constraint. Since the lowest cost set of sites with species coverage that is one below the maximum does not contain site 15, the leverage potential of this landowner is equal to this difference, which is $22,700.

Landowner cooperation influences leverage potential in several ways. From the perspective of the conservation group, landowner cooperation can only lead to leverage potential that is at least the same as the highest leverage potential of any individual landowner from the cooperative. This is apparent when the landowners of sites 13 and 15 act cooperatively. The leverage potential of this cooperative is \$22,700, the highest of either of the individual landowners. However, cooperation among landowners can lead to leverage potential that is considerably higher than when any single landowner negotiates with the conservation group. Cooperation between the landowners of sites 3 and 15 exemplifies such a situation. When these landowners form a cooperative and demand increasing payments, the conservation group must consider sets of sites that are more expensive and/or contain fewer species. In fact, this cooperative can increase costs until the conservation group is indifferent between the first-best and twelfth-best set of sites. Consequently, the leverage potential of this cooperative is large: they can gain \$61,900 as producer surplus. This figure is over 570% higher than the leverage potential of the landowner of site 3 acting individually and over 270% higher than the leverage potential of the landowner of site 15 acting individually.

Overall, when landowners seek to extract the highest payment possible from the conservation group, conservation outcomes can be diminished and site costs can be much higher than the landowner's WTA. Furthermore, cooperatives are able to demand increases in payments that are at least as great as the maximum increase of any individual member of the cooperative and, in some situations, can extract a substantially larger premium from the conservation group.

3.3.2 Leverage potential when varying numbers of landowners cooperate

I now turn to the calculation of the leverage potential of varying numbers of landowners, dropping the artificial three site constraint that was included for the illustrative case only.

The total leverage potential of the landowners displays a unimodal relationship with the size of the cooperative (Fig. 3.1a), reaching the maximum value when seven landowners act in cooperation. With this number of landowners demanding payments higher than the WTA, the mean leverage potential is 42% of the total cost of securing all prioritised sites when site costs are assumed to be at the level of the WTAs. Fig. 3.1b shows that as cooperative size increases, there is a linear decrease in the richness of species that the conservation group could protect should the negotiations result in them paying their WTP for the cooperative (where the WTP is achieved either on the basis of substitutability or the budget constraint).

The reason for the unimodal shape of leverage potential seen in Fig. 3.1a can be explained by Fig. 3.1c & d. Panel c shows the mean leverage potential from Fig. 3.1a along with the available surplus. The available surplus marks the amount of surplus that could in principle be gained by the cooperating landowners. If the cooperative could successfully negotiate payments to this level, the conservation group would be using their entire budget

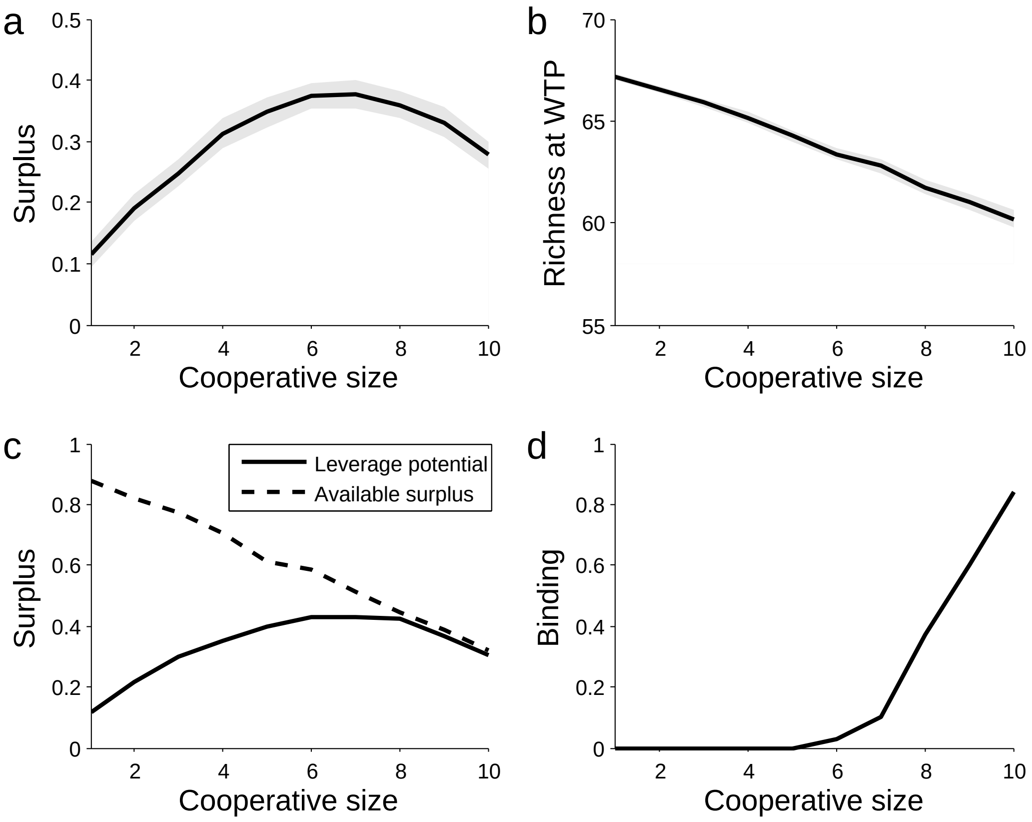


Figure 3.1: *Panel a:* Leverage potential as increasing numbers of landowners hold-out for higher payments. Leverage potential is expressed relative to the cost of procuring all prioritised sites at their WTAs. *Panel b:* The richness of species that could be protected if the conservation group paid their WTP for the leveraging landowners. *Panel c:* The mean leverage potential from Panel a along with the surplus that is available in principle, again expressed relative to the cost of securing all prioritised sites with costs at the WTAs. *Panel d:* The proportion of simulations in which the conservation group pay their entire budget for the cooperative. In all analyses 100 simulations were run. Shown are the mean figures with 95% confidence bands in Panels a and b.

to secure the cooperative and therefore no further increase in payments would be possible. The figure shows that the available surplus decreases as the cooperative size increases as a consequence of the minimum cost (summed WTAs) of the cooperative increasing as the number of members increases. Panel d shows the proportion of individual simulations in which the budget was the limiting factor for leverage potential rather than the substitutability of the cooperative. By this I mean that the conservation group is forced to pay their whole budget to procure the cooperative rather than being able to find a substitute solution that does not contain any member of the cooperative. Panel d makes clear that the negotiating power of the cooperative increases as its size increases. For fewer than six landowners leveraging, the conservation group can always find a substitute solution before they use their entire budget to secure the cooperative. For greater than six leveraging landowners, the number of individual simulations in which the conservation group exhaust their budget on the cooperative increases considerably.

Therefore, at low levels of cooperation, the negotiating power and consequently the leverage potential of the cooperative is low. For intermediate numbers of landowners cooperating, negotiating power is relatively high as is the available surplus. Cooperatives of these sizes can potentially demand a relatively large amount of this available surplus. For large cooperatives, although they have extremely high negotiating power, and can potentially gain almost all of the available surplus, this is low given the high minimum cost of the cooperative. The budget constraint thus negates some of the leverage potential of these landowners that could have been gained on the basis of substitutability. Consequently, leverage potential has a unimodal shape.

3.3.3 Conservation auctions

As noted, landowner leverage potential marks the upper bound on the price of providing conservation. To investigate where negotiations might conclude in the range of costs between this upper extreme and the lower extreme of the WTA, and to investigate how different negotiation structures influence the outcome, I now turn to the simulation of the conservation auctions. In the previous subsection, leverage potential was calculated upon the assumption that only the specified number of landowners leveraging demand payments above the WTA. In the auctions, however, all landowners attempt to augment bids over the rounds of the auction so as to maximise the surplus they receive from the conservation group

Fig. 3.2 presents the results from the different conservation auctions. The top row of the figure shows the results of the auctions that maximise species richness over sites. The bottom row shows the results of the auctions that maximise the occurrences of species. The results of the auctions that assume that land is homogeneous are contained in the appendix (Section 3.5) as they are qualitatively similar to those of the occurrences auction. Panels a & d of the figure show the surplus gained by the landowner cooperative in the species richness and species occurrences auctions, respectively. Panels b & e document the proportion of the total surplus of all landowners that goes to the cooperative in both auctions. And Panels c & f show the conservation outcomes of the different auctions. Finally, only results with no bid cap and a low bid cap are shown. The results for intermediate levels of the bid cap can be found in the appendix. It should be noted that the cap on the level of the bids relates to the overall bid and not the bids of the individuals within the cooperative.

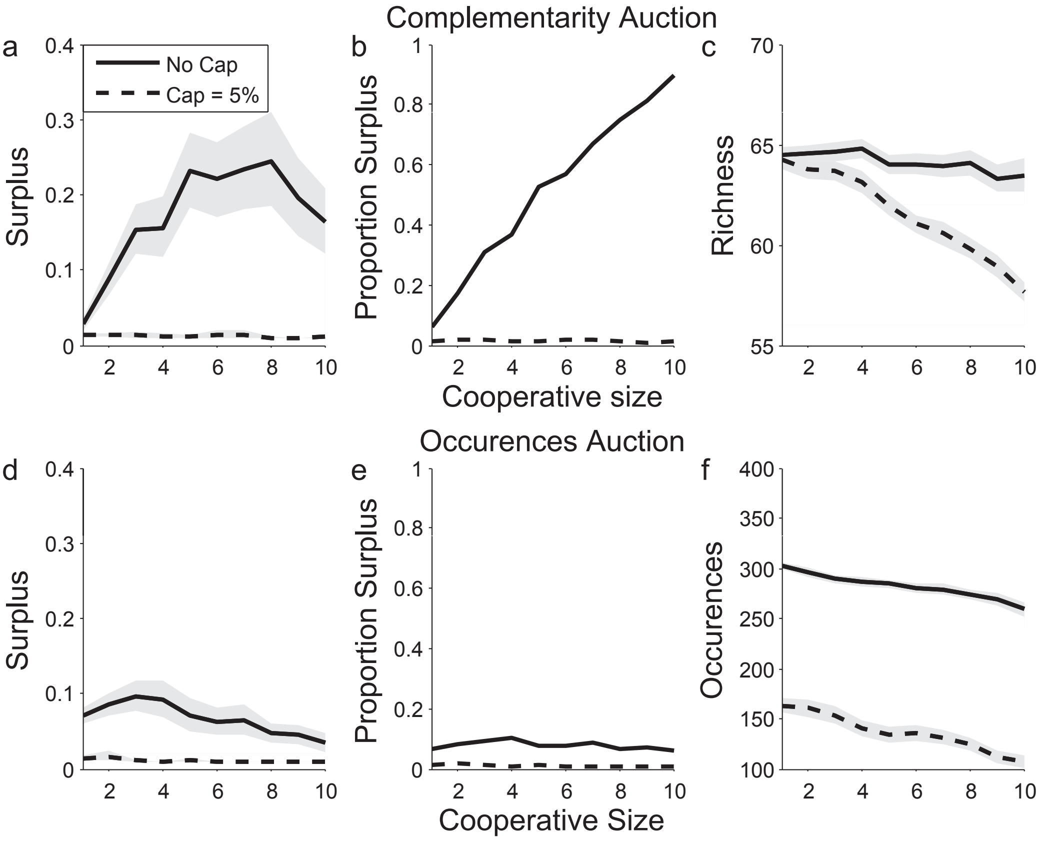


Figure 3.2: *Top Row:* The results from the auctions that maximise species richness. *Bottom Row:* The results from the auctions that maximise species occurrences. *Panels a & d:* The surplus gained by the cooperative after the final allocation round. *Panels b & e:* The proportion of the total producer surplus obtained by the cooperative after the final allocation round. In these four panels, the values are the cost of site(s) after the final allocation round relative to the cost of all winning sites at the level of the WTAs. *Panels c & f:* Conservation outcomes after the final allocation round. In all analyses 100 simulations were run. Shown are the mean figures with 95% confidence bands in Panels a, c, d and f.

Accounting for the complementary nature of sites can result in landowner cooperatives gaining a large amount of surplus in negotiations with the conservation group (Fig. 3.2a). This is dependent on cooperative size, with large amounts of surplus gained only by large cooperatives. In contrast, cooperatives obtain relatively small amounts of surplus in auctions that consider sites to be substitutes, such as the occurrences and homogeneous auctions (Fig. 3.2d). Thus, recognising site complementarity acts to incentivise the formation of cooperatives given that when eight landowners or fewer cooperate, they receive a greater amount of surplus than when acting individually (assuming an equal distribution of surplus among the cooperating landowners). On the other hand, those auctions that do not incorporate site complementarity disincentivise the formation of cooperatives: individual landowners gain the most surplus when they act alone. The fundamental difference between the auction types is how they determine the substitutability of sites. Recognising sites as partial complements reduces the scope for the conservation group to identify alternative sites when landowners demand increased payments compared to when sites are considered substitutes. Landowner cooperation further reduces substitutability, resulting in large cooperatives obtaining large amounts of surplus.

Furthermore, when sites are partial complements, cooperation leads to an uneven distribution of surplus among landowners. Small cooperatives gain only a small proportion of the total surplus whereas large cooperatives gain almost the entire producer surplus (Fig. 3.2b). Again, this is not witnessed in the auctions that consider sites to be substitutes, where the proportion of the total surplus obtained by the cooperative is insensitive to the number of members (Fig. 3.2e). The increased negotiating power of the cooperative when sites are considered partial complements not only increases the surplus that they are able to obtain but also forces the conservation group to choose investments alongside the cooperative that limit the loss of any additional surplus.

One feature of the auctions that can be used to negate the negotiating strength of landowner cooperatives in the auction that incorporates site complementarity is the bid cap. Placing a very low cap on the level of bids vastly reduces the amount of surplus that goes to the cooperative (Fig. 3.2a) and effectively equalises the level of surplus over cooperative size (Fig. 3.2a & c). Despite these seeming advantages, a low bid cap would result in limited conservation outcomes, dependent upon the size of the cooperative (Fig. 3.2c & f). A very low bid cap results in the conservation group being unable to select the cooperative, the biodiversity value and minimum procurement cost of which increases as its size increases. Thus, even though a low bid cap reduces the negotiating power of the cooperative, it forces the conservation group to reject sites of high biodiversity value.

Finally, the auction results demonstrate that conservation groups must pay costs in excess of the minimum level set by the site WTAs. This fact remains true irrespective of the structure of the auctions.

3.4 Discussion

As in Chapter 2, the results presented in this chapter show that it may be highly unrealistic to assume, as previous conservation planning studies have, that the cost of providing conservation is set at the landowner's WTA. In a realistic scenario for conservation investments, the gap between this lower bound on conservation costs and the upper bound of the conservation group's WTP indicates that conservation investments could be up to 42%

more expensive than anticipated. Consequently, conservation planning exercises that incorporate costs through estimates of landowners' WTA may significantly overestimate likely conservation outcomes.

The calculated estimates for the landowners' leverage potential represent a negotiated settlement that concludes at the conservation group's WTP. The likelihood is that most settlements will arrive at price levels somewhere between this upper bound on cost and the lower bound of the landowner's WTA. Key factors that influence where the negotiated payment level would fall in this range include the negotiation process that is followed and the information available to each party. Studies in mechanism design focus on determining efficient means to distribute conservation contracts in the presence of informational asymmetries. Research in this area has recently led to the advocacy of discriminatory-price auctions for the distribution of conservation contracts. Some studies have indicated that such auctions have the potential to result in considerable cost efficiencies (Stoneham et al. 2003; White and Burton 2005). However, in one of the first adequately controlled comparisons, Schilizzi and Latacz-Lohmann (2007) find that the advantages of such auctions over fixed price schemes are small in some cases and non-existent in others. Moreover, Jack et al. (2009) argue that designing tendering programmes in conservation instruments that seek to resolve property-scale variation in landowners' WTA, such as discriminatory-price auctions, may not always be helpful because they open the possibility for leveraging and hold-outs. The results of the auction simulations confirm this to be the case. In all auctions, landowners were able to increase costs above the minimum required payment level. More importantly, in order for the auctions to conserve biodiversity effectively, they had to be structured such that they ceded considerable amounts of surplus to landowners.

This latter result highlights the complexity of designing contract allocation mechanisms that are both economically efficient and biologically effective. To minimise landowner surplus, the conservation group had to use a low bid cap and thus accept limited conservation outcomes. To maximise conservation outcomes, the conservation group had to accept that landowners would gain substantial amounts of surplus. Moreover, situations in which biodiversity was protected effectively led to an uneven distribution of surplus among landowners, with large cooperatives obtaining the vast majority of the total surplus gained by all landowners. Thus, conservation programmes that seek an equitable distribution of surplus among landowners may be at odds with the biodiversity goals of those programmes.

Comparing the economic outcomes of the auctions with different conservation objectives produced one of the most striking results from the analyses. Recognising site complementarity produces an incentive for cooperation whereas those auction types that do not recognise complementarity create a disincentive for cooperation (Fig. 3.2a & d). In Chapter 2, I found that site complementarity increases the leverage potential of landowners when compared to objectives in which sites are substitutes. Taken alongside the fact that site complementarity allows cooperatives to gain substantial amounts of surplus, it appears that recognising complementarity can result in negative economic consequences for conservation groups. Complementarity can increase landowners' negotiating power and creates the conditions whereby landowner cooperation can further inflate conservation costs. The full biological and economic implications of differing conservation objectives must be explored in order that limited conservation funds can be used most effectively.

The contemporary paradigm for conservation planning is the prioritisation of multiple sites that provide the optimal amount of some conservation target. To inform analyses, studies have relied on estimates of landowners' WTA. However, as demonstrated, the potential exists for landowners to hold-out for payments far in excess of this minimum cost level. Furthermore, seeking to secure agreements with multiple landowners can result in the formation of agricultural cooperatives that can push costs further above the WTA. Importantly, conservation programmes focused on species richness, and which therefore value biodiversity most directly, are most vulnerable to hold-out behaviour.

3.5 Appendix

3.5.1 Homogeneous auctions

The bottom row of Fig. 3.3 gives the results from the auctions in which the conservation value of land is assumed to be homogeneous. As noted, these results are qualitatively similar to those from the auctions that assume that conservation values are spatially heterogeneous and the value of multiple sites is simply the sum of the values on the individual sites (i.e., the occurrences auctions, middle row Fig. 3.3). The reason why these results are similar is that both of these auctions view sites as substitutes whereas when the objective is to maximise species richness, sites are partial complements rather than substitutes.

3.5.2 Various bid cap levels

The top two rows of Fig. 3.3 give the results from the auctions in which the objective is to maximise species richness (top row) and in which the objective is to maximise species occurrences (bottom row) for various values of the bid cap.

As can be seen, there is close correspondence between the auctions with no bid cap and those where the cap is at 20% of the budget. This is a function of the distribution of site WTA values and the conservation group's budget. The best-fit distribution for the site per annum, per hectare WTA values has a support [5.5 124]. These values are then scaled by the farm area, the mean of which is approximately 100 hectares, and by 10, as I

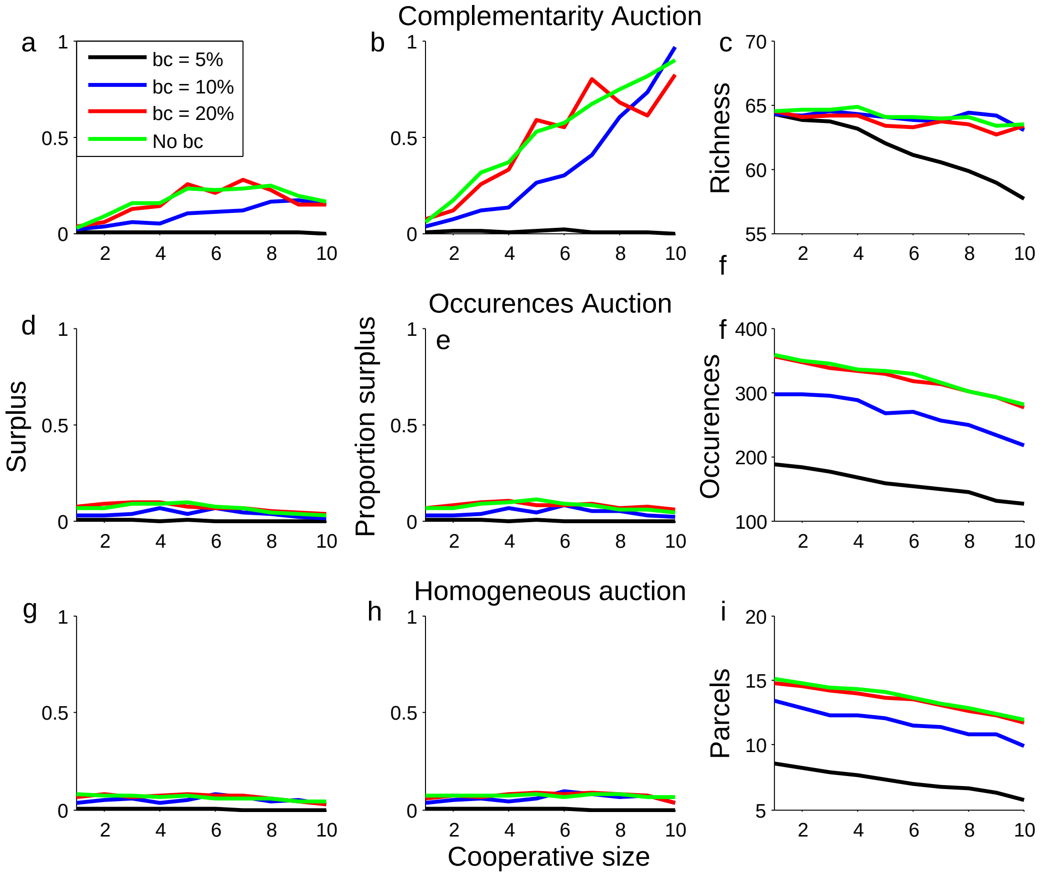


Figure 3.3: *Top Row:* The results from the auctions that maximise species richness. *Middle row:* The results from the auctions that maximise species occurrences. *Bottom Row:* The results from the auctions that maximise the number of parcels selected. *Panels a, d & g:* The surplus gained by the cooperative after the final allocation round. *Panels b, e & h:* The proportion of the total producer surplus obtained by the cooperative after the final allocation round. In these four panels, the values are the cost of site(s) after the final allocation round relative to the cost of all winning sites at the level of the WTAs. *Panels c, f & i:* Conservation outcomes after the final allocation round. In all analyses 100 simulations were run. Shown are the mean figures.

assume that contracts last 10 years. Furthermore, the budget available to the conservation group is ten times the per site, per decade mean site WTA, which comes to approximately \$550,000. Therefore, the maximum percentage of the budget that a landowner could bid is approximately $(124 \times 100 \times 10)/550000 \approx 22.5\%$. Thus, any bid cap above 22.5% is effectively equivalent to no bid cap and, as would be expected, the results for the 20% and no bid cap auctions are very similar.

3.5.3 Optimisation procedure

To determine the set of sites that results in the highest species richness at the lowest cost, I defined a two part linear programme.

Step 1: The first part is a classic maximum coverage problem (MCP) (Church and ReVelle 1974; Polasky et al. 2001), which determines the set of sites that results in the maximum species coverage subject to a budget constraint.

Step 2: Using the optimal richness found in Step 1, the second part of the programme finds the minimum cost set of sites that results in this level of coverage.

This two-step optimisation can be represented mathematically as

Step 1: The MCP is a maximisation of the form

$$\max \sum_{i \in I} y_i \tag{3.1}$$

subject to

$$\sum_{j \in N_i} x_j \geq y_i, \text{ for all } i \in I \tag{3.2}$$

$$\sum_{j \in J} c_j x_j \leq B \tag{3.3}$$

where I is the set of species that can be covered, J is the set of candidate reserves, and N_i is the subset of J that contains species i. The optimisation acts on the binary variables x_j and y_i. Should $x_j = 1$ then site j is selected for conservation and is 0 otherwise. If $y_i = 1$ then species i is present on at least one of the selected sites and is 0 otherwise. c_j is the cost associated with selecting site j and B represents the conservation group's budget. Objective (3.1) states that species richness is maximised. Constraint (3.2) ensures that a species is considered protected only if it occurs on at least one of the selected sites and constraint (3.3) ensures that the sum of the costs of the selected sites does not exceed the budget.

Step 2: The minimum cost solution with the maximum species richness is then given by solving:

$$\min \sum_{j \in J} c_j x_j \tag{3.4}$$

subject to

$$\sum_{j \in N_i} x_j \geq y_i, \text{ for all } i \in I \tag{3.5}$$

$$\sum_{i \in I} y_i = Y \tag{3.6}$$

where all terms are as before except Y, which is the optimal value of the objective function (3.1). Objective (3.4) states that the cost of the selected sites is minimised while constraints (3.5) and (3.6) ensure that the selected sites have the maximum number of species found in step 1. This two part programme ensures that the maximum species richness subject to the budget is returned at the lowest cost.

3.5.4 Behavioural algorithms

The aspect of landowner behaviour captured in the auctions was learning. Landowner learning was determined by ex-post rationality and risk profile. Landowners were assumed to be rational, revenue maximising agents. Therefore, a winning bid was not reduced and a losing bid was not increased. Whether or not a bid is augmented was in part dependent upon the landowner's risk profile. Empirical surveys have shown that farmers are generally risk averse (Latacz-Lohmann and Van der Hamsvoort 1997). However, the evidence is not conclusive, with some empirical research pointing to general risk neutrality (Gasson and Potter 1988). I therefore assumed that 45% of landowners were risk averse, 45% risk neutral and 10% risk seeking. The risk profile of cooperatives was taken as the mode of the risk profiles of the individual landowners. I further assumed that landowners never bid below their WTA. Augmentation to bids are made by the following probabilistic rules

- If a landowner/cooperative submits a winning bid, the bid in the following round will either be maintained or increased by 10%
- If a landowner/cooperative submits a losing bid, the bid in the following round will either be maintained or decreased by 10%

If a landowner/cooperative was risk averse, there was an 80% chance the bid would have been maintained if the previous bid was successful and an 80% chance it would have been reduced if the previous bid was unsuccessful. For a risk neutral landowner/cooperative these values fall to 50% and for a risk seeking landowner/cooperative the percentages fall to 20%.

Chapter 4

Continuous investments and the ability of landowners to gain surplus from conservation contracts

4.1 Introduction

The results in both Chapters 2 and 3 indicate that voluntary conservation investments have the potential to be considerably more expensive than suggested by estimates of landowners' willingness-to-accept (WTA). As a result, those studies which assume conservation costs at these minimum levels may be overestimating likely conservation outcomes.

In the first two chapters investigating these issues I assumed that the objective of the conservation group was to maximise a conservation target by enrolling a subset of available sites in a conservation scheme. Optimisations of this nature have a binary structure: the conservation group and landowner have the choice of enrolling the site or not. However, it is often the case that conservation investments can be varied continuously. For example, decisions over how much investment to devote to a region, how much time to spend on a conservation activity or how much land to enrol in an agri-environment or easement programme span a continuum of values from zero to the available maximum. An important difference between these two forms of investment relates to the substitutability of sites. When the decision variable is enrol/do not enrol the site, only differences between sites are important. When the decision is represented by a continuous variable, consideration must be given to differing levels of conservation on individual sites as well as differences between sites. The altered nature of substitutability between the binary and continuous type investments could affect the ability of landowners to gain surplus in negotiations.

In this chapter, I investigate the maximum amount of producer surplus that landowners can obtain in continuous conservation investments. The conservation group must decide how much conservation effort to devote to each site. Conservation effort is a generic measure that encompasses all examples of continuous investments, such as those given above. Where applicable, I will specify what conservation effort represents. Landowners must determine the payment demand to make to maximise their profits. Such a structure involves competitive interactions between landowners. Specifically, making payment demands that are excessively high will lead to limited profits as the conservation group can choose to invest with other willing landowners. Landowners must therefore set prices to maximise profits in light of these competitive forces.

Many game theoretic models have been developed to analyse how those seeking to maximise profits should set prices in competitive contexts. The first such analysis was undertaken by Bertrand (1883) in which a model was developed for a duopoly with homogeneous goods and equal production costs. Many extensions of the basic Bertrand model have since been developed to more accurately reflect market conditions (e.g., Allen and Hellwig 1986; Athey and Bagwell 2001; Ledvina and Sircar 2011).

In this chapter, I construct Bertrand models in the context of conservation planning. To contrast with the best-case scenario presented in most writings on conservation costs in which it is assumed that conservation groups have complete information on private costs (e.g., Ando et al. 1998; Polasky et al. 2001; Carwardine et al. 2008; Stoms et al. 2011), I present a worst-case scenario (the worst-case in the absence of landowner cooperation). This is achieved by assuming that every landowner has complete information on the cost of conservation of all other landowners. The presented models have more realistic assumptions than the classic Bertrand model. Rather than being homogeneous, conservation benefits are assumed to be differentiated substitutes. I also allow for asymmetric costs with variation in landowner opportunity costs. Finally, I model both duopolies and triopolies to analyse how competition in the context of continuous conservation investments influences the maximum producer surplus that landowners can obtain in negotiations with conservation groups.

4.2 Mathematical models and solutions methodologies

4.2.1 Problem formulation

In this problem formulation it is assumed the conservation group identifies a set of sites $I = \{i \mid i = 1, \ldots, n\}$ of conservation interest. Function V_i describes the conservation benefit of investing in site i. Throughout it is assumed that V_i is concave, twice differentiable and monotonically increasing. To simplify the analyses, I also assume that conservation benefits accrue additively over sites. The level of conservation effort on site i is represented by x_i and the vector $\boldsymbol{x} = (x_1, \dots, x_n)$ represents the effort level on all sites. The opportunity costs to the landowner of accepting a unit increase in conservation effort is c_i and the total conservation effort over all sites is constrained by the conservation group's budget, B. Finally, let $I_a \in I$ represent the set of landowners who accept at their opportunity costs and $I_d \in I$ represent the set of landowners who demand payments in excess of their opportunity costs. These assumptions lead to the following optimisation problem for the conservation group:

$$\max_{\boldsymbol{x} \in \mathbb{R}^n_+} \sum_{i=1}^{n} V_i(x_i)$$

subject to

$$x_i \geq 0, \text{ for all } i \in I$$

$$\sum_{i \in I_a} c_i x_i + \sum_{j \in I_d} (c_j + \Delta c_j) x_j \leq B$$

where Δc_j is the increase above opportunity costs that the landowner of site $j \in I_d$ demands. The Lagrangian, $\mathscr{L}$, for this problem is

$$\mathscr{L}(\boldsymbol{x}) = \sum_{i=1}^{n} V_i(x_i) - \lambda_1 \left(\sum_{i \in I_a} c_i x_i + \sum_{j \in I_d} (c_j + \Delta c_j) x_j - B \right) + \sum_{i=1}^{n} \lambda_{i+1} x_i$$

I illustrate for the case of an interior optimal solution (boundary solutions are discussed in the appendix (Section 4.6)). Necessary and sufficient conditions on an optimal solution are then

$$\frac{\partial \mathscr{L}}{\partial x_i} = 0, \text{ for all } i \in I \tag{4.1}$$

$$\sum_{i \in I_a} c_i x_i + \sum_{j \in I_d} (c_j + \Delta c_j) x_j \leq B \tag{4.2}$$

Eqns. (4.1) & (4.2) determine the optimal level of conservation effort on each of the sites given the payments demanded by the landowners. That is, they determine the demand functions for each site. Let these demand functions be represented by

$$x_i = f_i(\boldsymbol{\Delta c}_d), \text{ for all } i \in I$$

where the vector $\boldsymbol{\Delta c}_d$ represents the increase above the opportunity costs demanded by the landowners in the set I_d.

Landowners must determine the payment demand to make that will return them the highest profit. The profit functions of the landowners are

$$\begin{aligned}\pi_i &= \Delta c_i x_i \\ &= \Delta c_i f_i(\boldsymbol{\Delta c}_d) \\ &= \Pi_i(\boldsymbol{\Delta c}_d), \text{ for all } i \in I\end{aligned} \tag{4.3}$$

Using these profit functions, functions that specify the landowner's best reaction given the payment demands of the other landowners can be determined. Such reaction functions are derived by solving the profit-maximising conditions,

$$\frac{\partial \Pi_i(\boldsymbol{\Delta c}_d)}{\partial \Delta c_i} = 0, \text{ for all } i \in I \tag{4.4}$$

The intersection of these reaction functions gives the Nash-Bertrand (NB) equilibrium, the solution in which no landowner can increase his surplus by unilaterally changing his price demand,

$$\Delta c_i^{NB} = \underset{\Delta c_i}{\arg\max}\, \Pi_i(\Delta c_i, \boldsymbol{\Delta c}_{d-i}^{NB}), \text{ for all } i \in I \tag{4.5}$$

where $\boldsymbol{\Delta c}_{d-i}^{NB}$ represents the NB equilibrium solutions of all landowners except landowner i.

Implicitly assumed in the above formulation is a "worst-case" assumption described in Section 4.1: landowners have complete information on private costs of conservation or, in other words, each landowner knows the profit functions of the other landowners.

4.2.2 Solution methods

In the analyses, I employ an iterative method for determining Nash equilibria based on the Nikaidô-Isoda function (Nikaidô and Isoda 1955). This algorithm, known as the relaxation algorithm (Krawczyk and Uryasev 2000; Contreras et al. 2004), starts with an initial guess for the NB equilibrium and then in each iteration a landowner is selected and the price demand that returns his maximum surplus is calculated (Eqn. (4.4)) with the price demands of the other landowners fixed at the level in the previous iteration. This process is continued until the current solution converges to the NB equilibrium. The algorithm is exemplified in 4.3 and full details are given in the appendix.

4.2.3 Relationship between conservation effort and biodiversity improvements

To illustrate model predictions, I assume a particular form for the function V describing the relationship between the conservation effort devoted to a site and the resulting biodiversity

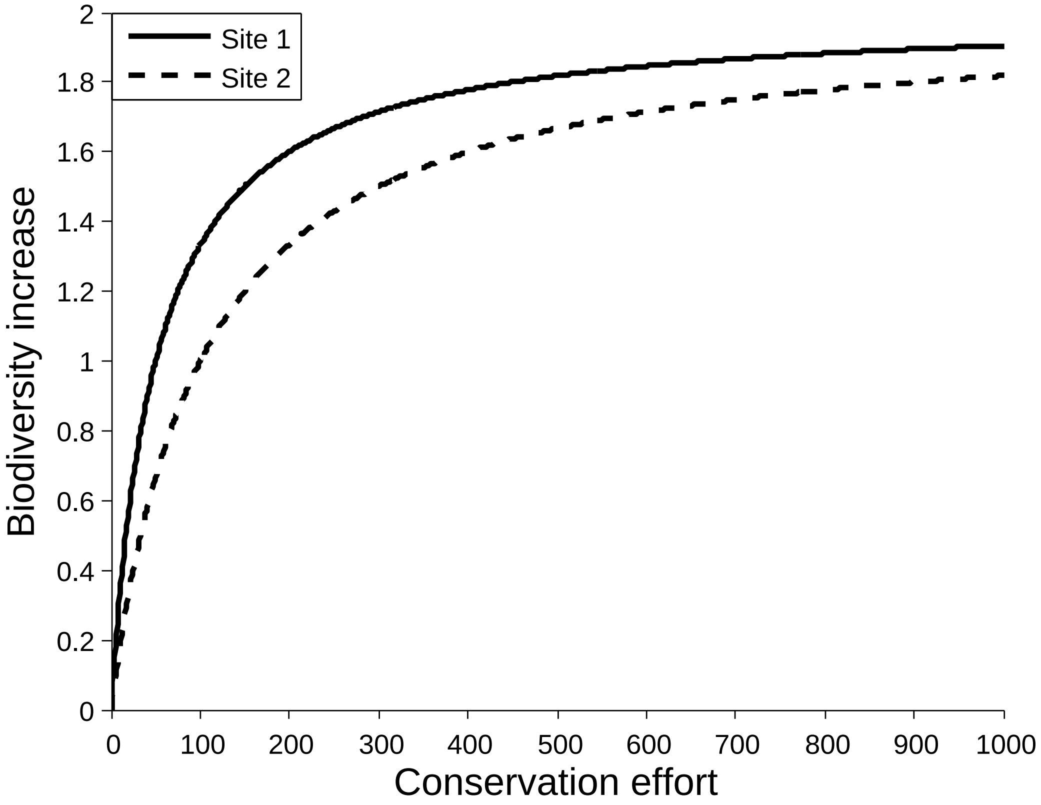


Figure 4.1: Two examples of Holling Type-II functions, the general functions used to model the relationship between conservation effort and biodiversity improvements. Parameters: $[r_1, h_1, r_2, h_2] = [0.04, 0.5, 0.02, 0.5]$.

improvements. I use a Holling Type-II function that is characterised by showing diminishing biodiversity improvements for increasing conservation effort on a particular property. The general form of the Holling Type-II function is,

$$V(x) = \frac{rx}{1+rhx}, \quad r,h > 0$$

Two examples of Holling functions, which will be utilised in Section 4.3, are shown in Fig. 4.1.

4.3 Illustrative examples

I now present the simple case of a hypothetical duopoly to exemplify the mathematical formulation and show how solutions are found. I also consider aspects of the model that affect the distribution of surplus among the landowners.

4.3.1 Calculation of NB equilibria

I begin with the simplest scenario in which one landowner demands a payment in excess of his opportunity costs and the other does not. By Eqn. (4.1), a condition on the optimal level of conservation effort, (x_1, x_2), that the conservation group should devote to the two sites is

$$\frac{V_1'(x_1)}{c_1+\Delta c_1} = \frac{V_2'(x_1)}{c_2} \tag{4.6}$$

Inserting general Holling-Type II functions into Eqn. (4.6), the demand function for site 1 can be derived as a function of Δc_1,

$$\begin{aligned} \frac{r_1 c_2}{(1+r_1 h_1 x_1)^2} &= \frac{r_2(c_1+\Delta c_1)}{(1+r_2 h_2 x_2)^2} \\ \Rightarrow x_1 &= \frac{\beta_1(1+r_2 h_2 x_2)-1}{r_1 h_1} \end{aligned} \tag{4.7}$$

where $$\beta_1 = \left(\frac{r_1 c_2}{r_2(c_1+\Delta c_1)}\right)^{1/2}$$

By the budget constraint, $x_2 = (B-(c_1+\Delta c_1)x_1)/c_2$. Substituting this value into Eqn. (4.7) and rearranging gives the demand function for site 1 as

$$x_1 = \frac{\beta_1(c_2+r_2 h_2 B)-c_2}{c_2 r_1 h_1 + \beta_1 r_2 h_2(c_1+\Delta c_1)} \tag{4.8}$$

Now let us assume that the relationship between conservation effort and biodiversity improvement on the sites are given as in Fig. 4.1, namely

$$V_1(x_1) = \frac{2x_1}{50+x_1} \text{ and } V_2(x_2) = \frac{2x_2}{100+x_2}$$

I further assume for the moment that the opportunity costs per unit of conservation effort on both sites are US\$300 and that the conservation group allocates a budget of \$150,000 for investment in the two sites. Both the optimal level of conservation effort devoted to site 1, x_1, and landowner 1's profit, $\Delta c_1 x_1$, can now be plotted as a function of the payment demanded in excess of the opportunity costs, Δc_1.

Unsurprisingly, as the landowner demands increasing costs for the acceptance of conservation measures, the optimal level of conservation effort on his site decreases (Fig. 4.2a). The total profit available to the landowner is unimodal (Fig. 4.2b). When the landowner accepts at his opportunity costs ($\Delta c_1 = 0$), he obtains no profit from the conservation investment. As he increases his payment demand, the total profit rises steeply. However,

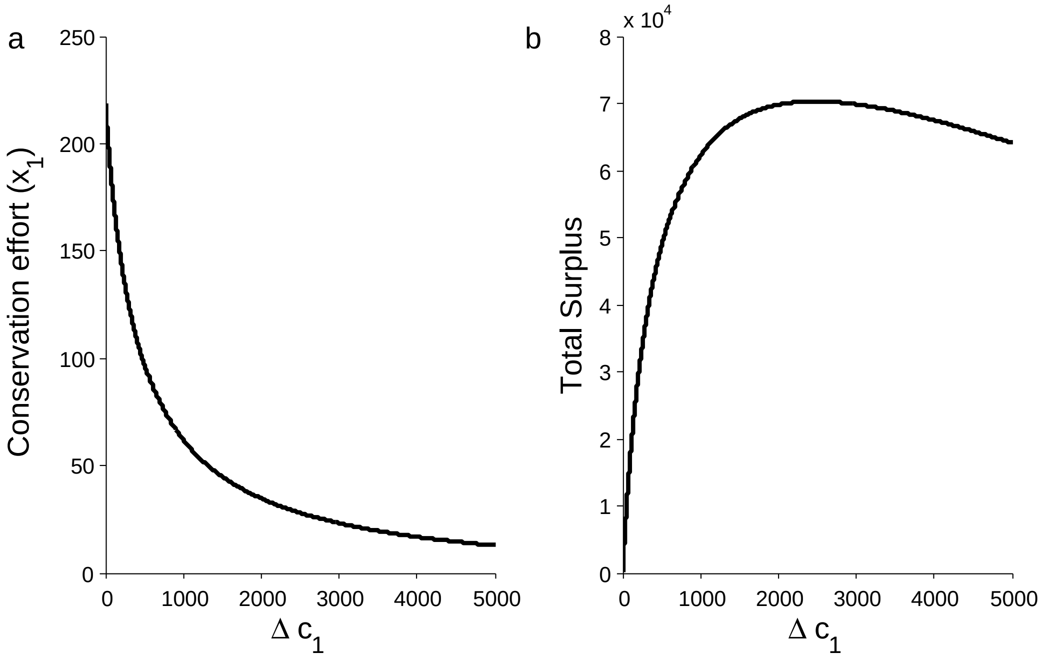


Figure 4.2: *Panel a:* The optimal conservation effort, x_1, that the conservation group should devote to site 1 as the landowner demands payments larger than his opportunity costs, Δc_1. *Panel b:* The total surplus of the landowner as he demands payments larger than his opportunity costs.

for large payment demands, the available surplus diminishes due to the very small level of effort that the conservation group is willing to devote to the site. For these parameters and assumptions, the optimal price demand that the landowner should make per unit of conservation effort is Δc_1 = \$2,479, more than 8 times higher than his opportunity costs. With this payment demand, the total surplus obtained by the landowner is \$70,306 or 47% of the conservation group's budget.

I now extend the example to the situation where both landowners compete for surplus. General profit functions for landowners 1 and 2 in this scenario are, respectively, (the derivation of these functions is contained in the appendix)

$$\pi_1 = \frac{\Delta c_1(\beta_2(r_2 h_2 B + c_2 + \Delta c_2) - (c_2 + \Delta c_2))}{r_1 h_1 (c_2 + \Delta c_2) + \beta_2 r_2 h_2 (c_1 + \Delta c_1)} \tag{4.9}$$

$$\pi_2 = \frac{\Delta c_2(\Delta c_1 B - (c_1 + \Delta c_1)\pi_1)}{\Delta c_1 (c_2 + \Delta c_2)} \tag{4.10}$$

where $$\beta_2 = \left(\frac{r_1(c_2 + \Delta c_2)}{r_2(c_1 + \Delta c_1)}\right)^{1/2}$$

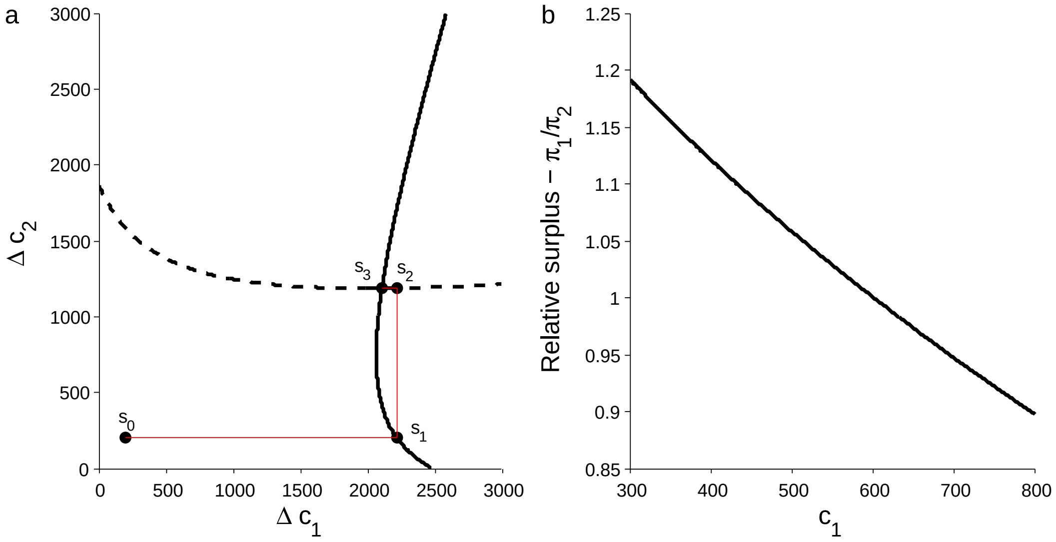


Figure 4.3: *Panel a:* Reactions curves for landowner 1 (solid line) and landowner 2 (dashed line). Also shown are the iterations of the relaxation algorithm that is used to find the Nash-Bertrand equilibrium solution. *Panel b:* Relative surplus of landowner 1 compared to landowner 2 as the opportunity costs of site 1 (c_1) is increased.

Reaction curves for the two landowners in this example are shown in Fig. 4.3a. When landowner 2 makes no payment demand in excess of his opportunity costs, the situation is the same as that analysed previously in this section and, as would be expected/hoped, the optimal price demand is as found (i.e., $\Delta c_1 = 2,479$ when $\Delta c_2 = 0$). The intersection of the two reaction curves marks the point at which neither of the landowners can increase his surplus by unilaterally changing his price demand or, in other words, the intersection is the NB equilibrium.

The equilibrium solution can be found by inspection of the figure in a two parcel system. Nonetheless, to illustrate the relaxation algorithm that will be used for the case study in Section 4.4 where it will not be possible to find the solution by inspection, I use the algorithm to find the solution here and the iterations are shown on Fig. 4.3a. Let $\boldsymbol{\Delta c}^s = (\Delta c_1^s, \Delta c_2^s)$ denote the solution of the relaxation algorithm at the s-th iteration. Starting at $s = 0$ with the guess $\boldsymbol{\Delta c}^0 = (200, 200)$, the algorithm assumes that Δc_2^0 remains fixed at 200 and determines landowner 1's best response, which is $\Delta c_1^1 = 2,218$. Therefore, $\boldsymbol{\Delta c}^1 = (2218, 200)$. The same procedure is now repeated for landowner 2 and the algorithm gives $\boldsymbol{\Delta c}^2 = (2218, 1188)$. This process is then continued until the solution converges on the equilibrium, which in this case occurs after 3 iterations, returning $\boldsymbol{\Delta c} = (2108, 1188)$. Inserting these values in Eqns. (4.9) & (4.10) gives the equilibrium surplus for landowners 1 and 2 of \$87,703 and \$55,751, respectively. Interestingly, introducing competition for surplus by allowing both

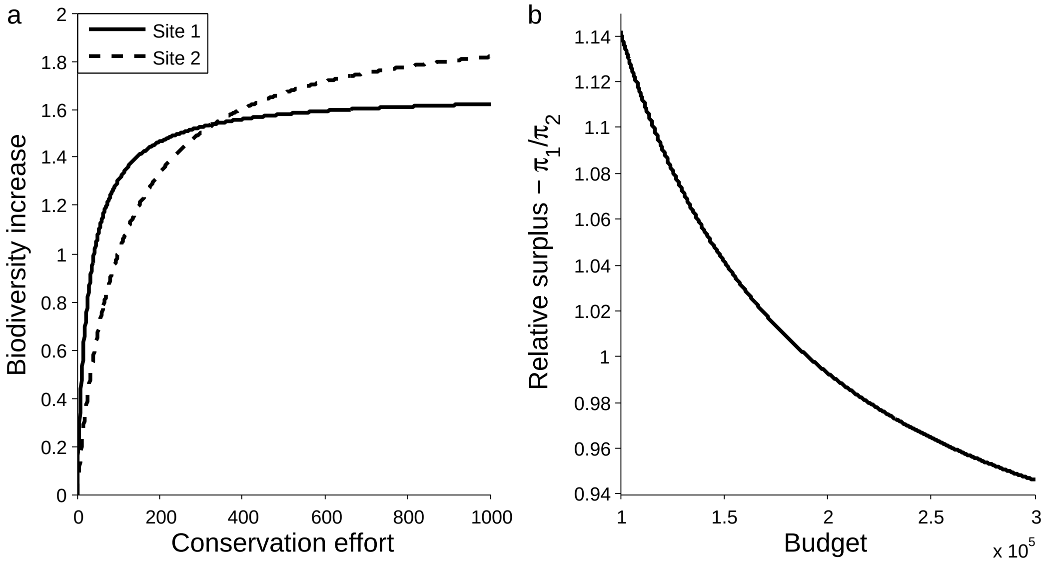


Figure 4.4: *Panel a:* The relationship between conservation effort and biodiversity increase. Parameters: [r_1, h_1, r_2, h_2] = [0.06, 0.6, 0.02, 0.5]. *Panel b:* Relative surplus of landowner 1 compared to landowner 2 as the conservation group's budget increases. Both sites' opportunity costs are fixed at \$300 per unit of conservation effort.

of the landowners to maximise their profits increases landowner 1's profit by approximately 16%.

4.3.2 Factors that affect the distribution of surplus

The distribution of surplus among landowners depends on the relationship between the relative biodiversity value of the sites, the landowners' opportunity costs and the conservation group's budget. In the previous example, landowner 1 obtained considerably higher surplus that landowner 2. This is unsurprising given that site 1 offers the higher conservation benefit irrespective of conservation effort (Fig. 4.1) and both sites have the same opportunity costs. However, should the opportunity costs of site 1 increase, the relative attractiveness of site 2 as a conservation investment would increase and landowner 2 would consequently be able to gain relatively higher surplus (Fig. 4.3b). In fact, should the opportunity costs of site 1 increase to approximately \$600 or higher, landowner 2 is able to obtain the higher surplus of the two landowners despite his site being of lesser conservation value (Fig. 4.3b).

The relationship between the conservation group's budget and surplus levels becomes apparent when the relative conservation value of sites is affected by the level of conservation effort. An example of such a system is shown in Fig. 4.4a. Here, site 1 returns the greater increase in biodiversity at low levels of effort whereas site 2 is more beneficial to the conservation group at high levels of effort. Consequently, at low budget levels the greater biodiversity value of site 1 at low levels of conservation effort results in the landowner of site 1 obtaining a larger amount of surplus (Fig. 4.4b). As the budget increases, the conservation group can devote greater levels of effort to the sites and the relative conservation value of site 2 increases. As a result, landowner 2 can gain relatively greater surplus such that for budgets greater than $200,000 landowner 2 obtains the higher profit.

4.4 Peak District case study

I now determine the maximum surplus that landowners can gain in negotiations with a conservation group in a case study from the Peak District of the UK. I investigate how competition among landowners for surplus influences the surplus gained and how conservation outcomes are affected when landowners seek to maximise their profits.

4.4.1 Data for model parameterisations

Data needed to parameterise the models is not available in exactly the required form. In fact, the unobservable nature of landowners' true WTA means that it will always be problematic to estimate. However, a recent study by Armsworth et al. (2012) provide data with many of required characteristics. These authors used optimisation modelling to estimate landowners' WTA given sets of restrictions on farming activities.

Specifically, as part of the UK Research Council's Rural Economy and Land Use programme, biodiversity and socio-economic data were collected from 44 farms in the Peak District of the UK. The biodiversity data records the density of avian species of conservation concern over the in-bye portion of the farm holding. The socio-economic data record quantities of and prices for inputs and outputs in production, including the per hectare rental price of land (Acs et al. 2010 – questionnaire published in Armsworth et al. (2012)).

Armsworth et al. (2012) used the data to parameterise models that predict the amount of foregone farm income required to achieve a given improvement in some biodiversity indicator under the assumption that landowners are profit maximisers. The authors divided the study region into three subregions based on abiotic characteristics (e.g., elevation and wetness gradients) and characteristics of the farming systems. The optimisation models were then parameterised using average biodiversity and economic values across farms in each of the subregions, giving three representative farm models in total.

Using these models, Armsworth et al. (2012) produced trade-off curves relating biodiversity benefits of conservation activities (V) to the overall cost of securing those benefits when paying at landowners' WTA ($c_i x_i$, see Fig. 2 in Armsworth et al. (2012)). Unfortunately, the data do not break that down further to measure landowners' WTA in terms of the per

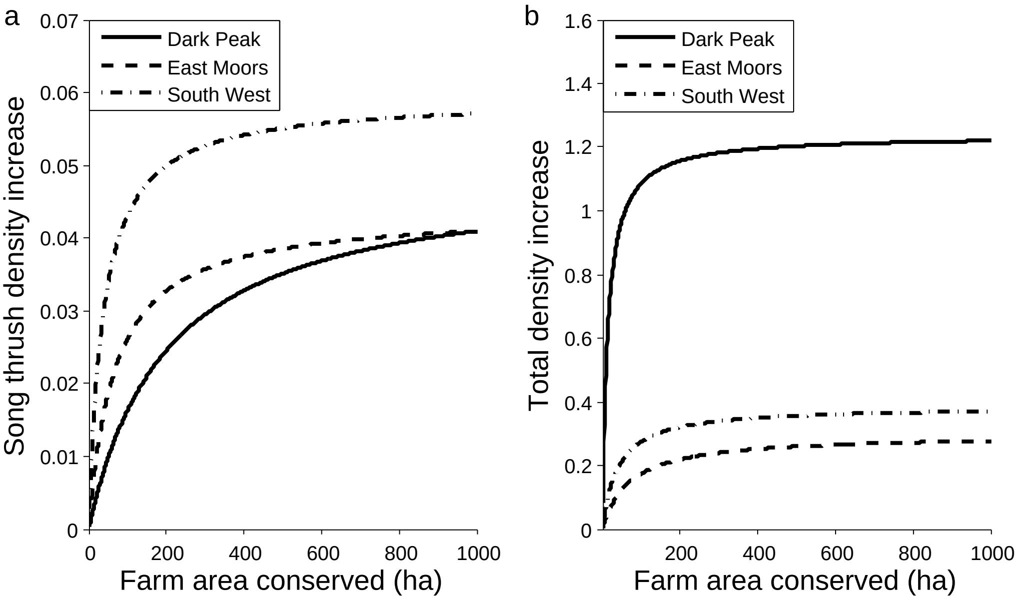


Figure 4.5: Best-fit Holling Type-II functions for the Peak District data; farm names are given in the legend. Song Thrush (Panel a) and total species (Panel b) density increase for farm area conserved. Parameters for Panel a are $[r_{DP}, h_{DP}, r_{EM}, h_{EM}, r_{SW}, h_{SW}] = [0.02, 0.2, 0.06, 0.2, 0.2, 0.2]$ and for Panel b = [0.08, 0.8, 0.01, 3.4, 0.01, 2.6]. Opportunity costs per hectare are: Dark Peak = \$183.6, East Moors = \$448.2 and South West = \$808.2.

unit cost of different conservation activities (c_i), which would be particularly helpful for parameterising the models of landowners' leverage potential in this chapter. Indeed, it is not clear how the authors would do so given that the conservation activities are a combination of agricultural practices (changing stocking rates of sheep and cows, changing fertiliser application rates and changing frequency with which rough grazing areas are cut for silage), with the particular combinations changing along the trade-off curves.

Despite these limitations, the trade-off curves show diminishing returns for increasing conservation investment and spatial variation in landowners' opportunity costs across the three representative farms, the key features of data required for the models in this chapter. Therefore, to use these data, I assume that conservation effort is measured by the area of farm used for conservation and use the heterogeneous rental prices for land to provide an independent estimate of landowner's opportunity costs.

Recognising these admittedly strong assumptions, I fit Holling Type-II functions to the relationship between farm area used for conservation and the improvement in the density of a single species, Song Thrush (*Turdus philomelos*), and the density of all recorded bird species. The fitted functions are shown in Fig. 4.5, with the estimate of opportunity costs of conservation on each farm given in the caption.

When the objective is to maximise the increase in Song Thrush density over the three farms (Fig. 4.5a), the site that offers the highest conservation benefit (South West) also has the highest opportunity costs; the site that ranks second in terms of conservation benefit (East Moors) has the second highest opportunity costs; and the site that offers the lowest conservation benefit (Dark Peak) has the lowest opportunity costs. Therefore, while it is difficult to predict the distribution of surplus among the landowners, I expect the range to be small. In contrast, when the objective is to maximise the increase in total species density (Fig. 4.5b), the landowner from the Dark Peak region should have the greatest ability to increase his payment demand above opportunity costs as his site offers the highest conservation benefit at the lowest opportunity cost.

4.4.2 Results

The results from the case study are only minimally affected by the level of the budget. Therefore, I document results when the budget is set to 500 times the mean, per hectare opportunity costs of land over the three sites (≈ \$255,000) with the results for other budget levels contained in the appendix (Section 4.6). I also refer readers to the appendix for the derivation of landowners' profit functions in a three parcel system.

Fig. 4.6 displays the results from the case study when the conservation objective is to maximise the increase in Song Thrush density over the three farms. In this example, the landowner of the South West farm, the site with highest conservation potential but highest opportunity costs, has the greatest ability to gain surplus. When this landowner only demands payments greater than opportunity costs, he can gain \$85,190 (524% of opportunity costs) as surplus from the conservation agreement. The landowner from the East Moors, the site with the second highest conservation potential but second highest opportunity costs, can gain the lowest surplus (\$61,088 or 428% of opportunity costs). As anticipated, however, there is little variability in landowner surpluses in this instance: the minimum and maximum achievable surplus among the landowners differs only by around 18%.

In contrast to the illustrative example of Section 4.3, landowners' ability to secure payments in excess of opportunity costs decreases marginally as more landowners compete for surplus (Fig. 4.6a). Nonetheless, when all three landowners demand payments higher than opportunity costs, the total surplus that could be ceded by the conservation group is \$199,199 (≈ 78% of the budget). Consequently, conservation outcomes could be severely diminished: the maximum landowner surpluses translate into conservation outcomes that are 50.7% lower than when landowners accept conservation investment at their opportunity costs.

Fig. 4.7 shows the results from the case study when the conservation objective is to max-

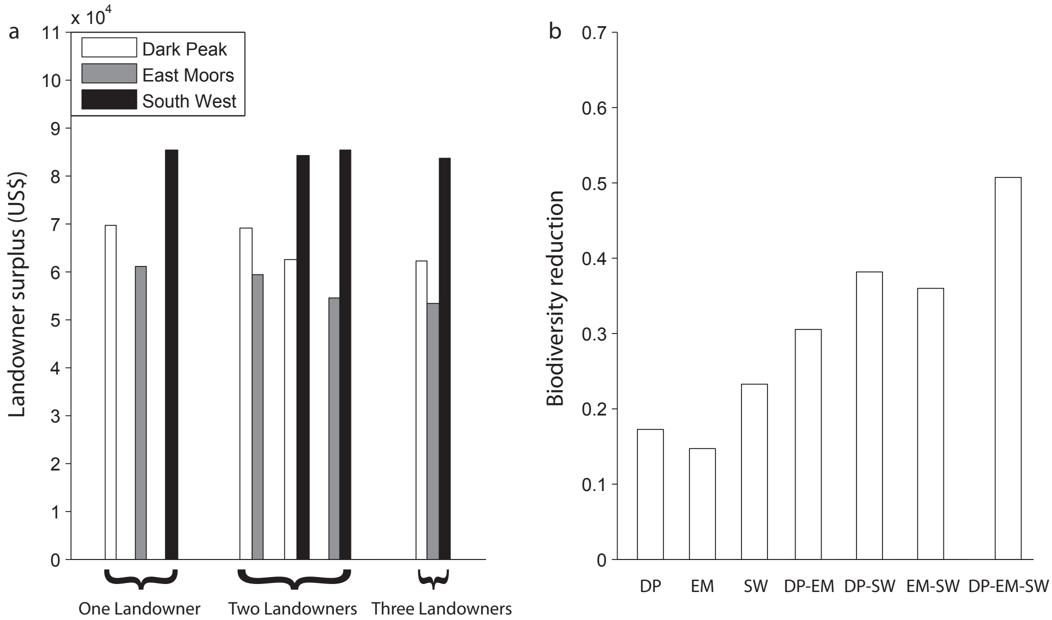


Figure 4.6: Results from the Peak District case study when the conservation objective is to maximise the increase in Song Thrush density over the three farms. *Panel a:* The profit obtained by the individual landowners when one, two and all landowners maximise their profits. *Panel b:* The reduction in conservation outcomes when landowners maximise their profits as a proportion of the outcome that would be achieved when landowners accept at their opportunity costs. Here, the *x*-axis labels give the initials of the region of the representative farms. So, for example, DP-EM indicates that the landowners from the Dark Peak and East Moors region maximise their profits.

imise the increase in total species density. Reference to Fig. 4.5b shows that the landowner from the Dark Peak region offers a significantly more attractive investment, both in terms of conservation value and opportunity costs, than the other two landowners in this case. As a result, the ability of this landowner to gain surplus is much greater than the other landowners (Fig. 4.7a). When only a single landowner demands payments greater than opportunity costs, the landowner from the Dark Peak can gain $163,038 (6,905% of opportunity costs) in profit, around twice as much as both the South West and East Moors landowners. When all three landowners seek to secure the highest payment possible, the negotiating strength of the Dark Peak landowner results in the potential for $223,715 to be lost by the conservation group as landowner surplus (88% of the budget) of which the landowner from the Dark Peak region receives 78%.

As increasing numbers of landowners demand payments in excess of opportunity costs, different dynamics are evident in this example than in either the illustrative example or previous case study example shown in Fig. 4.6. Here, the landowner of the Dark Peak farm

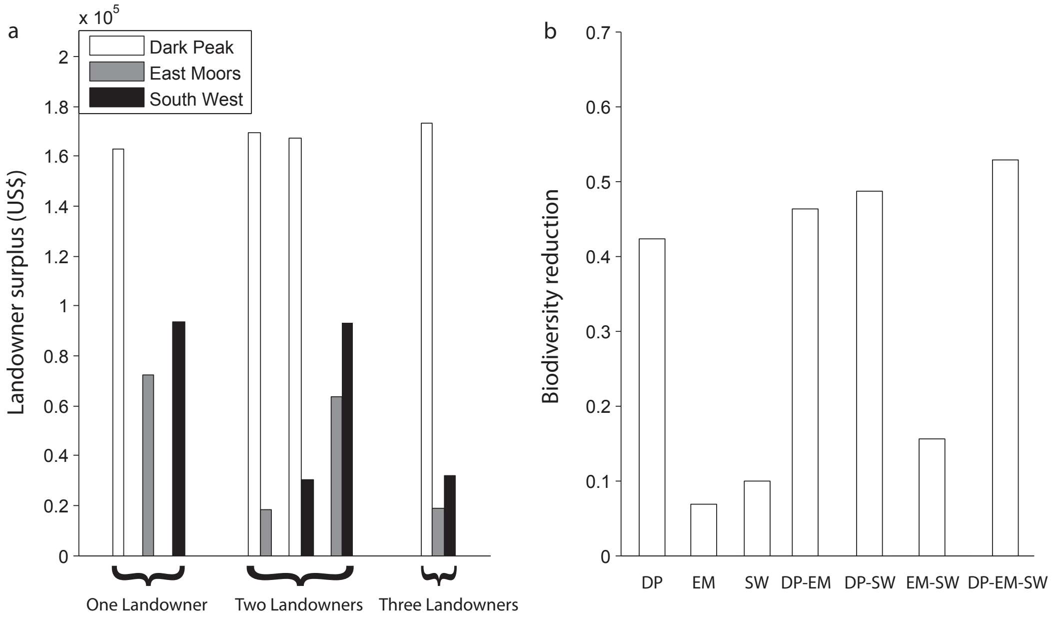


Figure 4.7: Results from the Peak District case study when the conservation objective is to maximise the increase in total species density over the three farms. *Panel a:* The profit obtained by the individual landowners when one, two and all landowners maximise their profits. *Panel b:* The reduction in conservation outcomes when landowners maximise their profits as a proportion of the outcome that would be achieved when landowners accept at their opportunity costs. Here, the *x*-axis labels give the initials of the region of the representative farms. So, for example, DP-EM indicates that the landowners from the Dark Peak and East Moors region maximise their profits.

can gain greater surplus when other landowners also demand higher payments. However, both of the other landowners can gain less surplus when others also demand higher payments, and this reduction is considerable when the Dark Peak landowner seeks to maximise payments. For example, the South West landowner could gain \$93,792 as surplus when the other landowners accept conservation investment at opportunity costs. However, should the Dark Peak landowner also demand payments greater than opportunity costs, the maximum surplus obtainable by the South West landowner falls by around two-thirds.

The negotiating strength of the Dark Peak landowner can significantly depress conservation outcomes. When this landowner is the sole individual who seeks to maximise his profits, the increase in total species density achieved through the investments is 42.3% lower than when the cost of conservation is assumed to equal the landowners' opportunity costs (Fig. 4.7b). This figure rises to 52.8% when all three landowners are profit maximisers.

4.5 Discussion

In this chapter, I investigated how much profit landowners could potentially gain through the acceptance of conservation investments where those investments were continuous. The results from the analyses confirm those found in Chapters 2 & 3 where investments were binary in nature. Hence, the informational advantage that landowners hold over conservation groups and the voluntary nature of the negotiations means that conservation costs may greatly exceed the minimum level set by landowners' opportunity costs in a wide variety of conservation scenarios. As in previous chapters, the extent to which these worst case outcomes for conservation are realised depends on the chosen contract allocation mechanism. Here, I have again explored the upper bound that is set by the conservation group's WTP to provide the contrasting benchmark to the overly optimistic focus on landowners' WTA in previous conservation studies (e.g., Ando et al. 1998; Polasky et al. 2001; Carwardine et al. 2008; Stoms et al. 2011).

In the previous chapters investigating these issues, I modelled the situation where the conservation group and landowner had the binary choice of enrolling a site or not. In those analyses, the maximum landowner surplus ranged from approximately 8-260% of opportunity costs. In the analyses of this chapter, landowners could gain much larger amounts of surplus: the maximum potential surplus was over 6,000% of opportunity costs. Several factors may account for the disparity in landowner surplus found here and in the preceding two chapters. First, for analytical clarity/tractability, in this chapter I limited the investigations to duopolies and triopolies. In the previous chapters, however, there were 44 sites in which the conservation group could potentially invest. Therefore, the conservation group had a much larger supply of sites and thus many more potential substitutes than was the case here. As a consequence, the ability of any particular landowner to gain surplus could be severely diminished in comparison to the situation analysed in this chapter. Second, the nature of site substitutability is altered when the investment is continuous as compared to binary. Rather than being forced to enrol the site or not, continuous investments allow the conservation group to augment the amount of conservation effort devoted to a particular site given the landowner's payment demand. The amount of surplus available to landowners may be larger in this case because it can be optimal for the conservation group to invest in a site whose landowner demands high payments, albeit at low levels of effort, rather than disregarding a site completely. Landowners may therefore be able to increase costs substantially more than would be the case if the possible level of conservation effort was fixed.

As in the preceding chapters, here I also found that different conservation objectives can result in landowners obtaining considerably different amounts of surplus. For example, when the conservation objective focused on Song Thrush and the landowner of the Dark

Peak region was the sole landowner who attempted to maximise profits, his maximum surplus was \$69,575 (Fig. 4.6a). In the same situation when the objective focused on total species density, the Dark Peak landowner's surplus increased by over 230% to \$163,038 (Fig. 4.7a). Conservation groups must be aware of how different conservation objectives affect the ability of landowners to gain surplus. As shown, when landowners have the potential to gain large profits from the conservation investment, conservation outcomes can be considerably lower than when assuming costs at the level of landowner opportunity costs (Figs. 4.6b & 4.7b).

Throughout the analyses I found that the distribution of surplus among the landowners depended on a host of different factors. For example, even though the landowner of the East Moors farm had the second most valuable conservation site when the conservation objective focused on Song Thrush (Fig. 4.6a), this landowner obtained the least surplus because his site had relatively high opportunity costs. Moreover, the illustrative examples of Section 4.3 highlighted that the level of the conservation group's budget could determine which landowner had the greatest potential to gain surplus as a consequence of different budget levels affecting the relative conservation value of the sites. Therefore, it is essential in conservation planning exercises to consider all biological and economic factors that affect the ability of landowners to hold-out for higher payments. Disregarding these issues could result in conservation outcomes that are substantially less than expected.

To calculate the landowners' maximum potential profit from the conservation investment, I assumed that landowners had complete information on the private costs of conservation. This means that landowners are aware of the profit functions of the other landowners and can therefore set their payment demands with no uncertainty. In reality, landowners would not have complete information: they would likely be uncertain about the identity and/or profit functions of any other landowners with whom the conservation group was negotiating, for example. This uncertainty could consequently mean that the payment demands made by the landowners were lower. The estimates for the landowners' maximum profits bound the range of potential conservation costs and highlight that the assumption that landowners will accept conservation investment at their opportunity costs may be unwarranted. Nonetheless, analyses in which landowners have incomplete information would allow for investigation of how informational asymmetries that favour the conservation group influence the ability of landowners to gain surplus.

4.5.1 Assumptions of the formulation

In the formulation of the optimisation problem (Section 4.2), I made two crucial assumptions. The first of these was that biodiversity benefits accrue additively over sites. Depending on the nature of the conservation objective, this assumption may or may not be met. I investigated the situation where the conservation group was focused on increasing the density of a single species and multiple species, both of which are additive objectives. However, if the objective of the conservation investments recognised site complementarity (Margules et al. 1988; Pressey and Nicholls 1989; Church et al. 1996; Justus and Sarkar 2002), when focusing on species richness, for example, then benefits would accrue non-additively. In Chapters 2 & 3, I found that objectives that incorporated site complementarity can increase the ability of landowners to gain surplus in comparison to objectives that focus on additive benefits. A possible explanation for this phenomenon was revealed in Chapters 2. There, I found that the amount of landowner surplus was positively correlated with site irreplaceability (Pressey et al. 1994; Ferrier et al. 2000) and irreplaceability increases when incorporating site complementarity. Whether these dynamics between additive and non-additive benefits would be evinced with continuous investments remains an open question.

Second, I assumed a linear cost structure in which the cost of the investments is directly proportional to conservation effort (Eqn. (4.2)). This is a very common assumption in conservation planning studies. Nonetheless, it is likely an oversimplification of conservation costs (Ando and Shah 2010 and see Chapter 5). The ability of landowners to gain surplus from conservation investments with more sophisticated cost structures warrants investigation.

4.5.2 Conclusions

In this chapter I have shown in conservation investments of a continuous nature that the assumption that landowners will accept a conservation agreement at their opportunity costs is suspect. Landowners have the potential to demand payments far in excess of this cost level, depending on the nature of the substitutability of investments both across sites and on individual sites. The most important implication of this is that conservation research that incorporates costs through estimates of opportunity costs is likely to overestimate conservation outcomes. For conservation science to deliver that which it says it can, accurate estimates for the cost of conservation measures must be determined.

4.6 Appendix

4.6.1 Relaxation algorithm

The Nash-Bertrand (NB) equilibrium solutions were found by employing an algorithm based on the Nikaidô-Isoda function (Nikaidô and Isoda 1955). To describe this algorithm, I first define a NB equilibrium in a slightly different manner to that given in Section 4.2. Let the set of possible price demands in excess of opportunity costs that a landowner could make be denoted by Y_i for landowner $i \in I$ and let the price demands made by all landowners be $\boldsymbol{y} = (y_1, \ldots, y_n)$. Now let $\boldsymbol{y}^* = (y_1^*, \ldots, y_n^*)$ denote the NB equilibrium. At this equilibrium,

$$\pi_i(\boldsymbol{y}^*) = \max_{(y_i|\boldsymbol{y}^*)\in Y} \pi_i(y_i|\boldsymbol{y}), \text{ for all } i \in I \tag{4.11}$$

Eqn. (4.11) states that at the equilibrium every landowner has maximised the surplus he can achieve for acceptance of the conservation measures given the equilibrium price demands of the other landowners.

The Nikaidô-Isoda function is given by

$$\psi(\boldsymbol{y}, \boldsymbol{z}) = \sum_{i=1}^{n} (\pi_i(z_i|\boldsymbol{y}) - \pi(\boldsymbol{y}))$$

Each summand of this function determines the increase in profit that a landowner will gain by changing his price demand from y_i to z_i while all other landowners maintain their price demands. The Nikaidô-Isoda function therefore calculates the sum of these increases in profit as each landowner changes his price demand. By definition, at the NB equilibrium no landowner can increase his profit unilaterally. Therefore, the element $\boldsymbol{y}^*$ is a NB normalised equilibrium solution if

$$\max_{z\in Y} \psi(\boldsymbol{y}^*, \boldsymbol{z}^*) = 0$$

A NB normalised equilibrium solution is a NB equilibrium solution if certain concavity conditions hold (Contreras et al. 2004).

The optimum response function, which maximises the Nikaidô-Isoda function, is given by

$$R(\boldsymbol{y}) = \arg\max_{z\in Y} \psi(\boldsymbol{y}, \boldsymbol{z})$$

where, $\boldsymbol{y}$ & $R(\boldsymbol{y}) \in Y$. $R(\boldsymbol{y})$ gives the set of price demands that landowners should make to unilaterally maximise their profits (Contreras et al. 2004). The optimum response function can therefore serve as the basis of the relaxation algorithm that determines the NB equilibrium solution (Krawczyk and Uryasev 2000; Contreras et al. 2004). Starting with an initial guess for the NB equilibrium, $\boldsymbol{y}^0$, the relaxation algorithm of the optimum response function, $R(\boldsymbol{y})$, is

$$\boldsymbol{y}^{s+1} = (1 - \alpha_s)\boldsymbol{y}^s + \alpha_s R(\boldsymbol{y}),\ s = 0, 1, 2, \ldots \tag{4.12}$$

where $0 < \alpha_s \leq 1$. The averaging of Eqn. (4.12) ensures convergence of the algorithm under certain conditions (Uryasev and Rubinstein 1994; Krawczyk and Uryasev 2000). In each iteration of the algorithm, a landowner is selected and the price demand that returns his maximum surplus is calculated with the price demands of the other landowners fixed at the level in the previous iteration. This process is continued until the current solution converges to the NB equilibrium.

4.6.2 Profit functions when both landowners demand payments greater than opportunity costs in a duopoly

From Eqn. (4.1), if both landowners demand payments greater than opportunity costs in a duopoly then,

$$\frac{V_1'(x_1)}{c_1+\Delta c_1} = \frac{V_2'(x_1)}{c_2+\Delta c_2} \tag{4.13}$$

Inserting general Holling-Type II functions into Eqn. (4.13), the demand function for site 1 can be derived as a function of Δc_1,

$$\frac{r_1(c_2+\Delta c_2)}{(1+r_1h_1x_1)^2} = \frac{r_2(c_1+\Delta c_1)}{(1+r_2h_2x_2)^2}$$

$$\Rightarrow x_1 = \frac{\beta_2(1+r_2h_2x_2)-1}{r_1h_1} \tag{4.14}$$

where $$\beta_2 = \left(\frac{r_1(c_2+\Delta c_2)}{r_2(c_1+\Delta c_1)}\right)^{1/2}$$

x_2 can then be eliminated from Eqn. (4.14) using the budget constraint, which implies that $x_2 = (B-(c_1+\Delta c_1)x_1)/c_2$. After this, the profit functions, $\pi_i = \Delta c_i x_i$, can be used to give Eqns. (4.9) and (4.10) as

$$\pi_1 = \frac{\Delta c_1(\beta_2(r_2h_2B+c_2+\Delta c_2)-(c_2+\Delta c_2))}{r_1h_1(c_2+\Delta c_2)+\beta_2r_2h_2(c_1+\Delta c_1)}$$

$$\pi_2 = \frac{\Delta c_2(\Delta c_1B-(c_1+\Delta c_1)\pi_1)}{\Delta c_1(c_2+\Delta c_2)}$$

4.6.3 Derivation of profit functions in a three-parcel system

I derive here landowner profit functions in a three-parcel system, the results of which were used in the Peak District case study. While more complicated, the process is essentially the same as outlined for a duopoly. Assuming an interior optimal solution (issues pertaining to boundary solutions follow), from the Lagrangian and the budget constraint the following conditions on the optimal solution can be derived

$$V_i'(x_i) - \lambda_1(c_i + \Delta c_i) = 0, \text{ for } i = 1,2,3 \tag{4.15}$$

$$\sum_{i=1}^{3}(c_i + \Delta c_i)x_i = B \tag{4.16}$$

By Eqns. (4.15)

$$\frac{V_1'(x_1)}{c_1 + \Delta c_1} = \frac{V_2'(x_2)}{c_2 + \Delta c_2} = \frac{V_3'(x_3)}{c_3 + \Delta c_3} \tag{4.17}$$

Inserting general Holling Type-II functions into Eqn. (4.17) and rearranging gives,

$$x_i = \frac{\beta_{i+1}(1 + r_{i+1}h_{i+1}x_{i+1}) - 1}{r_i h_i}, \text{ for } i = 1,2$$

where $$\beta_3 = \left(\frac{r_2(c_3 + \Delta c_3)}{r_3(c_2 + \Delta c_2)}\right)^{1/2}$$

And by budget constraint (Eqn. (4.16)),

$$x_3 = \frac{B - (c_1 + \Delta c_1)x_1 - (c_2 + \Delta c_2)x_2}{c_3 + \Delta c_3}$$

There are now three equations in three unknowns and so the system can be uniquely solved to give the demand functions in terms of model parameters ($x_i = f_i(r_i, h_i, c_i, \Delta c_i, B)$, for $i = 1,2,3$). This requires considerable algebraic manipulations, which I spare the reader. After having determined the demand functions, I use the equations $\pi_i = \Delta c_i x_i$ to find the profit functions. After such manipulations, the profit functions are found to be

$$\begin{aligned}\pi_1 = \quad & \frac{\Delta c_1(r_2h_2(c_3 + \Delta c_3) + \beta_3 r_3 h_3(c_2 + \Delta c_2))(1 - \beta_2^{-1})}{\beta_2^{-1} r_1 h_1(r_2 h_2(c_3 + \Delta c_3) + \beta_3 r_3 h_3(c_2 + \Delta c_2)) + \beta_3 r_2 r_3 h_2 h_3(c_1 + \Delta c_1)} \\ + \quad & \frac{\Delta c_1 r_2 h_2(\beta_3(c_3 + \Delta c_3) + \beta_3 r_3 h_3 B - (c_3 + \Delta c_3))}{\beta_2^{-1} r_1 h_1(r_2 h_2(c_3 + \Delta c_3) + \beta_3 r_3 h_3(c_2 + \Delta c_2)) + \beta_3 r_2 r_3 h_2 h_3(c_1 + \Delta c_1)}\end{aligned}$$

$$\pi_2 = \frac{\Delta c_2(r_1 h_1 \pi_1 - \Delta c_1(\beta_2 - 1))}{\Delta c_1 \beta_2 r_2 h_2}$$

$$\pi_3 = \frac{\Delta c_3(B\Delta c_1 \Delta c_2 - \Delta c_2 \pi_1(c_1 + \Delta c_1) - \Delta c_1 \pi_2(c_2 + \Delta c_2))}{\Delta c_1 \Delta c_2(c_3 + \Delta c_3)}$$

4.6.4 Boundary solution conditions

Optimal boundary solutions are those in which a subset of landowners receives no investment and consequently no surplus. One can derive such solutions using the Kuhn-Tucker (K-T) conditions (Kuhn and Tucker 1951). The K-T conditions for a three parcel system with Holling Type-II benefit functions are

$$\begin{aligned} \frac{r_i}{(1+r_ih_ix_i)^2} - \lambda_1(c_i+\Delta c_i) + \lambda_{i+1} &= 0, \text{ for } i = 1,2,3 \\ (c_1+\Delta c_1)x_1 + (c_2+\Delta c_2)x_2 + (c_3+\Delta c_3)x_3 &= C \leq B \\ \lambda_1 \geq 0, C \leq B, \lambda_1 C &= 0 \\ \lambda_{i+1} \geq 0, x_i \geq 0, \lambda_{i+1}x_i &= 0, \text{ for } i = 1,2,3 \end{aligned}$$

Using the first three conditions, candidate optimal solutions can be found. Such solutions are optimal solutions if they satisfy the remaining conditions and certain concavity conditions hold.

Boundary solutions can also be found using the relaxation algorithm. In such circumstances, the algorithm would return an equilibrium solution of zero surplus for at least one landowner. Interestingly, the equilibrium solution for all landowners given by the algorithm would not be a Nash equilibrium in such circumstances and the K-T conditions show why. Consider optimal boundary solutions of the form $(x_1, x_2, 0)$; the K-T conditions for such a solution are shown below.

$$\begin{aligned} \frac{r_1}{(1+r_1h_1x_1)^2} - \lambda_1(c_1+\Delta c_1) &= 0 \qquad (4.18) \\ \frac{r_2}{(1+r_2h_2x_2)^2} - \lambda_1(c_2+\Delta c_2) &= 0 \qquad (4.19) \\ r_3 - \lambda_1(c_3+\Delta c_3) + \lambda_4 &= 0 \\ (c_1+\Delta c_1)x_1 + (c_2+\Delta c_2)x_2 &= C \leq B \\ \lambda_1 \geq 0, C \leq B, \lambda_1 C &= 0 \\ \lambda_4 \geq 0, x_3 \geq 0, \lambda_4 x_3 &= 0 \end{aligned}$$

Assuming such a solution satisfies the K-T conditions, the profit functions would be derived from only Eqns. (4.18) & (4.19) as these are the only equations which have active variables (i.e., x_1 and x_2 are part of the system but x_3 is not). Consequently, the Nash equilibrium for a three-player game in which one landowner receives no investment is calculated with the same theory used for a two-player game. This result holds generally. Although such

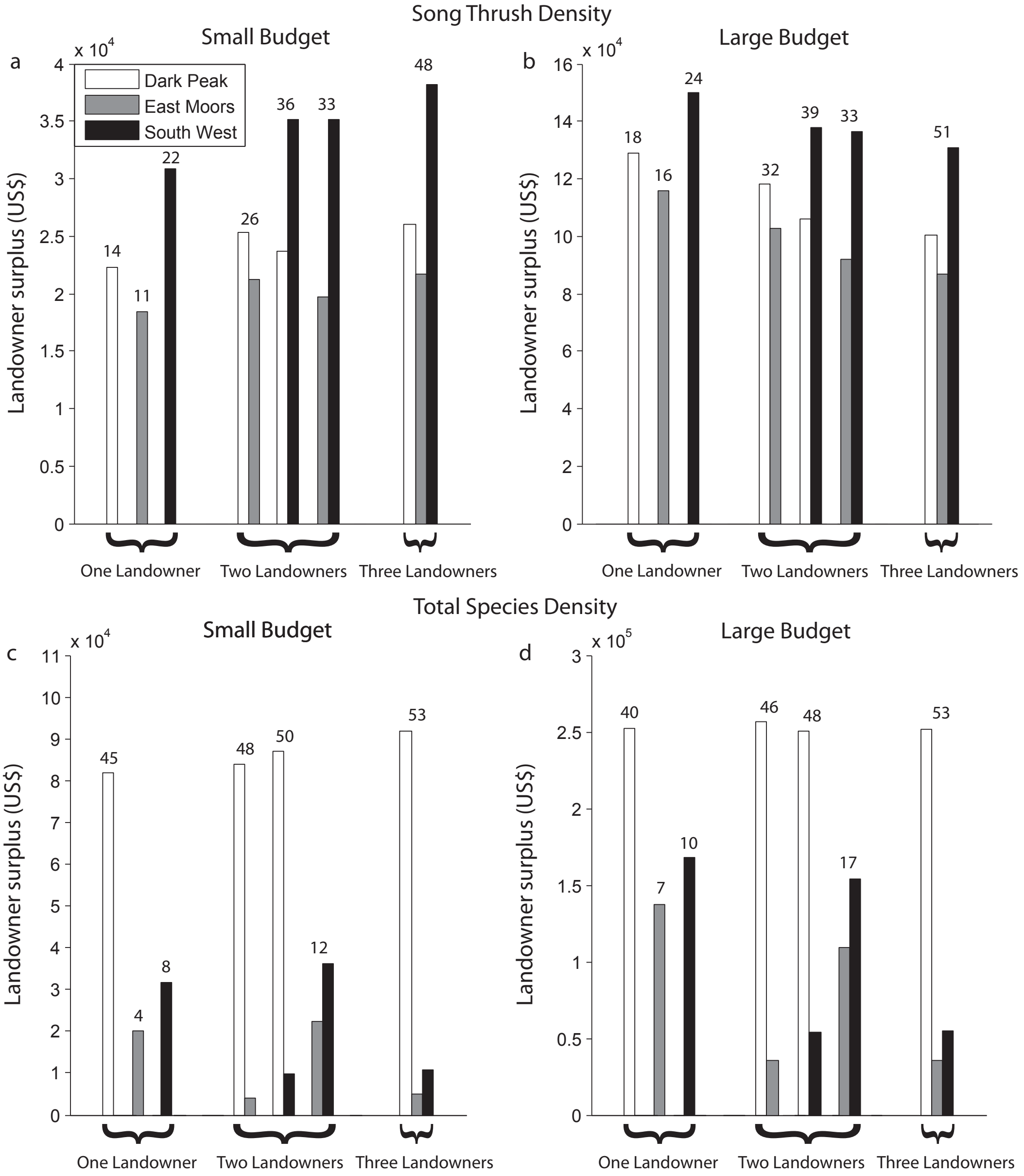


Figure 4.8: Landowner surpluses for small (Panels a & c) and large (Panels b & d) budgets when the conservation objective is to maximise the increase in Song Thrush density (Panels a & b) and total species density (Panels c & d). The consequences of landowners obtaining the documented levels of surplus are indicated above the bars; the numbers show the percentage decrease in conservation outcomes when landowners maximise profits compared to when they accept investment at opportunity costs.

circumstances did not arise in the examples in this chapter, this must be borne in mind when applying the relaxation algorithm.

4.6.5 Case study results for various budget levels

In Section 4.4, I stated that the results from the case study were only minimally affected by the level of the budget. To demonstrate this, here results are given for the same analyses undertaken with a small budget, equal to 250 times the mean per hectare cost of land over the three farms ($\approx \$127,000$), and a large budget, equal to 750 times the mean, per hectare cost of land ($\approx \$382,000$).

Results when the conservation objective is to maximise the increase in Song Thrush density over the three farms are shown in Fig. 4.8a & b. As can be seen, the distribution of surplus among the different landowners is essentially the same with a small, medium or large budget (a medium budget refers to the results in Section 4.4). For example, the landowner from the South West region can always obtain the highest surplus while the landowner from the East Moors can always obtain the least. There are minor differences among the different budget levels, however. For example, the relationship between the number of landowners demanding payments higher than opportunity costs and the surplus of individual landowners is not wholly consistent neither are the biodiversity outcomes that result from landowners obtaining surplus. These differences are very small, however, on the order of a few percent.

Results when the conservation objective is to maximise the increase in total species density over the three farms are shown in Fig. 4.8c & d. The same patterns are evident as in Fig. 4.8a & b: the distribution of surplus among the different landowners is essentially the same over the different budget levels with small quantitative differences in surplus levels and biodiversity outcomes.

Chapter 5

Cost structures in conservation planning

5.1 Introduction

How to optimally select conservation reserves, known as the reserve site selection (RSS) problem, is a central area of conservation research. The RSS problem lies at the intersection of biology and economics: not only should sites be selected that contain a representative sample of biodiversity and promote long-term species persistence (Margules and Pressey 2000) but limited conservation budgets mean that this objective must be achieved in a cost-effective and economically efficient manner (James et al. 1999; Balmford et al. 2003; Costello and Polasky 2004). To ensure that conservation planning exercises return reserve networks with these features, relevant biological and economic factors should be accounted for in model formulations.

The RSS problem has been expressed through two archetypal models: cost minimisation problems, most notably the set covering problem (SCP) (Ando et al. 1998; Rodrigues et al. 2000; Moore et al. 2003) and budget constrained maximisation problems, most notably the maximum covering problem (MCP) (Church and ReVelle 1974; Polasky et al. 2001; Gaston and Rodrigues 2003). The mathematical formulations for the SCP and MCP are shown below, where $I = \{i \mid i = 1,\ldots,m\}$ is the set of species that can be covered, $J = \{j \mid j = 1,\ldots,n\}$ is the set of candidate reserves and N_i is the subset of J that contains species i. x_j and y_i are binary variables. Should $x_j = 1$ then site j is selected for conservation and is 0 otherwise. If $y_i = 1$ then species i is present on at least one of the selected sites and is 0 otherwise. c_j is the cost associated with selecting site j and B is the conservation group's budget. The SCP selects a minimum cost set of reserves that covers all species (or other conservation feature) in a given landscape at least once. The reality of limited conservation funds means that covering all species in a landscape, even at minimum cost, is often not feasible. The MCP recognises such limitations by maximising the

Set Covering Problem

$$
\begin{aligned}
& \min \sum_{j \in J} c_j x_j && (5.1)\\
\text{s.t.} \quad & \sum_{j \in N_i} x_j \geq 1, \text{ for all } i \in I
\end{aligned}
$$

Maximum Covering Problem

$$
\begin{aligned}
& \max \sum_{i \in I} y_i \\
\text{s.t.} \quad & \sum_{j \in N_i} x_j \geq y_i, \text{ for all } i \in I \\
& \sum_{j \in J} c_j x_j \leq B && (5.2)
\end{aligned}
$$

number of species covered by the reserves subject to a given budget. The attractiveness of these formulations is that they focus on cost-effectiveness and economic efficiency (but see Chapter 6). However, both problem formulations produce reserves designs that lack spatial coherence given that solutions depend only on the spatial distribution of biodiversity and site costs (Fig. 5.1a). While such reserve designs may represent the biodiversity of a region adequately, species persistence may be jeopardised due to the lack of spatial considerations (Cabeza and Moilanen 2001).

There are many spatial features of reserve design that have been argued to promote species persistence and many models have been developed to achieve desirable spatial attributes. For example, ensuring that reserves within a network are not separated by large distances can facilitate recolonisation should species become locally extinct in one or more reserves (Diamond 1975). Genetic diversity may also be conserved when reserves are located in close proximity to one another by allowing for inter-population breeding (Williams et al. 2005). RSS models have been developed by Nalle et al. (2002a), Nalle et al. (2002b) and Önal and Briers (2002) that attempt to minimise the sum of pairwise distances between reserves (Fig. 5.1b). The shape of a reserve network can also be critical for the ability of species to persist. A reserve network that has a large boundary length compared to total area can suffer from pronounced edge effects whereby the effective size of the reserve is diminished due the unsuitability of habitat adjacent to non-protected sites (Williams et al. 2005). Reserve network shape has most usually been controlled by minimising the compactedness of the reserve network. For example, McDonnell et al. (2002) and Fischer and Church (2003) define models that minimise the reserve network's total boundary length (Fig. 5.1c). Several other spatial features of reserve design have featured prominently in the literature, including reserve connectivity (e.g., Cerdeira et al. 2005; Önal and Briers 2005; Cerdeira et al. 2010; Alagador et al. 2012; see Fig. 5.1d) for arguments that parallel those for limited inter-site distances, and the development of reserves with core protected areas surrounded by buffer zones to limit edge effects (e.g., Williams and ReVelle 1996; 1998; Clemens et al. 1999; see Fig. 5.1e). The mathematics of many of these models and many others are reviewed in Williams et al. (2005).

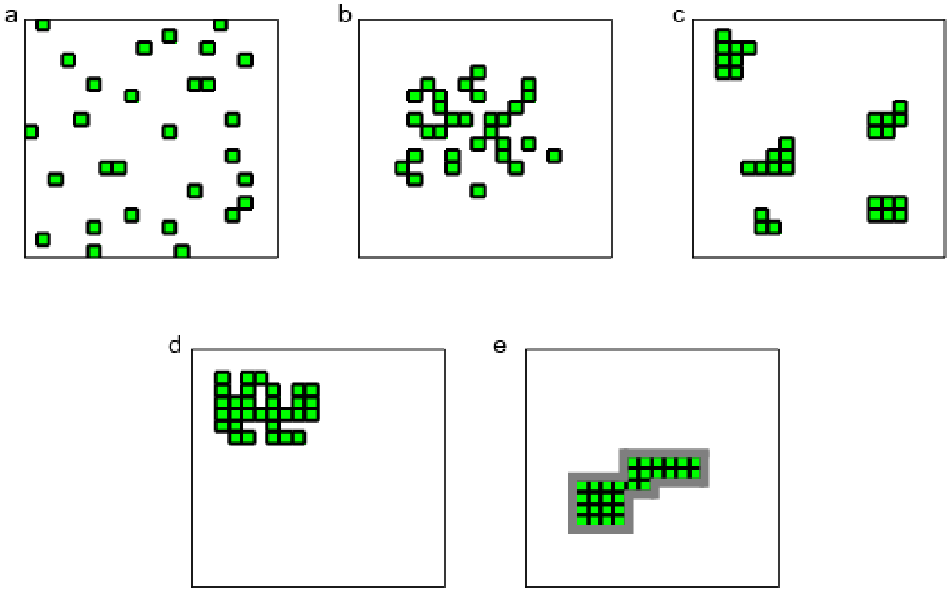


Figure 5.1: Reserve designs that arise from various reserve site selection models. Selected sites are shown in green. *Panel a:* Reserves selected with the MCP. Sites lack spatial coherence. *Panel b:* Minimising the sum of pairwise distances between reserves. *Panel c:* Minimising the total boundary length of reserves. *Panel d:* Maximising the contiguity of reserves. *Panel e:* Reserves with core habitat area and a buffer zone shown in grey.

These examples highlight that the biological consideration of how species persist in reserve networks has led to significant advancements in the basic models used for site selection. Reinforcing the bias highlighted in Chapter 1, commensurate focus has not been paid to developing RSS models with more sophisticated economic formulations. In fact, in the vast majority of RSS models the same cost structure has been applied as in the SCP and MCP. As can be seen in objective 5.1 of the SCP and constraint 5.2 of the MCP, this cost structure assumes that each site has a fixed, pre-determined cost and that the cost to secure multiple sites is simply the sum of the costs of the individual sites. This cost structure is likely an oversimplification of the economics of reserve site selection. In this chapter, I present initial ideas and results that will partly populate a synthetic review of cost structures in conservation planning. I model two distinct conservation planning problems that incorporate alternative cost structures and review two areas of conservation planning where complex economic formulations have been incorporated to evaluate how economic aspects of conservation affect the optimal spatial distribution of conservation reserves.

5.2 Conservation planning with alternative cost structures

5.2.1 Example problems

In this section I model two conservation planning problems with alternative cost structures. I compare the results from these models to those in which the assumption is made that the cost to conserve multiple sites is the sum of the cost of the individual sites (hereafter referred to as the basic cost structure). As conservation budgets are limited, I focus on budget constrained maximisation problems, as characterised by the MCP. To investigate how cost structures affect reserve site selections generally, in addition to modelling both example problems as maximum coverage problems, I also model them as knapsack problems (Hajkowicz et al. 2007). This problem is formulated mathematically as

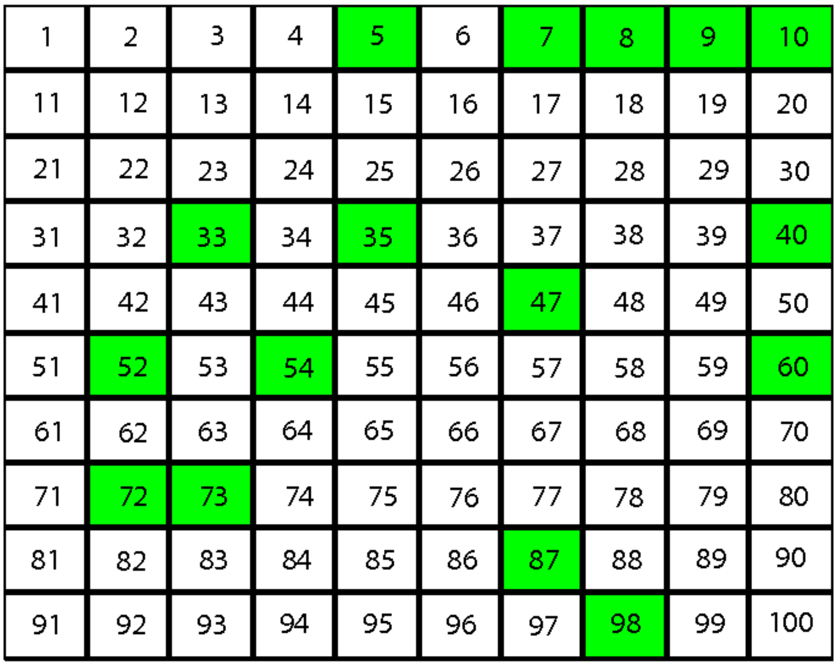
$$\max \sum_{j \in J} v_j x_j$$

subject to

$$\sum_{j \in J} c_j x_j \leq B$$

where all terms are as before except v_j, which is the conservation value of site $j \in J$. The difference between the MCP and knapsack problem is the assumed nature of ecological interactions over sites. The MCP incorporates site complementarity, meaning that the conservation benefit gained by protecting a site is dependent upon the conservation value of other protected sites. The most common example of complementary benefits is species richness. With the knapsack problem, on the other hand, conservation benefits are additive rather than complementary. This means that conservation benefits simply sum over selected sites and therefore that the benefit gained by protecting a site is independent of the conservation value of other protected sites. Common examples of such benefits are the abundance of a rare species or recreation benefits from an agri-environment scheme. By modelling both additive and complementary benefits problems with alternative costs structures, I am able to investigate how overlooked aspects of conservation planning affect reserve selection over a wide range of conservation scenarios.

To avoid confusion, throughout the example problems I use the following definitions: a site is a parcel of land; a reserve is a parcel of land under conservation management; and a reserve network is a group of land parcels under conservation management.



1	2	3	4	🟩 5	6	🟩 7	🟩 8	🟩 9	🟩 10
11	12	13	14	15	16	17	18	19	20
21	22	23	24	25	26	27	28	29	30
31	32	🟩 33	34	🟩 35	36	37	38	39	🟩 40
41	42	43	44	45	46	🟩 47	48	49	50
51	🟩 52	53	🟩 54	55	56	57	58	59	🟩 60
61	62	63	64	65	66	67	68	69	70
71	🟩 72	🟩 73	74	75	76	77	78	79	80
81	82	83	84	85	86	🟩 87	88	89	90
91	92	93	94	95	96	97	🟩 98	99	100

Figure 5.2: Solution to a hypothetical reserve selection problem. The sites selected as reserves are shown in green.

Problem 1 - Economies of scale in conservation management costs

The first example problem that I investigate has a cost structure that incorporates an area-cost relationship. Several studies have shown that conservation management costs show economies of scale or, in other words, that the cost to add a hectare of land to a large reserve is less than the cost to add an additional hectare to a small reserve (Auden and Hirons 2002; Balmford et al. 2003; 2004; Strange et al. 2007). Using data from 78 reserves in Yorkshire, Armsworth et al. (2011) quantify this relationship and show that management costs scale with the square root of reserve area. The authors also find that conservation management costs are much larger than acquisition costs when funded on an endowment basis. Consequently, the cost to manage a reserve network in this scenario would be approximated by

$$\sum_{k\in K}\Big(\sum_{m\in M} c_m a_{k,m}\Big)^{1/2} + \sum_{n\in N} c_n a_n^{1/2} \tag{5.3}$$

where c_m is the constant of proportionality for reserve m, $a_{k,m}$ is the area of reserve m that is a member of cluster k and a_n is the area of reserve n that is not in a cluster. A cluster of reserves is defined to be a collection of reserves that are contiguous. Fig. 5.2, which shows the solution to a hypothetical reserve selection problem, provides an example. For simplicity, the spatial configuration of sites has been abstracted from the problem such that, even though site area is heterogeneous, the results can be shown on a homogeneous grid (this simplification will be maintained throughout the section). Sites are contiguous if they share a border horizontally or vertically but not diagonally. Therefore, reserves 7, 8, 9 and 10 form a cluster as do reserves 72 and 73. Assuming that the proportional constants equal 1 for all reserves, the combined management costs for these two clusters would therefore be

$$(a_{1,7}+a_{1,8}+a_{1,9}+a_{1,10})^{1/2}+(a_{2,72}+a_{2,73})^{1/2}$$

whereas the cost to manage the other reserves not in clusters would simply be

$$a_5^{1/2}+a_{33}^{1/2}+a_{35}^{1/2}+\dots$$

Given that for any set of positive numbers, A,

$$\sum_{i\in A} a_i^{1/2} \geq \Big(\sum_{i\in A} a_i\Big)^{1/2}$$

all else being equal, reserves that are clustered will be less expensive to manage than those that are isolated. Therefore, a reasonable hypothesis for this conservation planning problem would be that when management costs show economies of scale, reserves are more highly clustered than when costs do not scale with reserve area as with the basic cost structure.

Problem 2 - Locate conservation offices and reserves simultaneously

The second conservation planning problem that I model deals with locating conservation reserves and management offices. Typically reserve selection analyses assume that reserves can be located independently of management offices. However, staff who monitor and manage reserves will often not be based on site but will travel among reserves via a regional management office. Given that staff time can be the largest single component in conservation management costs (Armsworth et al. 2012), the cost to conserve a site is likely an increasing function of distance from the nearest office. The locations for management offices and conservation reserves must therefore be simultaneously chosen such that there is an optimal balance between conservation benefits and distance.

Incorporating the location of conservation offices into the optimisation formulations requires an additional choice variable. Let this variable be z_j, where $z_j = 1$ if site j is selected as a management office and is zero otherwise. The full formulations for both the additive and complementary benefits versions of this problem are shown below. All terms are as before with the addition of constraint (5.4) & (5.7), which state that there must be N_o sites

Additive Benefits

$$\max \sum_{j \in J} v_j x_j$$

$$\text{s.t.} \quad \sum_{j \in J} z_j = N_o \tag{5.4}$$

$$\sum_{j \in J} x_j + z_j \leq 1, \text{ for all } j \in J \tag{5.5}$$

$$\sum_{j \in j} (c_j(r_j, d_j)x_j + c_j(o_j)z_j) \leq B \tag{5.6}$$

Complementary Benefits

$$\max \sum_{I \in I} y_I$$

$$\text{s.t.} \quad \sum_{J \in N_i} x_j \geq y_i, \text{ for all } i \in I$$

$$\sum_{j \in J} z_j = N_o \tag{5.7}$$

$$\sum_{j \in J} x_j + z_j \leq 1, \text{ for all } j \in J \tag{5.8}$$

$$\sum_{j \in j} (c_j(r_j, d_j)x_j + c_j(o_j)z_j) \leq B \tag{5.9}$$

assigned as conservation offices, constraint (5.5) & (5.8), which state that no single site can be assigned to be a reserve and an office, and constraint (5.6) & (5.9), the new cost constraint. In this constraint, reserve cost is a function of the cost to acquire the site as a reserve, r_j, and the distance from the nearest office, d_j. The cost of an office site is a function only of the cost to acquire the site as an office, o_j. The total cost of offices and reserves cannot exceed the budget, B. This constraint could conceivably take various forms depending on how conservation costs are affected by distance from the nearest office. I assume the simplest formulation in which those costs scale linearly with distance.

Given that the cost to designate a site as a reserve increases with distance from the conservation office in this problem, a sensible hypothesis is that when cost is incorporated in this form, reserves should generally encircle the locations of the offices with few reserves located at a large distance from every office.

Methods - Genetic algorithms

For both example problems, genetic algorithms (GA) are used to obtain solutions (Goldberg 1989). The GA is a heuristic solution technique for generating solutions to discrete or combinatorial optimisation problems. The method mimics the process of evolution by natural selection. While GAs cannot guarantee optimal solutions, they allow for large scale and nonlinear problems to be tackled, a feature which is necessary for the analyses in this chapter.

Solutions using a GA are found as follows. First, a random population of solutions is generated and the "fitness" of each solution is calculated. For these analyses, the fitness of a solution is the conservation value gained through reserve selections with ties differentiated by cost. Solutions are then modified through random mutations (i.e., the status of a site is changed from reserve to non-reserve or vice-versa) and random crossovers (i.e., part of one solution replaces part of another solution) and fitness levels are re-evaluated. At each iteration of the algorithm the fittest solutions from the population are not modified. As the algorithm proceeds, the fitness of solutions is expected to increase. The algorithm can be terminated when either a maximum number of iterations or a satisfactory fitness level has been achieved. For the analyses presented here, the algorithms were terminated after 1000 generations, which in all cases was sufficient to achieve convergence (defined as no improvement in fitness for at least 100 generations). For both problems, mutation rate was set at 1%, meaning that for 400 sites, the status of 4 on average would be changed; and crossover rate was set at 80%, meaning that 80% of new solutions were formed by crossing two solutions from the previous iteration.

Data generation

Biodiversity, site area and site cost data were simulated for the analyses using the Peak District data that has been described and used in the three preceding chapters. As in Chapter 3, biodiversity data were produced by first determining the best-fit distributions for data regarding avian species on the farms (Dallimer et al. 2009). For the analyses with additive benefits, site conservation value was taken to be total species density and thus the objective was to maximise the sum of species densities over selected sites. For the analyses with complementary benefits, maximising species richness over selected sites served as the conservation objective. Random presence or absence matrices that accounted for the nestedness of natural populations were constructed for these analyses in the same manner as described in Chapter 3.2.

Site area and cost data were produced by fitting the best-fit distribution to data documenting the size and per annum rental values of the same properties on which the biodiversity data was collected (Acs et al. 2010). For the economies of scale in management costs problem, I assumed that all proportional constants were equal to 1 ($c_i = 1$ for all sites in Eqn. 5.3) and used the best-fit distribution to randomly assign each site an area. For the office-reserve location problem, each site was randomly assigned a per annum rental cost and I assumed that the cost to acquire the sites was equal to the rental values over 10 years.

Results - Management costs scale with area

For both the example problems, a comparison is made between the complementary benefits problem with the basic cost structure and the problem with an alternative cost structure with both additive and complementary benefits. When formulated with a basic cost structure,

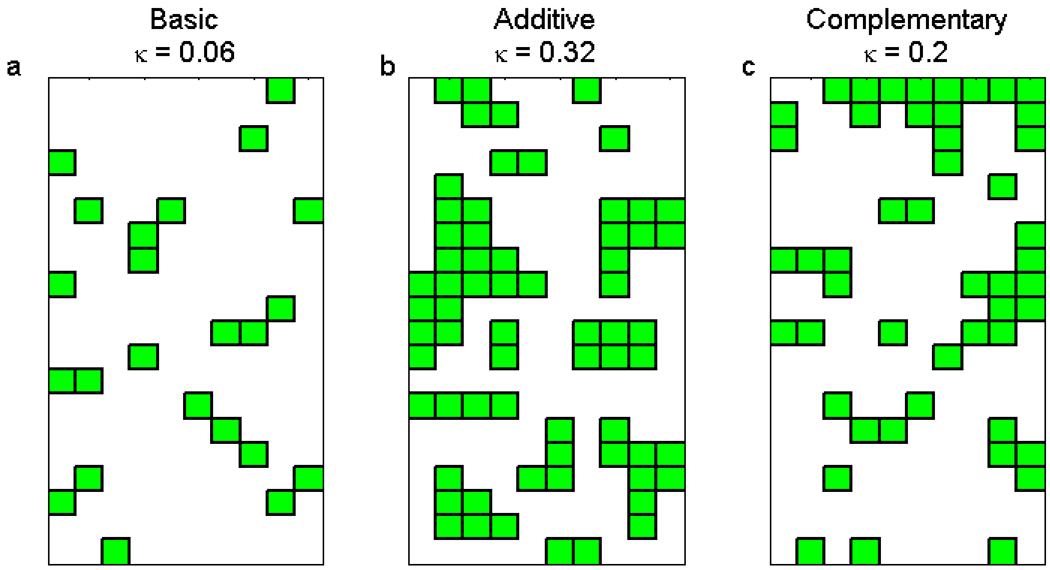


Figure 5.3: *Panel a:* Reserve selections with the basic cost structure and complementary benefits. Reserve selections when conservation management costs display economies of scale with additive benefits (Panel b) and with complementary benefits (Panel c). In all cases, there are 200 sites containing a total of 400 species and the budget available is the same. Selected reserves are shown in green.

model solutions are dependent only on the distribution of biodiversity and costs and thus there is little qualitative difference in outcomes for additive or complementary benefits.

To investigate the hypothesis that economies of scale in conservation management costs result in greater clustering of reserves than would be the case if management costs simply summed over reserves, I define a quantitative measure of clustering termed the clustering coefficient, κ, where

$$\kappa = \frac{1}{n}\sum_{i=2}^{m} i \cdot s_i$$

Here, s_i is the size of the family of clusters of size i, m is the maximum cluster size and n is the number of sites. This metric ranges from zero to one with zero indicating that all selected reserves are isolated and one indicating that all available sites are selected as reserves, forming one contiguous whole. Evidently, values close to one would require very large budgets. As an example, the hypothetical reserve selection solution shown in Fig. 5.2 has one cluster of size 2 and one cluster of size 4. Therefore, $\kappa = (2\times1+3\times0+4\times1)/100 = 0.06$

A typical simulation result is shown in Fig. 5.3. As expected, using the basic cost structure results in a reserve network that lacks any spatial coherence, with reserves appear-

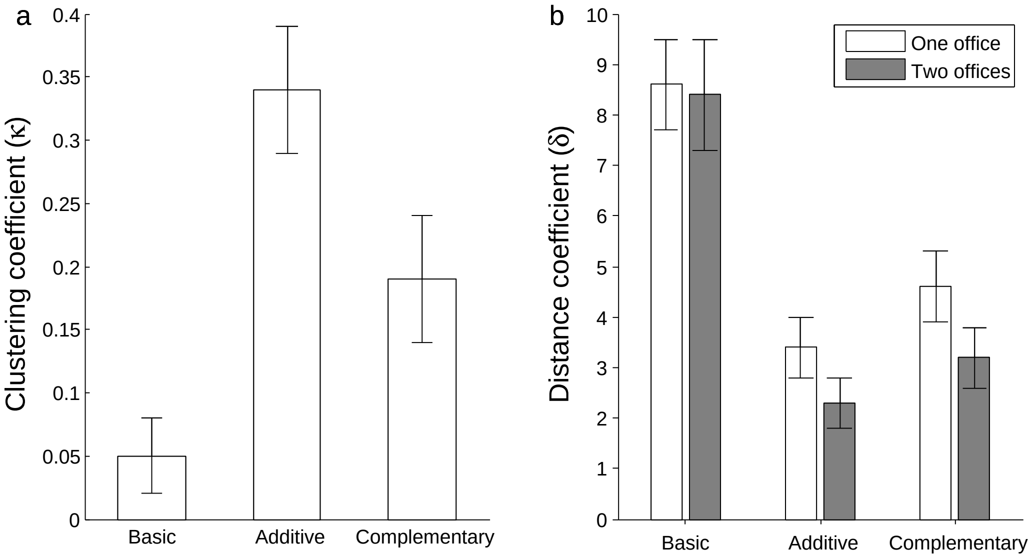


Figure 5.4: *Panel a:* Mean clustering coefficient from 100 simulations of the management costs problem with a basic cost structure and complementary benefits (“Basic”), when management costs show economies of scale with additive benefits (“Additive”) and with complementary benefits (“Complementary”). *Panel b:* Mean distance coefficient from 100 simulations of the office-reserve location problem with a basic cost structure and complementary benefits (“Basic”), when costs scale linearly with distance from the nearest office with additive benefits (“Additive”) and with complementary benefits (“Complementary”). Bars at shown at the 5th and 95th percentiles.

ing randomly distributed throughout the landscape (Fig. 5.3a). On the other hand, when economies of scale in conservation management costs are incorporated into the analysis, selected reserves are not randomly distributed but appear highly clustered (Fig. 5.3b & c). In this example, the clustering coefficient is at least 13 times greater when the nonlinearity in management costs is recognised compared to when it is not. Interestingly, the solutions shown in Fig. 5.3b & c appear similar to those obtained when minimising the total boundary length of reserves for biological reasons (Fig. 5.1c). When management costs scale with area, additive benefits result in considerably more clustering than does complementary benefits in this example. Also of note is the greater number of reserves selected as clustering increases given the marginally decreasing management costs.

To test whether these results hold generally, 100 simulations of the optimisation problem were run and the results are shown in Fig. 5.4a. The results confirm that when economies of scale inherent in conservation management cost are recognised and factored into reserve design mechanisms, selected reserves are considerably more clustered than when costs are incorporated with the basic cost structure. As in the example solution of Fig. 5.3, nonlinear

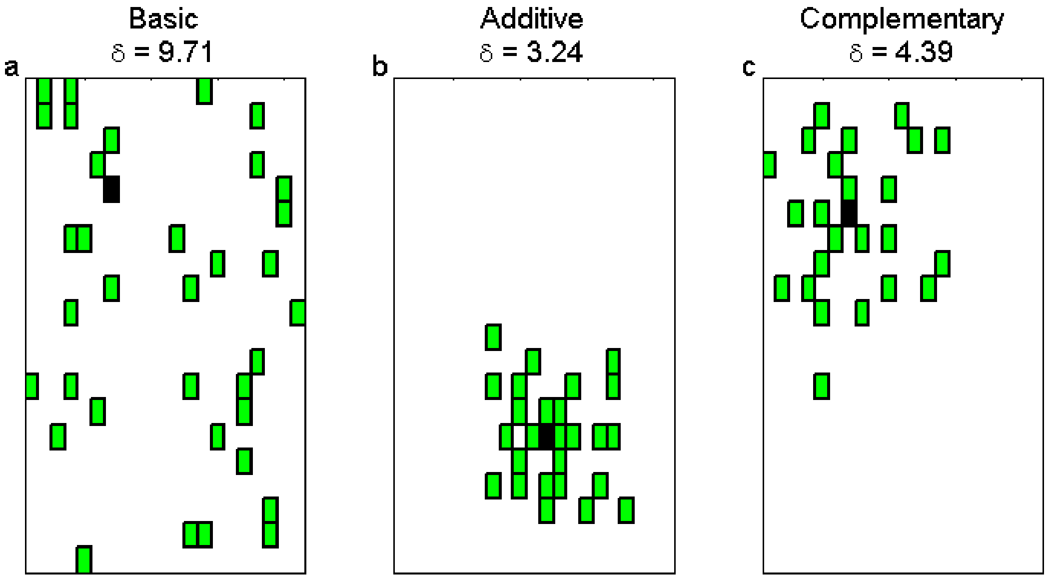


Figure 5.5: *Panel a:* Reserve selections with the basic cost structure in which there is no distance dependence on costs and complementary benefits. Reserve selections when conservation costs scale linearly with distance from the nearest conservation office with additive benefits (Panel b) and with complementary benefits (Panel c). In all cases, there are 400 sites containing a total of 400 species and the budget available is the same. Sites selected for reserves are shown in green and those selected as offices are shown in black.

management costs with additive benefits results in a higher level of clustering than does complementary benefits.

Results - simultaneously site offices and reserves

The hypothesis for the office-reserve location problem is that when conservation costs increase with distance from the conservation office, reserves should generally encircle the offices with few reserves located at a large distance from all offices. Here, I test this hypothesis by measuring the average distance of reserves to their nearest office. This is quantified by the distance coefficient,

$$\delta = \frac{1}{n}\sum_{i \in S}\sqrt{(x_i - x_{no})^2 + (y_i - y_{no})^2}$$

where S the collection of reserves, with $|S| = n$; (x_i, y_i) are the Cartesian coordinates of the reserve $i \in S$ (relative to some arbitrarily chosen origin); and (x_{no}, y_{no}) are the coordinates of the nearest office.

For visual clarity, the number of sites in the landscape is doubled from the previous

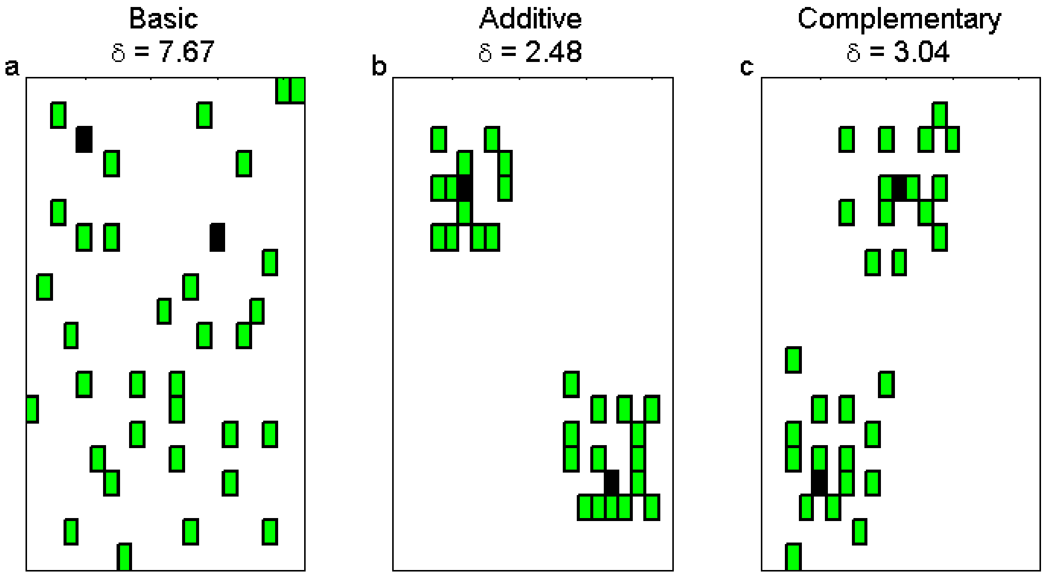


Figure 5.6: *Panel a:* Reserve selections with the basic cost structure in which there is no distance dependence on costs and complementary benefits. Reserve selections when conservation costs scale linearly with distance from the nearest conservation office with additive benefits (Panel b) and with complementary benefits (Panel c). In all cases, there are 400 sites containing a total of 400 species and the budget available is the same. Sites selected for reserves are shown in green and those selected as offices are shown in black.

example problem and problems with one or two conservation offices only are investigated. Typical simulation results are shown in Fig. 5.5 (one conservation office) and Fig. 5.6 (two conservation offices). When reserve cost is not dependent on office location, as with the basic cost structure (Figs. 5.5a and 5.6a), selected reserves form no spatial relationship with the conservation office(s). To the contrary, when conservation costs increase with distance from the conservation office (Figs. 5.5b, 5.5c, 5.6b & 5.5c) selected reserves are located near to office(s) with no reserve being located at a great distance. From these example solutions, the distance coefficients show that reserves are on average around 2-3 times closer to the conservation office when the distance dependence on conservation costs is incorporated in the analyses.

As in the economies of scale in management costs problems, differences emerge between the additive and complementary benefits formulations when the distance dependence on cost is recognised. In this case, complementary benefits seem to increase the average difference that a reserve is from the nearest conservation office (One office: Additive $\delta = 3.24$ vrs Complementary $\delta = 4.39$. Two offices: Additive $\delta = 2.48$ vrs Complementary $\delta = 3.04$). Differences also emerge between the problems that have one or two conservation offices:

an increase in the number of conservation offices reduces the average reserve distance. This remains the case for both the additive and complementary benefits problems (Additive: one office $\delta = 3.24$ vrs two offices $\delta = 2.48$. Complementary: one office $\delta = 4.39$ vrs Complementary $\delta = 3.04$).

To investigate whether these result patterns hold in general, 100 simulations were again run and the results are shown in Fig. 5.4b. The results confirm that when the increasing cost of conservation with distance from a regional management office is recognised and factored into the analysis, reserves are located considerably closer to the conservation offices than when this relationship is not recognised. Complementary benefits as opposed to additive benefits and increasing the number of conservation offices further decreases the average distance between offices and reserves.

5.3 Alternative cost structures in the conservation planning literature

Few studies in the conservation planning literature have either implicitly or explicitly incorporated more advanced/realistic cost structures, let alone explored their consequences on reserve network design. To further reinforce the critical nature of accounting for cost structures appropriately, I discuss two noteworthy areas of exception.

5.3.1 Land market feedbacks

Conservation planning studies that incorporate conservation costs commonly assume that land acquisitions do not affect land prices irrespective of the amount of land that is allocated for conservation. However, land acquisitions by conservation groups result in an overall increase in the local demand for land thus increasing its price, which in turn decreases and/or displaces development demand. As a consequence of the increased market price, however, some marginal landowners will be moved to sell their land who would have otherwise found it uneconomical to do so. These features of land markets result in a complex cost structure in which the cost to conserve a site depends on which other sites are selected for conservation, as with the economies of scale in management costs problem modelled in Section 5.2. In contrast to that problem, however, land market dynamics produce diseconomies of scale.

Armsworth et al. (2006) present the first model in the conservation planning literature that incorporates land market dynamics. An important feature of the authors' model is the role of "open land", which is land that is neither developed, and thus has no ecological value, nor reserved for conservation. Many conservation planning studies assume that open land cannot support any species. However, the authors point out that evidence suggests that many agricultural and urban areas provide important habitat for species of conservation concern. Making the assumption that open land has an ecological worth that is some fraction of the value of a conservation reserve, the authors show that land market dynamics mean that the amount of conservation that is achieved by land acquisitions is less that the acreage purchased. The authors further demonstrate that commonly pursued conservation strategies that fail to appreciate land market dynamics consistently perform worse than a strategy that recognises the effects of conservation acquisitions. Most soberingly, analyses in Armsworth et al. (2006) indicate that unappreciated land market dynamics can actually result in conservation doing more harm than good.

Following on from Armsworth et al. (2006), three recent studies have incorporated land market dynamics into conservation planning exercises (Dissanayake and Önal 2011; Jantke and Schneider 2011; Tóth et al. 2011). These studies show that failing to accurately account for the complex economic effects of conservation land acquisitions can result in highly inefficient reserve selections and the potential loss of biodiversity.

5.3.2 Human demands for conservation

The basic cost structure used in most conservation planning studies assumes that conservation budgets are fixed and predetermined. The implication of this assumption is that the funds available for conservation exist independently of conservation activities. To the contrary, Ando and Shah (2010) argue and provide evidence to show that individuals' willingness-to-pay (WTP) for conservation decreases with distance from the conservation project, a phenomenon the authors term "spatial value decay." This feature of conservation economics bares similarities to the distance dependence on cost in the office-reserve location problem modelled in Section 5.2. However, spatial value decay means that increasing distance from human populations reduces the budget available for conservation rather than increasing the cost. Nonetheless, the effect is equivalent in that the relative cost of conservation increases with distance.

The authors incorporate spatial value decay in two conservation planning problems: locating a single conservation reserve to maximise social welfare and designing a network of reserves to maximise an environmental service where fragmentation of the reserves diminishes the flow of the service. Without consideration of WTP for conservation, the optimal policy choices for these problems would be to locate the single reserve at the site of highest conservation value and design the reserve network with no fragmentation. Simulation results show that these policy prescriptions are rarely optimal, however, when cost structures that incorporate human demands for conservation are factored into the analyses. Social welfare is maximised when the single reserve is situated such that there is a trade-off between the ecological value of the site and the distance from human population centres. Similarly, reserve networks with a moderate degree of fragmentation are optimal for conservation as this locates reserves closer to human populations that are dispersed in the landscape.

5.4 Discussion

For biological considerations, the spatial configuration of nature reserves has been one of the core problems of conservation biology. Both the modelled problems and highlighted examples in Sections 5.2 & 5.3 show that reserve design also depends critically on the economics of conservation. Moreover, the results from Section 5.2 show that the effects on reserve design due to biological concerns can be replicated by those of an economic nature. While the focus of the conservation literature has been on biological factors (Naidoo et al. 2006), the analogous relationship between biology and economics in conservation planning should be self-evident. Many conservation planning analyses can essentially be reduced to the maximisation of some benefit to cost ratio. Mathematically then, it is unsurprising that biological effects on reserve design should have corresponding economic counterparts.

Synthesising over the examples in this chapter, Table 5.1 lays out the underlying principles of when reserves should be clustered or spread for biological and corresponding economic reasons. In general, conservation reserves should be clustered when the marginal gain in conservation benefits is increasing as reserve proximity increases. For example, as discussed in the Section 5.1, proximate reserves may increase conservation benefits by allowing species to persist as metapopulations, promoting recolonisation of reserves on which species become locally extinction and reducing inbreeding depression (Diamond 1975; Williams et al. 2005). Analogously, reserves will tend to cluster if the marginal cost of conservation is decreasing as reserve proximity increases, as demonstrated in the economies of scale in conservation management costs analyses of Section 5.2. On the other hand, there will be pressure to spread reserves throughout the landscape if the marginal gain in conservation benefits is decreasing as reserves become closer. An important example of

Table 5.1: General principles, along with particular examples from this chapter, for when biological and economic factors should lead to conservation reserves being spatially clustered or spread.

	Marginal Benefit	Marginal Cost
Clustered	Increasing e.g., Metapopulations	Decreasing e.g., Economies of scale
Spread	Decreasing e.g., Complementarity	Increasing e.g., Price feedbacks

this that was demonstrated throughout Section 5.2 is the complementary nature of conservation benefits when maximising species richness (Margules et al. 1988; Pressey and Nicholls 1989; Church et al. 1996; Justus and Sarkar 2002). Complementarity in this context means that the conservation value gained by adding a hectare of land to a reserve network is the number of species present on that hectare that are not already covered. Thus, fewer species will tend to be gained as more hectares are added given the high probability that widespread species are already represented. Consequently, maximising species richness will necessitate that many different habitats and ecosystem types are represented in the network, thus geographically spreading the reserves over the landscape. Equivalently, economic factors will lead to reserves being spread if the marginal cost of conservation is increasing with proximity. As discussed, land market price feedbacks have this effect when conservation acquisitions inflate local land cost (Armsworth et al. 2006).

Complex cost structures in conservation planning are likely to arise in many contexts besides those modelled and discussed in this chapter. For example, transaction costs, the costs associated with identifying sites of conservation interest and negotiating with landowners (Naidoo et al. 2006), are likely to scale with the size of the region in which the conservation group operates (McDonald 2009; Armsworth et al. 2012). Identifying important conservation areas when focused on protecting a geographically isolated species will be significantly less expensive than undertaking the same exercise to protect a species with a global distribution, for example. Many of the complexities in the cost structures discussed in the chapter have arisen due to the cost of one site depending on which other sites are selected for conservation. In a similar vein to land market feedbacks, such interdependences arise when conservation reserves have amenity values such that premiums are placed on land located in proximity to reserves (Thorsnes 2002). The effect of amenity values is thus to increase land prices in areas where conservation reserves are established, inflating the cost of other potential reserve sites in the same locality. Interdependences on cost can also arise as a consequence of knowledge transfers. Evidence of this was seen in Chapter 3. There, I found that agricultural cooperatives were able to skew the distribution of surplus among those landowners who received conservation investment. Specifically, the cost of any site (landowner's WTA + surplus) depended both on whether or not cooperatives were able to participate in the auctions and, if so, the extent of landowner cooperation (see Fig. 3.2). Similarly, landowners who enrol in conservation schemes may discuss their experiences with neighbouring landowners. The effects of these knowledge transfers could be highly complex and multidirectional. For example, some landowners could gain information regarding the conservation group's negotiating strategy or information level, allowing them to leverage for higher payments. On the other hand, landowners could share positive experiences of conservation with neighbours - effects of ancillary benefits such as ecotourism, for example - increasing the willingness of these neighbouring landowners to engage positively with the conservation group and accept lower payment levels than would otherwise have been case.

Finally, while each of the complexities that can arise in the cost structures of conservation investments has been analysed and/or discussed in isolation, many will act in concert. For example, while land market dynamics will result in diseconomies of scale, thus tending to spread reserves over the landscape, economies of scale in management costs will simultaneously act in the opposite direction. Identifying the elements of conservation costs that impact conservation outcomes most and bringing these elements together into a single cost structure is a daunting task much of which remains to be completed. Indeed, most studies that incorporate spatially explicit conservation costs focus on only one component of costs, typical acquisition costs, in the hope that the chosen component acts as a proxy for the total cost (Adams et al. 2009). As indicated here, however, different aspects of conservation costs potentially have very different consequences for spatial prioritisations.

5.4.1 Conclusions

In this chapter I have demonstrated, from a wide variety of conservation planning scenarios, that failing to appreciate realistic aspects of the economics of conservation can result in potentially ineffective and inefficient reserve network designs. Given that conservation is defined by both biological and economic factors, with each having analogous effects on reserve network design, it is essential that conservation science focuses as much attention on the costs of conservation as the biology.

Chapter 6

How does the distribution of biodiversity affect the trade-off between conservation costs and species coverage?

6.1 Introduction

As stressed and exemplified throughout this thesis, the problem of conservation planning is both biological and socio-economic. The preceding analyses have investigated the interaction between these two aspects of conservation, ranging from how landowners' leverage potential affects conservation outcomes to how cost structures can influence the optimal spatial distribution of conservation reserves. Mapping the trade-off between conservation improvements and the increasing costs of achieving those improvements is a direct and informative way to understand the relationship between the biological and socio-economic components of conservation. However, such trade-off analyses have had limited application in conservation planning studies. In one study, Polasky et al. (2001) map the trade-off between conservation cost and species coverage levels in a portion of the state of Oregon. They show, for example, that 75% of all terrestrial vertebrate species in this region can be covered at 1% of the cost of covering all such species. In a similar analysis, Ando et al. (1998) demonstrate that covering half of all species listed in the US Endangered Species Act in 1995 would cost 2% of covering all such species.

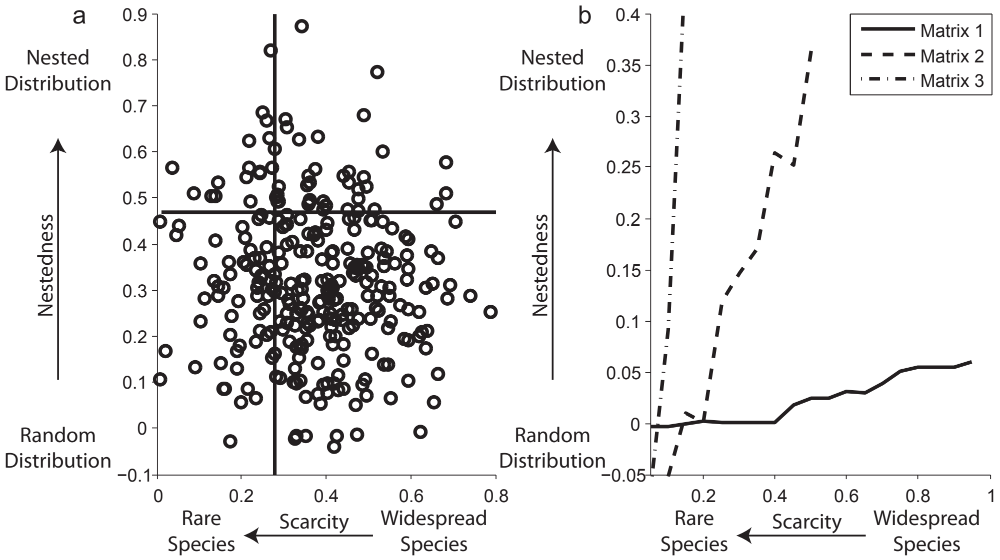
A drawback of these analyses, however, is that they focus on a single case-study, which has either limited geographical extent or a particular conservation focus (e.g., federally listed endangered species). This limits scope for general lessons to be drawn regarding the trade-off between conservation costs and biodiversity coverage. To begin to address this shortcoming, I analyse this trade-off over a range of biodiversity distributions. Specifically, I simulate data with varying levels of species nestedness and scarcity to generate a range of biodiversity distributions and then determine conservation costs and biodiversity coverage trade-off curves for each of the simulated datasets.

Both species nestedness and scarcity have profound implications for conservation biology. For example, if nestedness is strong, the few species present on less rich sites will be those that are found everywhere whereas uncommon species most in need of preservation will only occur on the most species rich sites (Patterson 1987). Wright et al. (1998) investigate the nestedness and scarcity among species assemblages covering a wide array of taxa from geographically diverse locations throughout the globe. They find that some level of nestedness is the norm. However, they also show that the level of nestedness and scarcity in species assemblages varies substantially from location to location (Box 6.1) with all combinations of high/low nestedness and high/low scarcity observed. Moreover, the choice of conservation targets will affect the distribution of species within a single assemblage. For example, focusing on species with differing levels scarcity can also alter the level of nestedness (Box 6.1).

To carry out the trade-off analyses, I develop a problem formulation specifically designed to calculate the trade-off curve between conservation costs and species coverage levels called the minimum cost coverage problem (MCCP). One problem formulation that has been used to map trade-off curves in previous studies is the maximum coverage problem (MCP) (Church and ReVelle 1974; Ando et al. 1998; Polasky et al. 2001; Gaston and Rodrigues 2003), the solution of which returns the maximum species richness that can be covered for a given budget. The MCP is not designed to calculate trade-off curves and its application results in the potential for solutions to be returned that are not minimum cost (Box 6.2). Alongside the trade-off analyses, I compare optimal solutions given by the MCP to those of the MCCP for various nestedness and scarcity structures in order to show the importance of testing competing solution methods against the full range of data characteristics to which those methods are likely to be applied (Pressey et al. 1999).

The remainder of the chapter is structured as follows: in Section 6.2, I define species nestedness and scarcity and describe how these parameters are varied to generate a range of species distributions; then, I detail the analytical method used to map the trade-off curves; finally, in Section 6.3, I present the results of the trade-off analyses and compare the cost-effectiveness of solutions given by the MCCP to those given by the MCP.

Box 6.1. Species distributions in natural populations



In their study of nestedness, Wright et al. (1998) use 291 presence/absence matrices made available by Atmar and Patterson (1995). In Panel a, I have shown the species nestedness and scarcity for each of these datasets. Here, I simply describe the levels of nestedness and scarcity. In Section 6.2, I outline how these parameters are quantified. As can be seen, natural populations exhibit a large variety of nestedness and scarcity structures. The horizontal and vertical lines indicate the parameter values analysed in Section 6.3 and are discussed in Section 6.2.

The choice of conservation focus will impact the distribution of species over sites within a single assemblage. To exemplify this point, in Panel b I have determined the level of species nestedness as the conservation objective changes from focusing on exceptionally rare species to include more widespread species. As the conservation targets change, the distribution of species can change in considerably different ways: including more widespread species among those identified for protection can have limited effect on nestedness (matrix 1); nestedness can change considerably but gradually as the conservation focus shifts (matrix 2); or nestedness can exhibit marked change with only a small change in the species identified for conservation (matrix 3). As I show in Section 6.3, the relationship between conservation outcomes and costs is crucially dependent on the nature of the distribution of species over sites and therefore on the choice of conservation objective.

6.2 Methods

6.2.1 Generation of biodiversity distributions and site costs

To capture the nestedness of presence/absence matrices, I use Wright and Reeves' (1992) C metric, where

$$C = \frac{N_c - E\{N_c\}}{\max\{N_c\} - E\{N_c\}}$$

N_c calculates the number of times that a species presence at a site correctly predicts its presence at richer sites, and sums these counts across species and sites. $E\{N_c\}$ is the expected value of N_c under the equiprobable species null hypothesis and $\max\{N_c\}$ is the value that N_c would take if the matrix were perfectly nested (see Wright and Reeves (1992) for further details). C varies from 0 to 1 where 0 indicates a random distribution of species over sites and 1 is a perfectly nested distribution[1]

Species scarcity is defined as the proportion of species presences over sites and it is calculated as the sum of species presences divided by the product of the number of species and sites. I will refer to this variable by S and it also varies from 0 to 1. When S is close to zero, the species presence/absence matrix is populated mainly by zeros, indicating a high level of scarcity. When S is close to one, the presence/absence matrix will consist mainly of ones and the matrix thus contains many widespread species.

As demonstrated in Box 6.1, all combinations of high/low nestedness and scarcity can occur. Therefore, random presence/absence matrices are generated that cover a large range of nestedness and scarcity values. This is achieved by independently varying either the level of nestedness or scarcity with the other parameter held constant at a default, “baseline” estimate. The baseline estimates are taken from field surveys of farmland avian species in the Peak District in the UK (Dallimer et al. 2009) ($C = 0.48$ and $S = 0.28$). Varying either C or S is then akin to generating species distributions along the horizontal or vertical lines shown in Panel a of Box 6.1. The number of species ($n = 81$) and sites ($n = 44$) are also held constant at the values from the Peak District data.

To construct random presence/absence matrices, I first created a perfectly nested matrix the column totals of which were set equal to those from the Peak District biodiversity data. Then, while maintaining the column totals, the disorder within the matrix was increased until the level of nestedness was as required. Maintaining column totals is necessary to ensure that matrices with different levels of nestedness contain the same levels of species rarity. To ensure column totals were held constant while increasing disorder, two columns were randomly chosen which either had the same number of incidences or whose number of

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[1] C can in fact take small negative values. This occurs when the distribution is less nested than would be expected with a random distribution. I do not consider such distributions in the presented analyses.

incidences differed by one. Then, from both of these columns a zero and one was randomly chosen and swapped. The C metric is sensitive to the ordering of species and sites and so it was also necessary to ensure that the matrix was maximally packed before calculating the level of nestedness.

A consequence of maintaining column totals was that the range of achievable nestedness values was limited. Nestedness is essentially a pattern in the presence/absence data. Constraining the column totals limits the ability to eliminate this pattern and produce a highly random species distribution. With the Peak District biodiversity data, I was only able to perturb the structure of the matrix such that $C \gtrsim 0.4$. Hence, in Section 6.3, results are presented for the nestedness values $C = \{0.4\ 0.65\ 0.9\}$.

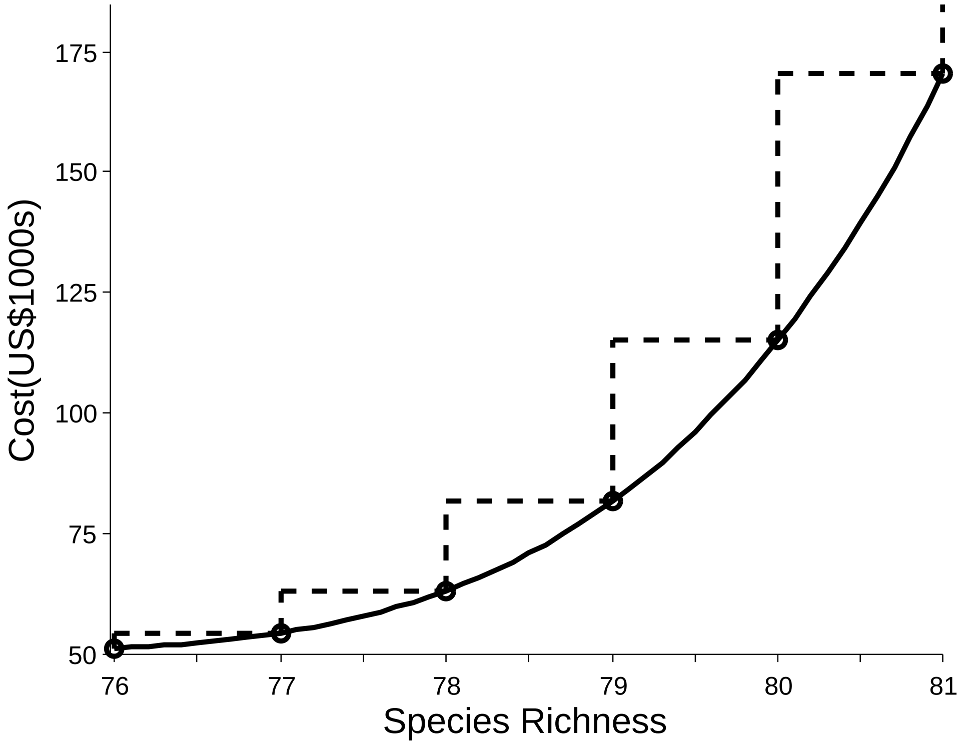
As one is altering the total number of species presences when varying the level of species scarcity, it is not possible to maintain the levels of species incidences. Therefore, I employed the simplest possible procedure when varying the level of species scarcity. Starting with a generic, perfectly nested matrix, which had a structure similar to a row reversed, upper triangular, binary matrix with the required level of scarcity, ones and zeros were randomly swapped until the level of nestedness was equal to that of the Peak District data. This allowed for presence/absence matrices to be produced with scarcity values anywhere between 0 and 1. Results are therefore presented for $S = \{0.2\ 0.4\ 0.6\}$ as these values conform to the majority of actual distributions shown in Panel a of Box 6.1.

Cost vectors were randomly generated using economic data from the same 44 properties on which the field surveys took place and was gathered at the same time (Acs et al. 2010). As described in earlier chapters, the measure of cost recorded in these data was the annual, per hectare rental value of the properties. The best-fit distribution to these rental values was a generalised β-distribution on the interval [5.5 124], with distribution parameters $\alpha = 0.66$ and $\beta = 0.91$. Having randomly drawn the per hectare rental values from this distribution, I multiplied them by the farm area. All costs are given in USD at 2006 values. No covariation structure was imposed between biodiversity and site cost. As the presence/absence data have high dimensionality, it is not clear how to define relevant patterns of biodiversity to cost covariation.

6.2.2 Calculation of the trade-off curves

One method that has been used in previous studies mapping the trade-off between conservation costs and species coverage levels is the MCP (Ando et al. 1998; Polasky et al. 2001). The objective of the MCP is to maximise species coverage subject to a budget constraint. The MCP is not designed to determine trade-off curves and using it to that end requires that the budget be systematically altered to determine the minimum cost solution at each level of richness.

Box 6.2. The difference between the MCP and the MCCP



The MCP and MCCP differ mainly in what they consider to be optimal solutions. In the figure above, minimum cost solutions (the circles lying on the curve) are shown for a variety of species coverage levels. The MCCP considers only such minimum cost solutions to be optimal. However, the MCP would consider any solution on the vertical lines above the minimum cost solution to be an optimal solution for that level of species coverage. For example, 80 and 81 species can be covered at a minimum cost of US$115,000 and $170,000 per annum, respectively. If the MCP was used with a budget constraint anywhere between these two figures, *any* solution that conserved 80 species would be considered optimal. To find the minimum cost solution that returned 80 species with the MCP, the budget would have to be systematically altered so as to perform an exhaustive search. Reformulating as the MCCP expedites this process and precludes the possibility that cost-ineffective solutions will be returned.

In this chapter, I define a linear programme, the MCCP, that is designed to map trade-off curves and thus overcomes some of the limitations of the MCP (Box 6.2). The MCCP is a minimum cost covering problem that bears similarities to the set covering problem (SCP) (Ando et al. 1998; Rodrigues et al. 2000; Moore et al. 2003). The objective of the SCP is to find the minimum cost set of sites that covers all species a given number of times (see Rodrigues et al. (2000) for the many different variants of the SCP). The MCCP finds the minimum cost set of sites that covers any specified level of richness (with the SCP one cannot find the minimum cost solution that covers 80% of species, for example. The MCCP formulation allows for such solutions to be found). By finding the minimum cost solution at each level of species richness, the MCCP can efficiently recover trade-off curves. The MCCP can be represented mathematically as

$$\min \sum_{j \in J} c_j x_j \tag{6.1}$$

subject to

$$\sum_{j \in N_i} x_j \geq y_i, \text{ for all } i \in I \tag{6.2}$$

$$\sum_{i \in I} y_i \geq s \tag{6.3}$$

where I is the set of species that can be covered, J is the set of candidate reserves and N_i is the subset of J that contains species i. The optimisation acts on the binary variables x_j and y_i. Should $x_j = 1$ then site j is selected for conservation and is 0 otherwise. If $y_i = 1$ then species i is present on at least one of the selected sites and is 0 otherwise. c_j is the cost associated with selecting site j and s is the required level of richness. Objective (6.1) minimises the cost of selected sites subject to the set of constraints (6.2), which ensure that a species is only counted as covered if it occurs on at least one selected site, and constraint (6.3), which ensures that at least the minimum level of species richness is covered.

Combining the approaches outlined in this and the preceding subsection allows for the analyses to be undertaken. First, random presence/absence matrices with a given nestedness and scarcity structure are generated along with random site cost vectors. Then the MCCP is applied to determine the minimum cost solution at each level of species coverage to recover trade-off curves. Repeating this process for various values of nestedness and scarcity reveals how trade-off curves change over a range of biodiversity distributions.

6.3 Results and discussion

6.3.1 Trade-off curves for varying biodiversity distributions

Fig. 6.1 displays the trade-off curves between conservation costs and species coverage levels for various species nestedness and scarcity structures. The results indicate that when the species distribution is less nested (lower values of C), the cost to cover a particular proportion of species is higher (Fig. 6.1a). For example, covering 90% of species costs

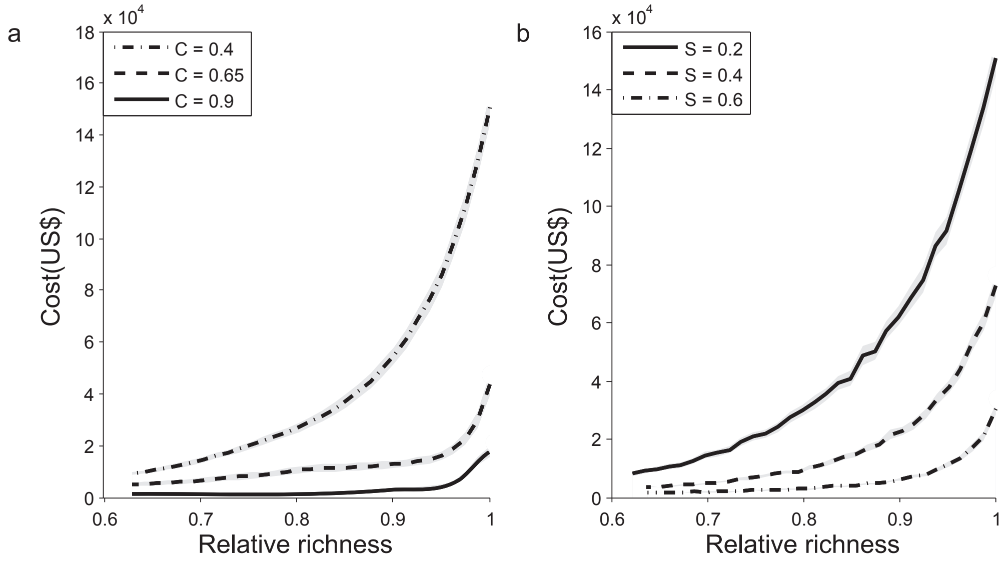


Figure 6.1: Trade-off curves between conservation costs and species coverage levels for differing values of species nestedness (Panel a), C, and scarcity (Panel b), S. All other parameters are at the baseline levels. 100 simulations were run in all analyses. Shown are the mean costs along with 95% bootstrap confidence bands. Species coverage is shown relative to the maximum possible coverage.

$54,700 per annum when $C = 0.4$. To cover the same level of richness when $C = 0.9$ would cost only $3,500 per annum. Similarly, the marginal increase in cost to cover increasing levels of species richness is higher when the distribution is less nested. For example, when $C = 0.4$ increasing coverage levels from 90% to 100% would increase costs by $95,600 per annum. When $C = 0.9$, the same increase in species coverage levels would cost an additional $14,200 per annum.

The relationship between conservation costs and levels of species coverage is also highly dependent on the scarcity of species over sites (Fig. 6.1b). When species are relatively scarce (lower values of S), the cost to cover a particular proportion of species is considerably higher than when the conservation targets include widespread species. For example, covering 90% of species costs approximately $62,100 per annum when $S = 0.2$ whereas when $S = 0.6$, this cost falls to $6,500 per annum. The marginal increase in cost to cover higher levels of species richness is also higher when a greater proportion of the conservation target species are scarce. For example, when $S = 0.2$ an additional $88,800 per annum is required to increase coverage levels from 90% to 100%. When $S = 0.6$, the same increase in species coverage increases costs by $23,900 per annum.

In all cases, the trade-off curves are convex indicating that the marginal increase in cost to cover larger numbers of species is increasing. Therefore, large increases in biodiversity coverage result in small increases in cost when the proportion of species covered is low. On the other hand, when a relatively large proportion of species are covered, increasing coverage further results in a large cost increase. One factor that leads to this increasing marginal cost for increasing coverage levels is the complementary nature of sites when the objective is to maximise species richness (Margules et al. 1988; Pressey and Nicholls 1989; Church et al. 1996; Justus and Sarkar 2002). Site complementarity means that the biodiversity value gained by adding a site to a reserve network is the number of species present on the site that are not already covered. Thus, fewer species will be gained as more sites are added given the high probability that widespread species are already represented within the network. This results in a convex trade-off curve. Economic factors will also lead to a convex trade-off curve given the tendency to prioritise low cost sites first.

6.3.2 Difference in cost-effectiveness of MCCP and MCP

I now investigate the difference in cost of the MCCP and MCP solutions. Reference to Box 6.2 shows that the upper bound on the difference between the cost of MCP and MCCP solutions for particular species coverage levels is determined by the minimum cost solution that contains one more species. I determine how this worst-case solution returnable by the MCP compares to the minimum cost solution given by the MCCP for a variety of species nestedness and scarcity structures.

For visual clarity, in Fig. 6.2 I document the relative cost difference between the MCP and MCCP when species coverage is 95% and 99%. As can be seen in Fig. 6.2a and b, as species coverage levels increase, the potential cost-ineffectiveness of the MCP rises in all cases.

The upward trend in the cost-ineffectiveness of the MCP compared to the MCCP as species coverage increases is a function of the shape of the trade-off curves. At relatively low levels of species coverage, the difference in the minimum cost of solutions that differ in richness by one species is small (Box 6.2, Fig. 6.1). Therefore, at these levels of richness, small increases in cost result in solutions with higher richness. However, as species coverage levels increase, the gradient of the trade-off curves increases, indicating an increasing gap in the cost of solutions that differ in richness by one species. Consequently, at high levels of species coverage, solutions covering a larger range of costs could be considered optimal by the MCP (Box 6.2), which increases the scope for inefficiencies in the MCP solution.

The nature of species nestedness and scarcity is crucial in determining the magnitude of the potential cost-ineffectiveness of the MCP. When the species distribution is less nested (low values of C), the worst-case solutions of the MCP are significantly more expensive than the minimum cost solutions compared to a highly nested distribution. When covering

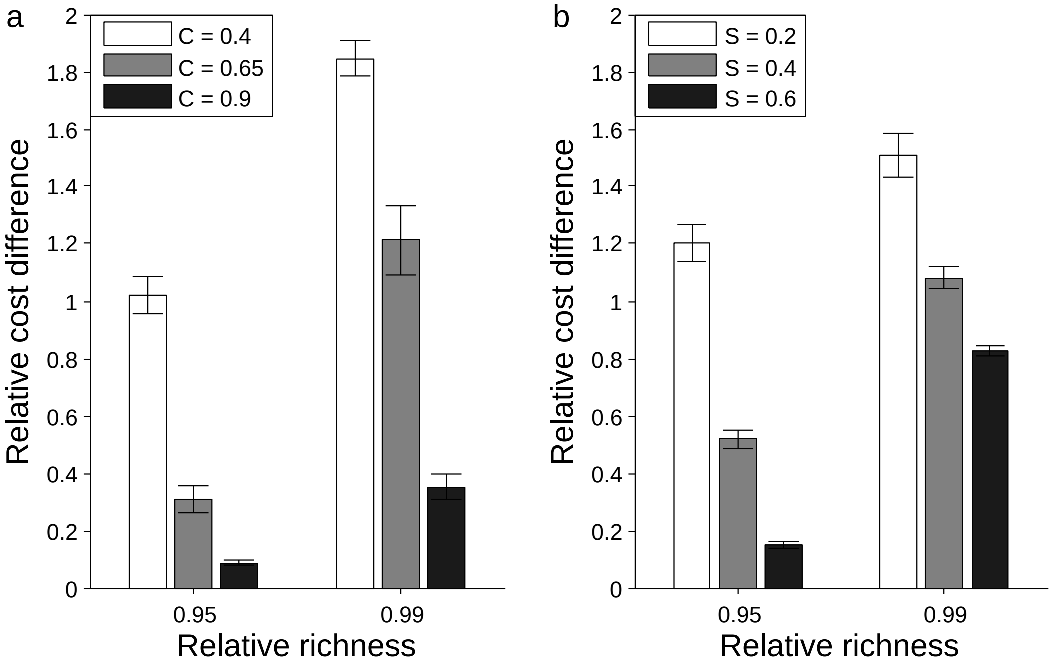


Figure 6.2: The difference between the minimum cost solutions given by the MCCP and the upper-bound on the cost of MCP solutions relative to mean site cost. Results are shown for species coverage equal to 95% and 99% for various levels of nestedness (Panel a), C, and scarcity (Panel b), S. All other parameters are at the baseline levels. 100 simulations were run in all analyses. Shown are the mean figures along with 95% bootstrap confidence bars.

99% of species with nestedness at $C = 0.4$, the worst-case optimal site selections suggested by the MCP are on average 1.85 times the mean site cost more expensive than the minimum cost solution. This figure falls to 0.35 when $C = 0.9$. When conservation targets are highly scarce (low values of S), the potential inefficiency of the MCP is significantly greater than when the targets include many widespread species. With coverage at 99% and $S = 0.2$, the worst-case site selections suggested by the MCP are on average 1.5 times the mean site cost more expensive than the minimum cost solution compared to 0.83 when $S = 0.6$.

These results emphasise the importance of testing the performance of reserve selection algorithms and related optimisation approaches against the full range of biodiversity and cost distribution data (Panel a, Box 6.1) to which they are likely to be applied, the approach I have taken in these analyses.

6.3.3 Conclusions

In this chapter I analysed how characteristics of the distribution of species over sites (nestedness and scarcity) affected the trade-off between conservation costs and species coverage levels. To undertake the analyses, I constructed a linear programme designed to reveal this trade-off. Perhaps the most important result from the analyses is that the relationship between conservation costs and conservation outcomes crucially depends on the distribution of species over sites. When making planning decisions, conservation organisations must be aware of how choosing different conservation targets (e.g., particularly rare species only versus all species) affect the degree of protection available for a given level of investment.

Chapter 7

The suitability of short or long conservation contracts under ecological and socio-economic uncertainty

7.1 Introduction

The limitations of conservation funding taken alongside the critical condition of the Earth's biodiversity means that it is necessary for conservation groups and agencies to wisely choose the locations on which they seek to secure land rights. From the 1980s onward, a large body of literature has grown to address this site-selection problem (e.g., Cocks and Baird 1989; Pressey et al. 1997; Ando et al. 1998; Moilanen et al. 2004; Dreschler et al. 2009). One drawback of early studies was that the solution was assumed to be static, in the sense that all desired sites could be secured instantaneously. However, the process through which land is protected is inherently dynamic. This temporal aspect of the acquisition process is caused by several factors. For instance, in any time period funding is rarely sufficient to allow conservation groups to protect all sites of conservation concern (Costello and Polasky 2004). Moreover, when and where sites will be available for conservation cannot be known in advance (Meir et al. 2004).

The dynamic nature of the acquisition process introduces several sources of uncertainty. Although the conservation group may be able to determine current site availability, future availability cannot be known with certainty. Further, conservation groups may be able to assess the current ecological value of sites but forces such as climate change or changing local land use can alter the future value of sites. Therefore, in order to make decisions that are robust to future change, conservation groups must be aware of these sources of uncertainty and factor them into conservation planning.

Models that deal with the dynamics of the acquisition process have recently been developed. For example, Costello and Polasky (2004), Meir et al. (2004), Strange et al. (2006) and Sabbadin et al. (2007) investigate site acquisitions where reserve networks are created through time and where there is a risk that unreserved sites will be lost to development. In these studies, acquired sites are considered protected in perpetuity with no further loss or gain in species richness. Similarly, if development occurs on unreserved sites it is considered irreversible with all species extirpated.

The assumptions that underlie the models developed in studies such as those cited above are particular to specific types of conservation investments or particular types of development. First, the assumption of perpetual protection is applicable if sites are acquired through fee-simple acquisitions or in well monitored and defended conservation easements (Fishburn et al. 2009a; Armsworth and Sanchirico 2008; Rissman and Butsic 2011). However, as described in Chapter 1, widespread use has also been made of fixed term contracts in agri-environment schemes (AES) and payment for environmental services schemes (PES). Second, an assumption that conservation values are lost completely and in perpetuity may be appropriate if sites become densely urbanised where this is incompatible with particular ecological functions. However, there are many situations in which conservation investment may not be possible for a temporary period only. For example, such situations arise when a landowner commits to a particular crop rotation or invests sunk costs in a particular technology that precludes conservation measures for a limited time. Furthermore, the availability status of sites will be affected by commodity prices. Such prices determine the forgone profits that will be lost should a landowner agree to undertake particular conservation measures. Thus, at times of high prices, a conservation group may not be willing or able to compensate for the high payment level demanded by the landowner. However, as prices fall, the site may become available to the conservation group once again. Such factors can therefore result in sites of high biodiversity value that may be unavailable for conservation investment in one time period becoming available for investment again in some future time period.

Fixed length contracts with private landowners that are used in conservation vary in their duration. Table 7.1 gives examples that highlight the large variety of contract lengths employed in AES and PES around the world. This variation in contract duration (with perpetuity being the logical extreme) begs the question of what the relative advantages and disadvantages are of short versus long contracts and whether there is an optimal contract duration or optimal distribution of contract durations for a particular set of circumstances. In this chapter, I examine the role that different kinds of uncertainty play in determining the relative advantages of short and long contracts. Specifically, I examine uncertainty over

Table 7.1: Variety of contract durations employed in AES and PES around the world

Conservation Scheme	Location	Length(s) (years)
Pagos de Servicios Ambientales[1]	Costa Rica	5,10 or 15
Environmental Stewardship Scheme[2]	UK	5 or 10
Conservation Reserve Program[3]	USA	between 10 and 15
Forest Carbon Sequestration[4]	Vietnam	20
Social Forestry Programme[5]	Indonesia	25 and extendable
Conservation International Forest Conservation[6]	Guyana	30

[1] Zbinden and Lee (2005), [2] DEFRA (2008), [3] Khanna and Ando (2009)
[4] Ha et al. (2008), [5] Pender et al. (2008), [6] Conservation International (2000)

future site availability and over future ecological condition. Elsewhere, Ando and Chen (2011) found that the optimal length of a conservation contract depends on a trade-off between the ecological effects of the length of the contract and the number of landowners enrolled. Longer contracts lead to greater ecological benefits but diminish the incentive for landowners to accept the contract. When analysing the optimal length of an agricultural carbon contract, Gulati and Vercammen (2005) found that the greater the reduction in farming intensity demanded by the contract, the shorter the period the landowner will accept conservation. However, neither of these initial studies parses the role of different kinds of uncertainty, nor do they consider the possibility of a portfolio of contract durations.

In general, contracts that are shorter are less expensive than those that are over a longer timescale, meaning more can be secured for a given level of investment. Short term contracts also spread the risk of wasting investments if protected sites themselves deteriorate in condition. Moreover, given that a larger number of short contracts would be available compared to longer agreements, they can increase competition among landowners for conservation investment, which can be advantageous in some programme designs (Latacz-Lohmann and Van der Hamsvoort 1997). However, short term contracts come with the attendant risk that landowners will decline to re-enrol on completion of the contract term. Empirical research has shown that the fate of land protected in temporary contracts is uncertain. For example, in the context of the Conservation Reserve Program (CRP), Skaggs et al. (1994) surveyed 811 participants in New Mexico regarding their post-CRP land use plans. Of those surveyed, 21% expressed the intention to go back to farming, 40% planned to re-enrol, with the remaining 39% undecided. Long term contracts reduce this “end of contract problem” (Whitby 2000) and ensure priority sites stay in conservation management for longer. In this chapter, I look at how differing levels of uncertainty affect the trade-off between these different pros and cons of different contract lengths. In Sections 7.2 and 7.3, I address these questions optimally on a small descriptive problem using stochastic dynamic programming

(SDP) (Mangel and Clark 1988; Possingham et al. 1993; Costello and Polasky 2004) then in Section 7.4, I address the same set of questions on a larger problem using heuristic methods.

7.2 Illustrative model

To formalise these ideas, I develop a model that captures some of the structure of fixed term contracts under uncertainty. In this first, illustrative model, the conservation group has only $J = 3$ sites of conservation interest to choose between. Each has been assessed and given an ecological score. A stochastic process affects the ecological score, which is discretised over 2 states. These states can be thought of as the site being in a good or degraded ecological condition. Transitions between states are dependent on whether or not a site is contracted, with sites not receiving conservation investment at greater risk of degradation. There is also a stochastic process affecting site availability. Unavailable sites can become available for conservation contracting and vice versa, independent of current conservation investment.

In the models, the short contracts last one time step while the long contracts last either two or three. I therefore model a situation in which, for example, a conservation group can use contracts of 5 years and 10 or 15 years.

To reduce the size of the state space, I assume a simple cost structure. Specifically, I assume that the overall budget available to the conservation group limits the investment choice to at most two concurrent short term contracts or one long term contract per time period. With this cost structure, the conservation group must pay landowners a high premium (or face much higher transaction costs) to secure a long-term agreement. I further assume that the budget is binding with no carry-over to future time periods. Variable budgets, carry-over and more complex cost structures can be included by adding more state variables.

Finally, I assume that conservation benefits accrue through time and that benefits are only derived on contracted sites. These assumptions would describe some ecosystem services such as access rights for recreation that accompany many public agency conservation contracts (e.g., in the Environmental Stewardship Scheme in the UK, landowners can receive payments for allowing public access to their land (Nature England 2010)) or water quality improvements over baseline levels (e.g., through payments to limit the use of fertilisers on properties (Cooper et al. 2009)). I relax some of these assumptions later and also review alternative formulations in the Discussion (Section 7.5).

7.2.1 Markovian structure and optimal rewards

The system is assumed to have a Markovian transition structure. The observable realisations are denoted by $\mathbf{x}_t$, where $\mathbf{x}_t \in \mathbf{S}_t$, the state space, and $|\mathbf{S}_t| = 2^{2J} + 2^{2J}J/2$. Each state describes a potential configuration of contract, ecological condition and availability status of the sites at time t. Single time step transitions between states are given by

$$\mathbf{x}_{t+1} = \mathbf{T}(\mathbf{x}_t, a_t)$$

where $\mathbf{T}$ defines the transition matrix. This gives the probability of moving from one realisation of $\mathbf{x}$ to another in one time step, dependent on making a particular investment decision, a_t.

The conservation group receives the reward R for taking investment a while the system is in state $\mathbf{x}$, plus the sum of expected future rewards. Thus, starting from time t, the overall reward, V, can be defined in recursive form as

$$V(A_t \mid \mathbf{x}_t) = R(a_t \mid \mathbf{x}_t) + \sum_{\mathbf{x}_{t+1}} \mathbf{T}(\mathbf{x}_{t+1} \mid \mathbf{x}_t, a_t) V(A_{t+1} \mid \mathbf{x}_{t+1})$$

This recursive equation allows for the determination of an optimal investment policy, A_t^*, that optimises V through use of the Bellman's equation, which in this case becomes

$$V^*(\mathbf{x}_t) = \max_{a_t} \left\{ R(a_t \mid \mathbf{x}_t) + \sum_{\mathbf{x}_{t+1}} \mathbf{T}(\mathbf{x}_{t+1} \mid \mathbf{x}_t, a_t) V^*(\mathbf{x}_{t+1}) \right\} \tag{7.1}$$

7.2.2 Curse of dimensionality

In Section 7.3, I use SDP to determine the sequence of decisions that results in the optimal value of Eqn. (7.1). Several studies, such as Richards et al. (1999), Westphal et al. (2003), Tenhumberg et al. (2004), Wilson et al. (2006) and McDonald-Madden et al. (2008) have applied this approach to deal with dynamics of ecological management decisions.

SDP offers vast improvements in solution times over brute force, forward solution methods. Nonetheless, the methodology suffers from what is known as the "curse of dimensionality" (Bellman 1957), whereby the state space rises exponentially as the number of sites increases linearly. For example, given the model structure, if I had chosen $J = 5$ the state space would have had 73,728 states compared to just 160 states with $J = 3$.

To overcome the limitation produced by this "curse", heuristic solution techniques have been developed for the site selection problem. While these methods cannot guarantee optimal results they have been shown to produce results that are close to optimal on relatively small selection sets (Costello and Polasky 2004; Vanderkam et al. 2007) and have been used in much larger selection problems (Nalle et al. 2002a; Turner and Wilcove 2006). In Section 7.4, I develop an informed greedy heuristic algorithm to test the results obtained from the optimal solutions on a selection problem containing 20 sites. To solve this problem optimally would have meant searching a state space with approximately 1.2×10^{13} elements.

7.2.3 Assumptions and analyses

As noted, solving site selection problems optimally requires consideration of a limited number of sites. Moreover, given that there are three stochastic processes in the model (availability, ecological conditions with investment and ecological conditions without investment), the parameter space has the capacity to be large. For example, for the initial three-site example, there are $3J = 9$ parameters. In order to manage this issue, I reduce the parameter space by a series of assumptions. In the analyses of Section 5.3, I initially analyse the situation where sites have homogenous probabilities of changing availability and ecological status and then introduce heterogeneity to test if this affects results. I assume further that conservation investment increases the probability that a site will begin the next time period in a good ecological condition by 0.2 over not contracting.

To allow the analyses to focus solely on the uncertainty of future site availability and ecological condition, the ecological value of sites in the objective function is assumed to be equal (i.e. when testing the effect of spatial ecological heterogeneity, I focus on heterogeneity in probabilities governing ecological state changes and not in human values attached to those state changes). I assume that a degraded site is worth 50% of a site in a good ecological condition and I document the rewards relative to the maximum that could be achieved through continuous investment on two good sites. This metric therefore compares the reward the conservation group can achieve given uncertainty relative to the situation of no uncertainty. I document the optimal contract selections from the stationary part of the optimal path for differing levels of uncertainty and I calculate the optimal rewards from Eqn. (7.1). The optimal rewards available to the conservation group are also calculated when they are constrained to use only short contracts or only long contracts. By comparing these values to the optimal solution of the full model that allows for both long and short contracts, I can investigate if any advantage is gained by having the availability of a portfolio of different contract durations.

7.3 Optimal solutions of illustrative model

While a full exploration is given to the effects on the choice of conservation contracts that different types of uncertainty play, Fig. 7.1 gives a brief overview of the main findings. The models show that when there is a zero or small probability that sites will be unavailable in the subsequent contracting period, short contracts are the optimal choice. As the probability that sites will become unavailable increases, there is a greater reliance on long contracts alongside a portfolio of different contract durations. Uncertainty in ecological conditions has no bearing on contract selection without the attendant risk that sites will become unavailable. When both sources of uncertainty are combined, uncertainty in site

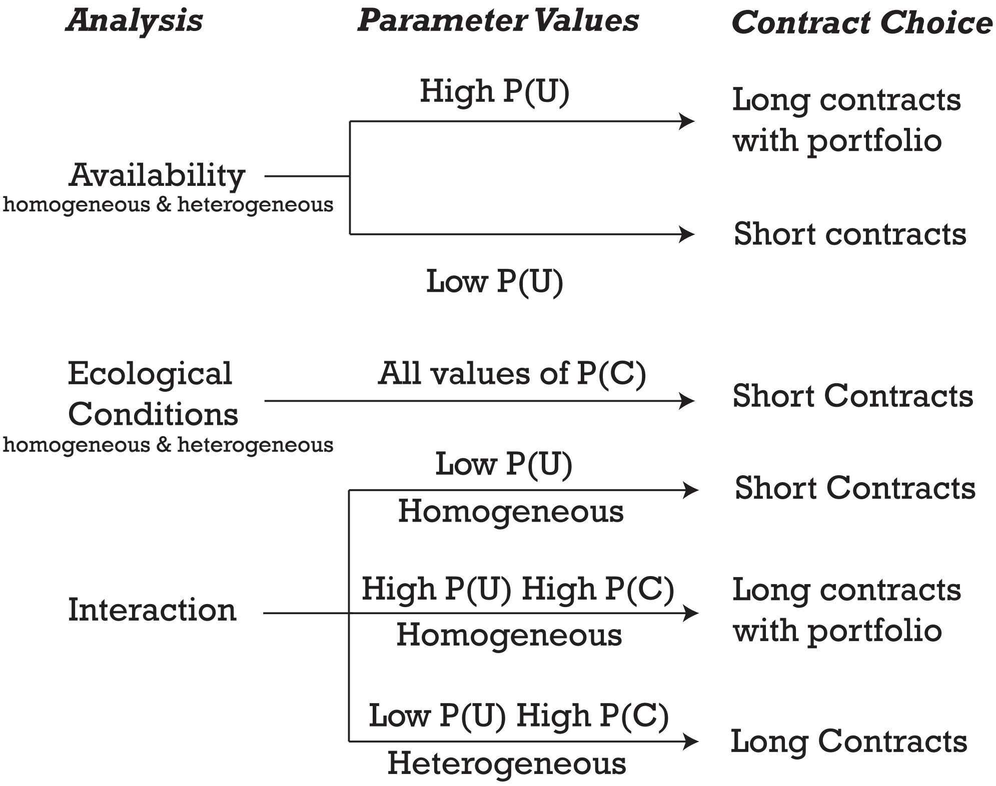


Figure 7.1: Summary of the headline results presented in the chapter. The analyses form three groups. Availability indicates that there is only uncertainty through time in the availability status of sites. Ecological Conditions indicates that there is only uncertainty through time in the ecological condition status of sites. Interaction indicates that there is uncertainty through time in both site availability and ecological condition. P(U) is the probability that sites will be unavailable at the beginning of the next time period. P(C) is the probability that sites will begin the next time period in a good ecological condition with conservation investment.

availability continues to be the most influential factor in setting the choice between short and long contracts. However, heterogeneity across sites in the probability of a change in ecological condition results in increasing use of long contracts. I elaborate on each of these findings below.

7.3.1 Uncertainty in site availability in isolation

In the first instance, I look at the role of uncertainty in site availability in isolation and assume that there is no uncertainty through time in the ecological condition of the sites (Fig. 7.1, first row). Therefore, for the optimal investment decisions, I analyse how these are affected by the initial state of the site ecological conditions as well as the probability that

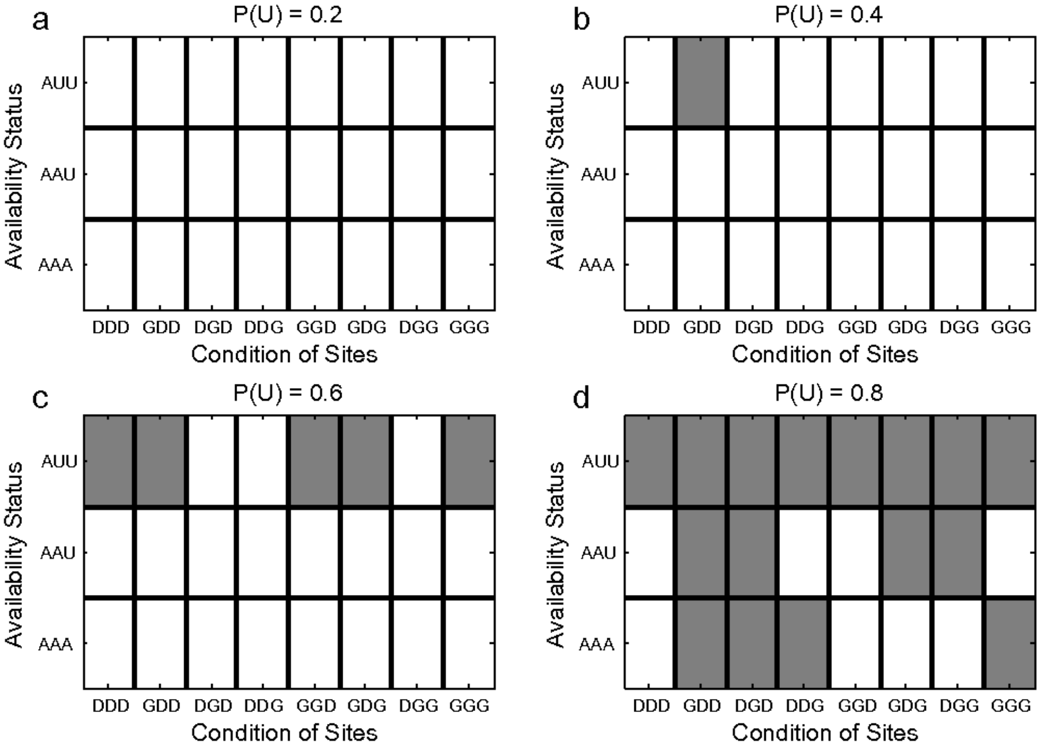


Figure 7.2: Short and long contracts chosen for differing probabilities that sites will be unavailable in the future, P(U). There is no variability through time in the ecological condition of sites.

sites will become unavailable for conservation investment. For the rewards, I illustrate the optimal expected values with the initial conditions of all sites in good ecological condition and available.

For the first analysis, I assume that all sites have the same probability of becoming unavailable in the next time period. Fig. 7.2 documents the state specific contract selections for various values of P(U). To simplify the presentation of results, the figure illustrates the investment decisions taken halfway through the time path when the optimal management strategy is stationary. The horizontal axis gives the ecological conditions of the three sites and the vertical axis shows their availability status. Should a particular configuration of site states be marked with white then short contracts should be chosen. If the sites states are marked with grey then a long contract should be chosen. For example, Panel d in Fig. 7.2 illustrates the optimal strategy on the stationary part of the path when the probability that sites will be unavailable is 0.8. The rectangle marked with GDD on the horizontal and AUU on the vertical indicates that site 1 is available and in good ecological condition whereas sites 2 and 3 are unavailable and in a degraded condition. This rectangle is shaded grey to indicate that a long contract should be chosen on site 1 in this particular combination of

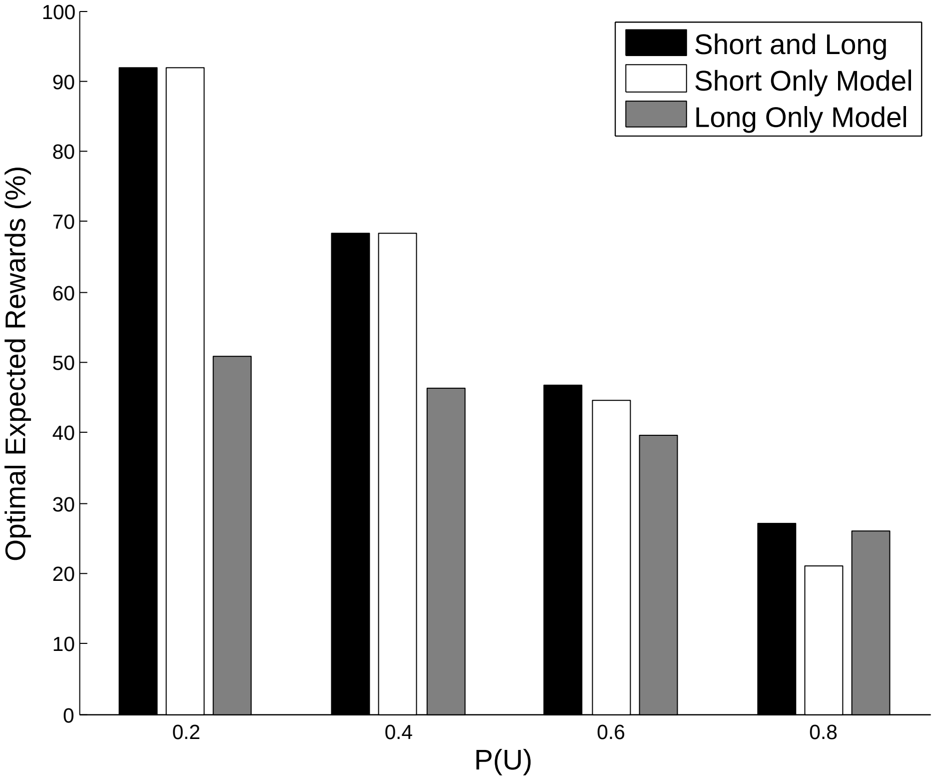


Figure 7.3: Optimal rewards when there is only uncertainty in future site availability. The results of three models are presented: the optimal rewards from the full model in which the conservation group can select short or long contracts on a given time step; and the optimal rewards when the group is constrained to choose either short or long contracts throughout all time periods. The optimal rewards are given relative to the rewards that could be gained with continuous investment on two sites in good ecological condition.

states of the three sites.

The results show that when there is a low probability that sites will be unavailable for conservation contracting in the next time period, short contracts are the optimal choice (Panel a). By assumption, in a given time period two contracts running concurrently offer the conservation group twice the benefits per time step of a single long contract. This cost premium required to secure a long-term agreement is not worth paying when there is little chance that sites will become unavailable in the future. However, when the probability that sites will become unavailable is high, the balance of costs and benefits reverses and long contracts are favoured in the vast majority of states (Panel d).

Turning to the optimal rewards, Fig. 7.3 highlights that as the probability that sites will become unavailable increases, the relative effectiveness of long contracts increases. When

P(U) is greater than 0.65, a long contracts only strategy gains more rewards than a short contracts only strategy. However, when the probability that sites will be unavailable in the future is moderate to high (P(U) ≥ 0.6), a portfolio of short and long contracts through time offers even greater rewards to the conservation group than being limited to either type.

Adding heterogeneity to the probabilities governing site availability in isolation (one site has a greater or lower likelihood of becoming unavailable next time period than the other two) does not change the overall results: long contracts are used only when all sites have a high probability of becoming unavailable.

7.3.2 Uncertainty in site ecological conditions in isolation

I now investigate the role of uncertainty in ecological conditions in isolation (Fig. 7.1, second row). There is no uncertainty in availability status. Similarly to Section 7.3.1, I investigate the optimal investment decisions for all initial conditions of availability status, assuming that all sites are in a good ecological condition in the first time period. For the optimal expected rewards, I again illustrate the case where all sites are initially available and in a good condition.

Irrespective of the particular values of site probabilities of changing ecological conditions or whether these probabilities are homogeneous or heterogeneous over sites, short contracts are preferable in all cases where a choice over which sites to protect is possible. When the likelihood that sites will begin the next time period in a good ecological condition increases, the optimal rewards from all models increase. However, the increase remains in constant proportion across the models, with the short contracts only model remaining as effective as the full model that allows for either short or long contracts, and the long contracts only model remaining half as effective as either of the other models.

7.3.3 Uncertainty in both site availability and ecological conditions

Homogeneity in probabilities over sites

Next, I analyse how the interaction of the uncertainty over future availability and future ecological condition affects the optimal investment decisions and the optimal expected rewards. Fig. 7.4 documents the state specific contract selections for various values of P(U) and P(C), where these values are homogeneous over sites.

The analysis of Section 7.3.1 demonstrated that long contracts become the optimal choice when the probability that sites will be unavailable in the next time period becomes high. Fig. 7.4 shows that when there is uncertainty in both site availability and ecological condition, the probability that sites will become unavailable still needs to be high in order for long contracts to become the best choice. When the probability that sites will be unavailable in

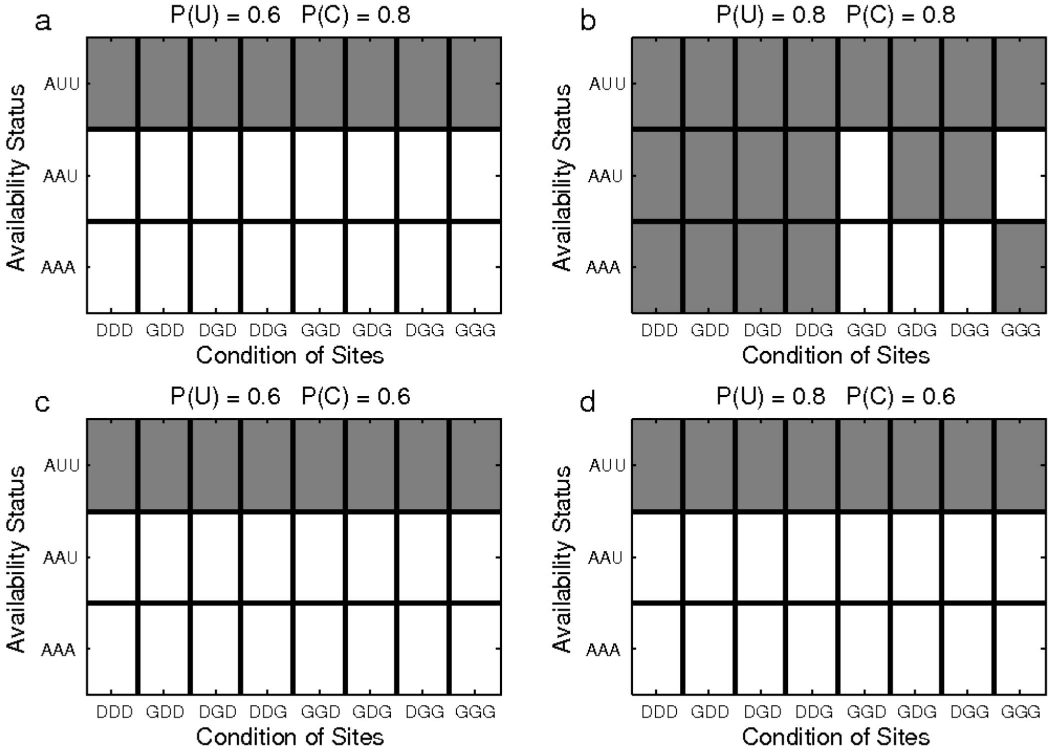


Figure 7.4: Short and long contracts chosen for differing probabilities that sites will be unavailable in the future, P(U), and that with conservation investment they will be in a good ecological condition, P(C). I assume that the probability of beginning the next time period in a good condition decreases by 0.2 in the absence of investment. Refer to Section 7.3.1 for a description of the structure of the figure.

the next time period is less than 0.6, short contract are always selected irrespective of the availability status of sites and the likelihood that sites will begin the next time period in a good ecological condition given conservation investment (P(C)). When the probability that sites will become unavailable rises above this threshold value, long contracts are always chosen when only one site is available (Fig. 7.4, top row in each panel). The rationale for this is straightforward. If the conservation group can protect only one site then it is advantageous to protect it for longer when the likelihood of it becoming unavailable in the next time period is high and there is a high probability that other sites will remain unavailable. For long contracts to become widespread over site states, it is also required that the probability that sites will begin the next time period in a good ecological condition with conservation investment is high (P(C) ≥ 0.8). In this situation, long contracts are used in almost every configuration of site state (Fig. 7.4b).

When there is uncertainty through time in both site availability and ecological condition, the expected optimal rewards from conservation investment are influenced by both sources

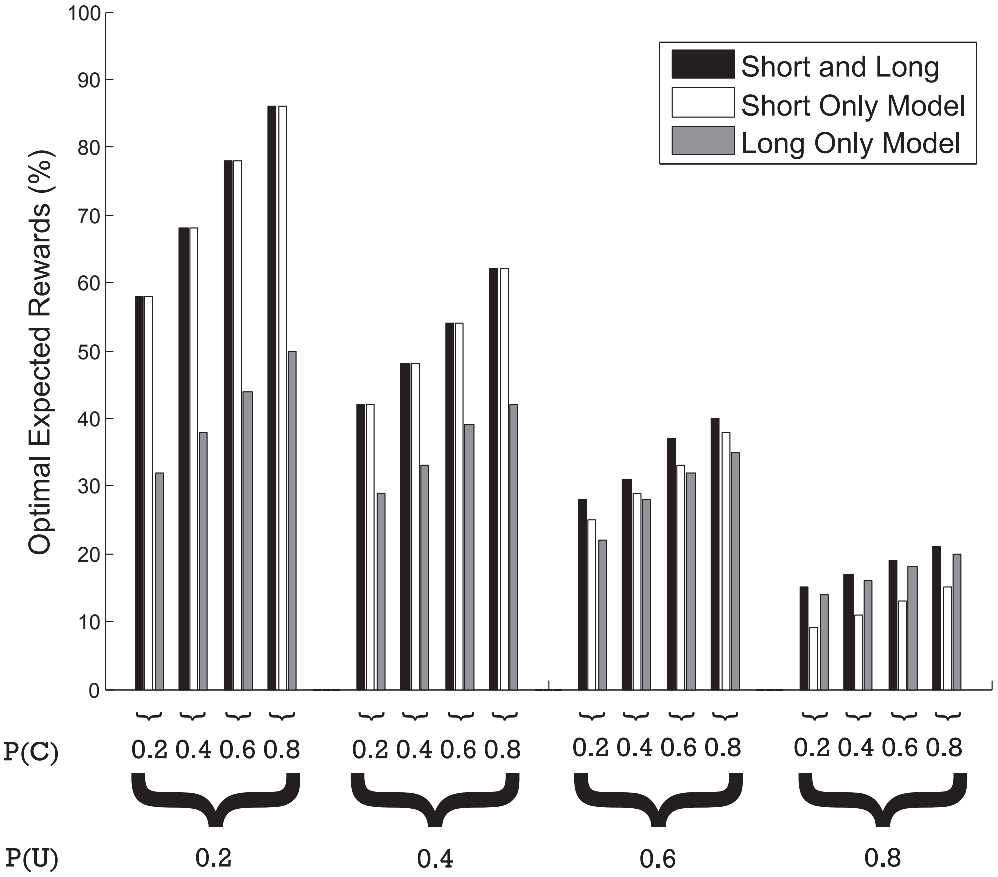


Figure 7.5: Optimal rewards for various combinations of probabilities that sites will be unavailable in the future, P(U), and that with conservation investment they will be in a good ecological condition, P(C). The results of three models are presented: the optimal rewards from the full model in which the conservation group can select either short or long contracts on a given time step; and the optimal rewards when the group is constrained to choose either a short or a long contract.

of uncertainty. Fig. 7.5 demonstrates that when both these sources of uncertainty interact, the optimal expected rewards from conservation investment are affected to a greater degree than when either source acts in isolation. For example, Fig. 7.3 shows that when P(U) is equal to 0.2 but there is no uncertainty in future ecological conditions, optimal rewards from the short and long contracts model and the short only model equals approximately 92% of the maximum available reward. The rewards from the the long only model in this case is around 51% of this maximum. Comparing this with P(U) equal to 0.2 in Fig. 7.5 shows that the optimal rewards are lower and span a wide range of values when uncertainty in ecological conditions also applies. Furthermore, Fig. 7.5 shows that uncertainty over future

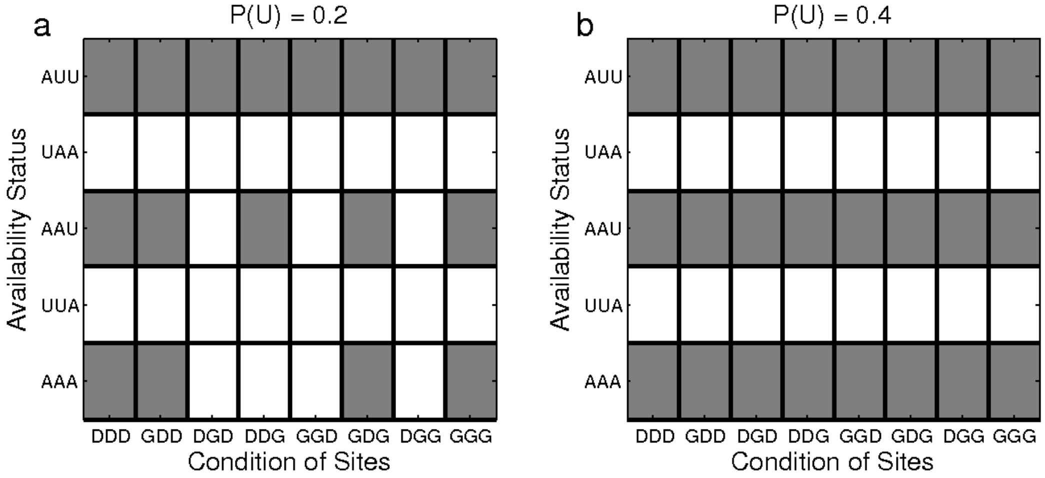


Figure 7.6: Short and long contracts chosen for differing probabilities that sites will be unavailable in the future, P(U). Sites are now heterogeneous in the probability of beginning the next time period in a good ecological condition. With investment, this probability for site 1 is 0.8 and for sites 2 and 3 is 0.4. I again assume that the probability of beginning the next time period in a good condition decreases by 0.2 in the absence of investment. Refer to Section 7.3.1 for a description of the structure of the figure.

site availability has a much greater impact on the rewards from conservation investment than the uncertainty regarding ecological conditions. As in Section 7.3.1, when the probability that sites will be unavailable in the next time period is moderate to high ($P(U) \geq 0.6$), it is again optimal for the conservation group to strategically employ contracts of differing lengths depending on the current state.

Heterogeneity in probabilities over sites

I now introduce heterogeneity in the probability of ecological change over sites while retaining homogeneity in the site probabilities of changing availability status (Fig. 7.1 third row, third branch). Specifically, I assume that when in a conservation contract, site 1 has a 0.8 probability of being in a good ecological condition in the next time period. Sites 2 and 3 have a significantly lower probability of beginning the next time period in a good ecological condition. With conservation investment, this probability is 0.4. The assumption of a 0.2 decrease in the probability of being in a good ecological condition in the next time period without investment remains for all sites.

Fig. 7.6a shows that even with a small probability that sites will become unavailable (P(U) = 0.2), taking a long contract on the site with better likely conservation outcomes (site 1) can be the optimal choice. When site 1 is the only site available, it is always optimal to secure it in a long contract (Fig. 7.6a, top row marked AUU), despite the low probability that sites 2 and 3 will remain unavailable in the next time period. Moreover, even when there exists the possibility of selecting two short contracts, selecting a single long contract on site 1 is the best choice for some states (Fig. 7.6a, middle and bottom rows marked AAU and AAA, respectively). When the likelihood that sites will be unavailable in the next time period increases to 40%, in any situation where site 1 is available, the optimal investment decision is to take this site in a long contract (Fig. 7.6b, top, middle and bottom rows). Thus, when any site offers markedly better conservation outcomes and there is any risk that it may become unavailable, securing it for a longer period can be more beneficial to the conservation group than accumulating greater instantaneous rewards with shorter contracts.

Finally, adding heterogeneity to the probabilities governing site availability does not change the overall results: a long contract is chosen on the site with a higher probability of positive ecological outcomes even when there is a small probability that this site will become unavailable and irrespective of the probabilities that other sites will be available

7.4 Contract selections on a larger set of sites

I now analyse whether contract selection is affected by the number of sites of conservation interest. To achieve this, I use a heuristic algorithm to undertake similar analyses to those of Section 7.3 on a larger selection problem. The algorithm makes contract selection decisions by assessing the value of all possible current investments plus the expected value of potential investments in the next two time periods. The investment(s) that give the highest expected value over the three time steps are chosen. In this way, the conservation group is choosing the best contract in light of the immediate uncertainty regarding site availability and ecological conditions. However, unlike the optimal model, they do not consider the whole time path over which decisions are being made.

The conservation group now has 20 sites to choose among. As with the optimal model, all sites contribute equally to the objective function when in a particular ecological state, with degraded sites 50% as valuable to the conservation group as a site in a good ecological condition. The same budget constraint applies, allowing for two concurrent short contracts or one long contract. In each run of the simulation, I calculate the number of long contracts chosen after 50 time steps and then repeat this process for 1000 simulations.

7.4.1 The effect of site availability on contract length

I undertake three analyses to investigate the effect of site availability on contract selection. I first look at the situation where there is homogeneity over sites in the probability that they will be unavailable in the next time period. Then I look at the situation where the

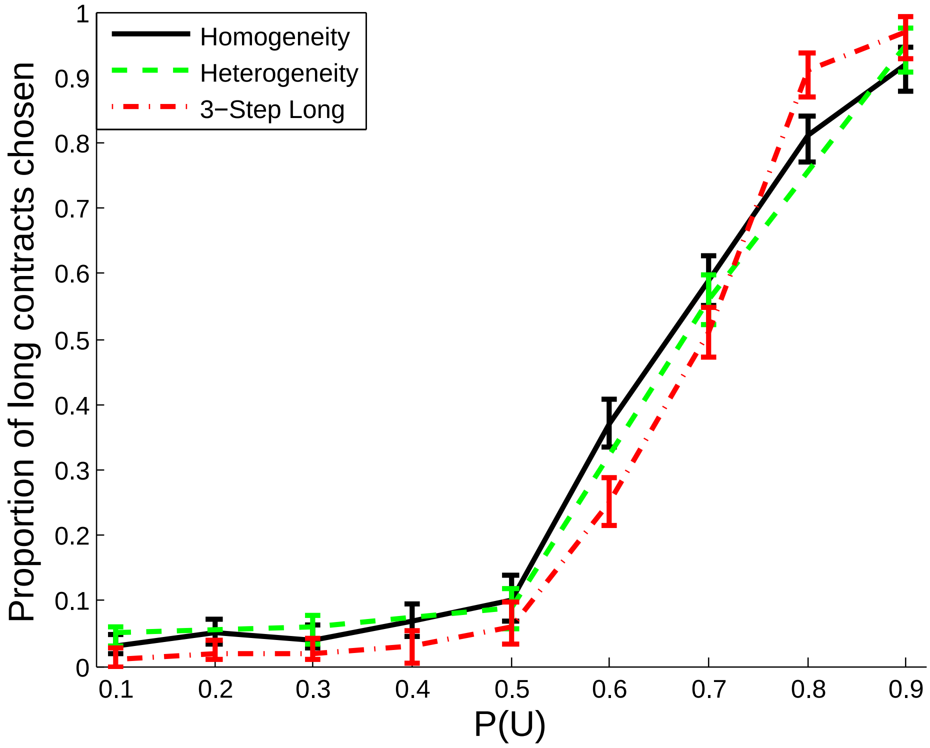


Figure 7.7: The proportion of long contracts chosen as the probability of sites being unavailable in the next time period, P(U), increases for the three analyses outlined in Section 7.4.1. In all three, 1000 simulations were run at each probability and the proportion of long contracts chosen was calculated. Presented is the mean number with error bars set at the 5th and 95th percentiles. It should be noted that the different number of error bars between the homogeneous and heterogeneous analyses is a function of the heterogeneous analysis being over probabilities with a range of 0.2 whereas in the homogeneous analyses the probability was increased by 0.1 from 0.1 to 0.9.

probability is heterogeneous over sites but is constrained to lie in an interval with a range of 0.2. I examine cases where the likelihood that sites will become unavailable can be described as low, moderately low, moderate, moderately high and high. Finally, I combine homogeneity with an extended long contract to test if a greater difference in the durations of the available contracts offers any advantages.

Fig. 7.7 displays the results and shows that an increasing probability of sites being unavailable in the next time period leads to an increased use of long contracts. In contrast to the optimal model (Fig. 7.2), however, even when this probability is very low, some long contracts are chosen. Also highlighted is the fact that homogeneity and heterogeneity over sites in the uncertainty relating to site availability results in effectively the same outcomes.

When long contracts last three times steps rather than two, they may be marginally less beneficial when the probability that sites will be unavailable is low and marginally more beneficial when this probability is high, but differences are small.

7.4.2 The effect of ecological condition on contract selection

With the optimal model, short contracts were always chosen when there was only uncertainty in site ecological conditions. I find exactly the same result when the number of candidate sites is increased to 20. Similar analyses were undertaken to those in Section 7.4.1. I undertook two analyses with site homogeneity. In the first, long contracts lasted two time steps and in the second, the long contract was extended to three time steps. In both these analyses, I retained the assumption that with conservation investment there was a 0.2 increase in the probability that sites would begin the next time period in a good ecological condition compared with no investment. I also undertook an analysis where sites were heterogeneous in the probabilities of positive conservation outcomes. To introduce heterogeneity, I assigned a random number of sites to be “good” conservation sites. With conservation investment, these sites had a probability of beginning the next time period in a good ecological condition (P(C)) ranging from 0.3 to 0.9, again with the assumption of a 0.2 decrease in this probability in the absence of investment. The “bad” conservation sites all had the probability of beginning the next time period in a good ecological condition of 0.2 with investment and without that probability was 0.1. Irrespective of the analysis undertaken or the given probabilities, short contracts were the optimal choice as I found in Section 7.3.1.

7.4.3 Contract selections when there is uncertainty in both future site availability and ecological condition and the influence of site heterogeneity

In the three-site optimal model, when there was uncertainty regarding both future site availability and ecological condition, contract selection was primarily set by site availability (Fig. 7.4). Adding heterogeneity in the uncertainty of future site conditions in Section 7.3.3, however, resulted in a larger impact on contract selection from this source of uncertainty. In this section, I test these results on the larger selection problem. Heterogeneity in ecological conditions was introduced as in Section 7.4.2 and I retain homogeneity over sites in the likelihood of future availability.

Both the homogeneity and heterogeneity analyses documented in Fig. 7.8 indicate that the primary driver setting contract length is the uncertainty regarding future site availability. However, when the probability that sites will be unavailable is moderate to high (P(U) ≥

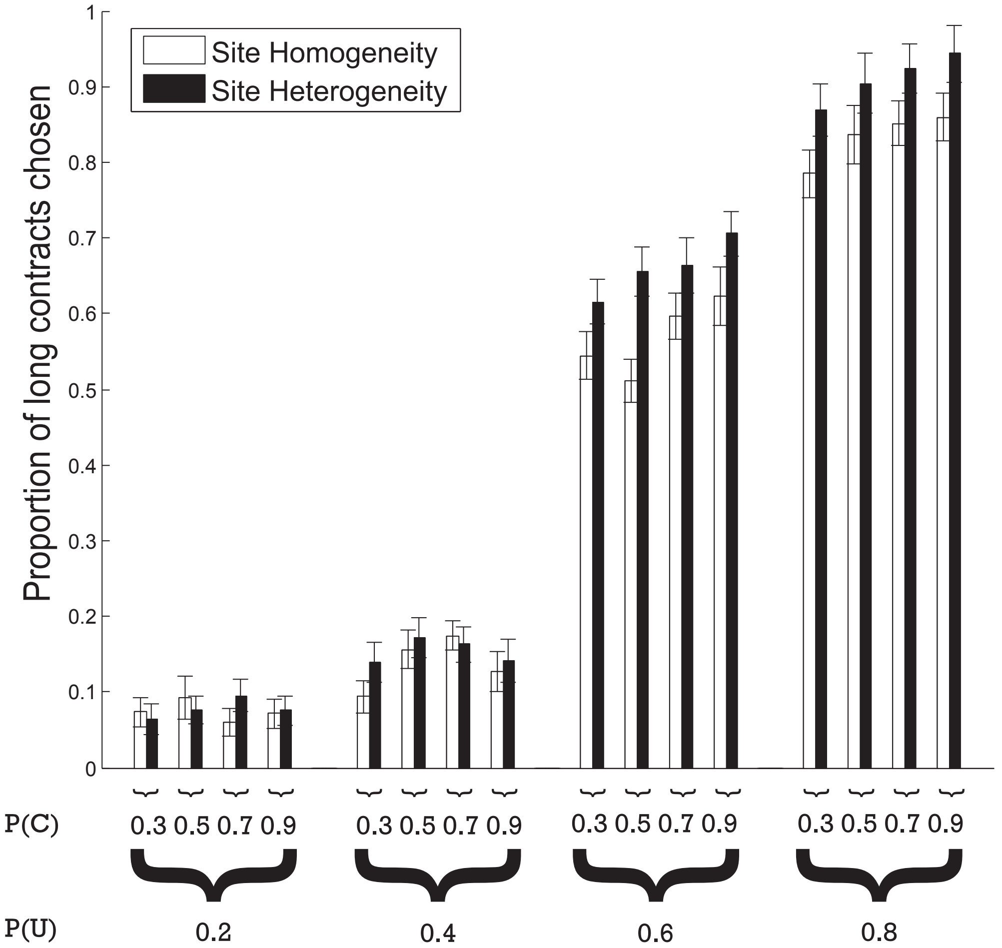


Figure 7.8: Proportion of long contracts chosen for differing probabilities that sites will be unavailable in the next time period, P(U), and that with conservation investment they will begin the next time period in a good ecological condition, P(C), for the homogeneous and heterogeneous analyses outlined in Section 7.4.3. Presented is the mean figure with error bars set at the 5th and 95th percentiles.

0.6), heterogeneity over sites in the probabilities determining likely future ecological status results in an increase in the use of long contracts compared to when these probabilities are homogeneous. In contrast to the optimal model (Fig. 7.6), heterogeneity in ecological conditions only affects contract selection when the likelihood that sites would be unavailable in the future is moderate to high. The underlying reason for this is likely due to differing number of sites between the optimal and heuristic models. When there are 20 sites of conservation interest and the probability that they will be unavailable in the next time period is small, the conservation group can have confidence that at least some of the "good" sites will be remain available for conservation. This leads to the primary use of two short term contracts. In the optimal model, given that there was only one good site, it made sense to secure that site in a long contract even when there was even a very small possibility that the site could become unavailable. Nonetheless, both models are consistent in the conclusion that heterogeneity over sites in the uncertainty regarding future ecological conditions produces an increase in the use of long contracts as opposed to when sites are homogeneous in this source of uncertainty.

7.5 Discussion

In this chapter, I used both optimal and heuristic models to examine the suitability of either short or long conservation contracts in light of ecological and socio-economic uncertainty. The analyses consistently pointed to several important results (Fig.7.1). First, the most important factor setting the choice of conservation contract was the likely availability of sites in the next time period. Second, contract choice was relatively insensitive to uncertainty over future ecological conditions. And finally, when the probability that sites would be unavailable was moderate to high, a portfolio through time of short and long contracts was more advantageous than relying solely on either type.

The fact that uncertainty in site availability was more influential in setting contract choice than uncertainty in ecological conditions to some extent reinforces the conclusions of other recent works that show that socio-economic factors can be as or more important in determining conservation priorities than ecological factors (Ando et al. 1998; Naidoo and Iwamura 2007; Bode et al. 2008; Perhans et al. 2008). Nonetheless, both the three-site optimal and twenty-site heuristic analyses indicated that heterogeneity in the probabilities governing changes in ecological conditions can lead to a greater reliance on longer conservation contracts. Thus, conservation programmes that are highly targeted and recognise ecological heterogeneity among candidate sites, such as Higher Level Stewardship in the Environmental Stewardship Scheme (ESS) in England, should involve longer contracts. In contrast, schemes that are focused on conserving the overall landscape and that are less resolved in how they assess properties, such as the Entry Level Stewardship in the ESS, could rely on shorter contracts. However, the findings suggest that the imperative for using long contracts is reduced when there are a large number of sites of good ecological quality that can act as substitutes for conservation investment.

A portfolio of different contract types was seen to offer better rewards than when only one type of contract was available (Figs. 7.3 & 7.5). This strategy is commonly overlooked in practice yet the ability to use contracts of differing lengths can offer advantages to conservation groups when decisions are made to be robust to future change. However, employing a large number of different contract types would have higher transaction costs (Box 1.1), which would limit the number of different contracts durations used. Nonetheless, some flexibility in the nature of conservation contracts might lead to better outcomes than rigidly sticking to one type irrespective of the situations faced.

This study is the first attempt to analyses how different sources of uncertainty set the choice between fixed term conservation contracts of differing lengths. While it may be the first in the conservation planning literature, a number of studies of labour markets have sought to understand how economic uncertainty sets contract length. Theoretical works by Grey (1978), Canzeroni (1980) and Dye (1985) and later empirical studies by Vromen (1989) and Murphy (1992) found that high levels of uncertainty should lead to shorter contracts. The proposition that underlies this phenomenon is simple. An increase in economic uncertainty increases the probability that unforeseen contingencies will arise during the course of the contract. This leads to the negotiating parties favouring a short contract (Grey 1978; Murray 2000). In the models presented in this chapter, the conservation group is least able to predict future availability when P(U) = 0.5. In these conditions, there is general indifference between long and short conservation contracts, in contrast to these studies in labour economics. Instead, the advantages of either short or long contracts are most pronounced for low or high values of P(U), when the conservation group can predict future availability of sites with more confidence.

7.5.1 Assumptions and extensions

In the models developed in this chapter, the manner in which the conservation group derives benefits from the sites on which it invests rests on several assumptions. First, I assumed that conservation benefits accumulate through time in the objective function. This assumption will be appropriate for describing some conservation programmes, such as those seeking to sustain a regular supply of some ecosystem service (e.g., public access to land on contracted sites or site specific measures to improve water quality). However, other conservation programmes may be better represented by objectives based on the condition of the system at the terminal time only, such as those focused on ensuring long-term persistence of a rare species. Furthermore, I assumed that benefits accumulate additively in space, or in other words that the benefit derived from contracting on multiple sites is simply the sum of the benefits on the individual sites. However, should there be a non-linear spatial accumulation of benefits, the imperative for different contract durations would be affected. If benefits were to accrue super- or subadditively in space, the desirability of more short contracts versus fewer longer agreements would increase or decrease respectively. Also, I incorporated spatial heterogeneity in sites only through the probabilities regarding future status. An extension of the model would be to incorporate spatial heterogeneity in the ecological value of sites in the objective function. Finally, I assumed that no benefit was derived from non-contracted sites. This assumption is consistent with many conservation studies and programmes. Nonetheless, a number of conservation planning studies have shown that assuming that non-contracted sites can support some ecological functions can have an impact on planning decisions and conservation outcomes (e.g., Polasky et al. 2005; Armsworth et al. 2006).

The cost structure of the model determines the number of contracts of differing durations that the conservation group can apply. The central assumption that defined the cost structure was that the annual rental rate on a two time period contract (10 years) was twice that of a one time period contract (5 years). This is clearly a gross simplification and was made to reduce the size of the state space, while also recognising that longer contracts are likely to be more costly than shorter ones. However, with this assumption, the conservation group must pay a very high price premium and/or face much higher transaction costs to secure long term agreements. An obvious alternative assumption would have been to assume an equal annual rental payment and small transaction costs making annual costs of short and long term agreements similar. Overall, this would make longer agreements more desirable than predicted by the model. However, both assumptions seem somewhat exaggerated and it is likely that the real world would fall somewhere in between. Either cost structure also assumes that annual monitoring and enforcement costs do not vary greatly between short versus long term contracts. More complex cost structure could be included in the model through, for example, contract re-enrolment bonuses and increasing payments schedules for time spent contracted. Both of these cost structures incentivise landowners to keep their land under conservation management for longer and have been shown to lead to better conservation outcomes and increased economic efficiency (Gulati and Vercammen 2006).

Other important model assumptions concern the transition probabilities governing changes of ecological condition. I assumed these probabilities were constant through time and adopted a Markovian structure. With this structure the assumption is made that only the current condition of a site and the transition probability that specifies the likely change in site condition influence the future condition of the site. However, other aspects of the model reflect the gradual improvements of ecological condition on sites through time with conservation investment. For example, for $P(C) \geq 0.5$, the difference in the likelihood of a site remaining in a good ecological condition with conservation investment compared to the likelihood of a site remaining in a good ecological condition without conservation investment has a unimodal relationship with the number of consecutive periods for which it is enrolled. This model feature is discussed further in the appendix (Section 7.6).

Finally, I focused throughout on fixed-term contracts that last 1 and 2 or 3 time periods. This situation conforms to one in which the conservation group uses contracts of 5 or 10/15 years. However, many conservation contracts with private landowners in the USA and elsewhere are made in perpetuity through conservation easements. The model could easily be extended to capture these alternative forms of investment. Moreover, by including the use of permanent acquisitions alongside fixed term agreements, the model could be used to determine if a portfolio of these different contract structures can result in higher conservation benefits than the uniform application of one type or another.

7.5.2 Conclusions

The use of conservation contracts of different lengths should be in part determined by both socio-economic and ecological factors. The results of this chapter show that uncertainty in the configuration of tomorrow's landscape should influence the length of contract that a conservation group chooses to employ. The future availability of sites must be considered carefully when making conservation plans as this factor can have a dramatic effect on the benefit of contracts with different durations. Uncertainty in ecological conditions has a smaller effect. But uncertainty over future ecological conditions can influence contract duration when acting alongside uncertainty in availability and with heterogeneity over sites. Conservation groups should also consider the merits of flexibility in contracting arrangements through a portfolio of different contract types so that they can shape their investments to maximise conservation outcomes. Given the paucity of conservation resources, it is essential that the large sums spent on fixed term contracts are used in the most efficient ways. This efficiency can only be achieved through an appreciation of an uncertain future.

7.6 Appendix

In the models, I assume that conservation benefits accumulate additively in space with constant probabilities that sites will change status through time. I also assumed a constant difference between the probabilities of positive ecological outcomes with and without conservation investment of 0.2. These factors lead to a particular form for the accumulated benefit gained by a site remaining in a conservation contract over T contract periods. For example, the probability of any particular site remaining in a good ecological condition throughout all the T periods with conservation investment is $P(C)^T$; while the probability of any site remaining in a good ecological condition throughout all the T periods without conservation investment is $(P(C) - 0.2)^T$. In this context, for $P(C) > 0.5$, the difference in the likelihood of a site remaining in a good ecological condition with conservation investment compared to the likelihood of a site remaining in a good ecological condition without conservation investment has a unimodal relationship with T (Fig. 7.9b). Over the initial periods, there is an increasing likelihood of a site remaining in a good ecological condition throughout all the periods under contract when comparing investment with no investment.

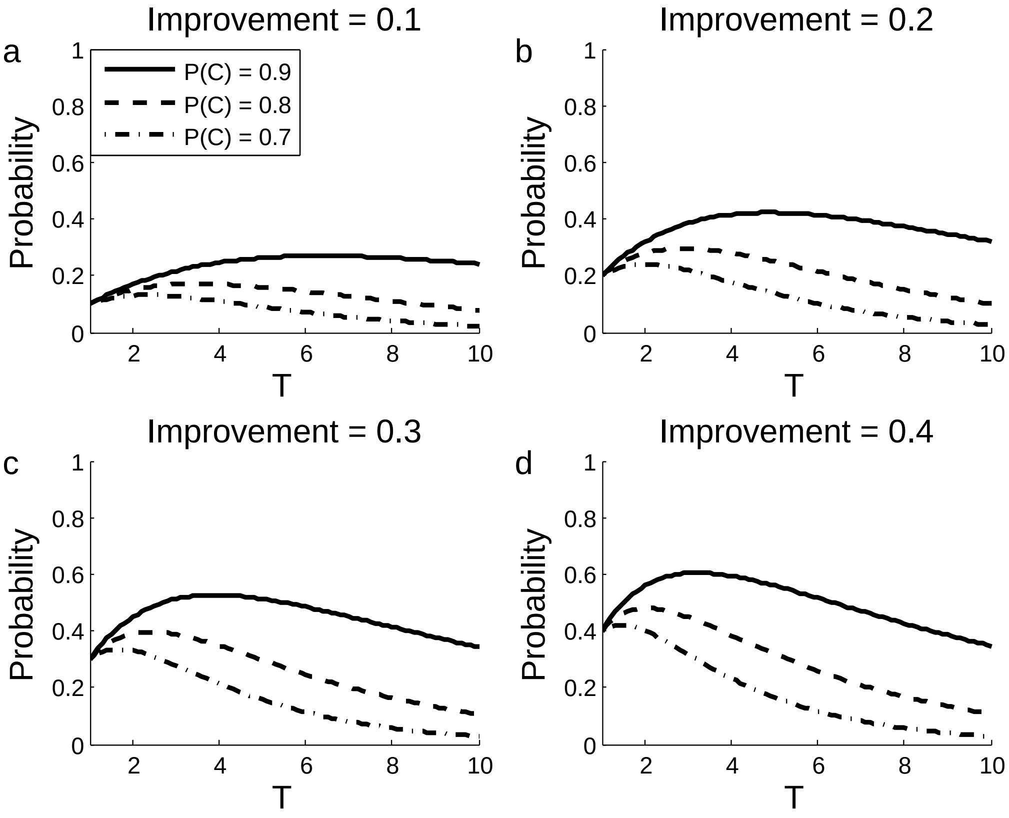


Figure 7.9: The accumulated benefit from remaining in a conservation contract for T periods compared to no investment over the same periods. *Improvement* indicates the difference in likelihood of being in a good ecological condition next time period with investment compared to no investment.

Critical in this context is the assumption that investment decreases the probability of degradation by 0.2. Should the impact of investment be higher, the benefit of conservation would be higher. Conversely, if conservation investment was less likely to lead to positive outcomes, the advantage gained by investment would be smaller (Fig. 7.9a,c,d).

Chapter 8

General conclusions

In this thesis, I have analysed several issues pertaining to the optimal design of voluntary conservation investments. The primary aims of these analyses has been to deepen our understanding of aspects of conservation contract design, critically analyse the implications of common assumptions in conservation planning, and to address unresolved areas in the formulation of conservation contracts. From these investigations several important lessons can be drawn.

Disregarding complexity can be counterproductive

Many of the analyses in this thesis have considered how simplifications in the formulation of conservation planning problems can affect outcomes. Chapters 2 to 4 evaluated the validity of the common simplifying assumption that conservation can be secured at a cost equal to a landowner's willingness-to-accept (WTA). In analyses that ranged from game theoretic investigations of the consequences of landowner competition to the simulation of discriminatory-price reverse auctions, the results consistently demonstrated that applying this assumption may significantly underestimate the cost of conservation. As a consequence, conservation science may be inadvertently overstating that which can be achieved through conservation measures, leading to a cycle of optimism followed by disenchantment as the benefits from investments fail to be realised. I feel that this is dangerous for conservation. Conservation initiatives depend on the general public for support. If conservation is seen to be failing, the willingness of the public to continue to fund conservation projects and the likelihood of governments being pressured to consider conservation science in policy deliberations may diminish. To contrast with the best-case scenario presented in most conservation planning studies, in these chapters I determined the worst-case outcome in which the cost of conservation was equal to the conservation group's willingness-to-pay. Recognising that conservation costs are complex and span a range of potential values allows for an investigation of the full spectrum of possible outcomes from conservation investments. I believe that a greater focus on analyses of these types would result in realistic expectations of what conservation can achieve for the levels of funding it receives. Presenting only the best-case scenario weakens the case for more funding and diminishes confidence in conservationists.

The role of complexity was central to the investigations of Chapter 5. While substantial research has been undertaken to incorporate complex biological aspects of conservation planning into analyses, commensurate focus has not been paid to the economics of conservation (Naidoo et al. 2006), leading to such factors often being incorporated simplistically. In Chapter 5, I investigated how complexities in the cost structure of conservation investments affect the optimal spatial distribution of reserves. I compared these results to those in which a very simple cost structure was applied, one that is utilised almost ubiquitously in conservation planning studies, and which assumes that the cost to secure multiple sites is simply the sum of costs of the individual sites. The results from this chapter showed that the application of a simple cost structure can result in reserve designs that are markedly different from those produced when realistic aspects of conservation costs are recognised. Within conservation there is increasing attention on the economics of planning problems. However, important issues, such as the nature and consequences of complex cost structures, remain neglected. Given the perilous state of biodiversity and the shortfalls in funding, efforts to ensure that biologically and economically realistic models of conservation investments are constructed should be increased. It is only through analyses that tackle the inherent complexity of conservation that effective and efficient plans will be devised.

A uniting principle behind many of the alternative, more complex cost structures that were investigated in Chapter 5 was that the cost of securing a site for conservation could only be known in light of which other sites were also selected. Such interdependencies were evident in the economies of scale in management costs problem, for example. There, the cost to manage a reserve depended upon whether or not it would be located adjacent to another reserve. The cost structure that results from landowners leveraging for higher payments in Chapter 3 similarly displays cost interdependencies. In these analyses, the final cost of a site (landowner's WTA + surplus) was shown to depend on which other sites were selected for conservation. These results clearly highlight that the cost of conservation is not simple or fixed, that it cannot be known a priori, and that it can depend on complex interactions across potential reserve sites.

The complexity that results from finite length conservation contracts has also been largely ignored by conservation research. Many site selection analyses have assumed that when a site is selected for conservation, it simply remains under conservation management in perpetuity with no future loss or gain in conservation value. However, as described in Chapter 1, finite length contracts between landowners and conservation organisations have been widely applied globally and receive enormous amounts of funding. To begin to investigate the optimal length of conservation contracts, in Chapter 7 I constructed a model through which I was able to determine the circumstances in which short, long or a portfolio of different contract lengths was the best choice. Unsurprisingly, the results from that chapter indicated that the simple application of a single contract length irrespective of the situation faced is rarely optimal. To the best of my knowledge there are only two other studies that deal with issues relating to conservation contract length, one of which is only tangentially related to biodiversity conservation (Gulati and Vercammen 2006; Ando and Chen 2011). As with the results from Chapter 7, both of these studies indicate that the choice of contract length can play a fundamental role in determining conservation outcomes. These issues must be fully explored so that vital conservation funding is not wasted through injudiciously designed contracts. More importantly, conservation science must, at a minimum, analyse all the major facets of conservation investments and not allow a prevailing analytical paradigm to bias the nature of research.

Formulations must account for uncertainty

The role of uncertainty featured in several of the analyses presented in the thesis. The basis of the chapters that dealt with landowners' leverage potential was that conservation groups cannot know a landowner's WTA with certainty. Starting from this point of uncertainty, I investigated the consequences for conservation, as discussed above. Similarly, when determining the optimal length of conservation contracts, as in Chapter 7, one immediately faces several sources of uncertainty, such as the future availability of sites for conservation and their ecological condition. The results from these four chapters indicate that when uncertainty is factored into analyses, conservation plans and/or outcomes can differ markedly from those suggested from a formulation that assumes that conservation practitioners have complete information. Given that this latter situation is almost never realised, grappling with these and other sources of uncertainty and understanding what they mean for the application of conservation must be a priority for conservation science.

Variation in the socio-economic aspects of conservation can be more important than variation in indicators of biodiversity

As conservation science has begun to recognise the need to account for the socio-economic components of conservation alongside the biological aspects, the question arises: does variation in socio-economic or biological factors have greater impact on conservation outcomes. As mentioned in Chapter 1, several studies that have considered this issue have concluded that variation in socio-economic factors has greater impact on conservation (Ando et al. 1998; Naidoo and Iwamura 2007; Perhans et al. 2008; Murdoch et al. 2010). For example, Bode et al. (2008) find that conservation funding allocations are more sensitive to variation in site cost and threat levels than to variation in biodiversity. Results from this thesis reinforce findings such as these. In Chapter 7, the choice of conservation contract length was generally insensitive to variation in the ecological condition of sites yet dependent critically on variation in their availability status. Similarly, Chapter 2 indicated that a landowner's leverage potential is more sensitive to variation in the distribution of landowners' WTA than to variation in the distribution of biodiversity levels over sites. The incorporation into research of all the dimensions of conservation planning problems requires a truly interdisciplinary approach. Although this can be difficult to foster, the value of such collaborations is that they can offer insights into the problems of conservation biology that would otherwise remain hidden. The results from my thesis further stress the need for research into all important aspects of conservation.

Conservation objectives can impact economic as well as biological outcomes

It is perhaps obvious that a plan designed for the conservation of species richness will result in different biological outcomes than a plan designed to increase the abundance of a single, rare species. What is less obvious, however, is how differences in conservation objectives can affect the economics of voluntary conservation investments. This issue has been tackled in several of the chapters in this thesis and the results indicate that different conservation objectives can produce considerably different outcomes. The starkest difference emerged in the analyses that compared objectives that incorporate site complementarity and those that do not. Complementarity in this context means that the conservation value of a site depends on the conservation value of other conserved sites. This is the case when focused on species richness as the conservation value of a site is the number of present species not present on other reserves. In Chapters 2 and 3, I found that landowners' leverage potential could be considerably higher when the objective recognised complementarity as opposed to simple additive objectives, such as maximising species occurrences. Chapter 5 showed that the spatial distribution of conservation reserves could also be affected by the nature of the conservation objective. There, complementary benefits acted to spread reserves over the landscape to a greater degree than additive benefits despite economies from clustering. Complementary benefits differ from additive benefits in how they translate into site value. When focused on species richness as opposed to species occurrences, those sites that contain rare species or distinct habitats are valued substantially more by conservation groups. This greater desirability of such sites results in landowners having greater ability to demand higher payments and for sites to be spread to capture samples of those distinct habitats even when clustering is desirable. The conclusion from these results is that conservation objectives should not be decided on the basis of biology alone. I do not advocate a particular objective, or class of objectives, over any other. Rather, that when formulating objectives, the economic implications of any choice should be considered alongside what it will mean for biodiversity.

Site selection algorithms should be fully understood and applied with caution

One of the successes of systemic conservation planning has been the development of computer programmes that apply theoretically derived principles to reserve design problems. These "off-the-shelf" programmes require limited user understanding of the underlying mathematical techniques. This is undoubtedly a positive aspect of conservation as it allows scientists and practitioners with little mathematical training to undertake rigorous research/devise sound conservation plans and for esoteric ideas from the literature to impact on-the-ground conservation. However, the results from Chapter 6 call for a degree of caution. In that chapter it was shown that the use of the maximum coverage problem, a problem formulation that has had widespread use in conservation planning studies, could return solutions that were cost-ineffective. This ineffectiveness could be overcome if one was aware of the limitations of the formulation and knew that sequentially altering the level of the budget until the minimum cost solution with the maximum richness was found was the method to employ to ensure cost-effectiveness. Therefore, those who design conservation planning tools must ensure that the algorithms they implement are appropriate for all problems to which they are likely to be applied. Moreover, a focus on highlighting the limitations of particular methods within conservation would give researchers a better chance of choosing appropriate tools.

More fine-grain data is needed for conservation planning analyses

Finally, in all but one of the analysis chapters, the models I developed were parameterised by biodiversity and socio-economic data from a farm system in the Peak District of the UK. The reason for the recurring utilisation of this one data source was not some special characteristic of the data. Rather, it was because fine-grain data at the scale of conservation investments is not widely available. This is severe problem for conservation. Scientific principles gain validity, and the limitations of those principles are uncovered, through exposure to a range of data. Many conservation planning studies, including my own, revolve around a single case study. Consequently, it is difficult to draw general lessons from the results. This problem was identified in Chapter 6 where it was seen that the two previous studies that had investigated the trade-off between biodiversity coverage and conservation costs had relied on a single data set. I attempted to extend these analyses by investigating this trade-off over a wide range of conservation planning scenarios by varying the nestedness and scarcity within the Peak District data. The results indicated that the nature of a species distribution can have considerable bearing on conservation outcomes. Nonetheless, by necessity, these simulated data retained many of the features of the original data, again limiting the scope to extrapolate to different situations. An essential component of future conservation research should be to collect and disseminate data at the scale of conservation investments. These data will ensure that the models developed and the policy suggestions that follow from conservation research are tailored to the natural world as it is.

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