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The Economics of Wildlife Conservation

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1. Introduction

Wildlife populations have been adversely impacted by a multitude of human activities, though most ecologists argue that the clearing of forests and grasslands for urban areas and agriculture has had the greatest impact (Sala et al. 2000; Wilcove et al. 2000, MEA 2005). The economic argument for conserving wildlife is largely a public goods argument. A private landowner lacks the incentive to provide adequate habitat for wildlife species because much of the use value (hunting, bird-watching, ecosystem service regulation, etc.) and non-use value (existence of species) produced on landowner-provided habitat will accrue to other people. Therefore, government policies or non-governmental organization (NGO) programs that encourage the conservation of wildlife habitat may improve the efficiency of land use patterns on landscapes.

This chapter focuses on several economic issues pertinent to wildlife conservation efforts. Wildlife conservation activities include climate change mitigation, limits on freshwater withdrawals from watersheds, and efforts to reduce the spread of invasive species. However, the dominant wildlife conservation activity undertaken globally is the setting aside of land to provide wildlife habitat. Here we focus on unresolved economic issues related to the three primary means of establishing habitat set-asides: i) government regulation, ii) direct appropriation or purchase of habitat by governments or NGOs, and iii) payments to landowners for voluntarily altering land-use activities to be more wildlife friendly. Rather than provide a comprehensive literature review, our approach is to provide an in-depth discussion of representative economic research related to these three habitat conservation approaches. The research we review is selected to highlight what we believe to be some of the outstanding economic issues in wildlife conservation that deserve future research attention. Our main arguments are illustrated with several simple extensions to prior studies.

Government regulation is one approach to conserving habitat, and is typified by the United States' Endangered Species Act (ESA). Under the ESA, a species that is determined to be at risk of extinction is listed and afforded regulatory protection. For example, the ESA generally gives the US government the authority to regulate timber harvesting if it is expected that unmitigated harvest activity would threaten the persistence or the habitat of a listed species. We provide a simple extension to previous theoretical models to show that regulatory designs similar to the ESA can drive a wedge between privately and socially preferred behavior. Further, it can create cases where society in general prefers harming wildlife populations. Effective regulatory design must address the tensions that approaches like the ESA can create between societal wildlife goals and individual preferences. To that end the ESA must integrate rigorous ex-post evaluations of conservation outcomes and regulators must be willing to act on uncovered shortcomings.[2]

The direct purchase of habitat by governments and conservation organizations is an alternative to government regulation of wildlife populations. The purchase of habitat for set-asides can take several forms. For example, Norway has bought and retired Peruvian government debt in exchange for the establishment of reserve area in Peru (Hansen 1999). In fiscal year 2010 The Nature Conservancy spent $204 million on the purchases of conservation land and easements across the globe (TNC 2010). At the heart of direct purchase programs is the problem of selecting which land to purchase when conservation funds are scarce and not all desirable habitat can be protected. The literature devoted to finding the best use of funds for some biological objective has been termed “reserve-site selection (RSS)” or “systematic conservation planning (SCP)”, and has been developed by both economists and conservation biologists. Recent efforts to more accurately measure the biological benefit created by reserve networks have been dubbed return-on-investment (ROI) for conservation. We develop a new U.S.-wide reserve selection model and use it to argue that existing reserve selection approaches must i) properly specify the conservation benefits from a reserve system, and ii) incorporate realistic expectations of landscape dynamics outside of the selected network.

The final approach to setting aside habitat is to offer voluntary payments to landowners to alter their land-use practices. This approach is typified by Costa Rica's 1996 national forest law and the U.S. Wildlife Habitat Incentives Program, both of which pay landowners directly for improved habitat provision. Two dynamics make efficient design of voluntary payment programs difficult: i) landowners' willingness-to-accept (WTA) payments is private information, and ii) habitat benefits are spatially dependent, meaning benefits are a function of the spatial pattern of conservation across large landscapes of multiple landowners. The configuration of conservation across a landscape is difficult for agencies to control when WTA information is private because it is unclear ex-ante which landowners will accept payments. Further, when benefits of habitat conservation are spatially dependent it is difficult for agencies to identify the benefits that will result from a particular payment program. As such, we develop a simple example to argue that efficient conservation of wildlife with incentives must overcome the problem of eliciting private information on landowners' WTA. New empirical evidence from the state of Oregon is used to illuminate the importance of the WTA information and to illustrate the large efficiency gains from solving this information problem.

2. Command-and-Control Approach and the U.S. Endangered Species Act

The original version of the ESA, passed in 1973, prohibited an individual, corporation, or government agency from killing or destroying the habitat of a species listed under the Act (a "taking").[3] According to the law's original language, the imperative of saving endangered public goods trumped the private economic interests of landowners (McAnaney 2006). Therefore, just like the original versions of the US Clean Air and Clean Water Acts, the initial version of the ESA was a command and control policy with little regulatory flexibility and no compensation for landowner economic losses due to regulatory actions. Since 1978, however, the ESA has been amended several times and has become a more flexible or permissive policy than its original incarnation, especially when dealing with habitat on private land (Scott et al. 2006a).

The ESA's private land policies are vital to the success and cost of the Act as data suggests that over half of all listed species have at least 80% of their habitat on private property (Innes and Frisvolde 2009). One example of this increased regulatory flexibility is the availability of permits that allow landowners or developers to destroy listed species or its habitat as long as the applicant can convince the permitting wildlife agency that the "take" will not appreciably reduce the species' likelihood of recovery. In many cases, incidental take permits are only granted if the applicant agrees to install conservation measures somewhere on their land or contribute to a general conservation fund (Thompson 2006). Landowner activities necessary to acquire an incidental take permit are laid out in a Habitat Conservation Plan (HCP).

In section 2.1, we extend Polasky and Doremus' (1998) model of landowner-wildlife agency relationships to consider incidental take permits and HCPs and we argue that command-and-control regulation for wildlife conservation can create situations where both individuals and society prefer harming wildlife populations. In section 2.2, we review evidence of the effectiveness of the ESA and argue that it is reasonable for society to expect robust and recovering wildlife populations to result from command-and-control regulation given the well documented welfare losses associated with such policy approaches. Unfortunately, evidence of the efficacy of the ESA is mixed.

2.1 Landowner-wildlife agency relationships under the ESA

Polasky and Doremus (1998) model the interplay between a landowner that is contemplating developing her land and an endangered species regulating agency, where the agency is uncertain whether the landowner's plot contains listed species or their habitats. While Polasky and Doremus consider several potential relationships between the landowner and agency, including payments for conservation, here we mention the two cases that most resemble the relationship under the current version of the ESA. In one case, the agency forces a landowner to set aside their land for conservation if the agency can prove that such action and its effect on listed species' persistence produces value to society, given by S, is greater than the private economic value that will accrue to the landowner after development, given by *D* (assume land development costs are netted out of *D*). Assume *S* falls to 0 if the land is developed (we will relax this assumption in a modification of the Polasky and Doremus model below) and *U* is private returns from conservation (if any) where $D > U$. In this case, the burden of determining *S* is with the regulating agency. Assuming *S* can only be calculated by inspection of the land and that private property holders have the right to prevent any inspections, the landowner has no incentive to allow agency representatives on their land. This blanket refusal of inspection, while always privately optimal given that $D > U$, may be inefficient from society's perspective. Before development, a survey would be warranted from society's perspective if the expected net benefit generated by paying for information on *S* exceeds the benefits generated without the information,

$$(1-p)(S+U) + pD - C > D \Rightarrow \tag{2.1}$$

$$(1-p)(S+U-D) > C \tag{2.2}$$

where $1–p$ is the probability that the survey will find $S \geq D$ and *C* indicates the cost of the survey.

Under another relevant landowner-agency relationship framework explored by Polasky and Doremus, the landowner must prove $D > S$ before they can develop, or otherwise pay a fine *F* where $F > D$. First, the landowner will never develop without a survey, otherwise net returns will be negative ($D–F < 0$). Therefore, a profit-maximizing landowner will commission a survey before development if the expected net benefit of doing so outweigh the benefits of not doing so,

$$pD + (1-p)U - C > U \Rightarrow \tag{2.3}$$

$$p\left(D – U\right) > C \tag{2.4}$$

where *p* is the probability that the survey will find $D > S$. However, from society's perspective, a survey is only welfare enhancing if it is expected to reveal that *D* is significantly larger than *S*,

$$pD + (1-p)(S+U) - C > S + U \Rightarrow \tag{2.5}$$

$$p\left(D – S - U\right) > C \tag{2.6}$$

According to inequalities (2.4) and (2.6) the landowner is more likely to find it in his best interest to survey than society would.[4] In both of these cases, a wedge exists between private and socially preferred behavior.

2.1.1 Landowner-wildlife agency relationships with a Habitat Conservation Plan (HCP)

The current version of the ESA differs from the species conservation policies considered by Polasky and Doremus in several ways (also see Innes and Frisvold 2009 for a revision of the Polasky and Doremus model). First, a finding of a listed species or its habitat on a parcel does not mean it cannot be developed; instead it may only mean a restriction on certain development activities in the parcel. Second, regulators are supposed to limit habitat destruction on a parcel no matter the expected social value of the conservation behavior. Finally, a landowner can choose to limit some species harm and/or habitat damage when developing in exchange for an incidental take permit.

We modify the Polasky and Doremus model to reflect the current agency-landowner relationship where a landowner knows that her land contains listed species or its habitat but the agency does not necessarily know this. She can choose to fully develop the land and risk regulatory penalties, take the steps necessary to gain an incidental take permit, or not develop. *Ex ante* the private economic value generated by a developed parcel with an incidental take permit is uncertain because the landowner does not know what the regulating agency will require in exchange for a permit. An incidental take permit will require the landowner to institute some conservation action or implement a land use that results in less value than unfettered development. In addition, the landowner will incur some HCP negotiation and implementation costs. Let 0 to N indicate the range of private economic value generated on the parcel with a permit less permit transaction costs where $N < D$. Let n indicate the expected private value of the developed parcel with an incidental take permit. Further, let t and w indicate the expected public and private conservation value, respectively, generated by an HCP on the parcel. Because a HCP allows for some development the non-market value of a parcel with a HCP is not as great as the non-market values on an undeveloped parcel : $t < S$ and $w < U$.

By seeking an incidental take permit the landowner signals to the regulator that she has a listed species or habitat on her land. Even if the landowner does not signal the presence of a listed species or its habitat, development action by a landowner could trigger regulatory scrutiny and a judgment of a taking. Let p_S indicate the probability that the agency will become aware that a listed species does use or has used the parcel or the parcel does contain or did contain a listed species' habitat during parcel development. In this initial setup we will assume that p_S is known to the parcel owner and it cannot be affected by parcel owner behavior. In other words, p_s will be determined by the regulating agency's competence, budget, etc. [5] Again let D indicate the private economic value of parcel development where any land development costs are netted out.

If the parcel owner develops and the agency becomes aware of a taking then fine F is levied and we assume the agency will enforce a re-development plan similar to the one that would have been generated under an incidental take permit negotiation process. Therefore, the profit-maximizing landowner chooses his development path according to the following,

$$\max \ \{(1-p_S)D + p_S(n+w-F), n+w, U\}. \tag{2.7}$$

where the first term is the expected net private economic value associated with not approaching the agency to cooperate on an HCP, [6] the second term is the expected net private economic value of approaching the agency to cooperate on an HCP, and the third term is the private conservation returns to the landowner from not developing her land (we assume the private economic value of undeveloped land is 0). The landowner will approach the agency to cooperate on the design of a HCP if,

$$n \geq D - w - \left(\frac{p_S}{1-p_S}\right)F. \tag{2.8}$$

and

$$n \geq U - w. \tag{2.9}$$

In words equation (2.8) indicates the landowner will only come forward to develop a HCP in conjunction with the agency if p_S and F are large enough to bridge the gap between D (the value of unfettered development) and n (the value of development with an HCP).[7] Monetary compensation for cooperating landowners would enter inequality (2.8) on the left hand side, making cooperation on an HCP more likely. For simplicity we assume that n is always larger than $U - w$ or the expected value of development with an HCP is greater than the incremental private non-market benefit of no development versus development with an HCP.

On the other hand, landowner initiative on an HCP is efficient only if the social returns of this decision are greater than expected benefits of unfettered development,

$$n + t + w - C \geq (1 - p_S)D + p_S(n + t + w - C) \Rightarrow \quad (2.10)$$

$$n \geq D - t - w + C \quad (2.11)$$

and the benefits of non-development,

$$n + t + w - C \geq S + U \Rightarrow \quad (2.12)$$

$$n \geq S + U - t - w + C \quad (2.13)$$

where C is the regulatory agency's HCP finding, planning, and implementation costs. [8] For simplicity we assume n is always larger than the incremental benefit of not developing at all plus the regulatory agency's HCP planning and implementation costs (i.e., $n > S + U - t - w + C$). Therefore, social and landowner incentives on landowner initiated HCPs are aligned when $t - C = \left(\frac{p_S}{1-p_S}\right)F$. Otherwise, if $t - C > (<) \left(\frac{p_S}{1-p_S}\right)F$, then society is more likely (less likely) to prefer landowner initiative on HCPs than the private landowner.

2.1.2 The "Shoot, Shovel, and Shut-up" Incentive

As Polasky (2001), Lueck and Michael (2003), and others have pointed out, the probability of the regulatory agency detecting a taking can be lower than p_S for several reasons. For example, the parcel owner can attempt to prevent the wildlife agency from gleaning information about their land prior to development by blocking access, or they can destroy or alter potential habitat on their land ("shoot, shovel up, and shut-up") prior to regulatory attention. Specifically, let $\overline{p_S}$ be the landowner-influenced probability that the wildlife agency will become aware of listed species or its habitat on the parcel in the process of development, and $c(p_s, \overline{p_s})$ indicates the cost of obtaining $\overline{p_s}$ where $\overline{p_s} \leq p_s$, $c(p_s, \overline{p_s}) = 0$ if $\overline{p_s} = p_s$, and $c(p_s, \overline{p_s})$ increases as the landowner reduces $\overline{p_s}$. Let $\overline{p_s}^*$ indicate the $\overline{p_s}$ that maximizes the expected value of full development on the plot (the first term in the maximization function (2.7) less $c(p_s, \overline{p_s})$). The landowner will approach the agency to design an HCP if,

$$n \geq D - w - \left(\frac{\overline{p_s}^*}{1-\overline{p_s}^*}\right) F - c(p_s, \overline{p_s}^*) \qquad (2.14)$$

Because of lower odds of a takings discovery, the fine *F* that may have been large enough to convince the landowner to seek a HCP with exogenous p_s (inequality (2.8)) may not be high enough to engender the same reaction with endogenous $\overline{p_s}^*$; it will depend on the size of $c(p_s, \overline{p_s}^*)$. Again, the inclusion of landowner compensation in an HCP would make conservation cooperation much more likely as the left hand side of inequality (2.14) would be larger.

Finally, we can show that under certain conditions, privately optimal "shoot, shovel, and shut-up" behavior under the ESA, given by $\overline{p_s}^*$, can generate higher net social benefits than when the landowner does not influence p_S. *Ex ante* society will prefer "shoot, shovel up, and shut-up" behavior on the part of the parcel owner if it is expected to generate more in net social benefits than not engaging in it,

$$\begin{aligned}(1-\overline{p_s}^*)D + \overline{p_s}^*(n+t+w-C) - c(p_s, \overline{p_s}^*) \geq \\ \max \ \{(1-p_s)D + p_s(n+t+w-C), n+t+w-C, U+S\}\end{aligned} \qquad (2.15)$$

where the maximization term in equation (2.15) is the parcel owner's decision criteria in the absence of "shoot, shovel up, and shut-up" behavior (equation (2.7)). If,

$$(1-\overline{p_s}^*)D + \overline{p_s}^*(n+t+w-C) - U - S \geq c(p_s, \overline{p_s}^*) \qquad (2.16)$$

then inequality (2.15) *always* holds and "shoot, shovel up, and shut-up" behavior generates higher net social benefits than having the landowner not influence p_S.[9] In words, the lower that $\overline{p_s}^*$can be driven at a reasonable cost, and the higher that the unfettered development value is compared to the social returns from an HCP, the more likely it is that optimal "shoot, shovel, and shut-up" behavior is preferred by both the landowner and society in general.

To summarize, there are two main points from this section. First, under the current version of the ESA the regulating agency can encourage conservation cooperation by levying high fines for a taking by the landowner (or compensating landowners for lost private economic value). However, there is a point where the fine becomes too large from society's point of view because it encourages the development of an HCP that generates less in expected net social benefit than an uncooperative landowner. Second, because the social benefits of an HCP can be small compared to the value of development, net social benefits can be higher when the landowner reduces the odds of finding a HCP optimal or being punished for avoiding one ("shoot, shovel, and shut-up"). The fact that net social benefits can be higher with such perverse landowner behavior than without it highlights the misalignment of private, social, and regulatory incentives under the current version of the ESA.

2.2 How effective is the ESA?

Despite these incentive compatibility issues on private land, whether the Act as currently constituted is as a whole creating more societal benefit than cost is an open question and can only be determined by adding up all the market and non-market values created by the regulation and comparing these to the sum of the opportunity costs generated by the Act's restrictions (Rachlinski 1997). However, this monumental cost-benefit analysis (CBA) has not yet been undertaken by researchers. Given the difficulty in accurately monetizing non-market benefits, a CBA of the entire ESA may be impossible. An alternative measure of regulatory success is given by progress on regulatory goals. And in cases where cost of achieving these goals can be monitized, cost-effective goal-achievement would mean meeting goals at least cost (Shogren et al. 1999, Naidoo et al. 2006).

The goal of ESA regulators is to list species that might go extinct without intervention and then take actions such that these species can eventually be taken off the list due to sufficiently reduced extinction probabilities. Up to this point in time the ESA has failed miserably on this goal. As of June 2012, 2000 animals and plant species[10] were listed as endangered or threatened (607 of these species inhabit ranges completely outside of US territories). Since 1973 only 21 species have been delisted due to recovery (U.S. FWS 2009).

Of course the lack of recovered species does not mean that the Act has not had beneficial effect. Some have argued that many more listed species would have gone extinct without regulatory coverage (e.g., Schwartz 1999). It could also be that recovery sufficient for a de-listing takes several generations of regulatory attention. If so, short-term progress towards de-listing could be measured by change in the status of species over time, a metric tracked by the US Fish and Wildlife Service (FWS) (Rachlinski 1997, Male and Bean 2005, Kerkvliet and Langpap 2007). If we assign a 1 to species whose population is in decline, a 2 to species whose population is stable, and a 3 to species whose population is improving or recovered, then the average status score across 255 listed vertebrates was 1.71 in 1990, 1.74 in 1994, 1.75 in 1998, and 1.68 in 2002 (Kerkvliet and Langpap 2007). This trend seems to suggest that protection has done little to improve the overall status of these 255 species.

Other than some landowners having incentive to reduce the persistence probabilities of listed species (see above), scarce progress on de-listing could also be explained by too little spending on listed species' recovery activities (Miller et al. 2002). There is evidence that increased spending on listed species' recovery activities does promote progress toward delisting. For example, Kerkvliet and Langpap (2007) find that increased spending on a species is correlated with a lower likelihood that the FWS will classify that species' status as extinct or declining. However, the direction of causality is unclear: does increased spending lower the risk of extinction, or is more money being directed to species that are less likely to go extinct? Taylor et al. (2005) argue that increased recovery spending is likely to promote delisting because the activities that they found most explain species' progress towards de-listing – published recovery plans, designated critical habitat, length of time listed, etc. – are positively correlated with more recovery spending, all else equal. Further, Ferraro et al. (2007) find that, on average, the conservation status of listed species with substantial recovery funding has improved over time compared to the contemporaneous conservation status of species with similar characteristics that are only candidates for listing and therefore are not subject to ESA protections and recovery funding. Provocatively, Ferraro et al. also find that the average conservation status of listed species with little or no recovery funding has deteriorated overall compared to the average status of similar candidate species. Why unfunded regulatory protection could lead to worse outcomes is still a matter of conjecture. Some argue that this trend can in part be explained by the incentives that private landowners have to engage in "shoot, shovel, and shut-up" behavior (Ruhl 1998). For "shoot, shovel, and shut-up" behavior to be detrimental to under-funded species it would mean that better funded species are better monitored and tracked on private land and that this acts as a deterrent to landowners destroying them and their habitat.

Presuming ESA funding will never be great enough to fund all recommended listed species' recovery activities, an endangered species agency has two reasonable constrained maximization objectives to choose from. One approach would be to spend recovery funds to maximize the number of species that are de-listed (Mann and Plummer 1995). In this case recovery funds would be directed towards species that could conceivably recover enough for de-listing with limited funding. This choice would leave little money for other listed species and therefore, could lead to an increased listed species extinction rate. An alternative approach would be to distribute recovery funds such that the sum of increase in persistence probabilities across all listed species is maximized. For many researchers this is the definition of cost-effective biological conservation (e.g., Possingham et al. 2002, Polasky et al. 2008). While this approach may not lead to many de-listings, it should limit the number of extinctions. Figure 1 illustrates both approaches.

[Figure 1 here]

Is there any evidence to suggest that either of these two constrained maximization objectives have been adopted by the FWS and other ESA regulatory agencies? Recovery funding is unequally distributed across listed species (see Figure 2) so there does appear to be some pattern to funding. Cash (2001) finds that species that are considered by scientists more likely to recover have received more in funding, all else equal. If we assume that these types of species are like species A in Figure 1 – recovery curves that increase rapidly and meet the de-listing criteria with limited funding – than this observed funding pattern supports an effort to prioritize de-listing of a few species. However, at the same time Cash (2001) also finds that species whose recovery is more likely to cause conflict with economic development goals have received more in funding, all else equal. Such a funding pattern is at odds with the basic tenants of cost-effective goal achievement. Metrick and Weitzman (1998) suggest that there is a strong preference among regulatory agencies for funding the recovery of charismatic species above and beyond what is warranted by recovery science. Such a funding pattern is consistent with the political economy story that regulators attempt to curry emotional support for the Act from the U.S. public rather than demonstrate efficiency. In addition, the allocation of up to 75% in recovery funds has been dictated by line items in appropriations legislation from Congress (Miller et al. 2002), and listed species' funding has been shown to depend on whether their range falls within political districts represented by Congressional representatives on the Department of Interior Subcommittees (Cash 2001, DeShazo and Freeman 2003, 2006).

[Figure 2 here]

2.3 Discussion

Forty years after its passage, opinion on the effectiveness and the net returns created by the ESA vary greatly. In 2003, then Assistant Secretary of the US Department of the Interior Craig Manson "said the 30-year-old environmental law is "broken" and should no longer be used to give endangered plants and animals priority over human needs."[11] Manson argues that the Act does not give regulators enough flexibility to balance economic and environmental tradeoffs. In addition, the listing process has been embroiled in lawsuits over the past decade. Environmental groups that have brought the lawsuits argue that the US government is not fulfilling its regulatory obligation to list all endangered species. Recently, the US government has, in response to environmental group pressure, agreed to decide whether listing is appropriate for 757 additional species by 2018 (pending approval by a federal judge).[12] Advocates of the Act, such as the Center for the Center for Biological Diversity, argue that the Act is essential and deserves strengthening.

Recent estimates indicate that US urban area will increase by 33 million hectares from 2001 to 2052 (Radeloff et al., 2011). Climate models predict accelerated climate change in the lower 48, which has the potential to drastically alter habitat and species geographic ranges on a large-scale (Lawler et al. 2009). Whether the ESA – and similar command-and-control regulatory approaches to species protection – can be effective in a rapidly developing and evolving landscape is questionable. To work, the ESA will need to provide landowners with a stronger incentive to conserve habitat than what is currently in place. We show that a landowner compensation system (or strongly enforced fine system) is one approach to providing this incentive under the ESA's current incidental take system. However, the more incentive that landowners are given to cooperate with authorities on listed species conservation, the more likely that net benefits to society will decrease if the decision to regulate private land activities is not a function of the opportunity cost of conservation. Our second argument is that any wildlife conservation program like the ESA must be subject to some type of ex-post evaluation and adjustment to compensate for some of the efficiency losses generated by a command and control policy. Such a process, however, requires researchers to develop appropriate counter-factual scenarios regarding how a species would fare in the absence of being covered in a conservation program. The recent econometric literature on program evaluation (e.g. Ferraro et al. 2007) has potential in this regard.

3. Purchasing Habitat for Conservation with Complete Information - Reserve-Site Selection

An alternative to government regulation of wildlife conservation is the purchase and setting-aside of privately-owned habitat. For a government or conservation organization involved in buying habitat a basic question is which land should be purchased when conservation funds and/or available land is scarce. In this section we present the basic reserve site selection (RSS) problem and highlight two largely unresolved issues of economic importance: i) specifying a quantitative environmental benefit function, and ii) how to incorporate baseline outcomes in the absence of reserve siting. We highlight these issues with a "return-on-investment" (ROI) approach using conservation siting across the US.

Whereas species form the set of the decision units under the ESA, selecting a set of undeveloped sites for habitat protection is the focus of RSS problems (RSS is often referred to as systematic conservation planning in the conservation biology literature; see Margules and Pressey 2000). In the rudimentary RSS problem the social planner selects a set of undeveloped sites to purchase such that the network of selected sites will provide additional habitat for as many species as possible given an area or habitat protection cost constraint (Ando et al. 1998).

$$\max_{s_j} \sum_i \sum_j s_j x_{ij} \tag{3.1}$$

Subject to

$$\sum_j c_j s_j \leq B \tag{3.2}$$

$$s_j \in \{0,1\} \tag{3.3}$$

where s_j equals 1 if site j is selected for habitat protection and equals 0 otherwise, j indexes all sites on the landscape, x_{ij} equals 1 if species i is known to use site j for breeding or feeding activities, c_j is the area of site j or cost of purchasing and establishing habitat on site j, and B is the social planner's areal or monetary budget. If site j that contains species x is selected then the species is considered "covered" or represented by the selected reserve network and the objective function value increases by one. Solutions to (3.1)-(3.3) typically include sites that are strongly complementary with one another in terms of species composition, not necessarily the sites that contain the most species (e.g., two neighboring sites that contain many species may contain the same species, making the selection of only one of the sites optimal). Because solving binary integer problems over a large choice set can be computationally difficult, heuristic methods for solving (3.1)–(3.3) and related problems have been developed. For example, a simulated annealing heuristic that can approximate solutions to a problem like (3.1)–(3.3) has been codified in a software package called MARXAN (Ball et al. 2009). MARXAN is a widely used in the conservation planning community.

Early work on RSS formulated (3.1)–(3.3) as an area-constrained problem where the planner was constrained by total land area rather than budget (Dobson et al. 1997; Camm et al. 1996; Church et al.

1996). Ando et al. (1998) relax the implicit assumption of uniform costs in the area-constrained problem and show that by setting c_j equal to the expected cost of purchasing an acre of habitat in county j and B equal to a protection area monetary budget, the same number of endangered species can be covered for less aggregate cost than Dobson et al.'s area-constrained solution to (3.1)–(3.3).

3.1 Issues in Reserve-Site Selection

Dobson et al. (1997) and Ando et al. (1998) assume that all species found in a selected county would benefit from a representative protected area within the county. However, this is unrealistic given the size of counties and the disparate habitat preferences of species located within a county. Over time the rudimentary RSS problem (3.1)–(3.3) has seen substantial refinement in representing the conservation benefits gained by selecting a site. One approach is to reduce the size of potential sites j so that each site contains uniform habitat (Haight et al. 2005, Polasky et al 2008). Alternatively, the objective function (3.1) can be re-specified such that additional habitat contributes to a more biologically complex metric. These more complex metrics include the sum of individual species persistence probabilities (Polasky et al. 2008) and some variant of the well-known biological species-area relationship (SAR) from the discipline of conservation biology (Rosenzweig 1995). Common across all of these approaches is that the biological score is more than just a sum of species covered by habitat area. Further, in many second generation RSS problems the biological metric is also a function of the portions of the landscape that are not protected.[13]

While all RSS problems assume that more habitat on a landscape increases the value of the objective function, the rate of and shape of objective value increase can vary substantially across second-generation RSS problems. For example, Wu and Boggess (1999) and Wu and Skelton-Groth (2002) argue that returns to resource conservation (e.g., improving water quality for recreational purposes, adding habitat to the landscape to increase biodiversity persistence) tend to display a "ʃ"-shape. Under such an assumption rapid increases in the benefits provided by additional resource conservation only occur once a threshold of minimum conservation has been reached; prior to that point, resource conservation only has a small effect on objective. Polasky et al. (2008) use such a "ʃ"-shape when explaining species' persistence probabilities across a landscape. While initial habitat conservation on the landscape increases species persistence probability at an increasing rate, rapid non-linear increases in species' persistence probabilities due to additional habitat only occurs once a threshold level of habitat is provided on the landscape. Eventually species response to additional habitat becomes saturated. At fairly high levels of habitat on the landscape additional habitat is relatively worthless. Conversely, the SAR, which specifies the number of species found on the landscape (richness) as a function of habitat provision, is strictly convex in conservation. In SAR-based RSS the first few units of habitat on the landscape add the most to the biodiversity objective maximization. Therefore, using a "ʃ"-shape objective function versus a SAR curve in a RSS problem over the same landscape with the same policy parameters can generate fairly different outcomes and/or solutions. In some cases, the same pattern of habitat conservation is selected by both types of objective functions but the gain in the relevant biodiversity score will be much more impressive with the SAR objective function. Or in other cases, the threshold effect in "ʃ"-shaped objective functions will mean very different patterns of habitat conservation when compared to the SAR generated landscape. For example, the threshold to rapid increases in species persistence due to additional habitat in Polasky et al. (2008) is reached more quickly if the initial habitat is clumped spatially on the landscape. A SAR-based analysis of the same landscape would not necessarily reward habitat clumping to the same degree.

Another issue that has seen recent attention is uncertainty across the RSS's parameters. For example, Haight et al. (2005) maximized expected species coverage given probabilistic geographic range maps. Such an approach can account for probabilistic shifts in species' range due to ongoing climate change (Araunjo et al. 2004; Pyke and Fischer 2005, Ando and Mallory 2012). In addition to biological uncertainty, some recent analyses have accounted for uncertain opportunity costs of conservation (e.g. Carwardine et al. 2010; Nelson et al. 2008; Lewis et al. 2011) and which habitat sites are actually available for purchase.

Landscape dynamics have also been incorporated in RSS problems. Much of the RSS literature assumes that sites not selected for protection will be lost to development. While this may be true in the long-run, it is not true in the short-run: valuable habitat not selected immediately for protection can persist indefinitely if current market conditions do not give the site's owner incentive to develop. Therefore, a dynamic planner with limited conservation funds each time step has to consider the likelihood that currently undeveloped sites will remain so indefinitely. In general, optimal RSS over time requires leaning towards purchasing the sites that are immediately threatened by development, even if they are not as biologically valuable as sites that are less threatened (Costello and Polasky 2004). Conversely, biologically valuable sites that are highly unlikely to be developed in the near future may be best left unprotected indefinitely since there is limited expected value in protecting them immediately.[14]

3.2 An Empirical Illustration using the Return on Investment (ROI) Approach

Recently, biological conservation journals have published a number of articles on the so-called return on investment (ROI) problem (e.g., Murdoch et al. 2007). The objective of the ROI problem is generally the same as the RSS problem: maximize the return (in biological terms) per unit investment (generally a conservation budget). However, ROI improves upon several features of the fundamental RSS problem. First, ROI is more explicit in incorporating baseline or business-as-usual threats to undeveloped area over time. Second, to make this approach applicable to conservation organizations and their rapid funding cycles, ROI approaches may select the sites that are expected to generate the greatest ROI at each decision time step and not the trajectory of site selection that will maximize ROI over the problem's entire time frame. Third, in addition to habitat purchases, ROI approaches can allocate effort across other conservation actions that can increase species persistence in an area (e.g. invasive species removal, fire suppression, fuel reduction, etc.; Murdoch et al. 2007). Finally, the ROI approach acknowledges some of the realities in conservation implementation, including the risk that purchased protected areas and the species they host may be lost due to unforeseen events, species extinction may reduce the biological value of a protected site in the future, conservation organizations with different objectives compete for the same sites (Bode et al. 2011), and that funds are not fully fungible across regions.

In this section we present an illustrative application that combines some of the conservation complications addressed in the dynamic RSS and ROI approaches with the traditional RSS problem given by (3.1)–(3.3). Similar to the early work of Ando et al. (1998), we solve equations (3.1)–(3.3) using the set of U.S. counties for the establishment of 1,000 hectare conservation reserves. Our dataset includes 1,066 vertebrate species in the continuous U.S. with detailed range maps. The data used is fully described in Withey et al. (2011). Finally, like most ROI literature, we assume the biological objective is convex in habitat.

Our goal is to illustrate the effects of i) diminishing "biological returns" to conserving land, ii) a baseline or business-as-usual future where not all land is at risk of development immediately, and iii) incorporating scale into a measure of biological benefits on solutions to the rudimentary RSS problem. First, we incorporate diminishing returns by favoring the selection of counties with less land already protected as of the late 2000s (CBI 2010). Define P_x as one plus the proportion of county x protected where higher P_x indicates greater existing protection (a completely unprotected county would have a score of 1 and a completely protected county would have a sore of 2). Second, the degree to which habitat is threatened is accounted for by targeting counties with higher rates of expected future habitat losses. Let T_x be a metric equal to one minus the proportion of natural land cover in county x developed between 1992 and 2001 (Fry et al. 2009). Therefore, T_x varies between 0 and 1 where a lower value means that development of habitat has been rapid in the immediate past and presumably will continue to be intense in the near future. Finally, given the large size of U.S. counties, we account for the fact that a 1,000 hectare reserve is unlikely to cover the range of relevant habitats within each county. We address this scale issue by selecting counties that have relatively more homogeneous land cover. Define D_x to range from 1 (high diversity of natural land cover types in county x) to 2 (no diversity of natural land cover types in county x) as of 2001 (Comer et al. 2003).

There are several ways we could add these selection criteria to the traditional RSS problem. For example, weight w_j could be added to objective function (3.1), $\max_{s_j} \sum_i \sum_j w_j s_j x_{ij}$, such that species that range over counties with lower *P*, lower *T*, and higher *D* values have higher *w*. Instead, our approach is to simply add constraints to the RSS problem such that the selected network has average county-level *P* and *T* values equal to or less than 30th percentile values for *P* and *T* across all counties (1.0013 and 0.9748 respectively) and an average *D* value equal to or greater than the 70th percentile value for D across all counties (0.8460). This means selected networks will have very little protected area already, are expected to experience significant development pressure in the immediate future, and will have much less natural land cover diversity than other counties. By lowering the maximum average *P*, *T*, and *D* value of the selected network we create solutions more consistent with ROI principles. The budget constraint is given by,

$$\sum_{x=1}^{X} s_x A_x C_x \leq B \qquad (3.4)$$

where $s_x = 1$ if area in county *x* is selected and equals 0 otherwise, A_x is the area selected in *x* for protection and is equal to 1000 ha for all *x* in this case, C_x is the average per hectare cost of undeveloped land in *x* as of 2001 (Withey et al., 2012), and *B* is the budget. For comparison, we also solve the traditional RSS problem (3.1)–(3.3).

In Figure 3 and Table 1 we present the solution to the two versions of the RSS problem for various budget levels. If the networks that form the traditional RSS curve score poorly on average *P*, *T*, and *D* values that are associated with high ROI, then the vertical gap between the ROI-influenced and traditional RSS curves can be interpreted as a measure of untenable species protection if one assumes that the ROI-influenced RSS networks are much more likely to increase the persistence probabilities of the covered species than those species covered by the traditional RSS networks. Consider the highlighted points in Figure 3A. Points ‘A’ and ‘a’ represent reserve networks that cost approximately $1,025,000. Table 1 indicates that the traditional RSS solution places conservation in areas that have experienced recent habitat loss of less than 1% (T=0.994), that have about 10% of their land already protected (P=1.098), and that have a fairly diverse land cover (D=0.634). In contrast, the ROI-influenced RSS places conservation in areas that have experienced higher recent habitat loss of more than 3% (T=0.969), that have no existing protected land (P=1), and that have less land cover diversity (D=0.85). An interpretation is that by mis-specifying conservation benefits, the traditional RSS drastically overestimates the number of 'protected species' by failing to account for diminishing returns and a baseline where much of the unprotected habitat remains on the landscape for the indefinite future.

[Figure 3 here]

An additional issue in specifying conservation benefits is the fact that effective coverage of species on a landscape is likely to require more than one additional habitat site. This is similar to Wu and Boggess' (1999) argument that returns to resource conservation tend to display a "∫"-shape. To begin to explore the ramifications of requiring more sites for species coverage, we re-run the ROI-influenced and traditional RSS problems where a species that has range over two or more counties needs to have range in at least two selected counties to be considered covered. Figure 3B gives the cost curves for this more restrictive approach. The requirement of a second site reduces the species covered by one-half to a third across the modeled budget levels. In Table 1 we compare the characteristics associated with all of the selected networks highlighted in Figure 3. Requiring two counties for coverage does little to change the overall characteristics of counties selected for protection at various budget levels. As with the one-county problem, the traditional RSS networks score poorly on the diminishing returns, threat, and natural land cover indices that indicate strong ROI. Therefore, the gap between the two frontiers in Figure 3B is likely to be indicative of overestimated species protection under the traditional RSS networks.

[Table 1 here]

3.4 Discussion

The problem of where to site nature reserves under a budget constraint has become a classic economic problem. In this section we optimally locate reserves across US counties to highlight two important features of the RSS problem that deserve further research attention. First, the solution of where to site reserves is greatly influenced by the specification of conservation benefits. While ecologists understand many principles about desirable wildlife habitat, much work remains on understanding how the conservation benefits of reserve creation are influenced by factors such as diminishing returns and species range considerations (the benefits of reserve creation become even more difficult to model if it is assumed that returns to habitat provision are increasing over one range of provision and decreasing over another). Second, reserve siting is greatly influenced by how the analyst treats baseline outcomes in the absence of reserve siting. Many regions are likely to see little loss of habitat in the absence of reserve creation, and so siting reserves in such areas is likely to be inefficient in the usual case of a scarce conservation budget. Continued emphasis on modeling and incorporating baseline landscape dynamics into RSS would generate substantial research value.

4. Conserving Wildlife with Voluntary Incentive-Based Payments

Many countries and government entities attempt to conserve wildlife and other ecosystem services through non-regulatory means, with voluntary payment programs (often termed payments for ecosystem services) being among the most popular approaches. In the US, the multi-billion dollar annual budget of the Conservation Reserve Program is an example whereby owners of agricultural land are offered voluntary payments to undertake conservation activities on their land. The efficient design of voluntary payments for wildlife conservation must overcome two principal challenges that will be discussed in this section. First, a landscape's ability to provide the habitat resources necessary to sustain a wildlife population is likely dependent on the spatial configuration of that habitat across many independent landowners. Second, landowners have private information regarding their willingness-to-accept (WTA) payments in exchange for adopting conservation measures on their land, and profit-maximizing landowners typically have no incentive to truthfully reveal their WTA to a conservation agency. It is the combination of spatial dependencies and private WTA information that makes designing efficient payment programs challenging and will be the focus of this section. The underlying argument of this section is that efficient design of voluntary incentives for wildlife conservation is essentially a problem of obtaining private information on landowners' WTA.

4.1 A simple example with spatial dependencies

A 1x4 parcel landscape is used to demonstrate the challenges of designing an efficient payment program with spatial dependencies and private information. Figure 4 illustrates this landscape. The WTA of each landowner to place their parcel in conservation is indicated at the top of the parcel, while the wildlife benefit in biophysical units of conservation is indicated along the bottom. The first number is the benefit when the parcel is conserved but no adjacent neighbor is conserved. The second number is the benefit when the parcel is conserved along with one adjacent conserved neighbor. The third number is the benefit when the parcel is conserved along with two adjacent conserved neighbors. Each parcel is assumed to generate biological benefits of zero if not conserved. On this landscape wildlife benefits exhibit spatial dependencies and conservation costs are heterogeneous.

[Figure 4 here]

Conservation costs may be heterogeneous because land quality varies across parcels or because landowners' WTA reflects different land management skills or other attributes associated with how they value their land. Wildlife benefits are heterogeneous across parcels because natural habitat may vary across the landscape, the ability to restore natural habitat may vary across the landscape, or the geographic range of some species may only comprise a subset of the four parcels. Finally, wildlife benefits exhibit spatial dependencies. In other words, benefits are increasing in the number of conserved neighbors. In general, species prefer larger contiguous patches of habitat than isolated, smaller patches.

If we assume that the value of biophysical benefits is $1/unit, the conserved landscape that maximizes net benefits can be determined by enumerating the total net benefits from all possible configurations. The maximum net benefit generated by parcel conservation on this landscape is $2, and arises when adjacent parcels A, B, and C are conserved ($8+$9+$5–$5–$8 –$7). The marginal benefit of conservation generated by parcel *i* in the optimal landscape configuration can be determined by calculating the total benefit from optimal conservation less the total benefit without parcel *i* in the conservation network. For example, the total benefit of conserving parcels A, B, and C (the optimal network) is $22, whereas the total benefit would only be $14 ($8+$6) if parcel C were not conserved. Therefore, the marginal benefit of conserving parcel C is equal to $8 ($22–$14), which is larger than parcel C's opportunity cost of $7. Parcel C is optimally conserved. If we measured parcel C's marginal benefit outside of the optimal network, for example a network where only parcels B and C are conserved, then C's marginal benefit in conservation has decreased to $7 ($11–$4 = $7). In this particular case, society is now indifferent to conserving C because its marginal benefit equals its WTA. When benefits are spatially dependent, the full marginal benefit of a conserved parcel can only be determined once the optimal landscape is known.

If the price of biophysical benefits is not known, an alternative formulation of the conservation problem would be to maximize biophysical benefits under a cost constraint. For example, all four parcels in figure 4 would be optimally conserved under a cost constraint of $28. In the cost-constrained formulation of the problem the marginal benefit of an optimally conserved parcel is a function of the cost constraint. For example, the marginal benefit of parcel C is 8 biophysical units under a cost constraint of $20 (8+9+5–8–6)[15], and 11 units under a cost constraint of $28 (8+9+6+6–8–6–4).

Regardless of whether the problem is formulated as a net-benefit maximization or cost-constrained optimization problem, an important conservation question is how to implement the optimal landscape with voluntary payments when WTA is known by the landowners but not by the conservation agency. As we saw above, under the net-benefit maximization problem the optimal pattern is to conserve parcels A, B, and C. Let us say a uniform payment of $8 was offered to parcel owners on the landscape in order to entice the owner of parcels A, B, and C to conserve their land. However, a payment of $8 could also induce parcel D to enroll, which is not optimal. As an alternative to a uniform payment program, an "agglomeration bonus" has been proposed (Parkhurst and Shogren 2002, 2007) as a means of giving a bonus payment to those landowners who jointly conserve their land along with a neighbor. However, the optimal size of the "bonus" would have to vary when marginal benefits from conserved land are heterogeneous across the landscape like they are in the illustrative example above. In such cases an agglomeration bonus program would require offering a menu of contracts where each landowner's bonus would depend on the exact configuration of the landscape. As an example of how complex the menu could get even with small landscapes, a 4x4 landscape has 65,535 possible conservation configurations. A Piguovian subsidy is a third implementation strategy. If the price of biophysical services is $1/unit, a Pigouvian approach would entail offering landowners payments equal to their land's marginal benefit from conservation. If there were no spatial dependencies, each parcel would be offered a monetary payment equal to the first number in the row of numbers that indicate the parcel's benefit in conservation. Only parcel A would accept this payment, and the optimal landscape in the absence of spatial dependencies could be conserved. However, this approach doesn't work with spatial dependencies: without information about landowners' WTA, which we had in our illustrative examples above, the regulator cannot solve the optimal landscape and determine each parcel's marginal benefit in equilibrium.

4.2 Empirical analysis of incentive policies under spatial dependencies and asymmetric information

We use data from a real landscape and extend the empirical analysis by Lewis et al. (2011) to illustrate two points discussed in section 4.1. First, we show how sensitive spatially dependent marginal benefits can be to changes in the optimal landscape. Second, we illustrate the importance of WTA information by examining the poor performance of second-best conservation policies. The recent study by Lewis et al. provides an empirical analysis of the efficiency of a series of second-best policies that operate when private WTA information is combined with spatially dependent benefits of conservation. The authors combine econometrically generated distributions of landowners' WTA with biological models of species persistence. The WTA distributions are estimated from observed plot-level land-use decisions over a 15-year period in the Willamette Basin of Oregon, USA (Figure 5). The estimated WTA distributions are used to simulate landowner responses to a variety of incentive policies, whereby landowners know their WTA, but the conservation agency does not. The biological model uses spatial landscape patterns generated by the econometric models and information on species' range and habitat compatibility as inputs and returns the sum of estimated persistence probabilities across a set of 24 terrestrial species of conservation concern (the landscape's biological score is normalized on a 0 to 100 scale where 100 means all species have a persistence probability of 100). In this application, the response of species persistence probabilities to additional conservation on the landscape is "ʃ"-shaped. The authors are able to compare outcomes from second-best policies with a first-best optimal policy. The optimal policy is estimated within a simulation by taking a random draw from the WTA distributions from each parcel, treating the draw as a known WTA value, and then selecting the conservation pattern that maximizes the biological score for a given level of opportunity cost. The opportunity cost constraint is treated as the sum of WTA across all conserved parcels.

[Figure 5 here]

Figure 6 presents the landscapes that maximize the biological score for a given opportunity cost. The conservation budget sums the WTA for each conserved parcel, where the landscape of random draws from the estimated WTA distributions for each parcel is held fixed. The five landscapes differ in terms of their conservation budget. As seen in the maps of Figure 6, small changes in the budget constraint can imply fairly large changes in the optimal conservation pattern, whereby parcels are both added and subtracted from the optimal conservation pattern as the budget constraint is relaxed. These results are driven by the spatial dependencies in the biology model and the particular landscape of WTA values. Different draws from the WTA distribution would also change the optimal conservation pattern.

[Figure 6 here]

Figure 7 presents marginal benefits (expressed as marginal biological scores) for a select set of sixteen parcels that are included in one or more of the optimal landscapes from Figure 6. As in the simple example in section 4.1, the marginal benefits of conservation for optimally conserved parcel *i* can be evaluated by examining the optimal biology score minus the landscape score without parcel *i* being conserved. The main point to be taken from Figure 7 is the fact that marginal benefits greatly depend on the optimal landscape, and so will differ as the budget constraint is changed. The other striking feature of Figure 7 is the magnitude of changes in marginal benefits that result from seemingly small changes in the conservation budget. This result falls from the highly non-linear nature of the spatially dependent biological benefit function ("∫"-shaped persistence probability function) and the potential of turning a fairly fragmented network of conservation sites into a much more connected network by strategically placing a few more conserved parcels on the landscape. All of this is indicative of the complexity of examining optimal landscapes for wildlife conservation.

[Figure 7 here]

Using the optimal landscape biology scores as a benchmark, Lewis et al. (2011) examine the performance of several alternative policy designs in which landowner WTA is assumed unknown by the regulator. First, a set of "least-cost" policies are evaluated in which uniform per-acre payments are offered to all landowners who meet particular eligibility requirements based on habitat type and size characteristics, and an agglomeration bonus. Relative to a baseline, none of the policies achieved even 25% (55%) of the optimal increase in the biology score at low budget levels (high budget levels). Second, a set of "benefit-cost" policies are evaluated, where "benefit" indices were constructed using the same set of habitat type/size and agglomeration characteristics considered above. While none of the "benefit-cost" policies achieved even 28% of the optimal increase in the biology score at low budget levels, the best-performing policy did achieve a more respectable 87% of the optimal increase at very high budget levels. Of course, it must be pointed out that one can conserve all available land as habitat if the budget is high enough. The underlying lesson from this analysis is that efficient conservation with spatially dependent benefits is extremely difficult in the absence of information on landowner WTA, and so efficient wildlife conservation with voluntary incentives should be treated as an information problem.

4.3 Related literature

There are several related literatures that evaluate and shed light on issues in conserving wildlife with voluntary incentives. Parkhurst and Shogren (2002; 2007) use experimental methods with students to examine possibilities regarding the agglomeration bonus. This set of papers generally finds that an agglomeration bonus can encourage clustered habitat, though the evaluated settings consists of only two to four landowners. Lewis et al. (2009) examine a second-best approach which divides landscapes into geographic sections consisting of multiple landowners each, whereby uniform afforestation payments are offered to all landowners within sections, while the payment amount differs by section. Their findings emphasize the optimality of corner-solutions, whereby it is optimal to either conserve all land in a section or none. Finally, there is a related literature on conservation auctions (Latacz-Lohmann and Van der Hamsvoort 1997; Stoneham et al. 2003; Cason and Gangadharan 2004; Kirwan et al. 2005; Schillizzi and Latacz-Lohmann 2007). While this auction literature focuses on information asymmetry issues with conservation programs, none of the auction designs examined are aimed at achieving truthful revelation of landowner WTA.

5. Conclusion

The challenge of how to slow the rate of decline in wildlife populations presents a significant public goods provision challenge to economists. The benefits from wildlife are generally non-market and largely accrue to individuals who do not own land that contains habitat. Governments and NGOs have addressed the conservation of wildlife habitat largely through i) land-use regulation, ii) habitat purchases, and iii) payments for voluntary conservation. This chapter synthesizes a set of outstanding economic issues that are necessary to understand the efficient design of wildlife conservation. While we highlight many of the issues that have been the focus in the literature over the past 15 years, we argue that many important issues remain to be explored in the economics literature. First, land-use regulatory design must provide direct conservation incentives for landowners or habitat destruction can be socially preferable, and researchers need to develop better methods for empirically evaluating regulatory outcomes and appropriately adjusting policy to partially compensate for the efficiency costs of regulation. Second, solving the problem of spending scarce conservation dollars on habitat purchases must devote more attention to the specification of a conservation benefit function and the specification of baseline landscape outcomes in the absence of habitat reserves. Finally, the efficient design of voluntary conservation payments must solve the problem of how to elicit landowner opportunity costs of conservation, as there are no current auction methods that have been successfully developed for this problem.

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Tables

Table 1

Reserve network	No. of species covered	Cost	No. of counties w/ a site	Average D	Average P	Average T
One County Cover						
A	439	1,034,800	5	0.850	1.000	0.969
a	656	1,022,429	6	0.634	1.098	0.994
B	522	1,913,140	8	0.861	1.001	0.971
b	743	1,909,807	9	0.545	1.141	0.993
Two County Cover						
C	255	821,313	6	0.858	1.000	0.977
c	461	814,650	8	0.514	1.090	0.993
D	376	1,799,383	9	0.850	1.001	0.976
d	583	1,799,004	11	0.579	1.137	0.993
All US counties	NA	NA	Mean	0.658	1.039	0.979
			Std. dev.	0.231	0.081	0.026

Figure captions

Figure 1: Potential recovery funding distribution strategies across listed species. Assume there are 4 listed species, named A, B, C, and D. Each curve represents how a species responds to recovery funding where the height of the curve indicates the species' indefinite persistence probability. In this case the marginal persistence value of recovery funding is diminishing across the entire range of funding. In some conservation contexts the persistence probability curves may initially increase in recovery funding and only after some threshold begin to decrease in recovery funding (a "ʃ"-shape curve; see Lamberson et al. 1992 and Wu et al, 2000).When persistence probability gets high enough a species is delisted. In this case, even with an unlimited budget, the agency could only fund the delisting of two species, A and B. Here assume the wildlife agency only has *X* dollars to spend on listed species recovery activities. Suppose *X*, if entirely spent on species A's recovery, would be just enough to fund its delisting. If the agency's objective is to maximize the number of species delisted it will provide *X* in recovery spending for species A. If the agency's objective is to fund as much of an increase in aggregate persistence probability as possible, it will give to species such that the marginal persistence value for each species is the same and the budget is exhausted. In this illustrative example this occurs at the funding levels x_A, x_B, x_C, and x_D where $x_A + x_B + x_C + x_D = X$.

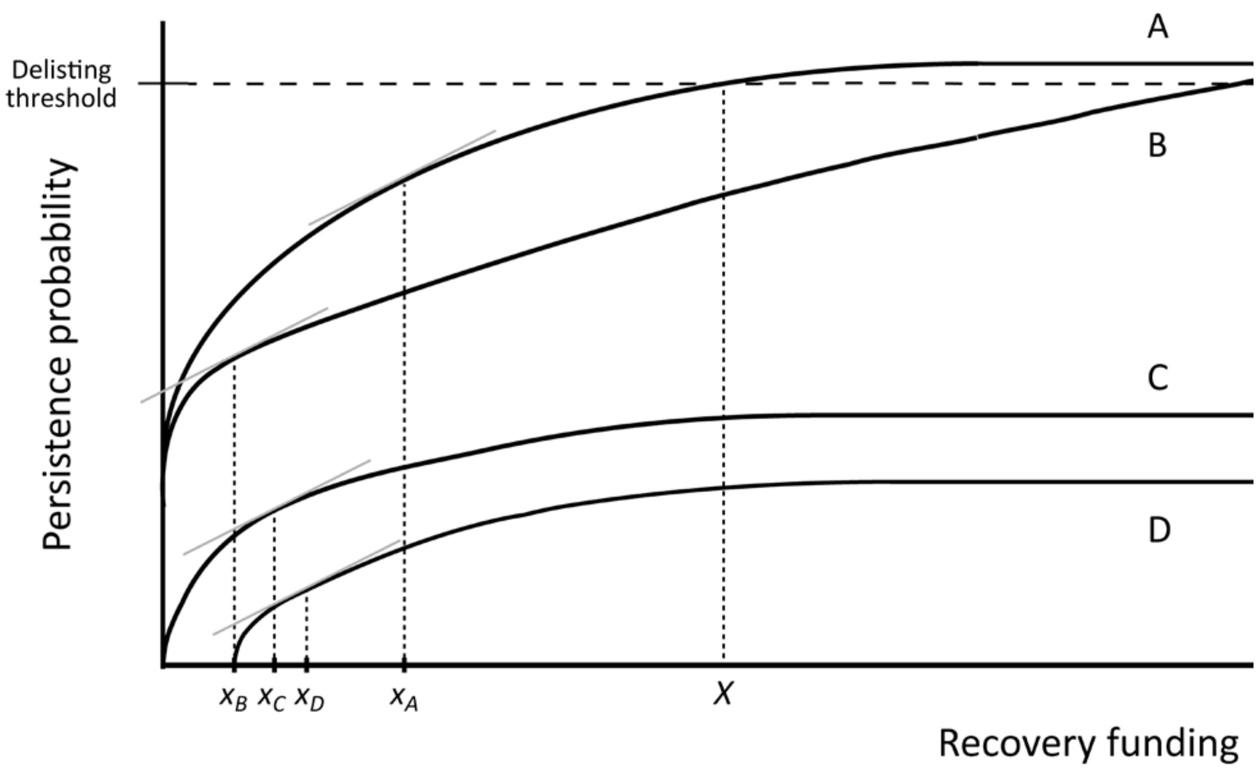


Figure 2: Cumulative recovery funding by US Federal and State agencies across listed species in fiscal year 2009 (not including land acquisition costs). Listed species are arraigned on the *x*-axis in order of recovery funding. The top 10 and 50 listed species in terms of fiscal year 2009 recovery funding received 34% and 85%, respectively, of all spending that year (U.S. FWS 2009).

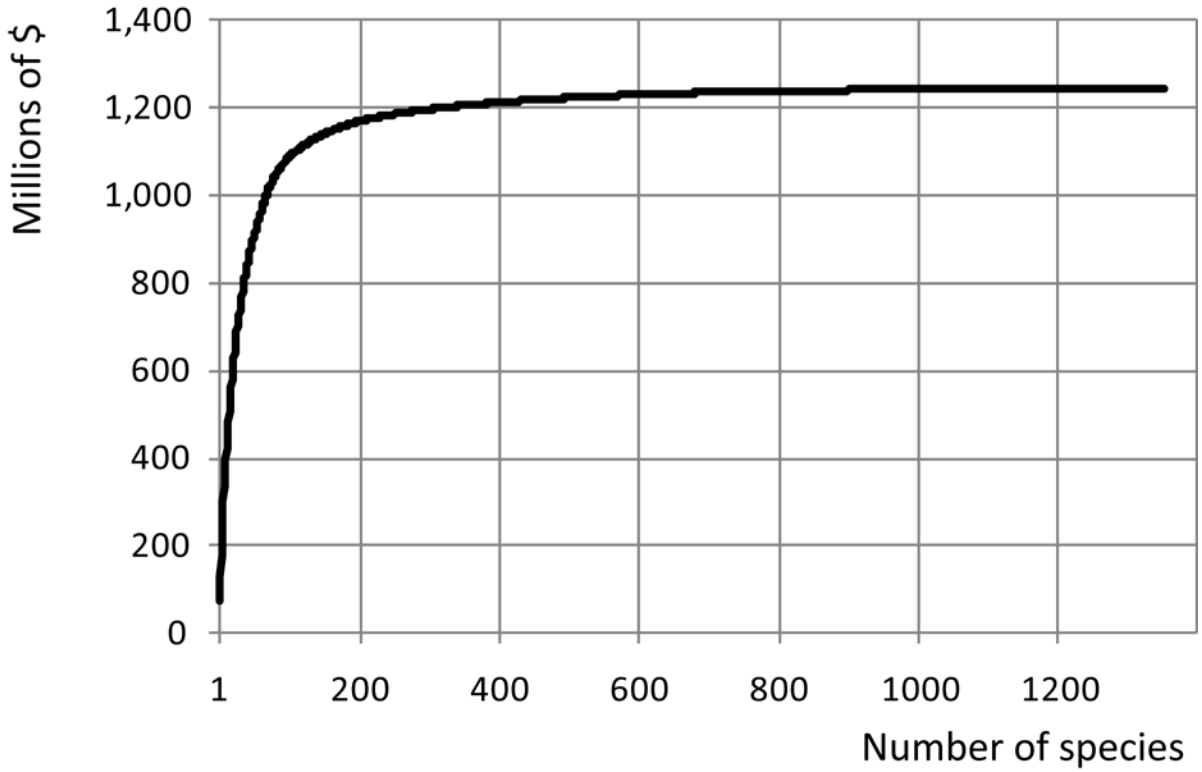
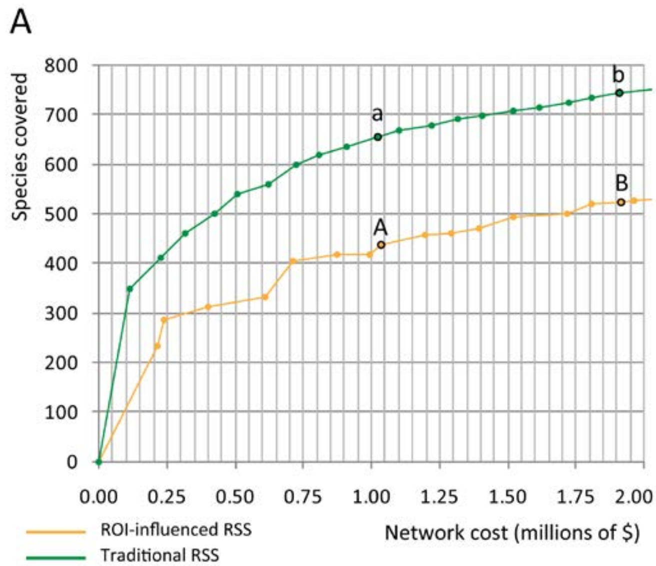


Figure 3: Species covered in networks selected by a traditional and ROI-influenced RSS problem at various budget levels. In Figure 3A a species is considered covered by a network if a 1000 hectare site is selected in at least one county that the species is known to range in. The traditional RSS networks represented by the frontier in Figure 3A were found by solving the problem (3.1)–(3.3). The ROI-influenced networks represented by the frontier in Figure 3A were found by solving the problem (3.1)–(3.3) with additional constraints that increase the network's ROI. In Figure 3B a species is considered covered by a network if a 1000 hectare site is selected in at least two counties that the species is known to range in (unless the species is endemic to a county, then it is covered if its home county is selected). Otherwise the traditional and ROI –influenced networks are selected in the same way as before. The faint lines in Figure 3B indicate the placement of frontier solutions from Figure 3A.



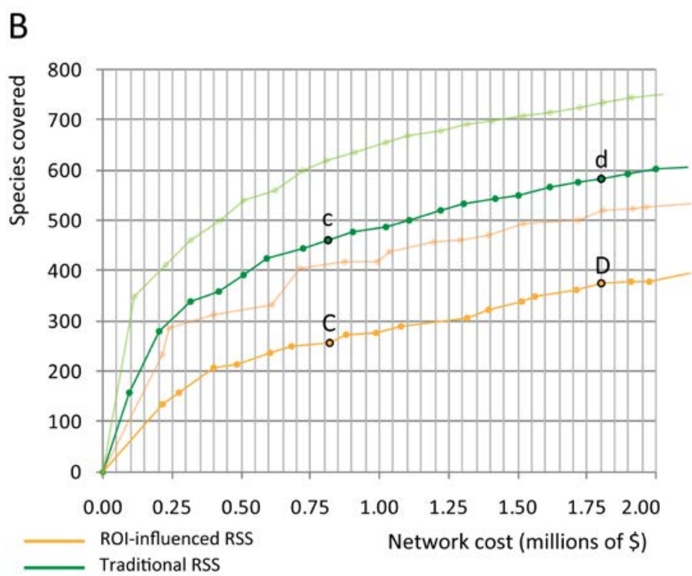


Figure 4: An example landscape with costs (top number in $) and biophysical benefits which depend on having zero, one, or two conserved neighbors (bottom numbers, in biophysical units)

Parcel A	Parcel B	Parcel C	Parcel D
$5 6 8	$8 4 6 9	$7 4 5 6	$8 4 6

Figure 5 – The Willamette Basin of Oregon, USA

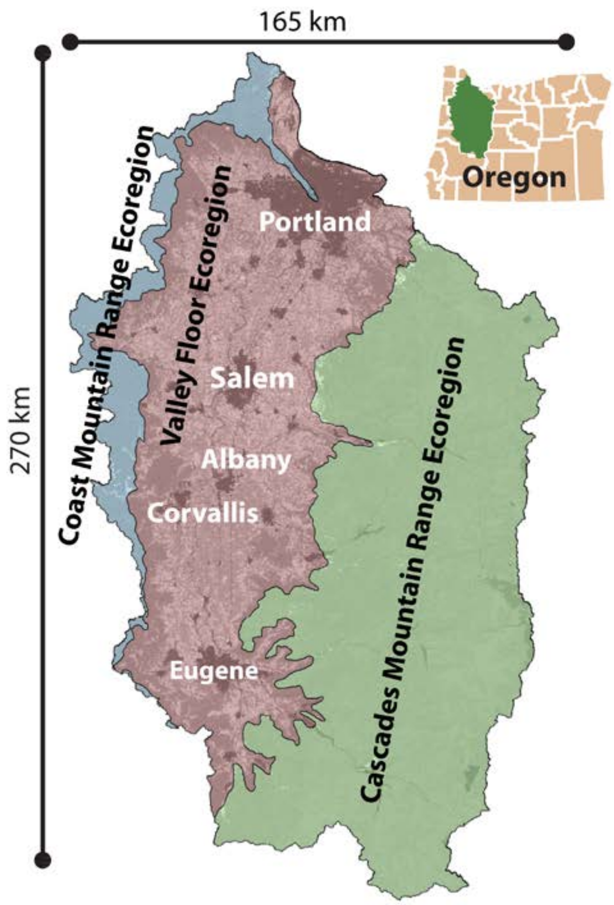


Figure 6: Optimal Conservation for the Willamette Basin. The top row gives conservation patterns that maximize the biology score (scaled from 0 to 100) for a given opportunity cost budget. Each mapped unit is a 500 hectare hexagon and is comprised of non-uniform parcels. The darker the shade of a hexagon the greater the fraction of the parcel space in the hexagon that is conserved. The bottom row of maps is the difference between two maps; map B less map A, etc., which shows how the distribution of conserved area changes from one landscape to the next. Areas with hotter colors represent hexagons that lost conserved area vis-à-vis the previous landscape and hexagons with darker shades of green have gained more conserved area. For example, if a hexagon has a score of -0.31 its fraction of conserved area has fallen by 0.31; in other words it has lost 0.31 x 500 = 155 hectares of conserved area (the greatest decline is 99%; the greatest increase is 98%).

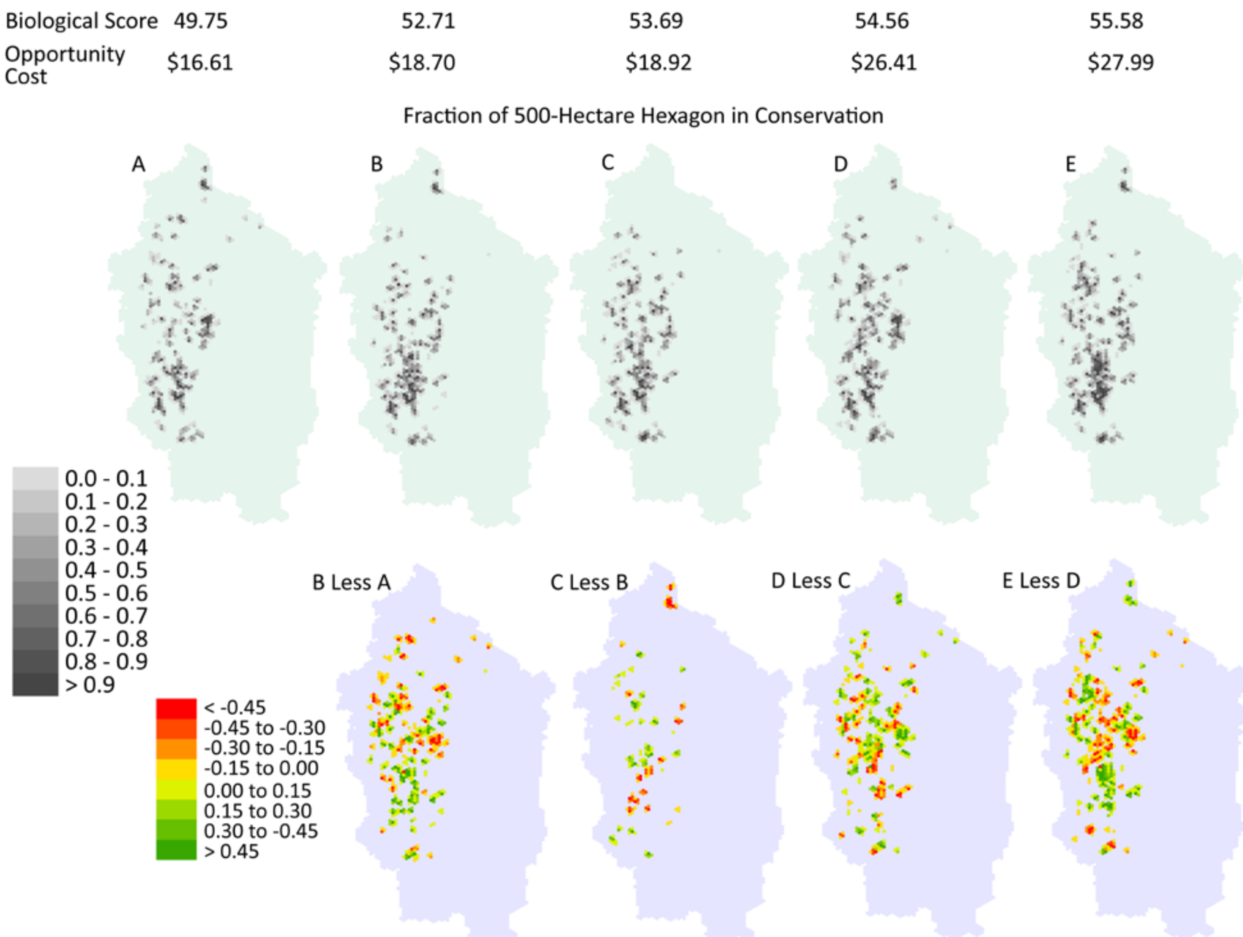
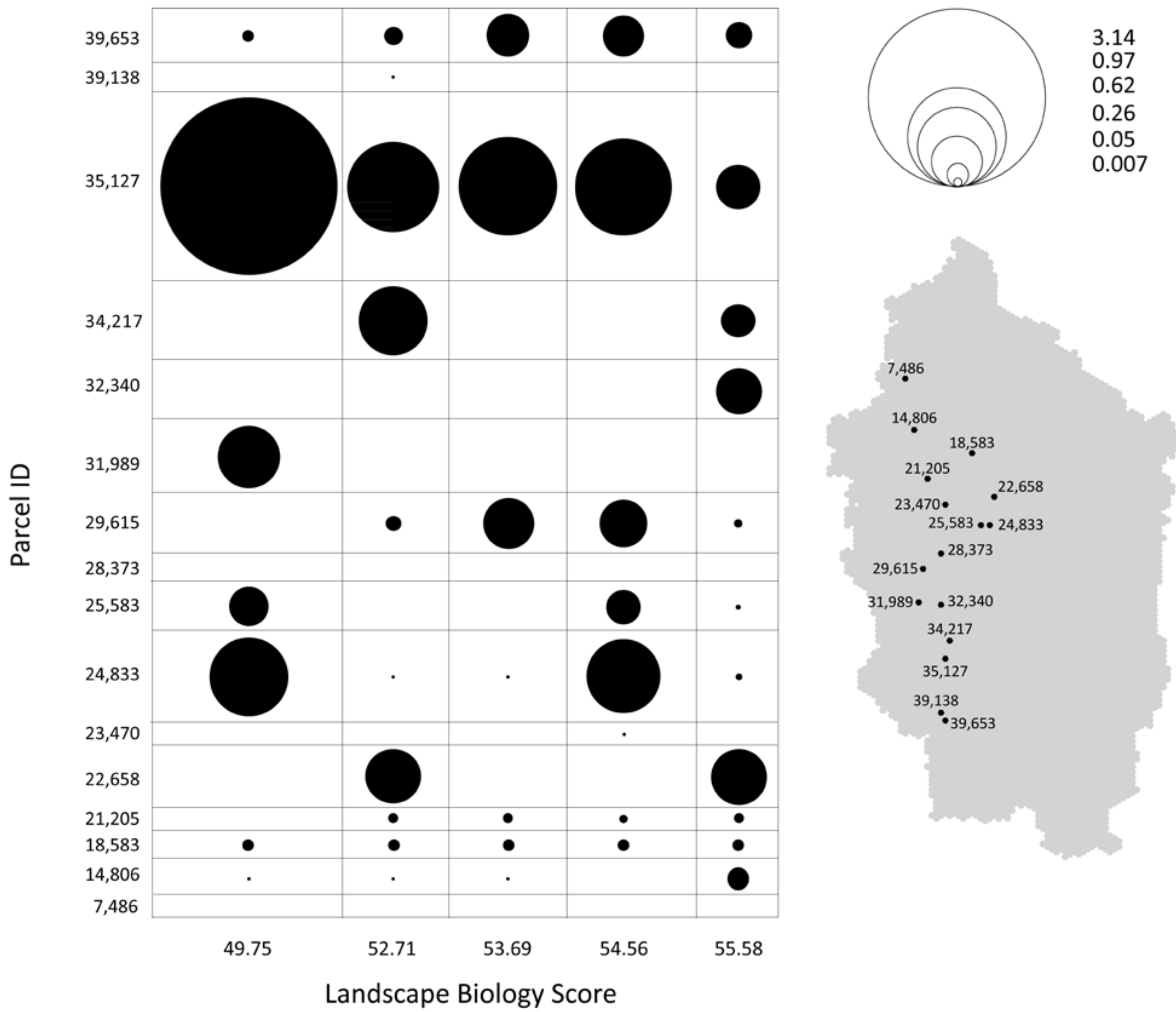


Figure 7 – Marginal Benefits for a Select Set of Optimally Conserved Parcels. Each column of bubbles gives the parcel's marginal biological score on a given optimal landscape (indicated by the biology score at the bottom of the figure). Blank cells either mean that the parcel was not part of the landscape's conservation network or its marginal biodiversity score was so small that it cannot even be represented by a visible point. The map on the right of the bubble diagram indicates each parcel's location on the landscape.



Endnotes

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[1] The authors are associate and assistant professors of economics at The University of Puget Sound and Bowdoin College, respectively. The authors acknowledge funding from the National Science Foundation's Collaborative Research Grants No. 0814424 (Lewis) and No. 0814628 (Nelson). Senior authorship is shared. Contact information: David Lewis, University of Puget Sound, 1500 N. Warner St., Tacoma, WA 98416, Ph: 253-879-3553, Email: djlewis@pugetsound.edu. Erik Nelson, Bowdoin College, 5000 South St., Brunswick, ME 04011, Ph: 207-725-3435, Email: enelson2@bowdoin.edu.
[2] Here we ignore another major type of government regulation associated with wildlife conservation, the direct appropriation of land. For example, in 1982 the Uganda government evicted approximately 4,500 families from land that became Lake Mburo National Park (Emerton 1999).
[3] An area is defined as habitat for a species if the species have been observed feeding or breeding on that land in the immediate past (Lueck and Michael 2003).
[4] Specifically, D has to be $C/p + U$ units greater than S for private and social incentives to align.
[5] There is some question as to how aggressively the ESA actually enforces takings on private land. In reality p_s may essentially be 0 for many private landowners.
[6] We assume that w can be reached on a piece of land that was developed but then was forced to institute some conservation due to the discovery of a taking. In reality the private nonmarket benefit on a parcel that was caught in a taking may not be reasonably restored to a nonmarket benefit level associated with the use of an HCP from the beginning.
[7] There is some question as to how aggressively the ESA actually enforces takings on private land. In reality F may essentially be 0 for many private landowners.
[8] We assume that t can be reached on a piece of land that was developed but then was forced to institute some conservation due to the discovery of a taking. In reality the public nonmarket benefit on a parcel that was caught in a taking may not be reasonably restored to a nonmarket benefit level associated with the use of an HCP from the beginning.
[9] There are other contingent cases where it is socially preferable for the landowner to engage in "shoot, shovel up, and shut-up" behavior. Inequality (2.15) also always holds if unfettered development or an HCP maximizes the right-hand-side of inequality (2.15) and $(1 - \overline{p_s}^*)(D + C - n - t - w) > c(p_s, \overline{p_s}^*)$. Inequality (2.15) also holds if an HCP maximizes the right-hand-side of inequality (2.15) and $(p_s - \overline{p_s}^*)(D + C - n - t - w) > c(p_s, \overline{p_s}^*)$. Contact author Nelson for a more detailed proof.
[10] Some listed species are actually subspecies while others are distinct populations of species (e.g., gray wolf populations in the northern Rockies versus Great Lakes). Here we refer to all listed entities as "species."
[11] Julie Cart, Species Protection Act "Broken": A Top Interior Officer Says the Law Should be Revised to Give Economic and Other Interests Equal Footing with Endangered Animals and Plants, L.A. Times, Nov. 14, 2003.
[12] Matthew Brown, "Deal struck to protect imperiled plants, animals", July 12, 2011, Associated Press.
[13] See Margules and Pressey (2000), Cabeza and Moilanen (2003), Moilanen et al. (2005), and Newbold and Siikamaki (2009) for other examples of RSS problems with more biologically meaningful objective functions.
[14] Examples of stochastic dynamic RSS are found in Costello and Polasky (2004), Snyder et al. (2005), and Haight et al. (2005).
[15] Biophysical benefits are maximized at a budget if \$20 when conserving adjacent parcels A, B, and C.