THREE ESSAYS IN RESOURCE ECONOMICS: PROTECTING NON-USE VALUES THROUGH ECOSYSTEM MANAGEMENT AND ESTIMATING RECREATIONAL DEMAND TO DETERMINE USE VALUES

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ABSTRACT

THREE ESSAYS IN RESOURCE ECONOMICS: PROTECTING NON-USE VALUES THROUGH ECOSYSTEM MANAGEMENT AND ESTIMATING RECREATIONAL DEMAND TO DETERMINE USE VALUES

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This dissertation consists of three essays that illustrate how the different ecological and economic tradeoffs accounted for in decision making affect estimates of value, with the ultimate goal of promoting more efficient ecosystem management. The first essay develops a bioeconomic model of managing excessive predation. A wildlife manager can reduce predation by removing the predator or by investing in exclosures to limit predator-prey interaction. Predator-prey dynamics are analogous to commercial exploitation of a renewable resource and predator removal and exclosures are analogous to entry fees and harvest quotas, respectively. Thus, the ecological model has relevance to more common human recreational hunting or commercial harvesting management problems. I find that using predator exclosures can yield a win-win outcome that increases both prey and predator populations. The model is parameterized and applied to the case of the Great Lakes Piping Plover, an endangered shorebird. I find that the solution to the Piping Plover problem is sensitive to the choice of economic values.

The second essay compares habitat creation and predator removal in conservation of the endangered Gaspésie Woodland Caribou. The caribou have suffered from habitat loss historically, but predation is currently a major limiting factor. In fact, land converted from caribou habitat to human use appears to favor increased predator survival. Using a bioeconomic model, I find that while increasing caribou habitat increases caribou survival and reduces caribou predation, a priori using caribou habitat as the only wildlife control is not first-best. The optimal management strategy involves a combination of predator removal and caribou habitat protection.

The third essay presents a model of the demand for Great Lakes fishing among Michigan recreational anglers. To control for travel cost endogeneity, two techniques are tested: alternative-specific constants and a control function. Both methods identify no evidence of travel cost endogeneity. Overall, model estimates predict that walleye, followed by chinook salmon, are the most valuable Great Lakes fish to Michigan anglers.

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INTRODUCTION

Ecosystems are valued in many different ways. There are use values (e.g., harvest values, stock-dependent eco-tourism values), which are interpreted from the observed choices of people. If these choices take place in markets, prices may reflect use value. Otherwise, these values may be measured econometrically. There are also non-use values for ecological resources, in which people are willing to pay to preserve resources they do not physically use. Some non-use values (e.g., stock-dependent existence values) can be measured econometrically. Other non-use values are ecosystem values that must be imputed from a combination of the estimated use and non-use values, key ecological relations and the manner in which the ecosystem is managed. Imputed values include the values of ecological services that stem from interactions between species as well as the total value of an ecological resource. Efficient ecosystem management requires the consideration of all these different values. It is therefore imperative that values are not ignored, poorly estimated or imputed incorrectly, or else the ability of economic analysis to inform ecosystem management choices may be diminished along with social welfare.

Ultimately, our ability to compute the relevant values depends on our ability to assess the tradeoffs implied by the choices made by individuals or resource managers. When economists use econometrics, values are estimated using statistical modeling that assesses the tradeoffs people make in their decision making, as reflected by observed behavior or stated preferences. When economists impute values to ecosystem services, applied modeling is used to examine the economic and ecological tradeoffs of ecosystem management decisions.

This dissertation illustrates how endogenous choices that account for important ecological and economic tradeoffs influence both imputed and econometrically-estimated values, with the ultimate goal of promoting more efficient ecosystem management. The approaches used here take into account factors that are often ignored in economic modeling, and thus can more accurately characterize use and non-use values in certain resource problems. In particular, two chapters address how the combination of estimated species' stock-dependent values, interspecies interactions and management options affect imputed species' values and hence the selection of efficient management strategies. The third chapter examines how endogenous choices not normally considered (i.e., location decisions affecting travel costs to recreational sites) in traditional valuation approaches may lead to improved estimates of recreational fishing values.

The first essay develops a bioeconomic model of a predator-prey system in which each species has estimated stock-dependent values (e.g, existence or eco-tourism values). These values, along with the management options, play a critical role in determining whether wildlife managers should impute a positive marginal value to the predator or view the predator as a nuisance species with negative marginal value. In turn, the imputed values are critical in the choice of controls to regulate species interactions. Whereas much of the economic literature on ecosystem management focuses on using only one type of conservation tool (primarily selective removal of animals or habitat preservation), I consider two different controls: predator exlcosures that reduce predation rates, and predator removal. I demonstrate that, when the predator has the type of stock-dependent values described above, using exclosures to regulate predation may preserve the values of both species, making a win-win outcome of more predators and more prey optimal.

The second chapter develops a bioeconomic model of a different predator-prey system in which exclosures are no longer an option, but for which habitat management affects predators and prey differently. Accordingly, I examine a different set of management controls: predator removal and habitat protection that simultaneously impacts the predator and prey. Habitat management, like exclosures, may ultimately reduce predation, but here the effect is indirect—the result of habitat-induced reductions in predator population levels rather than direct reductions in predation rates (the case with exclosures). I show that habitat protection, in conjunction with estimated stock-dependent values (e.g., existence or eco-tourism values), are critical in shaping ecosystem service values and that habitat protection can be used as an effective predator control. However, as habitat management reduces predator populations, a win-win outcome does not arise in this case. The result is that imputed predator values are reduced and there are fewer incentives to forego predator removal. The optimal management strategy may therefore involve a combination of predator removal and habitat protection.

The third chapter presents a model of the demand for recreational fishing in order to estimate use values for different species of fish. Similar demand models can also estimate non-use values. However, the traditional approach to modeling recreation demand assumes travel cost—a proxy for the price to recreation sites—is exogenous to the decision-maker. This may lead to biased measures of value when the travel cost is actually endogenous. To get unbiased estimates of value, I examine a corrective method known as the control function method. In the recreational fishing application, I find that travel cost is indeed endogenous, although the bias to estimates of fish and fishing site values is small.

CHAPTER 1

Controlling Excessive Predation with Predator Harvests and Predator Exclosures

1.1 Introduction

Ecosystems generally involve many complex interspecies interactions, including resource competition (Brock and Xepapadeas 2002; Tilman et al. 2005), mutualisms (Wacker 1999), and predator-prey relations (Ragozin and Brown 1985; Crocker and Tschirhart 1992; Ströbele and Wacker 1995; Hoekstra and van den Bergh 2005; Horan et al. 2011). Prior work has shown that human interventions may influence these stock-dependent interactions, so that management or exploitation of one species that does not account for interspecies interactions will generate spillover effects impacting other valued species. That is, human interventions may cause efficiency-reducing ecosystem externalities (Crocker and Tschirhart 1992).

Inefficiencies arising when humans do not intervene may be driven by species interactions. For instance, suppose people value greater species abundances, and no private incentives exist to intervene (i.e., assume harvests are not valued and habitat modification is prohibitively costly). Absent species interactions, each species would equilibrate at its carrying capacity, and this would be efficient: "nature's objective" of maximum species abundances coincides with human objectives. The distribution of species abundances is likely to be inefficient, however, if species interactions reduce the abundance of one or more valued species. These inefficiencies, arising because "nature's objectives" are affected by the interactions and hence diverge from human objectives (Brock and Xepapadeas 2002), are attributable to nature. This means ecosystem management may involve correcting both ecological and economic sources of inefficiencies.

Below, I explore ecosystem management to correct ecological inefficiencies. A key result is that recognizing the sources of ecological inefficiencies can help ecosystem managers select the right controls. Prior work on ecosystem management has focused primarily on setting the levels of controls to optimally account for interspecies interactions (Brock and Xepapadeas 2002; Ragozin and Brown 1985; Ströbele and Wacker 1995; Hoekstra and van den Bergh 2005; Crocker and Tschirhart 1992), with minimal discussion of which controls (e.g., harvest controls versus habitat interventions) promote greater efficiency. The selection of controls is not a trivial matter, however, especially for complex ecological systems (Horan et al. 2011).

I examine the selection of controls, and their optimal levels, by extending concepts from the joint-determination literature that seeks to more fully integrate economic and ecological systems (Tschirhart 2000, 2009; Brock and Xepapadeas 2002). An important strain of this literature anthropomorphizes species "behavior" to provide an economic interpretation for various ecological relations (Tschirhart 2000, 2009; Brock and Xepapadeas 2002; Tilman et al. 2005).[1] I expand on these concepts by showing how an individual animal's "behavior" creates ecological inefficiencies. For instance, predation creates detrimental impacts analogous to the externalities that arise when humans harvest open access resources: individual predators over-predate, reducing future prey stocks and thereby also reducing future predator fertility.[2] Humans

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[1] For instance, Tschirhart (2000; 2009) describes interspecies interactions as economic transactions involving energy flows among the interacting species, with the species "behaving" as if they were optimizing some biological objective (see also Brock and Xepapadeas 2002). Brock and Xepapadeas (2002) and Tilman et al. (2005) describe interspecies competition as being analogous to mechanistic resource competition among myopic humans. While not adopting an anthropomorphic perspective, Hoekstra and van den Bergh (2005), Ströbele and Wacker (1995), and Horan et al. (2011) explore problems where humans and (myopic) predators compete for the same prey.

[2] This is analogous to the notion that renewable resources under open access regimes can be interpreted as a predator-prey model in which humans are the predator (Ragozin and Brown 1985).

valuing one or more of these species are also affected. That interspecies interactions create ecological inefficiencies implies some species could be “regulated” to improve species’ efficiency. The literature on resource regulation can guide control choices to address these ecological inefficiencies (e.g., overharvesting). To this end, I show how specific ecosystem controls may be akin to particular regulatory instruments capable of correcting the relevant externalities in a human system.

The concepts are illustrated by the problem of conserving an existence-valued prey and predator species, where harvest values do not arise for either predator or prey (in contrast to virtually all the bioeconomic work on multi-species problems). This problem allows us to focus on reducing inefficiencies arising entirely from ecological sources. Moreover, the conservation of species that are threatened due to interspecies interactions, rather than overharvesting, is increasingly important (MEA 2005; Soulé et al. 2005). In particular, the multi-trophic nature of ecosystems is a key factor in species conservation (Soulé et al. 2005).[3]

As indicated above, relating un-managed predation to open access harvesting facilitates control choices and yields insight into their optimal values. License fees and harvest quotas can efficiently manage standard open access problems, and I find two controls can “regulate” predator-prey interactions in an analogous manner: predator removal to reduce the number of predators in the system, and predator exclosures to reduce the amount of prey each predator has access to (reducing predation per predator).

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[3] DeCesare et al. (2009) document the decline of over a dozen species from predation. Examples include predation by the Golden Eagle (*Aquila chrysaetos*) on the Island Fox (*Urocyon littoralis*) on the Channel Islands (Roemer et al. 2002), predation by gulls and falcons on the Piping Plover (*Charadrius melodus*), and predation by the Caspian tern (*Hydroprogne caspia*) on the endangered Snake River salmon (Antolos et al. 2005). A related issue is that interspecies competition can threaten predators. An example is the Bard Owl (*Strix varia*) displacing the Northern Spotted Owl (*Strix occidentalis caurina*) (Kelly et al. 2003).

I find that predator control utilizing predator exclosures can increase both the number of prey and predators under certain ecological conditions. This win-win outcome is akin to the standard result that efficient open access regulations simultaneously enhance economic welfare in the harvest sector and increase resource stocks.

The analogy to open access regulation is helpful but imperfect for two reasons. First, the type of values that ecosystem managers and open access harvesters place on the resource may be different. For example, open access harvesters may care only about the sale value of their catch while ecosystem managers may also be concerned with the noncommercial value of the stock of the resource. Second, investing in predator controls to regulate predator behavior generates real costs, whereas bioeconomic models generally assume negligible transactions costs from regulation. Together, these two distinctions affect the optimal mix of controls for predator management relative to the conventional open access setting. In particular, I find an exclosure-only strategy is optimal in some settings. Mathematically, this result is of interest because exclosures are non-targeted so as to affect both species (in contrast to targeted predator controls).

The analysis is applied to the case of the Great Lakes subpopulation of the Piping Plover, a small North American shorebird. In the mid-1980s there were only 28 Great Lakes plovers (Haig and Oring 1985). Since then the species has benefited from intensive management efforts, including productivity monitoring, predator exclosures, electrified fencing, predator removal, nest translocation, habitat enhancement, seasonal access closures and educational outreach (Gratto-Trevor and Abbott 2011). A significant portion of the birds' historical habitat is also now protected from development. However, the Great Lakes plover remains troubled by heavy loss rates to predators, especially small falcons known as Merlins (*Falco columbarius*) (V. Cavalieri, personal communication). This has led to the use of predator exclosures for protection. Predator exclosures are wire-mesh cages placed over the plovers' nests. The cages have gaps large enough for plovers to exit and enter, but small enough to stop predators. Alternatively, managers can remove predators directly, as they have in the past.

The next section develops the ecological plover-falcon model with predator removal and predator exclosures. A bioeconomic model of this problem is presented in section 1.3, and the economic-ecological tradeoffs arising in an optimal management regime are assessed. Numerical results are derived in section 1.4 for the Great Lakes Piping Plover. The final section concludes.

1.2 Ecology

1.2.1 Model with no controls

Consider an ecosystem composed of falcons and their prey, the plovers. The falcon population, denoted *Y*, changes over time according to

$$(1) \qquad \dot{Y} = g(X, Y)Y - \delta Y = \left(\varphi\left(\hat{Y} - Y\right) + \alpha\beta X - \delta\right)Y ,$$

where $g(\cdot)$ is falcon per-capita fertility, X is the plover stock, and $\delta > 0$ is the falcon mortality rate. The right-hand-side (RHS) term $\varphi(\hat{Y} - Y)$ is per capita growth absent plover predation, where $\varphi > 0$ and $\hat{Y} > 0$ are parameters. This growth, reflecting predation on other species, yields a carrying capacity of $\hat{Y} - \delta/\varphi > 0$ absent plover predation. The term βX is each falcon's plover predation, where β is a catchability parameter. Parameter $\alpha > 0$ converts captured plovers into new predators.

Growth of the plovers is described by

$$(2) \qquad \dot{X} = rX(1 - X / k) - \beta XY \ .$$

The first RHS term is net plover growth prior to falcon predation, where $r > 0$ is the intrinsic growth rate and k is the carrying capacity absent predation. The second RHS term is falcon predation.

1.2.2 Predation as an open access problem: ecosystem externalities and control measures

The predator-prey dynamics (1)-(2) are analogous to an open-access resource extraction problem with sluggish entry and exit (e.g. Smith 1968): equation (2) is the equation of motion for the harvested resource and equation (1) is the equation of motion for entry and exit of the harvesters. Under the entry-exit interpretation for (1), the net fertility relation $g(X,Y) - \delta$ is the predator's welfare measure analogous to per capita rents, where $g(\cdot)$ is the return to predation and δ is a fixed survival cost.[4] Specifically, in the relation $g(\cdot)$, $\alpha\beta X$ is analogous to total revenue from harvesting plovers, $\varphi \hat{Y}$ is net revenue from harvesting other species, and $-\varphi Y$ is a congestion externality related to predation on other resources. This perspective is in line with mechanistic models of open access, where the choice of each harvester is not modeled explicitly. Rather, harvest rate per harvester (β) is assumed to be fixed, and entry and exit of harvesters is modeled as a response to changes in rents. This is a common approach (see Clark, 2005).

The predator-prey dynamics embed two ecosystem forces that overexploit prey stocks in the same way that resource stocks are overexploited in conventional open access problems: (i) individual predators, each harvesting at the constant level βX, harvest too much because they have no incentive to preserve plovers, and (ii) too many predators enter the system, over depleting the prey stock and causing congestion. This latter effect arises as, just like profit-motivated firms, falcons enter (i.e., are born into) the system until rents are driven to zero. These two effects are akin to the externalities that arise under open access. Viewing the conservation

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[4] Likewise, evolutionary investments myopically maximize net fertility in evolution models (Rice 2004).

problem in this way yields insight into the types of wildlife controls to consider. Specifically, it is known that the two open access externalities can be corrected in conventional settings using harvest taxes (to correct both externalities) or a combination of per-harvester harvest limits/quotas (correcting excess harvests) and entry restrictions or license fees (correcting excess entry). The key is to identify wildlife controls that perform similar functions.

Two such controls are identified: falcon removal and predator exclosures.[5] Define falcon removal by *hY*, where *h* is the removal rate. The second control, predator exclosures, excludes a portion of prey habitat from predators.[6] Managers use this control to shield a proportion, *p*, of plovers from falcons while leaving the remainder susceptible to predation. With these controls, the falcon-plover dynamics (1)-(2) become

$$\dot{X} = rX(1 - X/k) - \beta XY(1-p). \tag{3}$$

$$\dot{Y} = \left(\varphi\left(\hat{Y} - Y\right) + \alpha\beta X(1-p) - \delta - h\right)Y \tag{4}$$

It is apparent from system (3)-(4) that *p* works like a quota and *h* works like a license fee. Just like quotas, exclosures reduce each falcon's predation from βX to $\beta X(1-p)$, reducing predation

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[5] Removal of plovers is not considered because reducing the plover stock harms both species and because it is assumed that (a) there are stock-dependent benefits (existence values) for both species, and (b) there are no positive use values for either species (see section 3.1). With these assumptions, it would never be optimal to remove plovers.

[6] Predator exclosures often fence off area(s) the prey frequents. In the case of Piping Plovers the areas fenced in are the birds' nests. Gaps in the fencing material are big enough to allow the plovers to move in and out of the exclosure but are inaccessible to predators like Merlins. Electrified fences have been used to this purpose for decades (Mayer and Ryan 1991). Practically speaking, exclosures are best suited against ground-based predators, although over-hanging lines can shield against avian predators and nets might be used in an aquatic environment. Evidence indicates fencing off terrestrial areas is effective in limiting the movements of predators (Moseby and Read 2006) and reducing predation (Lokemoen et al. 1882; Mayer and Ryan 1991; Bennett et al. 2009).

returns and leaving more plovers *in situ*.[7] Predator removal is analogous to a lump-sum license fee on harvesters, in that it reduces net returns by increasing the survival cost. The result is that *h* slows entry and speeds exit of falcons, the same as a license fee.

System (3)-(4) is analogous to a special class of open access problems where there are no variable production costs, so that an individual harvester's scale of production is not a concern. In this class of problems, and assuming a traditional setting where the manager costlessly implements regulations to maximize discounted rents from a single renewable resource, only one control–a quota or a license fee–is needed to achieve the efficient outcome. This is the case for (3)-(4) when $\varphi = 0$ (i.e., a single-species model with no congestion externalities) and there are no transactions costs of regulation (see Appendix A). The use of both instruments is efficient in the presence of a congestion externality, i.e., when $\varphi > 0$. These results also hold when social welfare is defined as discounted stock-dependent (existence) values rather than resource rents, although there will be greater marginal incentives to use exclosures when *Y* is valued directly (see Appendix A).

These results suggest that two controls may optimally manage both populations in the predator-prey model. However, the predator-prey model differs from traditional open access models in an important way that could impact this result. Specifically, the controls are not implemented costlessly, as shown in the bioeconomic model in section 3. The relative costs of the controls, viewed as transactions costs under the open access analogy, will influence the relative magnitude(s) of the control(s)–including whether more than one control is optimal.

The steady state of system (3)-(4) is examined to gain insight into the ecological impacts of these controls, with a particular focus on the impacts to the predator population as over-

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[7] Control *p* may also look like a harvest tax in (4). However, taxes do not affect prey dynamics (3), whereas *p* does.

predation is reduced. For simplicity, the analysis here assumes constant values for the controls *p* and *h* (this is relaxed in the bioeconomic analysis below). The steady state populations of *X* and *Y*, X^* and Y^*, are solved by setting (3) and (4) equal to zero:

(5) $$X^* = Max\left(\frac{k\varphi\left[r-\beta(1-p)\left(\hat{Y}-[h+\delta]/\varphi\right)\right]}{k\alpha\beta^2(1-p)^2+r\varphi},0\right) = Max\left(\frac{k\varphi\left[r-\beta(1-p)Y^*\,|_{X\to 0}\right]}{k\alpha\beta^2(1-p)^2+r\varphi},0\right)$$

(6) $$Y^* = \frac{r\varphi\left[\hat{Y}-(h+\delta)/\varphi+k\alpha\beta(1-p)/\varphi\right]}{k\alpha\beta^2(1-p)^2+r\varphi} = \frac{r\varphi Y^*\,|_{X\to k}}{k\alpha\beta^2(1-p)^2+r\varphi}$$

where $Y^*\,|_{X\to 0}=\hat{Y}-[h+\delta]/\varphi$ is the equilibrium predator population that would emerge in the special case where $X\to 0$, and $Y^*\,|_{X\to k}$ is the equilibrium predator population that would emerge in the special case where $X\to k$. Equation (5) indicates $X^* > 0$ when the prey's intrinsic growth rate exceeds the predation pressure that occurs as $X\to 0$. In what follows assume $X^* > 0$.

Taking derivatives of X^* and Y^*, predator removal (*h*) is found to unambiguously increase the equilibrium prey population but decrease the equilibrium predator population:

(7) $$\frac{\partial X^*}{\partial h} = \frac{k\beta(1-p)}{k\alpha\beta^2(1-p)^2+r\varphi} > 0$$

(8) $$\frac{\partial Y^*}{\partial h} = \frac{-r}{k\alpha\beta^2(1-p)^2+r\varphi} < 0$$

Predator exclosures (*p*) increase the long-run prey stock, but the effect on predators is ambiguous:

(9) $$\frac{\partial X^*}{\partial p} = \frac{k\varphi\left[\beta Y^*\,|_{X\to 0}\right]+2k\alpha\beta^2(1-p)X^*}{k\alpha\beta^2(1-p)^2+r\varphi} > 0$$

$$\begin{aligned}\frac{\partial Y^*}{\partial p} &= \frac{\left[k\alpha\beta^2(1-p)^2 + r\varphi\right] r\sigma\left[\partial Y^* |_{X\to k} / \partial p\right] + 2k\alpha\beta^2(1-p)r\varphi\left[Y^* |_{X\to k}\right]}{[k\alpha\beta^2(1-p)^2 + r\varphi]^2} \\ &= \frac{\left[2\beta(1-p)Y^* - r\right]k\alpha\beta}{k\alpha\beta^2(1-p)^2 + r\varphi} = \frac{\left(r - 2rX^*/k\right)k\alpha\beta}{k\alpha\beta^2(1-p)^2 + r\varphi}\end{aligned} \tag{10}$$

where the final equality in (10) stems from the steady state condition for X: $r(1 - X^*/k) = \beta(1 - p)Y^*$. Relation (10) indicates predator exclosures increase Y^* when $r > 2rX^*/k$, which occurs when $X^* < X^{MSY}$, where X^{MSY} is the plover population supporting the maximum sustainable yield of plovers to falcons.[8] When $X^* < X^{MSY}$, an increase in p increases both the plover stock and the yield to falcons, increasing the falcon stock. The interests of society and the falcon population coincide (qualitatively) in this case, so that regulating predation increases plover and falcon stocks. In contrast, the plover yield and the falcon stock decline for larger values of p when $X^* > X^{MSY}$.

These results echo Sanchirico and Wilen's (2001) findings that prohibiting harvests in part of a fishery (i.e., creating a reserve) can generate a "double payoff" of more fish and larger harvests. The size of the reserve and non-reserve are fixed in their model, but they show that a double payoff is possible if the rate at which the reserve stock disperses into the open-access zone is not too small relative to the growth rate of the reserve stock. In the model here, investing in exclosures is equivalent to reserve creation, and managers have control over the rate prey are consumed, which is akin to Sanchirico and Wilen's reserve dispersal rate (in this case, dispersal is of plovers becoming falcon prey). Thus, the conclusion is similar: if the rate of predation is not

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[8] This is likely the case for systems where predation significantly increases prey extinction risks, as there is evidence predators harvest their prey below MSY (see Beddington et al. 1978; Taylor 1981; Seip 1991).

reduced too much (i.e. the dispersal rate remains high enough), managers can achieve a double payoff.

Unlike Sanchirico and Wilen, whose marine reserve is costlessly established, predator controls have real economic costs. This increases the pertinence of the present model to open access problems, enriching the current understanding of open access resource regulation. This is because regulation of open access harvesting actually requires compliance and enforcement costs (Anderson 1989). These control costs are considered in the following bioeconomic section.

1.3 Bioeconomics

1.3.1 The manager's problem and necessary conditions

Suppose society derives stock-dependent benefits from each species. For simplicity, I refer to these benefits as existence values, although use values (e.g. bird watching) could also be a source of stock-dependent benefits. Existence values take the form $B_X(X) + B_Y(Y)$, where B_X and B_Y are increasing and concave in their respective stocks. Harvests do not generate positive use values for either species.

Conservation efforts are costly.[9] Predator removal costs take the Schaefer form (Clark 2005), $c_h R/Y$, where c_h is a parameter and $R=hY$ is the number of predators removed. Removal costs are increasing in the number of predators removed, but declining in Y as it becomes increasingly costly to locate and remove the last few falcons. Since $R = hY$, the cost of removal

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[9] Costs associated with the negative social perceptions of predator removal (i.e., a disutility associated with predator removal h) are not modeled. For plovers, there is evidence people are willing to support predator removal (Messmer et al. 1999), suggesting any social costs associated with predator removal are probably negligible in this case.

as a function of the rate of removal is $c_h h$. Exclosure costs take the form, $c_p p/(1-p)$, where c_p is a parameter. These costs are increasing and convex in p, as it becomes increasingly costly to find and protect remaining unprotected plovers as the proportion of exclosure protection increases. The form of the exclosure cost function is chosen because it ensures $p = 1$ is suboptimal as this implies infinite costs.

The social planner's problem is to maximize discounted social net benefits, denoted *SNB*:

$$\max_{h,p} \quad SNB = \int_0^\infty \left(B_X(X) + B_Y(Y) - c_h h - c_p p/(1-p)\right)e^{-\rho t}dt$$
$$s.t.\ \ (3),(4)\quad X(0) = X_0,\ \ Y(0) = Y_0,\ \ p \in [0,1],\ \ h \geq 0,$$

where ρ is the discount rate. Although the problem's ecology is analogous to open access dynamics, *SNB* differs in two ways from welfare in traditional open access problems: (i) falcon-related benefits depend on the stock, not on rents (net fertility), and (ii) there are real costs to implementing predator controls, whereas implementation of license fees and quotas in open access regulatory settings is generally assumed to be costless.

The current value Hamiltonian for the planner's problem is

$$(11)\qquad H(X,Y,h,p,\lambda_X,\lambda_Y) = B_X(X) + B_Y(Y) - c_h h - c_p p/(1-p) + \lambda_X \dot{X} + \lambda_Y \dot{Y},$$

where λ_X and λ_Y are the costate variables for plovers and falcons, respectively. The Lagrangean is $L = H + \mu p$, where μ is the Langrangian multiplier for the lower-bound constraint on p. Other constraints are treated implicitly. In what follows, the superscript * denotes an optimal trajectory.

The Lagrangean is linear in h, yielding a linear control problem in this variable. Accordingly, the optimality condition for h is (Clark, 2005):

$$(12)\qquad \frac{\partial L}{\partial h} = -c_h - \lambda_Y Y \begin{matrix} > 0 & iff & h^* \to \infty \\ = 0 & iff & h^* = h_{SV} \\ < 0 & iff & h^* = 0 \end{matrix} .$$

Condition (12) states that h is used as an impulse control when $\partial L/\partial h > 0$. Alternatively, no predator removal should occur if $\partial L/\partial h < 0$. The singular solution, h_{SV}, is optimally adopted when $\partial L/\partial h = 0$. The relation $\sigma(Y,\lambda_Y) = -c_h - \lambda_Y Y$ is known as the switching function (Clark, 2005), as this function determines when h optimally switches from one extreme to the other. Note that $\lambda_Y = -c_h/Y < 0$ when $\sigma(Y,\lambda_Y) = 0$. Thus, when h follows a singular (interior) solution, the shadow price of the predator is negative—i.e., the falcons are a nuisance. This is intuitive: managers only remove predators when the marginal predator has negative value. In contrast, if λ_Y is positive and large, so that the marginal predator is valued, $\sigma(Y,\lambda_Y) < 0$ and no predators are removed.

The necessary (Kuhn-Tucker) conditions related to p are

$$(13)\qquad \frac{\partial L}{\partial p} = \frac{-c_p}{(1-p)^2} + (\lambda_X - \alpha\lambda_Y)\beta XY + \mu = 0 \Rightarrow p^*$$

$$(14)\qquad \frac{\partial L}{\partial \mu} \geq 0; \quad \mu\left[\frac{\partial L}{\partial \mu}\right] = 0 .$$

The shadow price of plovers, λ_X, is always positive because $B_X > 0$ and plovers do not detrimentally affect falcons. When $\mu > 0$, then $p^* = 0$. This is optimal when $-c_p/(1 - p) + (\lambda_X - \alpha\lambda_Y)\beta XY < 0$, which occurs when the net marginal cost of falcon predation on plovers, $\lambda_X - \alpha\lambda_Y$, is sufficiently small—i.e., society derives little welfare from protecting a plover from predation. An interior trajectory for p is followed when $\mu = 0$, which can only occur when $\lambda_X - \alpha\lambda_Y > 0$.

This latter condition implies that $p > 0$ does not require $\lambda_Y < 0$, as was required for $h > 0$. Rather, exclosures could be optimal in cases when the marginal falcon is valued positively. Therefore, while $\lambda_Y > 0$ implies $h^* = 0$, it does not mean that predator management in general is undesirable.

Two adjoint conditions, $\dot{\lambda}_i = \rho\lambda_i - \partial L/\partial i$ (i=X,Y), are also necessary. I write these conditions in golden rule form (Clark 2005). The golden rule condition for the plover stock is

$$(15) \qquad \rho = r\left(1 - \frac{2X}{K}\right) + \left(\frac{B'_X}{\lambda_X}\right) + \left(\frac{\dot{\lambda}_X}{\lambda_X}\right) - (\lambda_X - \alpha\lambda_Y)\frac{(1-p)\beta Y}{\lambda_X}.$$

This relation equates the rate of return that could be earned elsewhere, ρ, to the net rate of return from conserving plovers (the RHS of (15)). The first RHS term is the marginal growth of plovers prior to falcon predation. The second RHS term is the marginal existence benefit of plovers. The third RHS term represents the capital gain or loss from changes in the plover stock. Finally, when $\lambda_X - \alpha\lambda_Y > 0$, the fourth RHS term is the net marginal cost of greater falcon predation in response to more plovers. When $\lambda_X - \alpha\lambda_Y > 0$, these costs are declining in p. Alternatively, when $\lambda_X - \alpha\lambda_Y < 0$ then the fourth RHS term is the net marginal benefit of predation and, from (14), it must be that $\mu > 0$ and so $p = 0$.

The golden rule condition for conserving the falcons is

$$(16) \qquad \rho = \left[\varphi\left(\hat{Y} - 2Y\right) - \delta - h\right] + \left(\frac{B'_Y}{\lambda_Y}\right) + \left(\frac{\dot{\lambda}_Y}{\lambda_Y}\right) - (\lambda_X - \alpha\lambda_Y)\frac{(1-p)\beta X}{\lambda_Y}.$$

When $\lambda_Y > 0$, the interpretation of (16) is similar to (15): the rate of return ρ is equated to the net return to falcon conservation. The first RHS term in (16) is the marginal growth rate of falcons prior to predation on plovers. Note that, from (12), $h^* = 0$ when $\lambda_Y > 0$ and so the rate of return to falcon conservation is not influenced by predator removal. The second RHS term is the marginal existence value of falcons. The third RHS term is the capital gain or loss. Finally, when $\lambda_X - \alpha\lambda_Y > 0$, the final RHS term is the net marginal cost of greater predation on plovers in response to more falcons. When $\lambda_X - \alpha\lambda_Y > 0$, these costs are declining in *p*, so that a larger *p* increases the return to conserving falcons. Alternatively, when $\lambda_X - \alpha\lambda_Y < 0$ then the fourth RHS term is the net marginal benefit of predation, with $\mu > 0$ and so $p = 0$. These results and those for (15) indicate that, when $\lambda_Y > 0$, it is only optimal to use exclosures when these increase the rate of return on both stocks.

The interpretation of (16) changes when $\lambda_Y < 0$. In that case, ρ represents the opportunity cost of pulling resources from elsewhere in the economy and using them to manage falcons as a nuisance. The RHS represents the rate of return to controlling nuisance falcons. This rate of return is increasing in the marginal growth of falcons prior to predation on plovers (the first RHS term). The second RHS term indicates the rate of return to nuisance control is decreasing in marginal existence values for falcons. The capital gain/loss term changes in sign when $\lambda_Y < 0$. Finally, consider the fourth RHS term. When falcons are a nuisance ($\lambda_Y < 0$), then the rate of return to falcon control is increasing in response to greater falcon predation on plovers at the margin (since $\lambda_X - \alpha\lambda_Y > 0$). Note that both predator removal, *h*, and predator exclosures, *p*, reduce the rate of return to nuisance control, suggesting there are diminishing returns to managing nuisance falcons.

1.3.2 Candidate management strategies

Conditions (12) – (14) imply the solution could be interior, in which $h^* = h_{SV}$ and $p^* > 0$, or a corner, in which one control is constrained while the other is free. Some combinations involving corner solutions can be discarded as candidate long-run strategies. For instance, $h \rightarrow \infty$ cannot persist for more than an instant or else $Y \rightarrow 0$ and removal costs become infinite; however, $h \rightarrow \infty$ can be used as an impulse control to move to a particular long-run trajectory. Also, $p = 1$ was ruled out as either a short-run or long-run solution. Finally, corner solutions with either $h = 0$ or $p = 0$ can be part of the long-run optimal trajectory, or they can be used to move the system to such a trajectory. The remainder of this section focuses on each of four strategies that can hold for a period of time: (A) no management ($h = p = 0$), (B) predator removal and exclosures ($h = h_{SV}$, $0 < p < 1$), (C) predator removal only ($h = h_{SV}$, $p = 0$), and (D) exclosure only ($h = 0$, $0 < p < 1$).

Strategy A: *No management (h = p = 0)*

The no-management strategy yields the system (1)-(2). It is currently believed that following this strategy indefinitely will lead to plover extinction. In fact, the numerical analysis below is parameterized as such (also see Appendix B). However, if falcons were eradicated prior to following the no-management strategy, then the no-management strategy would yield $X \rightarrow K$. Numerically, it is not possible to eradicate falcons in finite time when removal is defined as a rate, although a sufficient reduction in the population may be interpreted as eradication.

Strategy B: *Predator removal and exclosures ($h = h_{SV}$, $0 < p < 1$)*

To derive this candidate solution, first set $\partial L/\partial h = 0$ in (12) to obtain

$$(17) \qquad \lambda_Y(Y) = \frac{-c_h}{Y} \Rightarrow \dot{\lambda}_Y = -\lambda_Y \frac{\dot{Y}}{Y},$$

Next, substitute (17) into (16) and solve for λ_X:

(18) $$\lambda_X(X,Y,p) = \frac{B'_Y + (c_h/Y)(\rho + \varphi Y)}{\beta(1-p)X} \ .$$

Then use (13), with $\mu = 0$, to derive the following expression,

(19) $$\frac{c_p}{(1-p)^2} - (\lambda_X(X,Y,p) - \alpha\lambda_Y(Y))\beta XY = 0\,,$$

which implicitly defines the feedback relation $p = p^B(X,Y)$. Substituting $p^B(X,Y)$ into (18) yields $\lambda_X(X,Y,p(X,Y)) = \Phi(X,Y)$. Take the time derivative of Φ, $\dot{\Phi} = (\partial\Phi/\partial X)\dot{X} + (\partial\Phi/\partial Y)\dot{Y}$, and substitute $\dot{\Phi}$ and Φ into (15) to derive the feedback relation $h_{SV} = h^B(X,Y)$. Strategy B's dynamics are determined by substituting $p^B(X,Y)$ and $h^B(X,Y)$ into system (3)-(4).

Strategy C: Predator removal only ($h = h_{SV}$, $p = 0$)

To find the optimal trajectory in this case, set $\partial L/\partial h = 0$ in (12) and solve for λ_Y and $\dot{\lambda}_Y$ as in (17). Then substitute these relations into (16) to derive

(20) $$\lambda_X(X,Y) = \frac{B'_Y + (c_h/Y)(\rho + \varphi Y)}{\beta X}$$

which is just (18) with $p = 0$. Note that for $p = 0$ to be optimal, it must be that $\lambda_X(X,Y) - \alpha\lambda_Y(X,Y) < c_p/\beta XY$; assume this is the case. Take the time derivative of (20), $d\lambda_X(X,Y)/dt$, and substitute this relation and $\lambda_X(X,Y)$ into (15) to solve for $h_{SV} = h^C(X,Y)$. The dynamics for strategy C are determined by substituting $p = 0$ and the feedback solution $h^C(X,Y)$ into system (3)-(4).

Strategy D: *Predator exclosures only (h = 0, 0 < p < 1)*

For this case, use (13) to write p in terms of λ_X and λ_Y:

$$p = 1 - \sqrt{\frac{c_p}{(\lambda_X - \alpha\lambda_Y)\beta XY}}\,, \tag{21}$$

so that $p = p(X,Y,\lambda_X,\lambda_Y)$. From (13), note that $0 < p^* < 1$ only if $\lambda_X - \alpha\lambda_Y > c_p/\beta XY$. To find λ_X and λ_Y one solves the adjoint conditions associated with (15) and (16). An analytical solution is not possible, so the solution must be determined numerically. Actually, in this case the optimal management regime will be governed by four differential equations, $\dot{\lambda}_X$, $\dot{\lambda}_Y$, $\dot{X}$,$\dot{Y}$ (from (3)-(4) and (15)-(16), with $h = 0$ and $p = p^*$), rather than just $\dot{X}$ and $\dot{Y}$, as in cases A, B and C. As the two costates only affect the system through p, it is possible to describe the numerical solution (see the next section) in three dimensions as moving through (X,Y,p)-space, with $\dot{p} = dp(X,Y,\lambda_X,\lambda_Y)/dt$. Note that $X(0)$ and $Y(0)$ are given, whereas $\lambda_X(0)$ and $\lambda_Y(0)$–and hence $p(0)$ –are optimally determined. Hence, the solution effectively involves choosing $p(0)$ to place the system on the optimal trajectory (i.e., a three-dimensional saddle path), given the initial values of the state variables X and Y. The system then optimally follows the trajectory as determined by the differential system. This is akin to a traditional resource management problem involving a single management choice (e.g., harvests) and a single species.

1.4 Numerical Example

Piping Plovers are divided into three distinct subpopulations that nest on the Atlantic coast, the Great Lakes and the Great Plains. Their nesting habitat consists primarily of beaches that have been subjected to significant development and recreational use. Hunting in the early 1900s considerably reduced Piping Plover numbers, while rising beach use in the mid-1900s renewed these declines. Recent recovery efforts have led to partial recovery and the IUCN upgrading the species from Threatened to Near Threatened. However, the Great Lakes subpopulation (residing mostly in Michigan), which is the focus of the numerical analysis, remains precariously small (IUCN 2010) and is officially endangered under the Endangered Species Act (Gratto-Trevor and Abbott 2011).

Predation is now a significant limiting factor to recovery (Rimmer and Deblinger 1990). Population modeling predicts the Great Lakes Piping Plover will go extinct within the next century if predation rates are not reduced (Plissner and Haig 2000; Wemmer et al. 2001). The federal recovery plan for plovers outlines emergency anti-extirpation methods (USFWS 2003), including predator removal and protective nest exclosures (Gratto-Trevor and Abbott 2011). I examine these approaches, as studies indicate they can be successful (Mayer and Ryan 1991; Struthers and Ryan 2005). Piping Plover managers use protective exclosures as the primary anti-predator tool, while predator removal is used only marginally.[10]

The plover predator of most concern is the Merlin, a small falcon (V. Cavalieri, personal communication). Merlins have been consistently observed predating on plover adults and chicks. These falcons are protected as a threatened species under the Endangered Species Act of the State of Michigan (MNFI 2011). Their current population in Michigan is unknown, but their numbers have been increasing in recent years (V. Cavalieri, personal communication). Although

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[10] In the Great Lakes recovery plan, predator removal receives about 1/10th the funding of protective exclosures (USFWS 2003). Gratto-Trevor and Abbott (2011) find predator removal is used less extensively than exclosures in every meta-population management region, although Great Lakes plover managers use it more extensively than others. Part of the reason is concern that, despite evidence to the contrary, predator removal is perceived negatively by the public (USFWS 2003).

these falcons may roam between areas of plover habitat and the rest of the state, I assume a fixed subpopulation hunting in the vicinity of plover habitat, to keep the model tractable.

Economic and ecological parameter values for the benchmark scenario of the numerical analysis are listed in Table 1.1, with calibration of the model described in the Appendix. Functional forms for the model have already been described, with the exception of the social benefits derived from each stock. I assume $B_X(X) = u_X \ln(X+1)$ and $B_Y(Y) = u_Y \ln(Y+1)$, where u_X and u_Y are parameters. Benefits are increasing and convex and marginal benefits remaing finite when $X \rightarrow 0$, $Y \rightarrow 0$, so extinction does not imply negative social welfare.

1.4.1 Results for the benchmark scenario

The optimal solution is determined by examining each candidate strategy, A-D, in turn. The numerical solutions were derived using Mathematica 7.0 (Wolfram 2008).

Strategy A: No management ($h = 0$, $p = 0$)

The phase plane for strategy A is presented in Figure 1.1. The $\dot{X} = 0$ and $\dot{Y} = 0$ isoclines do not intersect in the positive orthant, indicating there is no equilibrium in which both species co-exist. The phase arrows indicate plover extinction is a globally stable outcome if strategy A is maintained without a switch to an alternative strategy. Despite this outcome, strategy A remains a candidate long-run strategy, because nothing in the formulation of the social planner's problem precludes plover extinction as a feasible equilibrium. If this strategy is chosen from the start, plovers go extinct, falcons attain an equilibrium population of 195, and *SNB* = \$724 million.

Strategy B: Predator removal and exclosures ($h = h_{SV}$, $0 < p < 1$)

The phase plane for this case is presented in Figure 1.2. The phase arrows indicate the direction of potential trajectories. By definition, all possible trajectories are switching curves for h since $\sigma(Y,\lambda_Y) = 0$ along each trajectory. Once on such a trajectory, there is no reason to switch off unless the trajectory enters some space where $h_{SV}(X,Y)$ becomes infeasible. This is indicated by the $h_{SV} = 0$ boundary: above this curve, $h_{SV}(X,Y) < 0$, and the singular solution is infeasible, while below the curve $h_{SV}(X,Y) > 0$. A similar logic holds for p, although a $p^*(X,Y) = 0$ boundary is not drawn because at this scale it is not distinguishable from the X-axis: above this curve, $p^*(X,Y) > 0$.

The phase dynamics are governed by the saddle point equilibrium at the intersection of the isoclines. This equilibrium, and the portion of the saddle path that converges to this outcome, lie above the $h_{SV} = 0$ boundary and are therefore infeasible. The only feasible trajectories are those that exist below the $h_{SV} = 0$ boundary. Trajectories that start below the boundary and then intersect the boundary can only be followed in the short-run, with strategy D ($h = 0$, $0 < p < 1$) then being pursued immediately upon reaching the boundary. All other trajectories that start below the boundary eventually intersect the X-axis, resulting in a long-run equilibrium of $Y \rightarrow 0$, $X \rightarrow K$. Extinction of Merlins can only be attained, however, if $h \rightarrow \infty$, implying infinite predator removal costs.[11] So these paths, and strategy B, can be discarded as candidates for the long-run optimal trajectory in this case.

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[11] In fact, $Y = 1$ before $h \rightarrow \infty$, which is effectively eradication. One could assume eradication is achieved when a strategy B trajectory leads to $Y = 1$ (so that further control is unnecessary), but this is also found to be suboptimal.

Strategy C: Predator removal only ($h = h_{SV}$, $p = 0$)

Figure 1.3 presents the phase plane for strategy C. The dynamics are governed by the saddle point equilibrium at the intersection of the isoclines. Unlike strategy B, the equilibrium and saddle path of strategy C lie below the $h_{SV} = 0$ boundary. However, the saddle path and virtually all trajectories in the state space lie above the $p^*(X,Y) = 0$ boundary (lying minimally north of the X-axis), indicating that some positive level of p is optimal. This contradicts the formulation of strategy C, so that this strategy is sub-optimal.

Strategy D: Predator exclosures only ($h = 0$, $0 < p < 1$)

Finally, consider strategy D, which is illustrated by the three-dimensional system presented in Figure 1.4. Although the solution for this strategy is characterized by a dynamic system in $(X,Y,\lambda_X,\lambda_Y)$-space, it is possible to graphically represent the solution in (X,Y,p)-space, as described earlier in section 1.3.2. The $\dot{X} = 0$ and $\dot{Y} = 0$ isoplanes are illustrated in Figure 1.4. The isoplanes intersect to form a locus of potential equilibria. In Figure 1.4, this equilibrium is where the saddle path intersects the $\dot{X} = 0$ and $\dot{Y} = 0$ isoplanes. The initial state, (X_0,Y_0), illustrated by a vertical line in Figure 1.4, has a unique saddle path. Indeed, there will be different paths for different initial states. The optimal strategy is to choose $p(0)$ to put the system on the saddle path and follow it to the equilibrium. Choosing the initial p to start the system on some other path leads to either $p \to 1$ and infinite costs (which cannot be optimal), $\sigma(Y,\lambda_Y) = 0$ (and a switch to strategy B, which is found to be not optimal), or $p \to 0$ (and a switch to strategy A). The last case, which involves an eventual switch to strategy A, merely delays extinction. Therefore, the only feasible long-run strategy D management program is to select $p(0)$ to place the system on the saddle path and then proceed along this path. Along this path, the plover population increases from 126 to 260, and the falcon population increases slightly from 195 to 197. I do not illustrate $p^*(t)$, as this value changes little over time, monotonically declining from approximately 0.901 to 0.886.

Numerically, strategy D yields *SNB* = \$993 million, which exceeds the value of strategy A. Compared to the no-management scenario A, strategy D yields a win-win outcome in that there are more plovers and falcons. In this case, the interests of society and falcons coincide, so that predator control improves both social welfare and the “welfare” of falcons. The result is that strategy D (exclosure-only) is the optimal management strategy overall. Note that the use of a single control differs from the traditional open access case described earlier, which indicated two controls were optimal. There are two reasons for this difference: in the current setting, predator removal involves real costs, and predators are valued directly so that predator removal implies an additional social cost.

1.4.2 Sensitivity Analysis

A parameter sensitivity analysis (Table 1.2) yields further insights. I first examine the role of Merlin existence values. Richardson and Loomis’ (2009) results, which were used to calibrate $B_Y(Y)$, are based on the assumption that the public considers the Merlins to be at risk. While Merlins are officially threatened in Michigan, the IUCN (2010) lists Merlins as a species of Least Concern and so the public may not consider them at risk. Suppose $u_Y = 1/2 \cdot u_{Y0}$ (where ‘$_0$’ denotes a benchmark value). In this case a win-win outcome yielding more falcons confers fewer benefits, implying fewer incentives to use exclosures. Indeed, strategy B ($h = h_{SV}$, $0 < p < 1$) is optimal in this case, with p slightly reduced and h slightly positive, resulting in a small increase in plovers and a 20 percent decrease in falcons (all relative to the benchmark).[12] Moreover, falcons decline compared to no-management. This makes sense: when falcons are valued less there is less opportunity cost to removing them, so predator removal is more likely to be optimal.

If the existence value of falcons is eliminated altogether, so that $u_Y = 0 \cdot u_{Y0}$, then the optimal solution involves even greater substitution of predator removal for exclosures. Specifically, it is optimal in this case to follow strategy C ($h = h_{SV}$, $p = 0$), so that the optimal set of controls is opposite that of the benchmark scenario. This was implied in the simple open access model discussed in section 1.2.2, where it was noted that predator removal (a license fee) was less likely to be optimal when predators had existence value (stock-dependent values). Note that the steady state $Y < 1$, so the optimal strategy is essentially one of falcon eradication.

The next scenario increases the plover carrying capacity, $k = 2 \cdot k_0$. It is still optimal to follow strategy D. At equilibrium, the rate of exclosures is slightly less ($p = 0.84$) than in the benchmark scenario, although the equilibrium plover population is now twice as high ($X^* = 519$) and there are more falcons ($Y^* = 199$). Fewer exclosures are optimal because the larger carrying capacity, which yields more plover growth, implies predation has a smaller marginal cost (i.e., $\lambda_X - \alpha\lambda_Y$ is smaller, though still positive) relative to the benchmark case.

The next two scenarios, $\hat{Y} = 1/2 \cdot \hat{Y}_0$ and $r = 2 \cdot r_0$, favor plovers so that plovers do not go extinct in the no-management scenario. Nevertheless, strategy D remains optimal in both cases.

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[12] The optimal long run trajectory is a movement along the saddle path to equilibrium. The system initially lies above this path, where $\sigma(Y,\lambda_Y) > 0$, and it is optimal to use an impulse harvest until the system reaches the path, when $\sigma(Y,\lambda_Y) = 0$. Then, the control $h = h_{SV}$ is followed and the system proceeds along the saddle path to equilibrium.

The primary quantitative changes from the benchmark are that a smaller $\hat{Y}$ significantly reduces the equilibrium falcon stock, while a larger r slightly increases the equilibrium number of plovers. Also, neither scenario yields a win-win outcome for the falcons. This is because the interests of falcons and society are less likely to coincide when plovers are not at risk of extinction. On the other hand, predator control can still be optimal even without extinction risks.

The next scenario, $\alpha = 2 \cdot \alpha_0$, doubles the sustenance that falcons get from predating on plovers compared with the benchmark scenario. Strategy D remains optimal, but the win-win outcome is more significant—the optimal equilibrium includes 199 falcons. Thus, falcons benefit more from the exclosure control when plovers are a more significant part of their diet, which is intuitive.

The last two scenarios increase the cost of the exclosure control, with $c = 2 \cdot c_{p0}$ and $c = 3 \cdot c_{p0}$. Once exclosure costs are high enough, the optimal management strategy switches to strategy B, with a lower p and a small, positive h. The equilibrium involves fewer falcons and plovers relative to the benchmark. Thus, as costs become excessive it becomes too expensive to regulate (control predation), so externalities (excessive predation) persist.

1.5 Conclusion

I analyzed how predator-prey relations create ecosystem inefficiencies that are akin to the inefficiencies that characterize anthropogenic overharvesting of valuable resources, and how management can more efficiently manage these resources. Using this perspective, I find that predator removal and predator exclosures are analogous to license fees and harvest quotas, which are capable of correcting the externalities in a standard open access system. Just as management of human hunters can benefit both humans and the exploited resource stock, I find management of predators can lead to a win-win situation in which both the prey and predator stocks increase. This is particularly true when over-predation would otherwise lead to prey extinction.

Although this win-win outcome has analogies in open access management, an important difference in the present context is that predator controls are costly, in contrast to the usual assumption that regulation of open access hunters is costless. These costs have implications for the choice of controls. Existence values related to the predator may also influence the optimal choice of controls, in contrast to existing analyses of open access problems involving humans. In the Great Lakes Piping Plover and Merlin application, an exclosure-only policy is found to be optimal when there are existence values for the predator population. The result arises, in part, because exclosures generate a win-win outcome in which regulation increases both wildlife populations. However, if predator existence values are sufficiently small or if the costs of exclosures are excessive, some predator removal optimally substitutes for exclosures and a win-win outcome may fail to materialize.

The numerical results have implications for managing open access resources, by showing how the optimal mix of regulatory instruments could be influenced by two types of values that are often ignored: regulatory transactions costs and non-market values related to the regulated industry. In particular, these latter values may arise for industries that become sentimentalized, such as with Maine’s “lobster culture” (Daniel et al. 2008). The inclusion of these values results in the optimal management strategy being parameter dependent, which demonstrates the qualitative importance of transactions costs and non-market values for policy.

1.6 Tables and Figures

Table 1.1. Benchmark parameter values.

Parameter	Interpretation	Value
r	Plover intrinsic growth rate	0.2
k	Plover carrying capacity with no predation	300
$\hat{Y}$	Falcon carrying capacity with no plover predation	248
X_0	Plover initial stock	126
Y_0	Falcon initial stock	195
α	Conversion of predated plovers into falcons	0.125
β	Predation parameter	0.0012
φ	Falcon Congestion parameter	0.001875
δ	Falcon mortality rate	0.1
u_X	Marginal plover value when $X = 0$	7,936,958
u_Y	Marginal falcon value with $Y = 0$	1,587,392
c_h	Removal cost parameter	140,000
c_p	Exclosure cost parameter	106,333.33
ρ	Discount rate	0.05

Table 1.2. Simulation results.

Scenario	**Optimal management**				**No management**	
	Strategy	(X^{∞}, Y^{∞})[a]	(h^{∞}, p^{∞})	SNB[b]	(X^{∞}, Y^{∞})	SNB[b]
Benchmark	D	(260, 197)	(0.00, 0.89)	1000	(0, 195)	727
Alternative parameters						
$u_Y = 1/2 \cdot u_{Y0}$	B	(264, 157)	(0.08, 0.87)	921	(0, 195)	643
$u_Y = 0 \cdot u_{Y0}$	C	(300, <1)	(0.41, 0.00)	871	(0, 195)	559
$K = 2 \cdot K_0$	D	(519, 199)	(0.00, 0.89)	1008	(0, 195)	763
$\hat{Y} = 1/2 \cdot \hat{Y}_0$	D	(274, 75)	(0.00, 0.81)	989	(151, 83)	910
$r = 2 \cdot r_0$	D	(273, 198)	(0.00, 0.85)	1003	(116, 204)	929
$\alpha = 2 \cdot \alpha_0$	D	(260,199)	(0.00, 0.89)	1052	(0, 195)	720
$c_p = 2 \cdot c_{p0}$	D	(245, 198)	(0.00, 0.84)	991	(0, 195)	727
$c_p = 3 \cdot c_{p0}$	B	(236, 183)	(0.03, 0.80)	980	(0, 195)	727

[a] The superscript $^{\infty}$ denotes the equilibrium value.

[b] All economic values are in present value US$ millions.

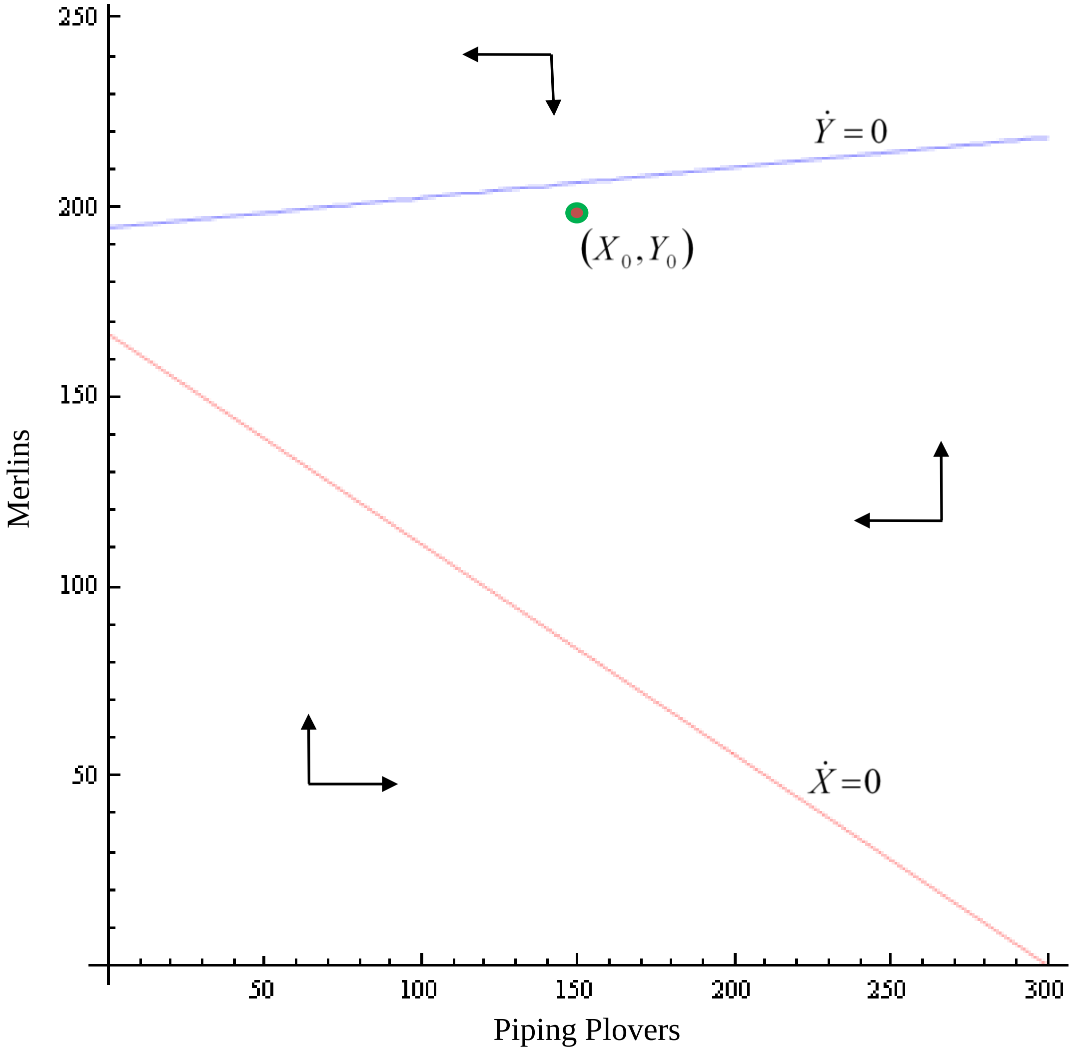

Figure 1.1. Phase diagram of strategy A in the benchmark scenario.

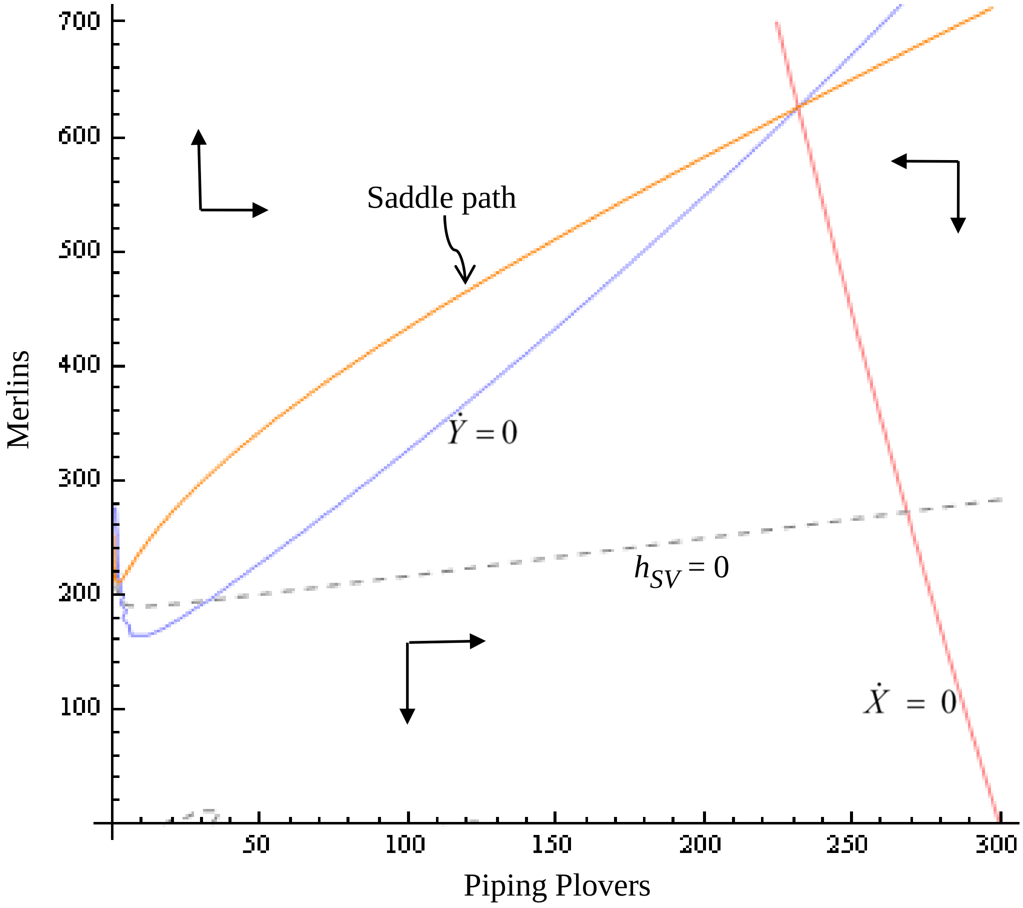

Figure 1.2. Phase diagram of strategy B in the benchmark scenario. The dashed line marks the $h_{SV} = 0$ threshold. Above this line $h_{SV} < 0$ and below the line $h_{SV} > 0$.

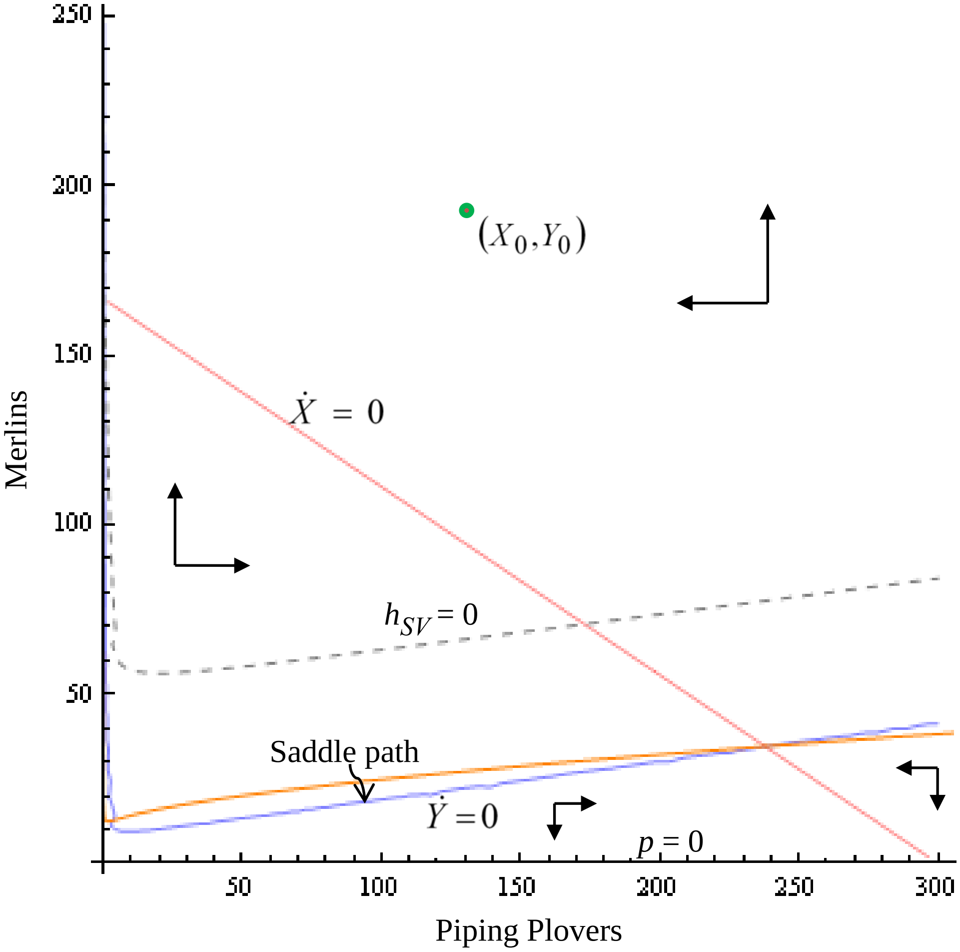

Figure 1.3. Phase diagram of strategy C ($h = h_{SV}$, $p = 0$) in the benchmark scenario. The dashed line marks the $h_{SV} = 0$ threshold. Above this line $h_{SV} < 0$ and below the line $h_{SV} > 0$. Running close to the X-axis is the $p = 0$ threshold. Above this line $p > 0$ and below the line $p = 0$.

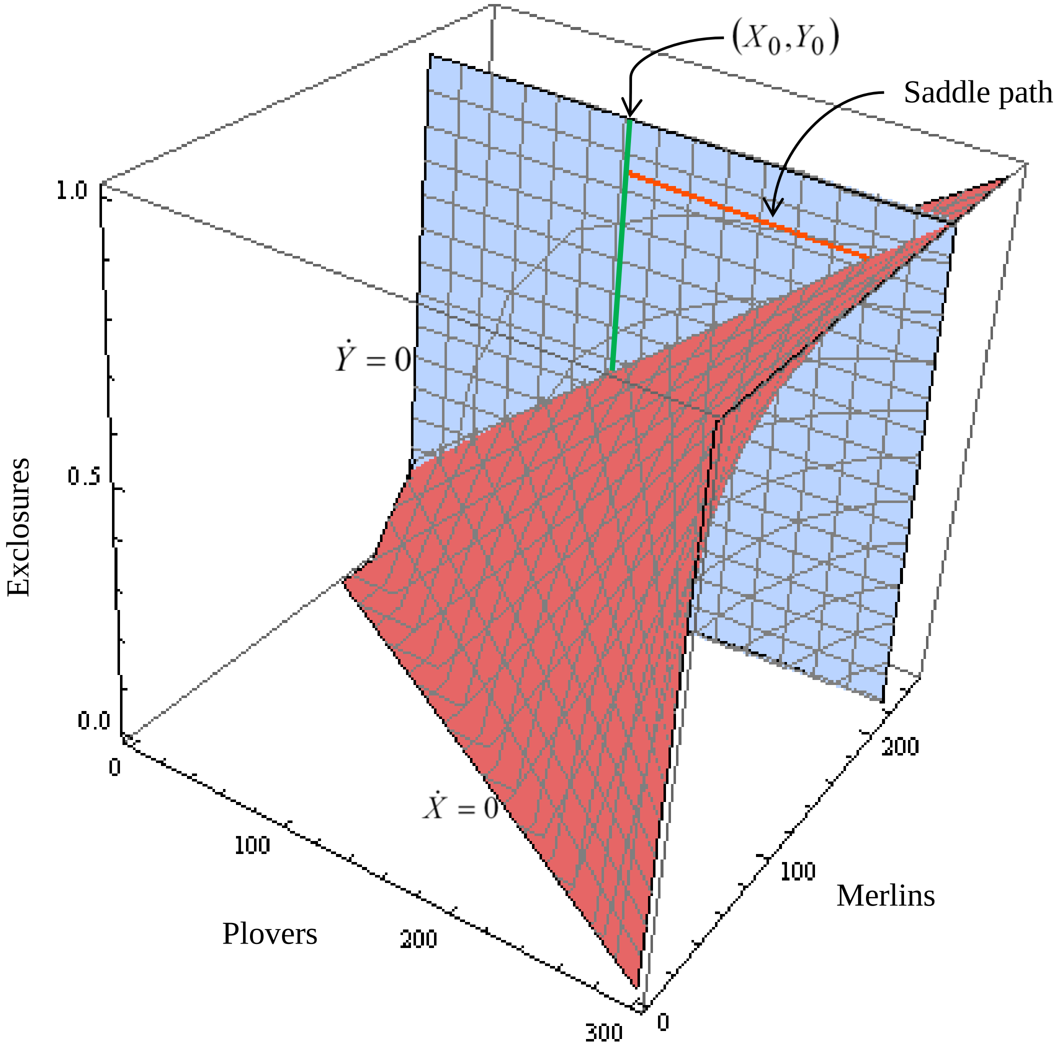

Figure 1.4. Graph of strategy D ($h = 0$, $0 < p < 1$) in the benchmark scenario.

1.7 Appendix

1.7.1 Bioeconomics and costless regulation

The main text described how the equation of motion for the predators, (4), is analogous to an entry-exit condition for resource harvesters in a regulated open access setting. Here, I set up and analyze the problem in such a setting, treating *Y* as harvesters, *X* as the resource, *p* as a proportional reduction in the harvest (i.e., a harvest limit/quota), and *h* as a licensing fee. As with traditional regulated open access models, I assume harvesters and resource managers both value the economic rents from harvesting activities. Managers may also have non-market (existence) values associated with the prey species, but not generally with harvesters of the prey. Denote non-market (existence) values associated with the prey by $B_X(X)$. Finally, I make the typical assumption that regulations (i.e., *h* and *p*) are implemented costlessly.

Given these assumptions, the planner maximizes the present value of economic (after-tax) rents, prey existence values, and tax receipts, subject to (3)-(4). The current-value Hamiltonian is

$$\text{(A1)}\qquad H = \left[\left(\varphi\left(\hat{Y} - Y\right) + \alpha\beta X\left(1 - p\right) - \delta\right)Y\right]\left[1 + \lambda_Y\right] + B_X\left(X\right) + \lambda_X \dot{X} - \lambda_Y hY \,.$$

Note the licensing fee *h* does not appear in the bracketed term in (A1) since *h* is a transfer payment and not a true cost. As *H* is linear in *h* and *p*, the optimality condition for *h* is given by (12) with $c_h = 0$, and the optimality condition for *p* is

$$\text{(A2)}\qquad \frac{\partial H}{\partial p} = \left(\lambda_X - \alpha\left[1 + \lambda_Y\right]\right)\beta XY \begin{array}{lll} > 0 & iff & p^* = 1 \\ = 0 & iff & p^* = p_{SV} \\ < 0 & iff & p^* = 0 \end{array} ,$$

The following adjoint equations are also necessary

$$\text{(A3)}\qquad \dot{\lambda}_X = \rho\lambda_X - \alpha\beta\left(1 - p\right)Y\left[1 + \lambda_Y\right] - \lambda_X\left[r\left(1 - 2X/k\right) - \beta Y\left(1 - p\right)\right] - B'_X \,,$$

(A4) $\dot{\lambda}_Y = \rho\lambda_Y - \left(\varphi\left(\hat{Y}-2Y\right)+\alpha\beta X\left(1-p\right)-\delta\right)\left[1+\lambda_Y\right]+\lambda_X X\beta\left(1-p\right)+\lambda_Y h$.

Consider whether both controls are used simultaneously. As $h^* \to \infty$ can only hold for an instant, focus on the singular solution for h. From (12) with $c_h = 0$, this requires $\lambda_Y = \dot{\lambda}_Y = 0$. With $\lambda_Y = 0$, (A2) implies $\lambda_X \geq \alpha$ must hold for $p^* > 0$. Consider the singular solution for p (given h is also singular), so that $\lambda_X = \alpha$ and $\dot{\lambda}_X = 0$. Insert these values into (A3)-(A4):

(A5) $X^* = k\left[B_X / \alpha + r - \rho\right]/2r$,

(A6) $Y^* = \left(\hat{Y} - \delta/\varphi\right)/2$.

The solution in (A5)-(A6) is a fixed (steady state) point, and so h^* and p^* are found by setting (3) and (4) equal to zero and setting $X = X^*$ and $Y = Y^*$. The two controls are used together: h^* addresses the congestion externality to maintain Y at the level that maximizes returns from “other resources” (implicit via $\varphi\hat{Y}$), while p^* ensures X is harvested efficiently. Note only one control would be optimal if not for the “other resource”. That is, a double singular solution does not exist when $\varphi = 0$, which is seen in that Y^* in (A6) becomes undefined when $\varphi = 0$. It is also easy to show that $h^* = h_{SV}$, $p^* = 1$ is not an optimal solution when $\sigma = 0$. Rather, only a single control is required when $\varphi = 0$. This is because harvests, $\beta(1-p)XY$, may be managed via p or by changing Y (via h), with neither control implying a different opportunity cost.[13]

A single control is also optimal when social welfare is instead defined as discounted stock-dependent (existence) values, even when $\varphi > 0$, because in this case there is no reason for a

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[13] Although h is ineffective in situations where the optimal strategy would be to let Y increase, the linear nature of the problem implies a most rapid approach path (i.e., $h = p = 0$) would be optimal in such instances.

second control to manage rents from the "other resource". Defining the existence value for harvesters by the increasing, concave function $B_Y(Y)$, the current-value Hamiltonian becomes

$$\text{(A7)}\quad H = B_X(X) + B_Y(Y) + \lambda_X \dot{X} + \lambda_Y \dot{Y}\ .$$

The optimality conditions for the controls are now (12), with $c_h = 0$, and

$$\text{(A8)}\quad \frac{\partial H}{\partial p} = (\lambda_X - \alpha\lambda_Y)\beta XY \begin{array}{lll} > 0 & iff & p^* = 1 \\ = 0 & iff & p^* = p_{SV} \\ < 0 & iff & p^* = 0 \end{array},$$

Consider whether both controls are used simultaneously, again focusing on the singular solution for h. From (12), $h^* = h_{SV}$ requires $\lambda_Y = 0$, and hence $\dot{\lambda}_Y = 0$. With $\lambda_Y = 0$, (A8) implies $\lambda_X \geq 0$ for $p^* > 0$. As $\lambda_X = 0$ will never arise for an optimally managed resource, a singular value for p is ruled out. The only option for a positive p is thus $p^* = 1$. To see if this can be optimal, consider the adjoint condition for Y for the proposed solution of $p^* = 1$, $\lambda_Y = 0$, and $\dot{\lambda}_Y = 0$:

$$\text{(A9)}\quad \dot{\lambda}_Y = \rho\lambda_Y - B'_Y - \lambda_Y\left(\varphi\left(\hat{Y} - 2Y\right) + \alpha\beta X(1-p) - \delta - h\right) + \lambda_X X\beta(1-p) = -B'_Y = 0$$

Given that $B'_Y > 0$, expression (A9) represents a contradiction. Hence, using both controls to manage the system (for more than an instant) is sub-optimal.

1.7.2 Derivations of the parameters

***Intrinsic growth rate*: *r*.** No published sources present a Piping Plover intrinsic growth rate. Courchamp *et al*. (2000) use a value of 0.2 for the Macquarie Parakeet, which went extinct due to predation. Similarities between the Macquarie Parakeet and the Piping Plover also extend to food preferences and size. Like the plover, the parakeet's habitat was near the shore and they fed on invertebrates. The parakeet was small, too, although somewhat larger than the plover (30cm vs 20cm in length). Due to this likeness, the value of 0.2 is used for the Piping Plovers.

***Carrying capacity*: *k*.** Plissner and Haig (2000) use $k = 300$ for the Great Lakes plover population, which is the value managers believe the current habitat can support (USFWSb 2009).

***Initial values*: X_0, Y_0.** The U.S. Fish and Wildlife Service (USFWSb 2009) estimated the Great Lakes population of Piping Plovers in 2009 at 126 individuals. The initial value for the predator population was set at its carrying capacity, that is, $Y_0 = \hat{Y} - \delta/\sigma$.

***Other Merlin ecological parameters*: $\hat{Y}$, β, α, φ and δ.** These values were not directly available due to a lack of published data. I chose values based on reasonable assumptions and simulations that reflect current knowledge of Merlins. The particulars are described below. The sensitivity analysis in section 1.4.3 examines the impact of various parameter choices on results.

Begin by specifying the Merlin carrying capacity when $X = 0$: $\hat{Y} - \delta/\varphi$. Doolittle and Balding's (1995) reported sightings in Michigan, Minnesota and Wisconsin imply a population density of 0.29 Merlins/mile^2. I focus on habitat in Michigan because most of the Great Lakes Piping Plovers nest in Michigan (Sleeping Bear Dunes National Park alone contains about one third of all nesting pairs in the Great Lakes, S. Jennings, personal communication). Suitable Merlin habitat in Michigan comprises about 7100 square miles (pine forests), so that if the Merlin population has recovered from historical lows in the 1960s at an average annual rate of 3.3% (Niven et al. 2004) the Michigan Merlin population would be 3900 in 2012. Only a fraction of the total Merlin population actually lives near Piping Plover habitat. A precise estimate of this fraction is not available. However, based on the relative size of Merlin and plover habitats, I assume five percent of the Merlin population, or 195 Merlins, resides in territory that overlaps with Piping Plover habitat. I take this value to be the Merlin carrying capacity when $X = 0$.

Next, specify the intrinsic growth rate for Merlins: $\hat{Y}\varphi - \delta$. Tanner (1975) estimates the maximum fertility of the European Sparrow Hawk, a similarly-sized predatory bird, to be 0.363. I adopt this value for the intrinsic growth rate of Merlins.

The third piece of information used for calibration is that plovers are small and occupy only a fraction of the Merlin diet, so the Merlin population benefits marginally (a small increase in fertility) from plover predation (V. Cavalieri, personal communication). This implies $\alpha\beta X_0 Y_0$ should be small, i.e. equivalent to only a few Merlins. Parameters are selected so that $\alpha\beta X_0 Y_0 \approx 3$.

Along with the conditions for carrying capacity ($\hat{Y} - \delta/\varphi = 195$), the intrinsic growth rate ($\hat{Y}\varphi - \delta \approx .363$), and sustenance ($\alpha\beta X_0 Y_0 \approx 3$), the final two parameters are set to satisfy a condition related to plover extinction. Specifically, Plissner and Haig (2000) and Wemmer et al. (2001) predict extinction of Plovers within 100 years if nothing is done. Assuming Merlins are the primary driver of extinction, parameters are selected so that a numerical simulation of (1)-(2) (using Mathematica 7.0, Wolfram 2008) satisfies the extinction conditions and the other conditions. The combination of $\hat{Y}$, β, α, φ and δ finally selected (Table 1,1) yielded approximately 195 Merlins and one Piping Plover by year 100 in the simulation, as well as $\hat{Y}\varphi - \delta = 0.365$ and $\alpha\beta X_0 Y_0 = 3.686$, closely satisfying each calibration condition listed above.

***Benefit values*: u_X, u_Y.** For species i ($i=X,Y$), u_i is calibrated using Richardson and Loomis' (RL's) (2009) *aggregate* willingness to pay (WTP) estimates (i.e.,WTP/ household times households) for a percentage increase in i. Specifically, their WTP in the context of our model is $WTP = B_i(i_0(1 + \%\Delta)) - B_i(i_0)$ where i_0 is some initial value. Let $B_i(i) = \upsilon_i \ln(1+i)$. Then $WTP = \upsilon_i[\ln(1 + i_0(1 + \%\Delta)) - \ln(1 + i_0)] \approx (\upsilon_i/100)\%\Delta$. Then I calibrate $\upsilon_i = 100\times WTP/\%\Delta$. For each species, I follow RL's example and assume $\%\Delta = 100$. The other independent variables of

RL's model take the following values: the survey response rate equals 100%; the species in question is a non-charismatic bird; the sample means of the RL data were used for the fraction of the survey done by mail, 0.851, and the fraction of respondents who were visitors, 0.231. Finally, I multiply υ_Y by 0.2 since only some Merlins in northern Michigan overlap with plovers. Finally, I calculate social WTP (u_i) by multiplying *WTP* by 0.35 million households (i.e., ≈ the population of northern Michigan).

***Cost of predator removal*: c_h.** The federal recovery plan for the Piping Plover estimates the cost of predator control (task # 1.222 in USFWS, 2003) to be $35,000, with the disclaimer that "Final costs contingent on areas and numbers of predators." I assume this cost is adequate for removing one quarter of the predator stock, so $c_h h = c_h / 4 = 35{,}000 \Rightarrow c_h = 140{,}000$.

***Cost of predator exclosures*: c_p.** For locating Piping Plover nests and identifying critical habitat, the recovery plan estimates a total cost of $319,000 (i.e., the sum of the costs for task #s 1.12, 1.16, 1.17, 1.18, 1.21, 1.221, 1.223, 1.31, 1.341, 4.1 in USFWS 2003). These figures are adequate for protecting 75% of the Great Lakes Piping Plovers (Gratto-Trevor and Abbott, 2011) and correspond closely to estimates provided by the National Park Service (S. Jennings, personal communication). This implies $c_p p_0 /(1 - p_0) = 3c_p = 319{,}000 \Rightarrow c_p = 106{,}333.33$.

REFERENCES

REFERENCES

Anderson, L.G. (1989). Enforcement Issues in Selecting Fisheries Management Policy. *Marine Resource Economics* 6: 261-277.

Antolos, M., D.D., Roby, D.E. Lyons, K. Collis, A.F. Evans, M. Hawbecker and B.A. Ryan (2005). Caspian Tern Predation on Juvenile Salmonids in the Mid-Columbia River. *Transactions of the American Fisheries Society* 134: 466-480.

Beddington, J.R., C.A. Free and J.H. Lawton (1978). Characteristics of Successful Natural Enemies in Models of Biological Control of Insect Pests. *Nature* 273: 513-519.

Bennett, C., S. Chaudhry, M. Clemens, L. Gilmer, S. Lee, T. Parker, E. Peterson, J. Rajkowski, K. Shih, S. Subramaniam, R. Wells and J. White (2009). Excluding Mammalian Predators from Diamondback Terrapin Nesting Beaches with an Electric Fence. Unpublished manuscript. Available online at http://drum.lib.umd.edu/handle/1903/9074.

Brock, W. and A. Xepapadeas (2002). Optimal Ecosystem Management when Species Compete for Limiting Resources. *Journal of Environmental Economics and Management* 44:189-220.

Clark, C.W. *Mathematical Bioeconomics*. Wiley, New York, 2005.

Courchamp, F., M. Langlais, and G. Sugihara (2000). Rabbits Killing Birds: Modelling the Hyperpredation Process. *Journal of Animal Ecology* 69: 154-164.

Crocker, T.D., and J. Tschirhart (2002). Ecosystems, Externalities, and Economies. *Environmental and Resource Economics* 2: 551-567.

Daniel, H., T. Allen, L. Bragg, M. Teisl, R. Bayer and C. Billings (2008). Valuing Lobster for Maine Coastal Tourism: Methodological Considerations. *Journal of Food Service* 19: 133-138.

DeCesare, N.J., M. Hebblewhite, H.S. Robinson and M. Musiani (2009). Endangered, Apparently: The Role of Apparent Competition in Endangered Species Conservation. *Animal Conservation* 13: 1-10.

Doolittle, T. and T. Balding (1995). A Survey for the Eastern Taiga Merlin (*Falco columbarius*) in Northern Minnesota, Wisconsin and Michigan. *The Passenger Pigeon* 57: 31-36.

Gratto-Trevor, C. and S. Abbott (2011). Conservation of Piping Plover (*Charadrius melodus*) in North America: Science, Successes, and Challenges. *Canadian Journal of Zoology* 89:401-418.

Haig, S.M., and L.W. Oring (1985). Distribution and Status of the Piping Plover Throughout the Annual Cycle. *Journal of Field Ornithology* 56: 334-345.

Hoekstra, J., and J.C.J.M. van den Bergh (2005). Harvesting and Conservation in a Predator-Prey System. *Journal of Economic Dynamics and Control* 29: 1097-1120.

Horan, R.D., E.P. Fenichel, K. Drury, and D.M. Lodge (2011). Managing Ecological Thresholds in Coupled Environmental-Human Systems. *Proceedings of the National Academy of Sciences* 108: 7333-7338.

IUCN Red List (2010). Table 4a: Red List Category summary for all animal classes and orders.

Kelly, E.G., E.D. Forsman and R.G. Anthony (2003). Are Barred Owls Displacing Spotted Owls? *The Condor* 105: 45-53.

Liski, M., P.M. Kort, and A. Novak (2001) Increasing Returns and Cycles in Fishing. *Resource and Energy Economics* 23: 241-258.

Lokemoen, J.T., H.A. Doty, D.E. Sharp and J.E. Neaville (1982). Electric Fences to Reduce Mammalian Predation on Waterfowl Nests. *Wildlife Society Bulletin* 10: 318-323.

Mayer, P.M., and M.R. Ryan (1991). Electric Fences Reduce Mammalian Predation on Piping Plover Nests and Chicks. *Wildlife Society Bulletin* 19: 59-63.

Messmer, T.A., M.W. Brunson, D. Reiter and D.G. Hewitt (1999). United States Public Attitudes Regarding Predators and Their Management to Enhance Avian Recruitment. *Wildlife Society Bulletin* 27: 75-85.

Michigan Natural Features Inventory (MNFI) . Michigan’s Special Animals. Available online http://web4.msue.msu.edu/mnfi/. Accessed 10-6-2011.

Millennium Ecosystem Assessment (MEA). *Ecosystems and Human Well-being: Biodiversity Synthesis*. World Resources Institute, Washington, DC, 2005.

Moseby, K.E. and J.L. Read (2006). The Efficacy of Feral Cat, Fox and Rabbit Exclusion Fence Designs for Threatened Species Protection. *Biological Conservation* 127: 429-437.

Munro, G.R. The Optimal Management of Transboundary Renewable Resources (1979). *The Canadian Journal of Economics* 12: 355-376.

Niven, D.K., J.R. Sauer, G.S. Butcher and W.A. Link (2004) Christmas Bird Count Provides Insights into Population Change in Land Birds that Breed in the Boreal Forest. *American Birds* 58: 10-20.

Plissner, J.H., and S.M. Haig (2000). Viability of Piping Plover *Charadrius melodus* Metapopulations. Biological Conservation 92: 163-173.

Ragozin, D.L., and G. Brown, Jr. (1985) Harvest Policies and Nonmarket Valuation in a Predator-Prey System. *Journal of Environmental Economics and Management* 12: 155-168.

Rice, S.H. (2004). *Evolutionary Theory: Mathematical and Conceptual Foundations*. Sinauer Associates, Inc., Sunderland, Massachusetts.

Richardson, L. and J. Loomis (2009). The Total Economic Value of Threatened, Endangered and Rare Species: An Updated Meta-Analysis. *Ecological Economics* 68: 1535-1548.

Rimmer, D. and R. Deblinger (1990). Use of Predator Exclosures to Protect Piping Plover Nests. *Journal of Field Ornithology* 61: 217-223.

Roemer, G.W., C.J. Donlan and F. Courchamp (2002). Golden Eagles, Feral Pigs, and Insular Carnivores: How Exotic Species Turn Native Predators into Prey. *Proceedings of the National Academy of Sciences* 99: 791-796.

Sanchirico, J.N. and J.E. Wilen (2001). A Bioeconomic Model of Marine Reserve Creation. *Journal of Environmental Economics and Management* 42: 257-276.

Seip, D.R. (1991). Predation and Caribou Populations. *Rangifer* 11: 46-52.

Smith, V.L. (1968). Economics of Production from Natural Resources. *American Economic Review*. 58: 409-431.

Soulé, M.E., J.A. Estes, B. Miller and D.L. Honnold (2005). Strongly Interacting Species: Conservation Policy, Management, and Ethics. *BioScience* 55: 168-176.

Ströbele, W.J. and H. Wacker (1995). The Economics of Harvesting Predator-Prey Systems. *Journal of Economics* 61: 65-81.

Struthers, K., and P. Ryan (2005) Predator Control for the Protection of Federally Endangered Great Lakes Piping Plover (*Charadrius melodus*) at Dimmick's Point, North Manitou Island. Proceedings of the 11th Wildlife Dammage Management Conference.

Taylor, R.J. (1981). Ambush Predation as a Destabilizing Influence Upon Prey Populations. *The American Naturalist* 118: 102-109.

Tilman, D., S. Polasky and C. Lehman (2005). Diversity, Productivity and Temporal Stability in the Economics of Humans and Nature. *Journal of Environmental Economics and Management* 49(2005): 405-426.

Tschirhart, J. (2009). Integrated Ecological-Economic Models. *Annual Review of Resource Economics* 1: 381-407.

Tschirhart, J. (2000) General Equilibrium of an Ecosystem. *J Theoretical Biology* 203: 13-32.

U.S. Fish and Wildlife Service (USFWS). Recovery Plan for the Great Lakes Piping Plover (*Charadrius melodus*). Dept. of the Interior, USFWS, Ft. Snelling, MN, 2003.

U.S. Fish and Wildlife Service (USFWS). Federal and State Endangered and Threatened Species Expenditures Fiscal Year 2009. Dept. of the Interior U.S. Fish and Wildlife Endangered Species Program, Washington, D.C., 2009a.

U.S. Fish and Wildlife Service (USFWS). Piping Plover (*Charadrius melodus*) – 5-Year Review: Summary and Evaluation. Dept. of the Interior U.S. Fish and Wildlife Service, Northeast Region, Hadley, Massachusetts and Midwest Region East Lansing Field Office, MI, 2009b.

U.S. Fish and Wildlife Service (USFWS). Summary of Listed Species Listed Populations and Recovery Plans, 2011. Available online at http://ecos.fws.gov/tess_public/pub/Boxscore.do.

Wacker, H. (1999). Optimal Harvesting of Mutualistic Ecological Systems. *Resource and Energy Economics* 21: 89-102.

Wemmer, L.C., U. Ozesmi, and F.J. Cuthbert (2001) A Habitat-Based Population Model for the Great Lakes Population of the Piping Plover (*Charadris mulodus*). *Biological Conservation* 99: 169-181.

Wolfram Research, Inc., *Mathematica 7*.0, Champaign, IL, 2008.

CHAPTER 2

Interspecies Management and Land Use Strategies to Protect Endangered Species

2.1 Introduction

Habitat loss is widely seen as the primary driver of biodiversity decline (Wilcove et al., 1998; MEA, 2005). In many countries, environmental regulations now provide legal protection for the undisturbed habitats of endangered species. Moreover, significant resources are spent on habitat rehabilitation and the promotion of private habitat stewardship. Nevertheless, the lack of suitable habitat has left many species "conservation reliant" (Scott et al, 2005). Species that are conservation reliant require continuous, species-specific intervention to reach and maintain a recovery goal (typically a population threshold). To reduce extinction pressures, species-specific intervention usually involves the control of other, non-endangered species that negatively impact the imperiled species. Scott et al. (2010) find that 66% of the recovery plans for species listed under the U.S. Endangered Species Act call for such controls, which includes the removal of predators, parasites and competitors. Managing interspecies interactions could become increasingly important as species continue to be subjected to habitat loss.

Managing endangered species, including conservation-reliant species, through habitat and species interactions is a form of ecosystem management. Prior work on ecosystem management has focused largely on direct controls, such as harvests, applied to individual species. This work has shown the need to adjust harvest levels to account for the effects of stock-dependent interspecies interactions (Brock and Xepapadeas, 2002), including predator-prey relations (Ragozin and Brown, 1985; Ströbele and Wacker, 1995) and interspecies competition

(Chaudhuri, 1986). Failing to take these interactions into account will result in ecosystem externalities (Crocker and Tschirhart, 1992).

The focus here is also on ecosystem management, but in addition to direct controls I consider the use of habitat protection as a control. Managing habitat is distinct from direct controls because it simultaneously affects multiple species in the same ecosystem. In particular, when humans engage in exploitation competition with animals for land resources, habitat is replaced by development, which affects interrelated species, such as predators and prey (Moon et al, 2010). Accounting for the simultaneous impacts of land use decisions yields important insights into ecosystem management. For example, I find that in some situations habitat protection can be an effective method of simultaneously increasing prey populations *and* controlling predators. This can make predator removal a redundant method of endangered species conservation, although a priori it is not clear if a habitat management-only strategy would be efficient.

Prior bioeconomic work has examined the management of exploitation competition through habitat management, although not in the context of an endangered species subject to predation. Swallow (1990) and Sanchirico and Springborn (2011) study the human-animal land use conflict as a development problem. Development yields economic rents but reduces habitat and the returns from a harvestable resource. These rents, though exploitation competition, encourage habitat loss, which may ultimately drive the harvested resource to extinction. Bednar-Friedl et al. (2012) study land use conflicts in the context of a park management problem. The park’s value increases with visits but visitors damage the habitat of, and thus compete with, an existence-valued endangered species. Bednar-Friedl et al. consider direct controls to aid the endangered species, but they find that optimal ecosystem management involves at least some reduction in visits. My approach to ecosystem management also focuses on habitat loss and the (opportunity) costs of habitat protection. However, my work is novel in that more than one species is directly affected by habitat management. Habitat management can affect different species in different ways. For example, habitat protection that helps conserve an endangered species may harm the endangered species' predators.

The conservation of the Atlantic-Gaspésie Woodland Caribou (*rangifer tarandus caribou*) motivates this investigation. The Gaspésie Woodland Caribou is the only remaining herd of woodland caribou living south of the St. Lawrence River. The population of woodland caribou has declined historically due to human encroachment and habitat loss. Presently, predation is a major limiting problem to the Gaspésie Woodland Caribou. Using optimal control theory, I identify the optimal level of predator removal and caribou-habitat protection for the Gaspésie Woodland Caribou. I find the first-best management strategy yields a caribou population that is conservation reliant. That is, although caribou-habitat protection can simultaneously benefit caribou and control predators, predator removal as a direct control remains an important, if secondary, ecosystem management tool.

The next section describes the caribou-predator dynamics that make up the ecological portion of the model. In the third section, I present the bioeconomic model and the conditions that define the optimal management strategy. The fourth section parameterizes the model to the case of the Gaspésie Woodland Caribou and discusses the numerical results; a sensitivity analysis is also provided. The final section concludes.

2.2 Ecological Model

The Gaspésie Woodland Caribou is the only population of woodland caribou living south of the St. Lawrence River. The species has been largely extirpated from the southern portion of its range. Historically, declines in the caribou population were due to human encroachment and habitat loss. Caribou are sensitive to human activity and will vacate habitat frequently used by people, which includes forestry and recreational activity (Recovery Plan, 2006). The Gaspésie Woodland Caribou population lives almost entirely within the Parc National de la Gaspésie, Quebec. Much of the impetus for creating the park was to protect caribou habitat. Within the park, recreational activities such as hiking are heavily regulated to minimize interference with the caribou herd (PNG, 2011). In 2000, the Gaspésie Woodland Caribou population was classified as endangered (Mosnier et al, 2010). Since 1983, the population has fallen from about 270 to less than 150, with predation a major limiting factor (Boisjoly et al, 2010). To meet recovery goals, the Gaspésie Woodland Caribou Recovery Plan has made active management via predator control a priority.

The conservation problem of the Gaspésie Woodland Caribou suggests two broad methods for its protection and the protection of other endangered species. The first is active management, such as predator control. The second is passive management, such as habitat protection (e.g. a conservation reserve program) or landscape investments that preserve elements of natural habitat consistent with human uses (e.g. forest retention; see Franklin, 1993). The Gaspésie Woodland Caribou currently receive implicit habitat protection through Parc National de la Gaspésie, which acts as a conservation reserve.[1] Placing land in a conservation reserve is

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[1] Adjacent to the park are wildlife management zones and privately-owned lands where recreational, forestry and mining activities are allowed. These activities are regulated within these wildlife management zones to promote sustainable ecosystem practices. However, it has been noted that sustainable timber harvesting can adversely affect caribou populations (Stone et al, 2008). Hunting and fishing are permitted within wildlife management zones. These activities

equivalent to caribou habitat protection. The size of the conservation reserve directly affects caribou *and* predator dynamics. Unlike caribou, predators thrive on land allocated for human-designated uses.[2] When caribou is not available as a source of food, the carrying capacity of the predator population is lower inside the conservation reserve than outside. Thus, predation rates are affected by the amount of land placed in the conservation reserve. With a sufficiently large conservation reserve, the predator population could decline to levels where predator control is no longer necessary to meet caribou conservation goals, and the Gaspésie Woodland Caribou would cease to be conservation reliant.

Maintaining a conservation reserve area outside the park boundaries would reduce the area suitable for human activities and increase the amount of habitat for caribou. Assume there is a fixed total amount of land available, L (in km^2), such that

$$L = L_M + L_R \tag{1}$$

where L_M is the area of natural resources managed for human use and L_R is the area placed in a conservation reserve.[3] The area of land set aside for a conservation reserve is treated as a control, so that managers can instantaneously change the size of the reserve. This is a reasonable approximation for the caribou because the land remains in a fairly natural state (i.e., it is not developed) under both alternative land uses. Indeed, the major difference between the two land

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limit caribou habitat outside the park. On the other hand, these recreation activities may support higher predator densities. For example, a dump containing hunting by-products is a source of food for coyotes (Mosnier et al, 2008).

[2] It is documented that some predator populations can increase in areas used by people. For example, coyotes may thrive in areas where forestry practices increase the population of prey, such as snowshoe hares (Boisjoly et al, 2010), while cowbirds can find more sustenance and thrive in areas of residential development (Borgmann and Morrison, 2010).

[3] In the case of the Gaspésie Woodland Caribou, this would include Parc National de la Gaspésie.

use states is that recreational activity (e.g., hunting) and some sustainable forestry is allowed in the managed state, whereas only limited recreational activity is allowed in the reserve state.[4]

The caribou dynamic can therefore be written in continuous time as

(2) $$\dot{C} = C\left[r_C\left(\alpha_M L_M + \alpha_R L_R - C\right) - \gamma P\right].$$

In the first term in brackets, growth takes the form of a Lotka-Volterra equation, $r_C(\alpha_M L_M + \alpha_R L_R - C)C$ where r_C is growth parameter, C is the stock of caribou and $\alpha_M L_M + \alpha_R L_R$ is the carrying capacity assuming no predation (Hannesson, 1983). Caribou carrying capacity net of predation is therefore determined by the amount of suitable habitat.[5] The parameters α_M, α_R determine the effect of land type on carrying capacity. Since L_R is better caribou habitat than L_M, $\alpha_R > \alpha_M \geq 0$. Predation is assumed to follow a Holling Type I functional response, so that predation is γPC where P is the stock of predators and $\gamma > 0$ is a parameter.

The predator dynamic is expressed in a fashion analogous to (2):

(3) $$\dot{P} = P\left[r_P\left(\beta_M L_M + \beta_R L_R - P\right) + \gamma C\right] - E\,.$$

Predator growth, net of caribou consumption, is $r_P(\beta_M L_M + \beta_R L_R - P)P$. The predator carrying capacity, in principle determined by the availability of other prey, is effectively determined by land use (Boisjoly et al, 2010). In contrast with caribou, for predators type-L_M habitat is better than type-L_R habitat, so $\beta_M > \beta_R \geq 0$. The effect of predation of caribou on the predator growth rate is γCP. The number of predators removed by managers is E.

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[4] Instantaneous adjustment in habitat will not be accurate for forms of habitat loss that are very destructive to the land and require many years to rehabilitate (i.e. “grow back”). In such cases, it would be more appropriate to model land in a conservation reserve as a state variable that can only change gradually over time.

[5] This is consistent with other bioeconomic models (e.g. Bulte and Horan, 2003).

The dynamics (2) and (3) are general enough to apply to many forms of interspecies interaction.[6]

The carrying capacities in (2) and (3) can be written as functions of a single control, the proportion of land in the conservation reserve $l = L_R/L$. Note that $L_M/L = (L - L_R)/L = 1 - l$. Hereafter I also model predator removal as a rate, h, where $h = E/P$. Thus, (2) and (3) can be written in the reduced form,

$$\dot{C} = C\left[r_C L\left(\alpha_M(1-l)+\alpha_R l-\frac{C}{L}\right)-\gamma P\right], \tag{4}$$

$$\dot{P} = P\left[r_P L\left(\beta_M(1-l)+\beta_R l-\frac{P}{L}\right)+\gamma C-h\right]. \tag{5}$$

Note, $\alpha_R - \alpha_M > 0$ and $\beta_R - \beta_M < 0$. Thus, assuming no caribou predation, the carrying capacity of caribou is increasing in the size of the conservation reserve while the carrying capacity of predators is declining in the size of the conservation reserve.

The impact of the controls l and h on the caribou and predators can be examined under steady state conditions. Define C^* and P^* as the steady state caribou and predator stocks. Setting equations (4) and (5) to zero, C^* and P^* can be derived as functions of the ecological parameters and management controls,

$$C^* = \frac{r_P\left(\gamma h + r_C L\left(\alpha_M(1-l)+\alpha_R l\right)-\gamma L\left(\beta_M(1-l)+\beta_R l\right)\right)}{r_C r_P + \gamma^2},$$

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[6] Interspecies competition would be modeled with (3) modified to $\dot{P} = P[r_P(\beta_M L_M + \beta_R L_R - P) - \gamma C - h]$. If the caribou and predator both benefited from conservation reserve land, then $\alpha_R > \alpha_M \geq 0$ and $\beta_R > \beta_M \geq 0$.

$$P^* = \frac{r_C\left(-h + r_P L\left(\beta_M(1-l) + \beta_R l\right) + \gamma L\left(\alpha_M(1-l) + \alpha_R l\right)\right)}{r_C r_P + \gamma^2}.$$

Operating a predator removal-based management regiment clearly benefits caribou: $\partial C^*/\partial h > 0$. However, the ecological cost of predator removal is fewer predators: $\partial P^*/\partial h < 0$.

Investing in a conservation reserve also benefits caribou. The relationship between l and C^* is positive because

$$\frac{\partial C^*}{\partial l} = \frac{r_P L\left(r_C(\alpha_R - \alpha_M) - \gamma(\beta_R - \beta_M)\right)}{r_C r_P + \gamma^2} > 0.$$

The steady state stock of caribou is increasing in l because caribou carrying capacity is strictly increasing in l while predator carrying capacity net predation on caribou is strictly declining in l. When l increases from zero to one, C^* increases by the amount $r_P(r_C(\alpha_R - \alpha_M) - \gamma(\beta_R - \beta_M))L/(r_C r_P + \gamma^2)$, which is positive and increasing in L.

The case of the Gaspésie Woodland Caribou notwithstanding, the predators' carrying capacity could be increasing in L_R, i.e. $\beta_R - \beta_M > 0$. In this case, the sign of $\partial C^*/\partial l$ is ambiguous. The sign of $\beta_R - \beta_M$ is therefore critical in determining the overall ecological impacts of a conservation reserve program. An endangered species may directly benefit from reserve creation, but predators will curb these gains when $\beta_R - \beta_M > 0$. Ignoring the conservation reserve's effect on predators could lead to inappropriate investment levels in reserve creation.

The effect of investing in a conservation reserve program has an ambiguous effect on P^*. This is because

$$\frac{\partial P^*}{\partial l} = \frac{r_C L\left(\gamma\left(\alpha_R - \alpha_M\right) + r_P L\left(\beta_R - \beta_M\right)\right)}{r_C r_P + \gamma^2},$$

which could be positive or negative. The quantity $\gamma(\alpha_R - \alpha_M) + r_P(\beta_R - \beta_M)$ is the predators' net marginal growth from the conservation reserve. The first term, $\gamma(\alpha_R - \alpha_M)$, measures the marginal change in caribou predation arising from the conservation reserve. This term is positive because increasing in the size of the reserve raises the stock of caribou the predators can catch. The second term, $r_P(\beta_R - \beta_M)$, measures the marginal change in other sources of sustenance for predators (i.e. predator carrying capacity net of caribou predation) from to the conservation reserve. This term is negative because less non-caribou prey is available inside the reserve. For the predators of the Gaspésie Woodland Caribou it is expected that $\partial P^*/\partial l < 0$. This requires $(\alpha_R - \alpha_M)\gamma < -(\beta_R - \beta_M)r_P$. In sum, the stock of predators is declining in the size of the conservation reserve if the increase in caribou predation is outweighed by the loss of other prey.[7]

The concern of Gaspésie Woodland Caribou managers is that predators will eventually drive the caribou to extinction. This implies the condition

$$(6) \qquad \frac{\dot{C}}{C} < 0 = \text{for all } C \geq 0 \Rightarrow$$

$$r_C L\left(\alpha_M\left(1-l\right) + \alpha_R l\right) - \gamma P\left(0\right) < 0,$$

where $P(0)$ is the stock of P when $C \to 0$. If Gaspésie Woodland Caribou can only survive on conservation reserve land, then $\alpha_M = 0$. In this case, (6) clearly holds when $l = 0$, making caribou

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[7] On the other hand, $\partial P^*/\partial l > 0$ requires $(\alpha_R - \alpha_M)\gamma_P > -(\beta_R - \beta_M)r_P$. This condition implies that a larger conservation reserve results in a higher steady state predator stock if the increase in caribou predation is sufficiently large. This is not expected to be the case for the Gaspésie Woodland Caribou and their predators.

extinction is inevitable. However, it may not be enough to have any $l > 0$ to prevent extinction. As long as $P(0) > 0$, an insufficiently large l implies $r_C L\alpha_R l < -\gamma P(0)$ and extinction still occurs. Extinction can only be prevented when l is large enough to overcome (6). This implies some minimum conservation reserve size may be necessary for caribou preservation.

2.3 Bioeconomic Model

2.3.1 Economics

Caribou provide significant values in Canada (Tanguay et al, 1995).[8] Hunting of the Gaspésie Woodland Caribou is not permitted, so only nonconsumptive values exist for this herd. Some visitors of Parc National de la Gaspésie enjoy use benefits from viewing the caribou, but nonuse value is likely the most important source of value (Tanguay et al, 1995). Caribou nonuse value is assumed to take the form $u_C \ln(C)$, where $u_C > 0$ is a parameter.

The primary predators of the Gaspésie Woodland Caribou are coyote and black bear. These species are treated jointly in the model as a single biomass of predators to simplify the exposition and because they respond similarly to the controls under consideration. Predators have existence value, which is assumed to take the form $u_P \ln(P)$, where $u_P > 0$.

The hunting level necessary to reduce predator populations may be intense (Mosnier et al, 2008). Private hunting is not likely to be sufficient so a predator control regime is considered necessary (Recovery Plan, 2006).[9] Predator removal costs are assumed to take the Schaefer form

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[8] In fact, the caribou is popular enough that it is featured on the reverse side of a standard Canadian quarter.

[9] The control strategies include aerial hunts in the winter and trapping in the spring and summer.

(Clark, 2005), $c_h E/P = c_h h$, where $c_h > 0$. Removal costs increase with the number of predators removed but decrease in the stock of predators, as the remaining predators become more difficult to locate.

Establishing or expanding the amount of critical habitat for the caribou—the conservation reserve—is costly. Society must forego the returns from using land for human activities. I assume the cost of a conservation reserve is $c_{R1}L_R + c_{R2}L_R^2 = c_{R1}Ll + c_{R2}(Ll)^2 = c_{l1}l + c_{l2}l^2$, where c_{R1}, $c_{R2} > 0$ and $c_{l1} = c_{R1}L$ and $c_{l2} = c_{R2}L^2$. The particular form of the cost function is actually quite general—it can be thought of as a second-order approximation to any convex cost function. The cost of the conservation reserve is conceived as forgone rents. These costs are expected to be convex in the size of the conservation reserve. The opportunity cost of displacing human activities is relatively cheap at first, displacing only economically marginal activities or perhaps none at all, but becomes increasingly costly, as the conservation reserve encroaches on land that is highly beneficial to people. For simplicity, I assume there are no transactions cost from changing *l*.

The management problem is to choose the level of predator removal and the size of the conservation reserve to maximize present value net benefits. Formally, this problem is,

$$\begin{aligned} (6) \qquad & \max_{h,l} \int_{t=0}^{\infty} \left[u_C \ln(C) + u_P \ln(P) - c_h h - (c_{l1} + c_{l2} l) l\right] e^{-\rho t} \\ & \text{subject to} \quad (4), \quad (5), \quad 0 \le h, \quad 0 \le l \le 1 \end{aligned}$$

where ρ is the discount rate. The constraints on the controls *h* and *l* are addressed implicitly in determining a solution.

2.3.2 Necessary Conditions

The Hamiltonian method is used to solve (6). The optimal choice of controls should maximize the Hamiltonian,

$$H = u_C \ln(C) + u_P \ln(P) - c_h h - (c_{l1} + c_{l2} l) l + \lambda_C \dot{C} + \lambda_P \dot{P}, \tag{7}$$

and any solution must satisfy the adjoint conditions,

$$\dot{\lambda}_C = \rho\lambda_C - \frac{u_C}{C} - \lambda_C \left(r_C L \left(\alpha_M (1-l) + \alpha_R l - \frac{2C}{L} \right) - \gamma P \right) - \lambda_P \gamma P, \tag{8}$$

$$\dot{\lambda}_P = \rho\lambda_P - \frac{u_P}{P} + \lambda_C \gamma C - \lambda_P \left(r_P L \left(\beta_M (1-l) + \beta_R l - \frac{2P}{L} \right) + \gamma C - h \right). \tag{9}$$

The costate variables λ_C and λ_P are the shadow prices of one more animal in stock C and P, respectively.

The Hamiltonian is linear in *h*, so maximization of (7) is a linear control problem in *h*. The optimality condition for *h* is therefore,

$$\frac{\partial H}{\partial h} = -c_h - \lambda_P P \begin{array}{lll} > 0 & iff & h^* \to \infty \\ = 0 & iff & h^* = h_{SV} \\ < 0 & iff & h^* = 0 \end{array}, \tag{10}$$

where h^* is the optimal choice of control. Condition (10) says that *h* is an impulse control when $\partial H/\partial h > 0$, which arises when the marginal cost is less than the marginal reduction in damages due to predator removal. No predator removal, $h^* = 0$, occurs when $\partial H/\partial h < 0$ and the marginal cost of removal outweighs the reduction in damages. When the special case of $\partial H/\partial h = 0$ occurs, i.e. a singular solution, *h* is set at its singular value, h_{SV}. Note that the singular solution implies $\lambda_P = -c_h/P$. Thus, when *h* follows an interior solution, the predator shadow price is negative.

This is intuitive, because managers only remove predators when an additional predator has negative value.

Assuming an interior solution, the optimality condition for *l* is,

$$\text{(11)} \qquad \frac{\partial H}{\partial l} = -c_{l1} - 2c_{l2}l + \lambda_C r_C LC(\alpha_R - \alpha_M) + \lambda_P r_P LP(\beta_R - \beta_M) = 0 \,.$$

Note that the shadow price of caribou, λ_C, will always be positive because caribou do not inflict any social or ecological damages. Using (11), *l* can be expressed as a function of λ_C, λ_P, *C* and *P*, defining $l^* = l(\lambda_C, \lambda_P, C, P)$. An interior solution, $l^* > 0$, requires $\lambda_C r_C C(\alpha_R - \alpha_M) + \lambda_P r_P P(\beta_R - \beta_M) - c_{l1}/L > 0$, where $\lambda_C r_C C(\alpha_R - \alpha_M)$ and $\lambda_P r_P P(\beta_R - \beta_M)$ are the marginal benefits of the conservation reserve on caribou and predators, respectively. This is possible if the *net* marginal ecological benefits of a conservation reserve, $\lambda_C r_C C(\alpha_R - \alpha_M) + \lambda_P r_P P(\beta_R - \beta_M)$ are sufficiently high. When predators are a nuisance, i.e. $\lambda_P < 0$, net marginal benefits of a conservation reserve will always be positive(since $\beta_R - \beta_M < 0$), which increases the likelihood that a conservation reserve will be optimal.[10] Thus, when $\lambda_P < 0$, not only is $h^* > 0$ possible but it is more likely that $l^* > 0$ is optimal.

The two adjoint equations, (8) and (9), can be used to derive a pair of golden rule conditions. For caribou this condition is

$$\text{(12)} \qquad \rho = \frac{\dot{\lambda}_C}{\lambda_C} + \frac{u_C}{\lambda_C C} + \left[r_C L\left(\alpha_M (1-l) + \alpha_R l - \frac{2C}{L} \right) \right] - (\lambda_C - \lambda_P)\frac{\gamma P}{\lambda_C} \,,$$

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[10] On the other hand, if $(\beta_R - \beta_M) > 0$ and $\lambda_P < 0$, net marginal conservation reserve benefits will tend to be small and could even be negative.

which equates the discount rate, ρ, to the net rate of return from conserving caribou. The net rate of return, on the RHS, is made up of four terms. The first term is the capital gain or loss from changes in the caribou stock. The second term is the marginal existence value of caribou, normalized by λ_C. The third term, in brackets, is the marginal growth in caribou before any predation. Here, it is seen that investing in a conservation reserve, *l*, increases the returns to caribou conservation. The final term is the cost of one more caribou to caribou predation, assuming $\lambda_C - \lambda_P > 0$, or the benefit of one more caribou to caribou predation if $\lambda_C - \lambda_P < 0$.

The golden rule condition for the predators is

$$\rho = \frac{\dot{\lambda}_P}{\lambda_P} + \frac{u_P}{\lambda_P P} + \left[r_P L \left(\beta_M (1-l) + \beta_R l - \frac{2P}{L} \right) - h \right] - (\lambda_C - \lambda_P) \frac{\gamma C}{\lambda_P}, \tag{13}$$

which has an interpretation analogous to condition (12) when $\lambda_P > 0$. The interpretation of (13) changes when $\lambda_P < 0$ and caribou managers must control a nuisance predator. Consider the RHS when controlling predators. The first term is the capital gain or loss from changes in the predator stock. The second term is the marginal existence value of predators, normalized by λ_P. The third term, in brackets, is the marginal growth in predators before any predation of caribou. The final RHS term is the benefit of one less predator to caribou predation. Note that investments in a conservation reserve, *l*, or in predator removal, *h*, reduce the rate of return from controlling predators.

2.3.3 Optimal Management Strategies

The optimal management strategy may be an interior solution, one of several corner solutions, or a combination of interior and corner solutions. Some strategies can be eliminated as possible solutions. Assuming L is sufficiently high such that all reasonable L_R are less than L, the corner solution $l = 1$ can be discarded as a feasible strategy.

The following strategies are feasible, although they can be ruled out as the long run solutions. This includes the strategy of employing maximum predator removal, $h \rightarrow \infty$, which, unless followed for an instant, implies the collapse of the predator stock and drives removal costs to infinity. Assuming a strategy of no conservation reserve and no predator removal ($l = h = 0$) leads to extinction, then this no management strategy should can only be followed temporarily because $l \rightarrow 0$ implies $C \rightarrow 0$ and $\ln(C) \rightarrow -\infty$. I refer to these strategies as short run-only strategies.

Generally, the following three strategies are the only possible long run—i.e. leading to a steady state—solutions: a conservation reserve with predator removal ($l^* > 0$, $h^* = h_{SV}$), predator removal and no conservation reserve ($l = 0$, $h^* = h_{SV}$) and a conservation reserve with no predator removal ($l^* > 0$, $h^* = 0$). Refer to these, respectively, as the dual control strategy, the predator removal-only strategy and the conservation reserve-only strategy.

The complete management solution could involve a combination of these long run and short run-only strategies. A solution which includes short run-only strategies will involve switches between strategies. These switches are defined implicitly in (10) and (11). For example, if a pulse removal, $h \rightarrow \infty$, is optimal it must be followed only for an instant, before switching to a strategy which has $h = h_{SV}$, i.e. a conservation reserve with predator removal ($l^* > 0$, $h^* = h_{SV}$) or just predator removal ($l = 0$, $h^* = h_{SV}$).

For the three candidate long run strategies, determining the optimal choice of controls is non-trivial. The solution to each of these strategies is discussed below. The first-best combination of strategies and the solution to managing the Gaspésie Woodland Caribou is then presented in the next section.

Dual control strategy ($l^* > 0$, $h^* = h_{SV}$)

This candidate strategy is determined by (8)-(11), setting $\partial H/\partial h = 0$ in (10). First, set $\partial H/\partial h = 0$ and solve for

(14) $\lambda_P(P) = -c_h/P$.

Condition (14) says that predators must be valued as a nuisance when managers are willing to incur harvest costs to control predation on caribou. Take the time derivative of (14), $d\lambda_P(C,P)/dt$.

Then substitute $\lambda_P(P) = \lambda_P$ and $d\lambda_P(C,P)/dt = \dot{\lambda}_P$ into (9) to derive to derive

(15) $$\lambda_C(C,P) = \frac{[-\lambda(P)(\rho + r_P P) + u_P/P]}{\gamma C}.$$

Consider the RHS of condition (15). The numerator implies that the marginal value of caribou in the optimal dual control strategy is determined by two components. The first component says that the marginal value of caribou is increasing in the nuisance value of predators. This is because caribou must be valued if investments in costly predator removal are optimal. The second component says that the marginal value of caribou is higher when there is a nonuse value to predators. This is because if it is optimal to remove predators even when they have non-use value, it must be that caribou are highly valued. Take the time derivative of (15), $d\lambda_C(C,P)/dt$.

Substituting $\lambda_C(C,P) = \lambda_C$ and $d\lambda_C(C,P)/dt = \dot{\lambda}_C$ into (8), it is then possible to derive h as a function of C and P: $h_{SV} = h^{DC}(C,P)$. For brevity the feedback rule for h is not presented here.

Recall that $\partial H/\partial l = 0$ can be used to write l in terms of C, P, λ_C and λ_P:

$$l^{*}=\frac{\left(\lambda_{C}r_{C}LC\left(\alpha_{R}-\alpha_{M}\right)+\lambda_{P}r_{P}LP\left(\beta_{R}-\beta_{M}\right)-c_{l1}\right)}{2c_{l2}}. \tag{16}$$

Examine the RHS of (16). Recall that $\alpha_R - \alpha_M > 0$ and $\beta_R - \beta_M < 0$. The first term reflects the value of reserve creation in support of caribou and is positive. The second term reflects the value of reserve creation for predator control and is also positive. Note that the second term implies that a conservation reserve in a system with nuisance predators should be larger than a reserve in a system without predators.[11] These additional reserve investments are made in effort to further control predators.

By inserting (14) and (15) into (16) the feedback relation for *l* is derived:

$$l^{DC}\left(C,P\right)=\frac{r_{C}Lu_{P}\left(\alpha_{R}-\alpha_{M}\right)/\gamma P+c_{h}r_{C}L\left(\rho+r_{P}P\right)\left(\alpha_{R}-\alpha_{M}\right)/\gamma-c_{h}r_{P}L\left(\beta_{R}-\beta_{M}\right)-c_{l1}}{2c_{l2}}. \tag{17}$$

$l^{DC}(C,P)$ is clearly decreasing in the costs of the reserve and increasing in the cost of predator removal. This is intuitive: managers should shift resources out of predator removal and into conservation reserve creation when the cost of predator removal increases or when the cost of the reserve decreases.

The optimal trajectory of the dual control strategy is defined by (4) and (5) with substitutions for $h = h^{DC}(X,Y)$ and $l = l^{DC}(C,P)$.

Predator removal-only strategy ($l = 0$, $h^{*} = h_{SV}$)

The trajectory for this strategy is found by solving (8), (9) and $\partial H/\partial h = 0$ in (10). $\partial H/\partial h = 0$ is

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[11] On the other hand, if the conservation reserve benefits nuisance predators on the margin ($\beta_R - \beta_M > 0$), there is less incentive to invest in conservation reserve creation in a system with predators than without.

used to solve for $\lambda_P(P)$ and $\dot{\lambda}_P$, which are then substituted into (9) to derive (15), which is unambiguously positive. Take the time derivative of (15), $d\lambda_C(C,P)/dt$, and substitute this and $\lambda_C(C,P)$ into (8) to solve for $h_{SV} = h^{PR}(C,P)$. Thus, the optimal rate of predator removal is defined by a feedback rule—a function of the current state values. For this strategy, the optimal trajectory is determined by (4) and (5) substituting $h = h^{PR}(X,Y)$ and $l = 0$.

This strategy is optimal only if the net marginal benefits of a conservation reserve are never positive ($\lambda_C r_C C(\alpha_R - \alpha_M) + \lambda_P r_P P(\beta_R - \beta_M) - c_{l1}/L < 0$). When is this likely? Examine the expression for λ_C in (15). This costate tends to be large, and therefore a predator removal-only strategy optimal, when predators have little marginal value. In this case, a manager should prefer to drive down the stock of predators using predator removal. The expression $\lambda_C r_C C(\alpha_R - \alpha_M) + \lambda_P r_P P(\beta_R - \beta_M)$ also tends to be small, and predator removal-only strategy optimal, when the marginal cost of predator removal is small.

Conservation reserve-only strategy ($l > 0, h^* = 0$)

This candidate strategy is determined by (8), (9) and (11). Condition (11) yields (14). With only the adjoint equations (8) and (9), analytically eliminating λ_C and λ_P from (14) to derive the feedback rule for l is infeasible. The optimal trajectory therefore defined by the implicit solution to (8), (9), and $l(C,P,\lambda_C,\lambda_P)$, as well as the state equations (4) and (5). This strategy is useful when caribou conservation is necessary but the predator is not viewed as a nuisance, $\lambda_P > 0$, or the predator is a nuisance but issues of cost make predator removal uneconomical, $\lambda_P > -c_h/P$.

2.4 Optimal Management of the Gaspésie Woodland Caribou

The Gaspésie Woodland Caribou is in historic decline. Although the population once spread from Quebec to New York, today the herd is restricted to Parc National de la Gaspésie, a conservation park. In the 1950s the population stood at around 750 individuals, falling to about 200 in the 1970s. Current estimates place the population at around 140 caribou. Human activity and predation are the major causes of the decline of the Gaspésie Woodland Caribou.

To protect the Gaspésie Woodland Caribou, in 1937 Parc National de la Gaspésie was established, with subsequent bans on hunting, forestry and mining within the park. After continued declines in the caribou population, in 1990, 1996 and 2001, predator removal was exercised, which temporarily improved caribou survival. In 1999, measures were taken to create additional caribou habitat outside the park boundaries. Management of park visitors now regularly occurs in order to further reduce human disturbance of the caribou. However, the Gaspésie Woodland Caribou population continues to decline.

The new Gaspésie Woodland Caribou recovery plan places the burden of recovery on continuous predator control. Implicitly, the caribou already receive significant aid in the form of conservation reserve investments (Parc National de la Gaspésie).

2.4.1 Parameterization

Parameterizing the bioeconomic model allows us to determine the optimal investment levels in predator control and the conservation reserve to protect the Gaspésie Woodland Caribou.

Parameter values are listed in Table 2.1. Most of the parameters are calculated from published sources or gray literature. See the Appendix for details. Exceptions include the predator growth rate parameter, r_P, the caribou carrying capacity of non-conservation reserve land, α_C (which general observation suggests is null), the cost of maintaining the conservation reserve at the expense of other economic activities (although a search of land values can yield some insights) and the discount rate. The robustness of the results to these assumptions is therefore tested with a sensitivity analysis.

2.4.2 Management Results

The optimal solution is determining by examining the three long run strategies. Numerical solutions were derived using Mathematica 7.0 (Wolfram 2008).

The predator removal-only strategy can be discarded a priori due to the assumption regarding non-conservation reserve land, i.e. $\alpha_M = 0$. Using only predator removal cannot be optimal in the long run because the caribou require $l > 0$ to survive. This leaves the conservation reserve-only strategy and dual control strategy as the feasible candidate long run strategies.

I find the conservation reserve-only strategy is subobtimal. This is implied by the dual control strategy result (see below), which constitutes a viable interior solution for all the states and controls. The conservation reserve-only strategy does have a unique and feasible interior steady state of $C^* = 27$, $P^* = 297$ and $l^* = 0.2098$. However, a check of the optimality condition for $h = 0$ reveals a contradiction. During any point the conservation reserve-only strategy is optimal, then $\partial H/\partial h < 0$—that is, increasing h away from zero to something slightly positive should reduce the Hamiltonian and therefore social net present value. However, at the conservation-reserve only strategy equilibrium I find $\partial H/\partial h > 0$. Since social net present value can be increased by some $h > 0$ amount, the conservation reserve-only strategy cannot be optimal.

The solution to (6) follows the dual control strategy. The optimal management plan is illustrated in Figure 2.1 as a phase plane. The trajectories of the dual control strategy are illustrated by phase arrows. Following any of these trajectories leads to infeasible values, except for a single path which leads to a steady state equilibrium. This path is a saddle path, and equilibrium occurs where the two nullclines ($\dot{C} = 0$, $\dot{P} = 0$) intersect.

The complete solution first involves a pulse harvest ($h^* \to \infty$) to initially remove a large portion of the predator population as well as a large decrease in the size of the conservation reserve. This shifts the system from point A to point B, on the saddle path. Then, the solution involves proceeding along the saddle path until, at point C, $C^* = 40$, $P^* = 229$, $h^* = 0.0628$ ($E^* = 14$) and $l^* = 0.2094$ ($L_R^* = 471$). Note that fewer resources are shifted into reserve creation here than in the conservation reserve-only strategy. Approaching the equilibrium, both the number of predators *and* caribou gradually decline.

Figure 2.2 illustrates the optimal levels of the controls as the system proceeds along the saddle path. After the initial shift to the saddle path, the size of the conservation reserve is established at 406 km^2. Over time, both the size of the reserve and the number of predators removed increases.

The solution I identify is in ways similar and dissimilar with the current approach of the Gaspésie Woodland Caribou managers. A dual control strategy that involves a sizable conservation reserve with significant predator removal is clearly optimal. This supports the general approach of caribou managers. A strategy of periodic predator culls, as historically performed by caribou managers, is not optimal.[12] Instead, the results support a management plan that involves continuous predator removal to protect caribou. Furthermore, the results indicate a much smaller conservation reserve is optimal than is currently provided for the caribou, and increasing the current amount of caribou habitat would reduce social welfare.

Overall, I find the optimal equilibrium number of caribou is small: it is far less than the current population of caribou and less than one-fourth of the recovery goal of caribou managers.[13] Although the caribou generate significant economic value, there is a high opportunity cost of placing land into the conservation reserve. On the other hand, the results could be misled by incorrect parameter estimates. This can be tested with a sensitivity analysis, to which I now turn.

2.4.3 Sensitivity Analysis

By altering the values of the various parameters of the model it is possible to gain further insights into optimal management strategies. The results of this sensitivity analysis are presented in Table 2.2. In the table the values of the controls are expressed in units rather than rates. The scenario at the original parameter values is referred to as the benchmark scenario. Most of the other scenarios are chosen to reflect plausible parameter changes.

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[12] Periodic culls can be optimal when there are substantial fixed costs in using a control (Clark, 2005). The model here assumes no fixed costs in running a predator removal strategy. It is not clear if Gaspésie Woodland Caribou managers face substantial fixed costs in organizing a hunting and trapping program for caribou predators.

[13] Indeed, $C = 40$ may not be a viable sustainable population level. It is not within the scope of the model to incorporate extinction risk due to exogenous events, such as a severe winter, or inbreeding depression, which are a concern for small populations. On the other hand, it may not be cost effective to conserve a significantly larger herd of caribou to mitigate these risks.

The first two alternative scenarios change biological parameters. When the predator growth rate falls, the optimal management response is to reduce the amount of predator removal. There is little visible effect on the size of the conservation reserve. Thus, the management response to shifts in the predator biology is primarily reflected in the predator removal strategy. The next scenario allows caribou to survive on land outside the conservation reserve, although land inside the conservation reserve still supports relatively greater caribou densities. Compared to the benchmark scenario, in this case the manager reduces investments in both conservation reserve creation and predator removal. This is an intuitive result, because the benefits of caribou conservation decline when caribou can survive on land not in a conservation reserve.

The remaining scenarios change economic parameters. Compared with the benchmark results, when the cost of predator removal increases the optimal response is shift resources out of predator removal and into reserve creation. Similarly, when the opportunity cost of conservation reserve land increases the optimal response is to shift resources out of conservation reserve creation and into predator removal.

When the predator has no social value, then the optimal management strategy involves eradicating predators. This may seem extreme, but when predators limit caribou survival and have little marginal existence value there is no justification for keeping predators around. The optimal conservation reserve size in this case is significantly less in the benchmark scenario. This is because once predators are eradicated the only benefit to investing in conservation reserve creation is increasing caribou carrying capacity (and not also decreasing the net predator carrying capacity of land; see section 2.3.3). Since the benefits of conservation reserve creation are less than in the benchmark scenario, the manager finds it optimal to reduce investments in the conservation reserve.

Not surprisingly, when the existence value of caribou is increased, it is optimal to increase investments in caribou conservation. Similarly, when the discount rate is reduced the manager should increase investments in caribou conservation. Compared with the benchmark scenario, in both these cases the optimal predator control rates and conservation reserve size increase significantly.[14] For the discount rate change, this is because future economic values are now more important, and the manager is less content to let the caribou population decline from its current population to a future, smaller stock level.

Of the eight scenarios tested, only two yield an equilibrium population of caribou that exceeds the conservation target for the Gaspésie Woodland Caribou. The first is the scenario that assumes caribou can survive on land outside of the conservation reserve, which is probably the least plausible of the alternative scenarios. The second is the scenario that increases caribou existence value, which is plausible. However, both scenarios include a conservation reserve significantly less than the size of the current Parc National de la Gaspésie conservation reserve for the Gaspésie Woodland Caribou. On the other hand, two scenarios have equilibrium conservation reserves close to the current size of the Parc National de la Gaspésie. These are the scenarios that consider a cost of land and a discount rate at half the benchmark values. Both may reflect the situation of caribou better than in the benchmark scenario. However, in either case the equilibrium stock of caribou is significantly less than the conservation goal and even less than current population estimates. Thus, unless the existence value of caribou is higher than current estimates, conserving only a small number of caribou is probably optimal.[15]

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[14] Both effects are consistent with other bioeconomic analyses of reserves. See Horan and Melstrom (2011).

[15] The benchmark existence value of caribou was calculated assuming only households in Quebec and New Brunswick valued the Gaspésie Woodland Caribou (see Appendix). If other

Changing the initial predator stock level

The dual control solution described in section 2.4.2 is unique to the case of the Gaspésie Woodland Caribou. To illustrate another solution strategy, I examine the case when the initial stock of predators is much lower than the stock used in the benchmark scenario ($P(0) = 200$ vs. $P(0) = 270$). As in the benchmark case, the solution involves proceeding along the saddle path to the steady state equilibrium. However, the initial starting point lies below the saddle path. This is indicated in Figure 2.3. Thus, in order to get to the saddle path, the solution must proceed along a trajectory different from that in the benchmark scenario. What is this trajectory? Note that below the saddle path (which defines the switching curve for h) $h^* = 0$ is optimal. Therefore, to reach the saddle path either the conservation reserve-only strategy ($l^* > 0$, $h^* = 0$) or the no management strategy ($l = 0$, $h^* = 0$) is optimal. I find that it is optimal to follow a trajectory from the conservation reserve-only strategy. This trajectory is indicated by the dashed line in Figure 2.3, starting at point A. The manager should choose l to keep the system on this trajectory until the saddle path is reached, at point B. Once at the saddle path, $l^{DC}(C,P) = l^{CR}$, where l^{CR} is the value of l following the conservation reserve-only strategy, and $h^* = h_{SV}$. Thereafter, the system proceeds along the saddle path to the equilibrium, at point C.

The choice of l (L_R) in the region where $l^{DC}(C,P) = l^{CR}$ is indicated graphically in Figure 2.4. The dashed line marks the path of the optimal conservation reserve-only trajectory, l^{CR}. The solid line represents $l^{DC}(C,P)$. Starting from point A, the conservation reserve-only trajectory is

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Canadian households strongly value Gaspésie Woodland Caribou conservation, investing more in caribou conservation will be optimal.

followed. At point B, the optimal l^{CR} trajectory lies tangent to the optimal $l^{DC}(C,P)$ trajectory. Once this point is reached, the conservation reserve-only strategy is abandoned in favor of the dual control strategy. Thus, there is no shift in conservation reserve size in moving from one strategy to the other.

This solution shows how to reach the equilibrium when the system initially lies below the saddle path. It is not possible to jump to the path using a pulse of predator removal, nor is it best to follow the no management strategy. Rather, I find the optimal solution avoids predator removal but includes conservation reserve investments, until the system reaches the saddle path. Once at the saddle path, it is optimal to switch to the dual control strategy in order for the system to proceed along the saddle path to equilibrium.

2.5 Conclusion

This paper analyzes predator control and habitat protection in an endangered species conservation problem. I show that managing habitat will simultaneously impact multiple species in an ecosystem. Habitat protection is therefore a unique management tool. It can be used to boost the stock of an endangered species, but it can also be used to control an endangered species' predators.

Studying the case of the endangered Gaspésie Woodland Caribou, it is possible to gain a number of new insights into ecosystem management. With a sufficiently large conservation reserve protecting its habitat, the caribou would cease to be conservation reliant. Predator removal would no longer be necessary to maintain a safe, sustainable population of caribou. On the other hand, placing land in a conservation reserve is costly. Many development opportunities have to be sacrificed to maintain a large reserve. This is largely why in the benchmark analysis the optimal management strategy utilizes a relatively small reserve.

Habitat management and species-specific, direct controls are both valid ecosystem management methods that can be used to conserve species. Although habitat loss is the primary reason most species are endangered, this does not mean habitat management is the best method to conserve endangered species. If human-induced land disturbance disproportionately favors some species at the expense of others—for example, due to excessive predation—then direct controls can be an effective conservation tool. Habitats can be restored, but direct controls may be comparatively cheap. For this reason, conservation-reliant species are likely to be a permanent if cost-effective feature of modern wildlife conservation.

2.6 Tables and Figures

Table 2.1. Parameter values

Parameter	Interpretation	Value
r_C	Caribou growth parameter	0.00011
r_P	Predator growth parameter	0.00089
α_M	Managed land effect on caribou	0
α_R	Reserve land effect on caribou	0.2
β_M	Managed land effect on predators	0.148
β_R	Reserve land effect on predators	0.074
γ	Predation parameter	0.000026
u_C	Benefits of caribou parameter	36,766,035
u_P	Benefits of predator parameter	3,986,302
c_h	Predator removal cost parameter	150,433
c_{R1}	Reserve cost parameter	154.89
c_{R2}	Reserve cost parameter	$c_{R1}/10$
ρ	Discount rate	0.05

Table 2.2. Sensitivity Analysis

Scenario	Optimal Strategy	Equilibrium Stocks (C^*,P^*)	Equilibrium Controls (E^*,L_R^*)	PVSNB (in millions $)
Benchmark	Dual Control	(40,229)	(14,471)	3,871
$r_P = r_{P0} \cdot 0.5$	Dual Control	(40,228)	(7,471)	3,871
$\alpha_M = \alpha_{R0} \cdot 0.5$	Dual Control	(220,312)	(1,345)	4,736
$c_h = c_{h0} \cdot 1.5$	Dual Control	(39,230)	(14,472)	3,871
$c_R = c_{R0} \cdot 0.5$	Dual Control	(105,259)	(3,826)	3,924
$c_R = c_{R0} \cdot 1.5$	Dual Control	(14,217)	(17,330)	3,852
$u_P = u_{P0} \cdot 0$	Dual Control	(86,<1)	(<1,430)	3,506
$u_C = u_{C0} \cdot 1.5$	Dual Control	(177,195)	(18,555)	6,260
$\rho = \rho_0 \cdot 0.5$	Dual Control	(105,153)	(18,706)	7,612

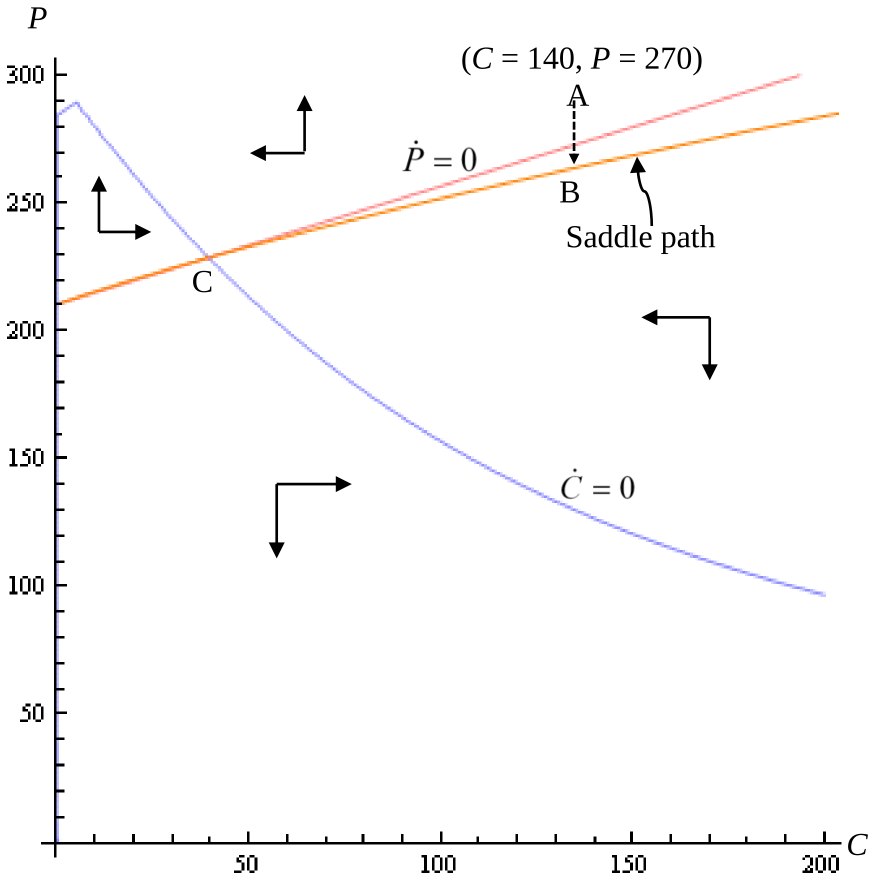

Figure 2.1. Solution diagram of the optimal management strategy, which is a dual control strategy ($l^* > 0$, $h^* = h_{SV}$).

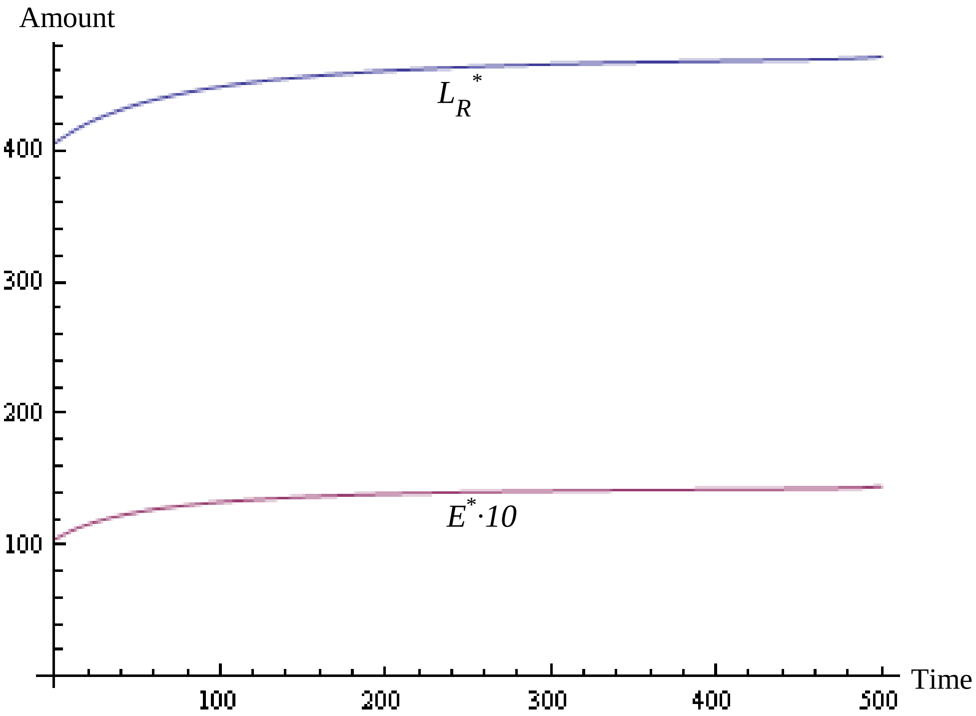

Figure 2.2. Time path of predators removed (x10) and conservation reserve size (in km^2) under the optimal management strategy.

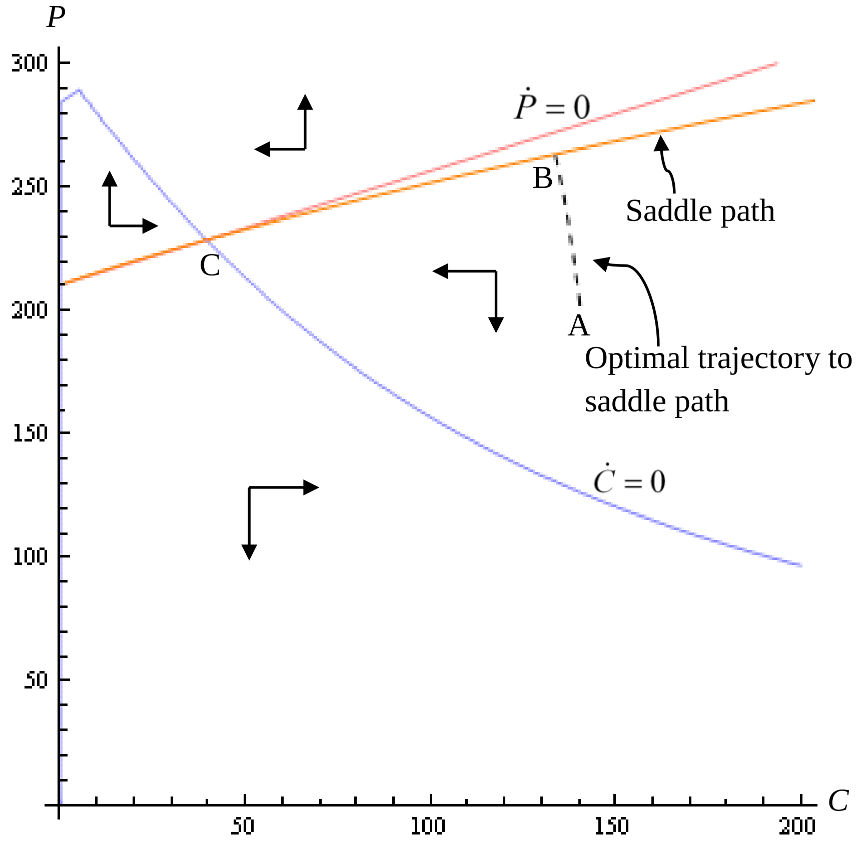

Figure 2.3. Solution diagram of the optimal management strategy when $C(0) = 140$, $P(0) = 200$, which is a dual control strategy ($l^* > 0$, $h^* = h_{SV}$).

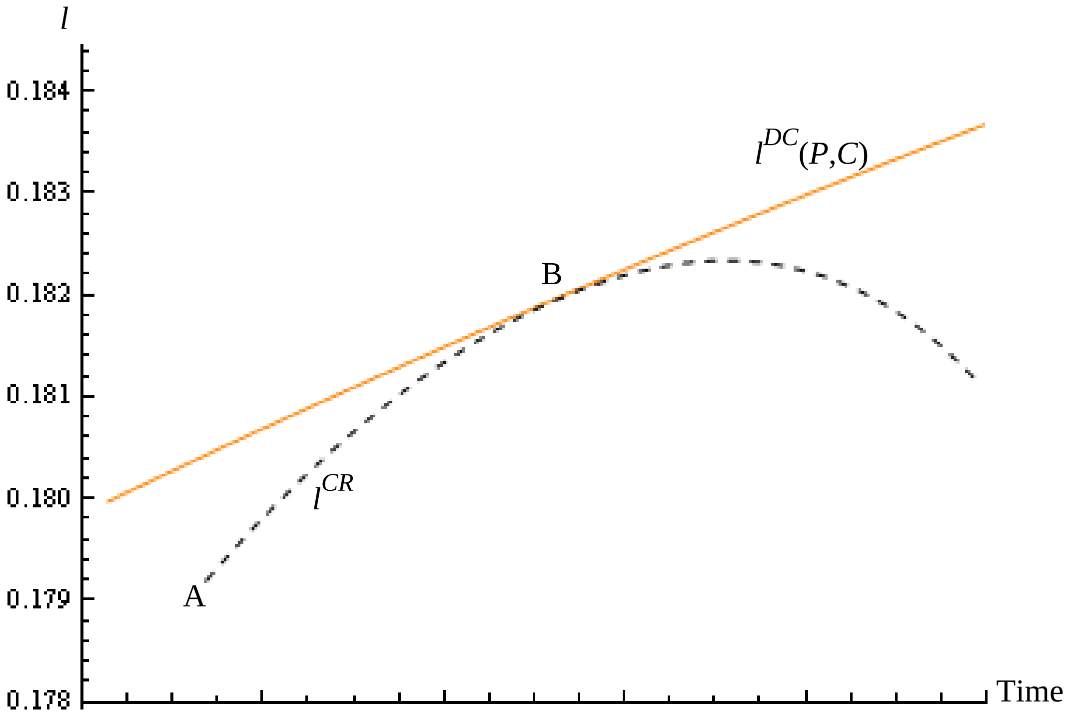

Figure 2.4. Time path of *l* trajectories following the saddle path (solid line) and the optimal short run conservation reserve-only strategy (dashed line).

2.7 Appendix

Total land: L. Land is measured in units of square kilometers. In the model, L = 2250, although in principle L is unlimited. This size is chosen because currently the Parc National de la Gaspésie conservation reserve is 802 km^2 (Mosnier et al, 2008), and calls to increase the size of the park include a 214 km^2 park enlargement with a 1234 km^2 buffer zone (Cadieux and Guay, 2010). This would imply a total reserve of approximately 2250 km^2, more than twice the size of the current park. It seems unreasonable to expect L_R to be greater than 2250 km^2 or that predators outside of the 2250 km^2 area to interact with caribou inside the conservation reserve, hence L = 2250.

Caribou growth parameter: r_C. Courtois and Ouellet (2007) construct a predator-prey model of wolves and Gaspésie Woodland Caribou. They use a logistic growth function and an intrinsic growth rate of .245 for caribou. The caribou growth dynamic of this paper uses a sigmoid growth function similar to that of Courtois and Ouellet. However, in Courtois and Ouellet caribou growth is modeled nonlinear in caribou carrying capacity, while it is linear here. To impute the analogous growth rate, .245 is divided by L. Thus, r_C = .00011.

Predator growth parameter: r_P. Predators are expected to exhibit a high growth rate from the consumption of prey that are not caribou and demand a high removal rate to reduce predator density (Sterling et al, 1983). Following population growth and harvesting trials, r_P = .00089 was selected.

Caribou carrying capacity from managed land: $\boldsymbol{\alpha_M}$**.** The general observation is that land managed for human use, even as a natural resource, is unsuitable habitat for caribou. This implies $\alpha_M = 0$.

Caribou carrying capacity from conservation reserve land: $\boldsymbol{\alpha_R}$**.** Courtois and Ouellet (2007) use a carrying capacity of 20 caribou per 100 km^2. This implies $\alpha_R = .2$.

Caribou predation rate and effect of caribou predation on predator growth: $\boldsymbol{\gamma}$**.** The parameter γ is calibrate from the following information. The extant number of predators is approximately 270, (coyote and bear; see Mosnier et al, 2008). The size of the current conservation reserve is 802 km^2. The sustainable population of caribou under the conditions of predation and the size of the conservation reserve is unknown, but St. Laurent et al (2009) show the smallest population of Gaspésie Woodland Caribou recorded is 96. Assuming a steady state where $L_R = 802$, $C = 96$ and $P = 270$, equation (2) implies $\gamma = -.000026$.

Predator carrying capacity from managed land and conservation reserve land: $\boldsymbol{\beta_M}$***,*** $\boldsymbol{\beta_R}$**.** To reduce the number of parameters that must be calibrated, the condition $\beta_M = 2\beta_R$ is imposed, which implies an square kilometer of managed land can support twice as many predators as a square kilometer of conservation reserve land. Assuming a steady state where $h = 0$, $L_R = 802$, $C = 96$ and $P = 270$, and where $\gamma = -.000026$ (see above), equation (3) implies

$$\begin{aligned}&0.00089 \cdot \left(2\beta_R 1448 + \beta_R 802 - 270\right) - 0.000026 \cdot 96 = 0\\&\Rightarrow \beta_R = 0.074\end{aligned}$$

and therefore $\beta_M = .148$. This calibration implies a predator density of about 15 predators per 100 km^2 on managed land and about 7 predators per 100 km^2 in a conservation reserve. These densities are within the bounds observed for forested black bear habitat, a major predator of caribou (Rogers and Allen, 1987).

Value of caribou and predators: u_C, u_P. Martín-Lopez et al. (2008) collect published estimates of willingness to pay for wildlife conservation. This includes values for woodland caribou ($44.71 in 2005 dollars) and coyotes ($5.49 in 2005 dollars). To impute the parameters, u_C and u_P, the number of households that could associate strong values for the Gaspésie Woodland Caribou population is estimated. The provinces of Quebec and New Brunswick are selected as the relevant geographic areas due to their proximity to the endangered Gaspésie Woodland Caribou population.

The current populations of Quebec and New Brunswick are approximately 7.9 million and .75 million, respectively. The average household size in Canada is 2.5, implying a total household population in the area proximate to the Gaspésie Woodland Caribou of 3.46 million. At current population levels and converting values to 2012 dollars, then,

$$u_C \cdot \ln(140) = 3460000 \cdot 52.51 \Rightarrow u_C = 36766035$$

$$u_P \cdot \ln(270) = 3460000 \cdot 6.45 \Rightarrow u_P = 3986302 \ .$$

Cost of predator removal: c_h. Smith et al. (1986) study the cost of a coyote control program. Coyotes are a major predator of caribou calves. Over three years of aerial hunting, they find the average cost of a hunt is $17,200 (1983 dollars). The average reduction in the coyote population per hunt is 26.33%. Adjusting to 2012 dollars, this implies

$$c_h \cdot 0.2633 = 39614 \Rightarrow c_h = 150433 \ .$$

Cost of conservation reserve: c_{R1}, c_{R2}, θ. The opportunity costs of taking land out of managed use and placed into a conservation reserve must include the lost benefits of activities such as for timber and hunting. This is calculated as a function of square kilometers of land. However, recreational benefit estimates are largely estimated on a per trip basis (Harshaw, 2000). Timber values are available per unit of land (often for old growth forests, e.g. van Kooten, 1995), but there is no published statistics available for the area around Gaspé.

The cost of conservation land is constructed as follows. The parameter θ is set at 2. Assume, at the current size of the conservation reserve, the annual benefit of one more hectare of managed land is \$250. From personal observation, this value implies a present value not outside the range of market values of forested land on the Gaspé peninsula. Then, converting the value per hectare into the value per square kilometer, the marginal cost of conservation reserve land is

$$c_{R1} + 2c_{R2}(802) = 25000 \; .$$

I assume $c_{R1} = 10c_{R2} = c_R$. This implies

$$c_R + .2c_R(802) = 25000 \Rightarrow c_R = 154.89,$$

so that c_{R1} = 154.89 and c_{R2} = 15.49.

REFERENCES

REFERENCES

Alexander, R.R. and D.W. Shields (2003). Using Land as a Control Variable in Density-Dependent Bioeconomic Models. *Ecological Modeling* 170: 193-201.

Bednar-Friedl, B., D.A. Behrens and M. Getzner (2012). Optimal Dynamic Control of Visitors and Endangered Species in a National Park. Environmental and Resource Economics 52: 1-22.

Borgmann, K. L. and M. L. Morrison (2010). Factors Influencing the Frequency of Nest Parasitism by Brown-Headed Cowbirds in the Northern Sierra Nevada. *Western North American Naturalist* 70: 137-143.

Boisjoly, D, J. P. Ouellet and R. Courtois (2010). Coyote Habitat Selection and Management Implications for the Gaspésie Caribou. *The Journal of Wildlife Management* 74: 3-11.

Brock, W. and A. Xepapadeas (2002). Optimal Ecosystem Management when Species Compete for Limiting Resources. *Journal of Environmental Economics and Management* 44:189-220.

Bulte, E.H. and R.D. Horan (2003). Habitat Conservation, Wildlife Extraction and Agricultural Expansion. *Journal of Environmental Economics and Management* 45: 109-127.

Clark, C.W. *Mathematical Bioeconomics.* Wiley, New York, 2005.

Chaudhuri, K. (1986). A Bioeconomic Model of Harvesting a Multispecies Fishery. *Ecological Modeling* 32: 267-279.

Courtois, R. and J.P. Ouellet. Modeling the Impact of Moose and Wolfe Management on Persistence of Woodland Caribou. *Alces* 43(2007): 13-27.

Crocker, T.D., and J. Tschirhart (1992). Ecosystems, Externalities, and Economies. *Environmental and Resource Economics* 2: 551-567.

Franklin, J.F. (1993). Preserving Biodiversity: Species, Ecosystems, or Landscapes? *Ecological Applications* 3: 202-205.

Hannesson, R. (1983). Optimal Harvesting of Ecologically Interdependent Fish Species. *Journal of Environmental Economics and Management* 10: 329-345.

Harshaw, H. W. (2000). *The Economic Valuation of Selected Canadian Non-timber Resources: An Examination of Valuation Methods and A Review of Results*. PhD dissertation, University of British Columbia.

van Kooten, G.C. (1995). Economics of Protecting Wilderness Areas and Old-Growth Timber in British Columbia. *The Forestry Chronicle* 71: 52-58.

Martín-Lopez, B., C. Montes and J. Benayas (2008). Economic Valuation of Biodiversity Conservation: the Meaning of Numbers. *Conservation Biology* 22: 624-635.

Millennium Ecosystem Assessment (MEA). *Ecosystems and Human Well-being: Biodiversity Synthesis*. World Resources Institute, Washington, DC, 2005.

Moon, D.C., J. Moon and A. Keagy (2010). Direct and Indirect Interactions. *Nature Education Knowledge* 1:9.

Mosnier, A., D. Boisjoly, R. Courtois, and J.P. Ouellet (2008). Extensive Predator Space Use Can Limit the Efficacy of a Control Program. *The Journal of Wildlife Management* 72: 483-491.

Cadieux, E. and J.P. Guay (2010). *Pour Une Meilleure Protection du Caribou de la Gaspésie*, Nature Quebéc.

Parc National de la Gaspésie Journal (PNG). Parcs Québec. English version, 2011-2012.

Ragozin, D.L. and G. Brown, Jr (1985). Harvest Policies and Nonmarket Valuation in a Predator-Prey System. *Journal of Environmental Economics and Management* 12: 155-168.

Gaspésie Woodland Caribou Recovery Plan (2002-2012) (Recovery Plan). Ministère des Ressources Naturelles et de la Faune, Secteur Faune Québec, 2006.

Rogers, L.L. and A.W. Allen. Habitat Suitability Index Models: Black Bear, Upper Great Lakes Region. U.S. Department of the Interior, Fish and Wildlife Service, Biological Report 82(10.144), 1987.

Sanchirico, J.N. and M. Springbor (2010). How to Get There from Here: Ecological and Economic Dynamics of Ecosystem Service Provision. *Environmental and Resource Economics* 48: 1-25.

Scott, J.M., D.D. Goble, J.A. Wiens, D.S. Wilcove, M. Bean and T. Male (2005). Recovery of Imperiled Species Under the Endangered Species Act: The Need for a New Approach. *Frontiers in Ecology and the Environment* 3: 383-389.

Scott, J.M., D.D. Goble, A.M. Haines, J.A. Wiens, and M.C. Neel (2010). Conservation-Reliant Species and the Future of Conservation. *Conservation Letters* 3: 91-97.

Smith, R.H, D. J. Neff and N. J. Woolsey (1986). Pronghorn Response to Coyote Control: A Benefit: Cost Analysis. *Wildlife Society Bulletin* 14: 226-231.

Sterling, B.W., W. Conley, and M. R. Conley (1983). Simulations of Demographic Compensation in Coyote Populations. *Journal of Wildlife Management* 47:1177-1181.

Stone, I, J. P. Ouellet, L. Siroisa, M. J. Arseneaua, M. H. St-Lauren (2008). Impacts of silvicultural Treatments on Arboreal Lchen Biomass in Balsam Fir Stands on Québec's Gaspé Peninsula: Implications for a Relict Caribou Herd. *Forest Ecology and Management* 255: 2733-2742.

Ströbele, W.J, and H. Wacker (1995). The Economics of Harvesting Predator-Prey Systems. *Journal of Economics* 61: 65-81.

Swallow, S.K. (1990). Depletion of the Environmental Basis for Renewable Resources: The Economics of Interdependent Renewable and Nonrenewable Resources. *Journal of Environmental Economics and Management* 19: 281-296.

Tanguay, M., W. Adamowicz and P. Boxall. *An Economic Evaluation of Woodland Caribou Conservation Programs in Northwestern Saskatchewan*. Project report 95–01. Department of Rural Economy, University of Alberta, Edmonton, 1995.

Wilcove, D.S, D. Rothstein, J. Dubow, A. Phillips and E. Losos (1998). Quantifying Threats to Imperiled Species in the United States. *Bioscience* 48: 106-115.

Wilcove, D. S. and Y.L. Chen (1998). Management Costs for Endangered Species. *Conservation Biology* 12: 1405-1407.

Wolfram Research, Inc., *Mathematica 7*.0, Champaign, IL, 2008.

CHAPTER 3

Valuing Great Lakes Recreational Fishing with Controls for Travel Cost Endogeneity

3.1 Introduction

How the demand for recreation trips is formulated can play a significant role in valuing recreation sites and site quality characteristics. In particular, if some trip and site characteristics important to the recreationist's visiting decision are ignored (e.g. water quality, presence of a boat ramp, substitute sites), researchers may misinterpret the value of other, observed site characteristics or the value of the site itself. How a researcher chooses to addresses the site characteristics not directly observed is therefore an important concern.

The omission of site quality characteristics can create endogenous site characteristics and bias in economic models. This problem has been widely noted in the recreation demand literature, both in single-site (e.g. Parsons, 1991; Weber et al, 2012) and multiple site (e.g. Hausman et al, 1995; Murdock, 2006) demand applications. In both cases, it is possible to value recreation sites and their characteristics using the travel cost method, assuming that travel cost is exogenous and is therefore uncorrelated with unobserved site characteristics. However, if key site quality characteristics are omitted, travel cost (or any other observed site characteristic for that matter), may become endogenous. Biased estimates of the travel cost effect in recreation demand modeling is a serious concern given the travel cost coefficient is the denominator in formulas for the valuation of sites and site characteristics.

This paper explores potential travel cost endogeneity in a site choice model of Great Lakes recreational fishing. Using a random utility framework and data on site characteristics and angler trips, it is possible to value the various Great Lakes recreational fishing locations and their amenities. However, these values will be biased if travel cost is correlated with unobservables in the error term. This may arise if anglers choose to live near fishing sites with favorable site characteristics, “selecting” a smaller distance and cost to visiting the sites. If any of the site characteristics influencing the angler’s residential location decision are omitted in the fishing site choice model, travel cost will be endogenous. One approach that could correct for this is the use of alternative specific constants (ASCs). This approach is explored here, but I also consider an alternative correction method that has not yet been used in recreation demand modeling. In particular, I employ a control function to correct for travel cost endogeneity. A control function is an instrumental variable method that, similar to two-stage least squares, uses a two-step estimation approach. The first step estimates a reduced form equation for the endogeneous variable (i.e. travel cost) with the observables and an instrumental variable as a regressor. I exploit the time variation in fuel prices to derive such an instrument for the Great Lakes recreational fishing model. The second step estimates the site choice model as usual except that the residuals from the first regression included as a regressor.

The application uses new data on Great Lakes recreational anglers. Catch rates unique to the timing of an angler’s trip are used to describe the fishing quality of alternative sites. Therefore, catch rates vary across anglers and alternatives, so the fixed-effects nature of ASCs does not prohibit estimation of the catch rate effects in the site choice model. The ASCs do correct for travel cost endogeneity, although I find the magnitude of bias is small. However, by soaking up the cross-site variation in catch rates, ASCs make identification of the catch rate parameters difficult. I show that the control function also corrects for travel cost endogeneity, but without the loss of cross-site variation to ASCs. In general, the control function may be a useful alternative to ASCs-based methods that correct for endogeneity in recreation demand models.

3.2 Background

Travel cost becomes endogenous when unobserved site characteristics are correlated with travel cost. This can arise when anglers choose where to live based on their fishing preferences. This issue was first raised by Parsons (1991) and Randall (1994), who noted that recreation preferences could influence the choice of residential location. If these preferences are driven by characteristics not captured in the recreation demand model, travel cost will be endogeneous.

There is good indication that travel cost to recreational sites is correlated with site amenities. In an empirical study of household preferences for metropolitan areas of the United States, Albouy (2009) finds that households are willing to pay to live near the coasts, including the Great Lakes. Deller et al. (2001) look at rural areas in the United States and find that a county's recreational infrastructure, water and land amenities are major determinants of rural population growth. With specific regard to anglers, Weithman and Haas (1982) report over one-fourth of anglers in Taney County, Missouri chose their home partly due to nearby fishing opportunities.

In theory, travel cost endogeneity is implied by hedonic analysis that values amenities through home prices. In hedonic analysis, individuals are assumed to freely choose where to live, selecting a home based on amenities that include the travel costs to work, shopping and recreation opportunities. In so far as commuting costs to activities—including fishing—are a significant determinant of residential location, any model that assumes travel costs are exogenous is inconsistent with household behavior. Phaneuf et al. (2008) do, in fact, find that access to recreation sites and water quality at recreation sites are important factors in residential location decisions. However, if the amenities that determine residential location are believed to be largely uncorrelated with the recreation site characteristics important to anglers, then travel cost endogeneity may be less of a concern. Additionally, if travel cost is correlated only with observed site characteristics and not unobserved characteristics, then travel cost will not be endogenous.

The handful of recreation demand studies that control for unobserved site characteristics correlated with travel cost (or other observed site characteristics) present an assortment of the various strategies that could be used to correct for endogeneity in different demand modeling contexts. Parsons (1991) uses a two-stage least squares strategy in a single site application. He finds that correcting for travel cost endogeneity leads to higher estimates of site value. Despite this finding, the matter of travel cost endogeneity received little attention until recently. This may be due to the limitations of two-stage least squares, which is not a valid correction method for endogeneity in the nonlinear models that most recreation demand modelers now use.[1]

An alternative method to two-stage least squares that is valid in nonlinear models is the inclusion of fixed effects. Fixed effects, also referred to as alternative-specific constants (ASCs), can capture the effects of unobserved characteristics and thus remove the correlation between the travel cost variable and the error term. Hausman et al. (1995) use fixed effects in this manner in a site choice model of recreational trips in Alaska.[2] However, following Hausman et al., the use of ASCs was avoided by most subsequent studies, perhaps because recreation data is frequently cross-sectional and the observed site characteristics are site-specific, and in site choice modeling ASCs absorb all variation (both unobserved and observed) that is site-specific. In this manner, ASCs can protect against travel cost endogeneity but their use prohibits identification of any

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[1] See Moeltner and von Haefen (2011) for a review of the methods that correct for endogeneity and, more generally, unobserved heterogeneity in recreational demand modeling.

[2] Hausman et al. refer to the ASCs in their trip allocation model as site-specific effects.

other variables measuring site-specific characteristics when estimating the site choice model—a problem avoided by Hausman et al., who could estimate the effects of site characteristics that changed over time, such as catch rates, because they had panel data on site visits.

To overcome this limitation of ASCs as a correction method for endogeneity, Murdock (2006) demonstrates that, after estimating a site choice model with ASCs, the coefficients of site-specific characteristics can be recovered in an auxiliary regression. In essence, this is possible because the information captured in the estimated ASCs shares a direct relationship with the site-specific characteristics. The framework for this method was originally developed by Berry (1994) and Berry et al. (1995) for models of differentiated product markets. Murdock presents the following two stage approach. First, estimate the choice model with ASCs. If travel cost is correlated with unobserved site-specific characteristics, the effect of travel cost in the choice model can then be estimated consistently. Second, regress the estimated ASCs on the observed site-specific characteristics. The effects of the site-specific characteristics on site choice are thus estimated in the second stage. The development of this bias-correction method has encouraged the recent use of ASCs in recreation demand models (e.g. Phaneuf et al, 2009; Timar and Phaneuf, 2009; Jakus et al, 2010).

Von Haefen and Phaneuf (2008) and Moeltner and von Haefen (2011) caution against the use of the two stage ASCs approach to correct for endogenous site characteristics. They argue that the effectiveness of the approach relies on features of the data and model, such as the use of a large choice set with a sufficient number of visits to each alternative. On the other hand, von

Haefen and Phaneuf (2008) show that the use of stated preference data can alleviate some of the drawbacks of the two stage ASCs bias-correction method.[3]

Alternatively, a control function approach can be used to correct for endogeneity. Early applications of a control function were performed by Smith and Blundell (1986) in a tobit model and Rivers and Voung (1988) in a probit model. More recent applications include Liu et al. (2011) and Ricker-Gilbert et al. (2011). Petrin and Train (2010) present the control function approach as an alternative to the bias-correction methods of Berry (1994) and Berry et al. (1995) in discrete choice models. The control function approach may therefore be a useful alternative to the two stage ASCs bias-correction method. A control function corrects for endogeneity by conditioning on the portion of the error in the model that is correlated with the observed covariates (Petrin and Train, 2006). Employing a control function is similar to standard two-stage least squares methods (which condition *out* the portion of the endogenous variable correlated with the error), but it remains valid in nonlinear models. It also can be used in situations where the two stage ASCs bias-correction method is unsound. As with two-stage least squares, a good instrumental variable is necessary to eliminate endogeneity (Petrin and Train, 2010). Control functions have not been used before in recreation demand modeling.

3.3 Model

3.3.1 Site Choice Model

A random utility model is developed in the context of a recreational angling site choice problem. Consider an angler’s utility function. On a fishing trip occasion, an angler maximizes utility by

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[3] Recent work by Abidoye et al. (in press) examine a Bayesian approach to recover the effects of site-specific characteristics when ASCs are used to control for unobserved site characteristics. Unlike the two stage estimation approach of Murdock (2006), the approach suggested by Abidoye et al. can be used in general applications of mixed logit models.

selecting the site which delivers the greatest level of utility subject to an income constraint. Let y_i be the income of angler *i*, p_{ik} the travel cost to site *k*, $\boldsymbol{q_k}$ a vector of quality measures that vary across sites and $\boldsymbol{q_{ik}}$ a vector of quality measures that vary across sites and anglers. The vector $\boldsymbol{q_k}$ is site-specific characteristics—that is, site characteristics common to all anglers—such as a dummy variable for whether a boat ramp is at the fishing site. In contrast, the vector $\boldsymbol{q_{ik}}$ includes site characteristics specific to an angler or the timing of the angler's trip, such as the average fishing catch rate at a site in the month the angler fished. It could include interactions between angler characteristics and site-specific characteristics, such as an indicator for boat ownership interacted with the shoreline length of a site. Assume utility is linear in its arguments and has a marginal utility of money that is constant across the choice alternatives. Then, after substituting in the budget constraint, angler *i*'s utility conditional on visiting site *k* is

$$(1) \qquad U_{ik} = \rho(y_i - p_{ik}) + \boldsymbol{\beta_1 q_k} + \boldsymbol{\beta_2 q_{ik}} + \varepsilon_{ik}.$$

where ρ is the marginal utility of money, $\boldsymbol{\beta_1}$ is the vector of parameters on site-specific characteristics, $\boldsymbol{\beta_2}$ is the vector of parameters on site-angler varying characteristics and ε_{ik} is the unobserved utility component. The error ε_{ik} is usually assumed to be independent of all other variables in the right-hand side of (1), but this may not be the case. In particular, travel cost endogeneity arises because p_{ik} is correlated with ε_{ik}, which is to say $cov(p_{ik},\varepsilon_{ik}) \neq 0$. Note that there could also be reason to think ε_{ik} is correlated with $\boldsymbol{q_k}$ or even $\boldsymbol{q_{ik}}$.[4]

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[4] I do not specifically address other potentially endogenous variables beyond travel cost. However, with suitable instruments, the control function method can correct for other endogenous variables in the same way that it corrects for travel cost endogeneity.

3.3.2 Alternative Specific Constants

Until recently, in applications with cross section data, using a full vector of ASCs prevented identification of the coefficients on the observed site characteristics in random utility models. Murdock (2006) shows that after estimating a choice model with ASCs, the ASC parameter estimates can be regressed on the site observables to recover the coefficients on the site-specific characteristics.[5]

Formally, the two stage ASCs bias-correction method yields the site utility

$$U_{ik} = \rho(y_i - p_{ik}) + \boldsymbol{\alpha_i} + \boldsymbol{\beta_2 q_{ik}} + \zeta_{ik}, \tag{2}$$

where $\boldsymbol{\alpha_i}$ is the vector of ASCs, which captures both observed and unobserved site-specific characteristics. So that the parameters in (2) can be estimated consistently, it is necessary and typical to assume $cov(p_{ik},\zeta_{ik}) = 0$, i.e. the idiosyncratic portion of the error is uncorrelated with individual travel cost (Murdock, 2006). This is the first stage of two stage ASCs method. In the second stage, the estimated ASCs are regressed on the site-specific characteristics, $\boldsymbol{q_k}$. The second stage can be estimated with ordinary least squares or, if $\boldsymbol{q_k}$ is endogenous, with two-stage least squares (Murdock, 2006).

The two stage ASCs method does have some limitations. For sites with very few visits $\boldsymbol{\alpha_i}$ is unlikely to be well identified, which makes identification in the second step difficult. In fact, it is not possible to estimate the constants for sites with no visits, so it is necessary to drop unvisited sites to implement the ASCs method. Identification of $\boldsymbol{\beta_1}$ in the second step is also difficult if there are relatively few sites in the choice set. Furthermore, poor estimation of the

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[5] With a number of *X* alternatives in the choice model, there will be *X*-1 ASC estimates and thus *X*-1 observations with which to predict site-specific effects.

constants $\boldsymbol{\alpha_i}$ for sites with few visits may affect estimates of $\boldsymbol{\beta_2}$ in the first step. It may also be inappropriate to perform the two step ASCs method in mixed logit models (Abidoye et al, in press).

3.3.3 Control Function

Employing a control function is similar to standard instrumental variables methods involving a two-step estimation procedure, such as two-stage least squares. Petrin and Train (2006) (see also Wooldridge, 2012) show that under basic assumptions one can include in a model with endogenous covariates an additional variable to condition for the portion of the error correlated with the observables.

For the Great Lakes sportfishing model described above, suppose travel cost can be expressed in reduced form as

(3) $p_{ik} = \boldsymbol{\gamma_q q_{ik}} + \boldsymbol{\gamma_z z_{ik}} + \mu_{ik},$

where $\boldsymbol{z}_{ik}$ is a vector of observed covariates and μ_{ik} is unobserved. The argument here is that p_{ik} is partly determined by the choices of anglers. In this case, $\boldsymbol{z}_{ik}$ are observables important to an angler’s travel cost to a fishing site but not a part of an angler’s site choice decision (i.e. utility at a fishing site), while the variable μ_{ik} contains elements in the angler’s residential location decision not observed by the researcher.

Assume μ_{ik} is independent of $\boldsymbol{q_{ik}}$ and $\boldsymbol{z_{ik}}$. Travel cost endogeneity arises because the unobserved component of travel cost, μ_{ik}, is correlated with the unobserved component of the site utility function, ε_{ik}. This can occur when a key site characteristic, such as scenic beauty, is important in fishing site choice, and in turn encourages anglers to live closer to the fishing site, but is unobserved by the researcher, so its effect becomes captured by both μ_{ik} and ε_{ik} However, an estimate of μ_{ik} can be had by estimating (3) and calculating the residuals, $\hat{\mu}_{ik}$. Inserting $\hat{\mu}_{ik}$ (i.e. the control function) into the random utility model, then

$$(4) \qquad U_{ik} = \rho(y_i - p_{ik}) + \boldsymbol{\beta_1 q_k} + \boldsymbol{\beta_2 q_{ik}} + \varphi\hat{\mu}_{ik} + \omega_{ik} .$$

Assuming $\hat{\mu}_{ik}$ is a good estimate of μ_{ik}, so the control function captures the effect in ε_{ik} that p_{ik} is correlated with, then $cov(p_{ik},\omega_{ik}) = 0$ and the parameters in (4) can be estimated consistently (Petrin and Train, 2010).

In short, the control function procedure consists of two steps. First, the endogenous variable of the primary model of interest is regressed against the observables and a vector of instruments. Second, the primary model is estimated but with the residual from the first regression as an additional variable.

As with the ASCs method, the control function method has some limitations. First, a control function requires an instrument. These instruments, $\boldsymbol{z_{ik}}$, must be correlated with p_{ik}, hence $\boldsymbol{\gamma_z} > 0$, but uncorrelated with ε_{ik}, $cov(\boldsymbol{z_{ik}},\varepsilon_{ik}) = 0$. Second, a control function for p_{ik} will only capture endogeneity correlated with p_{ik}. Thus, if $\boldsymbol{q_k}$ is endogenous it will be necessary to include additional controls. Finally, if the estimate of φ is statistically significant, the standard errors in the site choice model stage need to be corrected for two-stage estimation.[6]

3.3.4 Estimation and Welfare Analysis

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[6] This could be done using the bootstrap.

In (1), (2) and (4), assuming the error is independently and identically distributed extreme value yields the conditional logit model. The probability of individual *i* choosing alternative *k* is

$$(5) \qquad \pi_{ik} = \frac{e^{V_{ik}}}{\sum_{l=1}^{K_i} e^{V_{il}}},$$

where V_{ik} is the deterministic portion of the utility function and K_i is the number of alternatives in the choice set. For the traditional conditional logit $V_{ik} = \rho p_{ik} + \boldsymbol{\beta_1 q_k} + \boldsymbol{\beta_2 q_{ik}}$, for the conditional logit with ASCs $V_{ik} = \rho p_{ik} + \boldsymbol{\alpha_i} + \boldsymbol{\beta_2 q_{ik}}$ and for the conditional logit with control function $V_{ik} = \rho p_{ik} + \boldsymbol{\beta_1 q_k} + \boldsymbol{\beta_2 q_{ik}} + \hat{\mu}_{ik}$. The parameters of these three site choice models are estimated via maximum likelihood.[7]

In the second step of the ASCs method, ordinary least squares is used to estimate the site-specific effects. In the first stage of the control function approach ordinary least squares is used to estimate (3).

For welfare measurement, from the observed indirect utility function V_{ik} the expected per-trip compensating variation measure is

$$(6) \qquad E[CV] = \frac{1}{\rho}\left[\ln\left(\sum_{l=1}^{K} e^{V_k^1}\right) - \ln\left(\sum_{l=1}^{K} e^{V_k^0}\right)\right]$$

where V_k^0 is the conditional utility level prior to some change in price or quality, while V_k^1 is the level after the change. It is clear that biased parameter estimates will produce invalid site values. In particular, if the estimate of ρ is biased upwards, *ceteris paribus*, welfare estimates will be too

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[7] Specifications other than conditional logit were tested, including nested logit and mixed logit. The improvement in the fit of these two models over the conditional logit was extremely modest. Moreover, these various models are known to deliver similar welfare estimates (von Haefen, 2003).

small. On the other hand, if the travel cost coefficient estimate is biased downwards, *ceteris paribus*, welfare calculations will overstate the value of a site.

3.4 Data

3.4.1 Fishing site characteristics

Data were obtained from a 2008-2009 survey of anglers who fished in Michigan. Information from the survey included the location and date of anglers' most recent trips, targeted fish species, whether the purpose of was primarily to fish and how many days were spent fishing.[8]

The sample used in the application included only Michigan residents who took sportfishing trips to the Great Lakes. The sample was trimmed to anglers who took a day trip and traveled less than 200 miles one-way to their destination, which was considered the maximum feasible distance for a day trip. Accordingly, an angler's choice set included only sites within 200 miles of their home. Information of site quality measures was gathered independently of the survey. A list of the variables used in the models is presented in Table 3.1.

Travel costs were imputed from distance information, angler characteristics and gasoline prices. First, per-mile fuel costs were computed from Michigan monthly retail gasoline prices (per gallon, see EIA, 2012) divided by 23.1 (typical miles per gallon for cars; see AAA, 2009). For anglers who reported trailering a boat to the site, fuel costs were increased by 33%. Second, per-mile maintenance and depreciation costs were gathered from AAA reports (see AAA, 2008; AAA 2009). Third, the opportunity cost of travel time was computed as a 1/3 the angler's annual income divided by 2000, normalized for time by assuming an average driving speed of 45 miles

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[8] See Simoes (2009) for a detailed treatment of the survey and the sample.

per hour. Travel costs were calculated as per mile fuel costs, plus per mile maintenance and depreciation costs, plus the opportunity cost assuming a driving speed of 45 miles per hour.

Michigan Department of Natural Resources creel survey data were used to compute expected monthly per-hour catch rates at fishing locations on the Great Lakes for six fish species: chinook, coho, steelhead, lake trout, walleye and yellow perch. These are the most popular species among Great Lakes anglers. For the warm water species, the catch rates were defined as

$$catch\ rate_i = \frac{number\ of\ species\ i\ landed}{hours\ spent\ fishing\ for\ i}$$

for *i* = walleye and yellow perch. For the salmonids, the catch rates were computed over an effort level that summed all the time spent fishing for the species.[9] That is,

$$catch\ rate_i = \frac{number\ of\ species\ i\ landed}{\sum_j hours\ spent\ fishing\ for\ j}$$

for *i*, *j* = chinook, coho, steelhead and lake trout. From this information, 66 potential fishing sites were distinguished along Michigan's Great Lakes coast. Due to significant gaps in the data—missing average catch rates for several months for most sites—predicted average hourly catch rates by site and month were used. These predictions were calculated from a series of Tobit regressions. Each regression was specific to a fish species and region of the Great Lakes. Alternative methods of filling in missing data, including multiple imputation, were found impracticable.[10]

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[9] The Michigan Department of Natural Resources computes the salmon catch rates in this manner because it is relatively easy for anglers to target more than one salmon species at a time.

[10] Using an estimate of catch rates is necessary but introduces measurement error. In angling models, the size of this measurement error bias can be large (Morey and Waldman, 1998). However, Train, McFadden and Johnson (2000) point out that this does not mean the approach is inconsistent. In particular, the error variance declines as more data become available. Since the Michigan DNR use stratified random sampling, month-to-month, to creel fishing sites, and surveying at creeled sites is extensive, the bias should be limited. On the other hand, it is also

After expected catch rates were computed for each site, anglers' reported destinations from the survey were matched to the closest creeled fishing site. Since trips were taken at different times of the year, the expected catch rate at a fishing site depends on the month of the trip. Therefore, catch rates varied across sites *and* anglers. Using ASCs in the site choice model will capture part but not all of the variation in catch rates, which allows the catch rate parameters to be estimated in a site choice model with ASCs.

In this model, anglers who fish for the salmonids—chinook, coho, steelhead and lake trout—are distinguished from those who target the warm water species. These collections of species are referred to as product lines. This distinction is desirable because they constitute different types of fishing experiences, i.e. goods, for prospective fishing trips (Kikuchi, 2986; Jones and Lupi, 2000). In particular, catching salmonids in the Great Lakes involves fishing in certain temperature zones, usually from boats in deep water during the summer or from breakwaters and river mouths in the spring and fall. Perch and walleye can be caught in similar temperature zones, usually in the same location year-round. With 66 fishing locations, between the two product lines there are 132 alternatives.

A number of dummy variables are also included as covariates. This includes *Urban*, which takes the value one if the U.S. Census 2000 considers at least a portion of the zip code in which the fishing alternative is located urban. *Urban* is intended to be a measure of amenities at a site. *Highway* takes the value one if a state highway runs adjacent to the alternative. *Highway* is intended to be a measure of the remoteness of a site. *Bayorseaway* takes the value one if the alternative is located on a large bay or seaway (e.g. the St. Mary's River). *Bayorseaway* is intended to capture the preference for fishing in areas of the lake that are typically warmer.

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possible that the catch rate regressions make predictions that more accurately reflect angler perceptions of catch rates than would using true catch rates.

Bayorseaway is also interacted with a dummy variable indicating if the alternative is a warm water alternative (*Warmwater*), so that *Bayorseaway* can affect preferences for sites across the two product lines differently. This interaction term are intended to capture the difference in *Bayorseaway* preferences between anglers that take a trip to target warm water versus cold water species.

The indirect utility function for the traditional conditional logit is specified as

$$V_{ik} = \rho p_{ik} + \boldsymbol{\beta_{CR} CR_{ik}} + \beta_U Urban_k + \beta_H Highway_k + \beta_B Bayorseaway_k$$
$$+ \beta_{BWW} Bayorseaway_k \cdot Warmwater_k$$

where $\boldsymbol{CR}_{ik}$ is the vector of catch rates. The indirect utility function for the conditional logit with the control function is identical, except for the inclusion of the control function as a covariate. The indirect utility function for the conditional logit with ASCs is $V_{ik} = \rho p_{ik} + \boldsymbol{\alpha_i} + \boldsymbol{\beta_{CR} CR_{ik}}$; the second step in Murdock's two step method estimates $\alpha_i = \beta_U Urban_k + \beta_H Highway_k + \beta_B Bayorseaway_k + \beta_{BWW} Bayorseaway_k \cdot Warmwater_k$.

3.4.2 Instrumental Variables

To correct for travel cost endogeneity, a valid instrument must be correlated with p_{ik}, so that $\varphi \neq 0$ in (4), but uncorrelated with ε_{ik}, so that the instrument is exogenous to site choice. For the particular multinomial model used here, which is a conditional logit, it is also necessary that an instrument have cross-site variation, so that the control function is not differenced away in estimating the model.

The nature of the instrument used for the control function is based in the construction of the travel cost variable described in section 3.4.1. Note that the travel cost computed for each destination depend on the origin of the trip and, due to fuel prices, the month of the trip. Although the data set is not a panel, the different timing of trips across 2008-2009 captures several large exogenous changes in fuel prices that influence travel costs.

As an instrument, I use crude oil prices interacted with the distance, in miles, to fishing sites. This instrument clearly satisfies the condition that it be correlated with travel cost. It is not possible to test the exogeneity restriction. However, it is unlikely that crude oil price times distance is correlated with the unobserved site-specific effects of site choice model, for two reasons. First, unobserved characteristics, such as scenic beauty, are not likely to be a function of or determined by (current) oil prices and distances. Second, unobserved natural events in Michigan or the Great Lakes that may influence site choice are unlikely to impact and therefore be correlated with oil prices, since oil prices are determined on a global scale. I also include month-year fixed effects in the travel cost equation to control for exogenous shifts in demand.

3.5 Results

Four different site choice models were estimated. The results are presented in Table 3.2. The first estimated model is a traditional conditional logit, which ignores any potential travel cost endogeneity. The second is conditional logit that includes only the set of alternatives that received visits in the sample (there are 31 unvisited sites). The third is a conditional logit with ASCs, which also does not include any unvisited sites in the choice set. The parameters of site-specific characteristics are not estimated in the conditional logit with ASCs. However, following Murdock's two stage procedure, these parameters are estimated in an auxiliary regression. The final estimated model is identical to the first conditional logit except that it includes a control function.

In the traditional conditional logit model, the catch rate parameters are all positive, implying that, ceteris paribus, anglers are more likely to choose sites with high catch rates. The parameters on *Urban* and *Highway* are positive, so that all else being equal anglers are more likely to choose sites with urban amenities and that are not isolated. In contrast, the effect of the *Bayorseaway* is negative while the effect of *Bayorseaway·Warmwater* is positive, implying that anglers avoid cold water alternatives that lie in bays or seaways but are drawn to the warm water alternatives in bays or seaways. This may arise for two reasons. First, bays and seaways offer more protected waters for anglers using smaller boats, and these tend to be warm water anglers. Second, bays or seaways are better habitats for warm water species and anglers that prefer warm water alternatives are drawn to these locations; the opposite holds true for anglers that prefer cold water species. The results of the conditional logit on the sample that includes only visited alternatives are similar, although the precision of estimates is less compared with the traditional conditional logit. Thus, there is some loss of information when unvisited sites are dropped from the analysis.

The conditional logit with ASCs yields substantially weaker results compared with the two prior models. Although the catch rate parameters can be estimated in the presence of ASCs, the precision of the estimates is poor. Only the effect of the chinook salmon and walleye catch rates remain positive and significant. On the other hand, as expected, there is a large improvement in the fit of the model—the log likelihood is considerably smaller compared with the other conditional logits.

Table 3.3 presents the results of Murdock's second step, which regresses the ASCs on the site-specific characteristics (which in this case are only dummy variables). Compared with the other site choice models, only the effect of *Bayorseaway·Warmwater* retains the same sign and significance level. In modeling the demand for Great Lakes recreational fishing, these results, combined with the weak identification in the first step, reduce the appeal of using ASCs to control for travel cost endogeneity.[11]

Now consider the control function approach. The travel cost equation estimates are presented in Table 3.4. The results indicate a high level of explanatory power in the instrument. The statistical significance of the month-year dummy variables also indicates the role of other shocks (shifts in regional demand due to, e.g., changes in income) during this time to travel costs. An R-squared of 0.709 indicates that the explanatory power of the equation is large. Thus, the control function should be a good estimate of the part of travel cost correlated with the unobservables in the site choice model.

The results of the site choice model with the control function are presented in the final column of Table 3.2. The results are largely similar to the results of the traditional conditional logit, except for the new covariate. In particular, the control function coefficient is statistically significant, demonstrating the presence of travel cost endogeneity. Note that the travel cost coefficient in the model with ASCs is approximately equal to its counterpart in the model with the control function. This is because both models correct for travel cost endogeneity. However, the change in the travel cost coefficient in these two models compared with the traditional conditional logit is less than 5%.

Correcting for travel cost endogeneity increases, in absolute value, the travel cost coefficient. This bias would lead to welfare estimates that are too high. This is confirmed by comparing the welfare impacts of some plausible scenarios, reported in Table 3.5. Each welfare measure in the table is an angler’s per trip willingness to pay to forgo the scenario change,

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[11] Although the results are not reported here, significance levels of the estimated ASCs is also poor. This may explain the lack of identification in the second step.

averaged across all anglers in the sample. Consider the traditional conditional logit model and the model with the control function. The former predicts greater welfare impacts than the latter in every scenario. However, since the parameter estimates between the two models are largely similar, the bias in welfare predictions is small.

Actually, whether the model with ASCs or the control function is preferred will have a greater impact on welfare predictions than the issue of travel cost endogeneity per se. For site loss scenarios, the welfare discrepancy between the two models arises because of differences in the predicted site choice probabilities. For changes in catch rates, the discrepancy arises from differences in the coefficient estimates. For example, the chinook catch rate coefficient is 7.996 is the model with ASCs but 12.018 in the model with the control function. As a result, the average per-trip predicted willingness to pay to forgo a 50% decrease in chinook catch rates is lower in the model with ASCs ($2.14 vs. $3.32). Possibly, some of the catch rate coefficients in the model with ASCs differ from their counterparts in the other models because the ASCs also control any correlation between catch rates and unobserved site-specific characteristics. The loss of identifying power in the model with ASCs—because useful cross-site variation in catch rates is absorbed by the ASCs—makes it difficult to determine whether catch rates are truly endogenous.

Perhaps not too much should be made of the different points estimates. Comparing the welfare effects of the models, most of the predictions are not dramatically different from one another. This is particularly true for the site loss scenarios. Previous research also makes similar welfare predictions. Specifically, in a demand model of Lake Michigan salmon fishing in Wisconsin, Provencher and Bishop (1997) estimate that anglers are willing to pay about $37 ($52 in 2008 dollars) per trip. The models here predict Michigan anglers are, on average, willing to pay \$34-\$37 per trip to fish on Lake Michigan. The smaller dollar figure may be explained by the greater number of substitutable Great Lakes available to Michigan anglers.

3.6 Conclusion

This paper investigates controls for travel cost endogeneity in the context of a site choice model of Great Lakes recreational fishing. It was thought that travel cost could be endogenous if anglers selected their residential location conscious of the travel cost to nearby fishing opportunities. To identify and correct for this source of endogeneity two methods were used: ASCs and a control function. From the results I draw two conclusions. First, ASCs can correct for endogeneity but identification of the observed site-specific parameters in second stage estimation may be poor. Second, a control function can also control for travel cost endogeneity and preserve the traditional approach of recreation demand modeling.

In the Great Lakes sportfishing application, catch rates varied across sites and trips, which permitted estimation of the catch rate effects even with ASCs in the site choice model. Although this is a useful feature, I showed that using ASCs made it difficult to identify the catch rate parameters in the site choice model. Following the two step ASCs bias-correction method developed by Murdock (2006), I also showed that estimation of the parameters in the second step could be poor. These problems can be avoided by using a control function.

Control functions are a useful method to correct for endogeneity in nonlinear models. Until now, control functions have not been employed in recreation demand modeling. Although the control function used here corrected for travel cost endogeneity, control functions can be used to correct for any endogenous covariate. This makes the method flexible. However, an effective control function requires a good instrumental variable, which may not always be available.

In the application to Great Lakes recreational fishing, I find evidence that travel cost endogeneity biases the travel cost coefficient toward zero. The result is robust because both the model with ASCs and the model with the control function find the same level of bias. However, the bias is small so that ignoring travel cost endogeneity may not be a problem in valuing Great Lakes sportfishing. However, this conclusion may not extend to other recreational activities. Von Haefen and Phaneuf (2008), Jakus et al. (2010) and Abidoye (in press), in models of moose hunting, off-highway driving and lake recreation, respectively, all find that the travel cost coefficient is biased toward zero and that the magnitude of the bias is large. Thus, testing and correcting for travel cost endogeneity will continue to be important in recreation demand modeling.

3.7 Tables

Table 3.1. Variable descriptions

Name	Description	Mean	SD
Site Characteristics - Catch Rates			
CR chinook	Chinook (king) salmon per hour catch rate	0.028	0.055
CR coho	Coho salmon per hour catch rate	0.007	0.018
CR lake trout	Lake trout per hour catch rate	0.028	0.108
CR steelhead	Steelhead (rainbow trout) per hour catch rate	0.008	0.018
CR walleye	Walleye per hour catch rate	0.042	0.093
CR yellow perch	Yellow perch per hour catch rate	0.383	0.702
Site Characteristics - Other			
Urban	Site partially urban as defined by U.S. Census 2000	0.600	0.490
Highway	Adjacent to a state highway	0.945	0.229
Bayorseaway	Adjacent to a major Great Lakes bay or seaway	0.434	0.496
Warmwater	Warm water option of the site	0.500	0.500
Angler Characteristics			
Distance	Distance, in miles, to fishing site	125.113	55.112
Crude price	Crude price, in dollars/barrel, at month of trip	91.095	29.304
Travel cost	Mileage and opportunity cost of trip, in dollars	171.193	87.879

Table 3.2. Site choice model parameter estimates

	Traditional CL		CL – Only visited alternatives		CL with ASCs[a]		CL with CF	
Variable	Estimate	t-stat	Estimate	t-stat	Estimate	t-stat	Estimate	t-stat[b]
Travel cost	*-0.0348	-46.10	*-0.0345	-45.70	*-0.0359	-40.97	*-0.0357	-28.92
CR chinook	*11.759	14.10	*10.747	12.70	*7.996	4.09	*12.018	13.43
CR coho	*6.546	3.34	*6.236	3.19	6.802	2.05	*6.289	3.61
CR lake trout	0.327	0.62	-0.520	-0.88	-0.538	-0.45	0.338	0.80
CR steelhead	6.657	2.36	6.610	2.30	-4.835	-1.00	6.229	2.31
CR walleye	*3.927	12.62	*3.533	11.31	*1.915	4.26	*3.954	13.07
CR yellow perch	*0.318	6.79	*0.289	6.21	0.167	1.88	*0.303	7.15
Urban	*0.716	9.36	*0.642	8.47			*0.727	10.10
Highway	*1.801	6.88	*1.621	6.06			*1.799	6.60
Bayorseaway	*-1.091	-9.31	*-1.155	-9.92			*-1.134	-9.08
Bayorseaway Warmwater*	*1.748	15.24	*1.870	16.19			*1.753	14.71
Control Function							*0.011	3.26
Trips		2233		2233		2233		2233
Log likelihood		-4184.530		-4091.505		-3645.443		-4174.000

[a] Alternative specific constant estimates are withheld for brevity.

[b] Derived from standard errors calculated from bootstrapping with 250 replications.

*Significant at the 1 percent level.

Table 3.3. ASCs second-step regression

Variables	Estimate	t-stat
Urban	-0.469	-1.69
Highway	-0.298	-0.67
Bayorseaway	-0.157	-0.47
*Bayorseaway*Warmwater*	*1.260	2.94
Constant	-0.917	-2.11
R^2		0.135

*Significant at the 1 percent level

Table 3.4. Travel cost equation parameter estimates

Variables[a]	Estimate	t-stat
CR chinook	-3.354	-0.97
CR coho	-17.262	-1.73
CR lake trout	-1.542	-1.08
CR steelhead	20.799	2.02
CR walleye	2.573	1.28
CR yellow perch	*-1.482	-5.94
Urban	0.272	0.84
Highway	0.091	0.13
Bayorseaway	*-4.617	-11.19
*Bayorseaway*Warmwater*	0.623	1.13
January08	*-24.801	-9.42
February08	*-20.931	-7.72
March08	*-16.965	-6.10
April08	*-25.238	-9.77
May08	*-50.019	-26.93
June08	*-62.161	-36.43
July08	*-60.685	-36.55
August08	*-40.732	-25.08
September08	*-15.863	-9.82
October08	*13.310	8.05
November08	*37.803	19.87
December08	*58.344	28.41
January09	*62.999	34.75
February09	*65.689	37.47
March09	*40.851	21.25
April09	*41.156	23.01
May09	*39.095	22.53
June09	*20.687	12.13
July09	*22.582	11.75
August09	*14.431	4.57
September09	*35.483	13.78
October09	*67.224	11.12
*Crudeprice*Distance*	*0.014	476.88
Constant	*25.978	14.93
R^2		0.709

[a] The omitted category in the set of month-year dummy variables is all trips taken prior to January, 2008.

*Significant at the 1 percent level

Table 3.5. Welfare change scenarios in average WTP ($) per trip occasion across all trips

Scenario	Traditional CL	CL – Only visited alternatives	CL with ASCs	CL with CF
CR chinook 50% decline	3.34	3.13	2.14	3.32
CR coho 50% decline	0.50	0.48	0.44	0.47
CR walleye 50% decline	7.08	6.50	4.50	6.95
CR yellow perch 50% decline	3.25	3.00	1.62	3.03
Loss of Lake Erie	3.24	3.19	4.66	3.13
Loss of Lake Huron	20.24	21.07	23.84	19.74
Loss of Lake Michigan	37.59	37.48	34.37	37.08
Loss of St. Clair and Detroit Rivers and Lake St. Clair	24.68	24.95	20.71	24.06

REFERENCES

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Abidoye, B.O., J.A. Herriges and J.L. Tobias (in press). Controlling for Observed and Unobserved Site Characteristics in RUM Models of Recreation Demand. American Journal of Agricultural Economics.

Albouy, D. (2009). What are Cities Worth? Land Rents, Local Productivity, and the Value of Amenities. NBER Working Paper No. 14981.

American Automobile Association (AAA). *Your Driving Costs: 2008 Edition*.

American Automobile Association (AAA). *Your Driving Costs: 2009 Edition*.

Berry, S.T. (1994). Estimating Discrete-Choice Models of Product Differentiation. *RAND Journal of Economics* 25: 242-262.

Berry, S., J. Levinson and A. Pakes (1995). Automobile Prices in Market Equilibrium. *Econometrica* 63: 841-890.

Deller, S.C., S.T. Tsung-Hsiu, D.W. Marcouiller and D.B.K. English (2001). The Role of Amenities and Quality of Life In Rural Economic Growth. *American Journal of Agricultural Economics* 83: 352-365.

U.S. Energy Information Administration (EIA). *Michigan Total Gasoline Through Company Outlets Price by All Sellers (Dollars per Gallon)*. Available online at http://tonto.eia.gov/dnav/pet/hist/LeafHandler.ashx?n=PET&s=EMA_EPM0_PTC_SMI_DPG&f=M.

Hausman, J.A., G.K. Leonard and D. McFadden (1995). A Utility-Consistent, Combined Discrete Choice and County Data Model Assessing Recreational Use Losses Due to Natural Resource Damage. *Journal of Public Economics* 56: 1-30.

Jakus, P.M., J.E. Keith, L. Liu and D. Blahna (2010). The Welfare Effects of Restricting Off-Highway Vehicle Access to Public Lands. *Agricultural and Resource Economics Review*, 39: 89-100.

Jones, C.A. and F. Lupi (2000). The Effect of Modeling Substitute Activities on Recreational Benefit Estimates. *Marine Resource Economics* 14: 357-274.

Kikuchi, H (1986). Segmenting Michigan’s Sport Fishing Market: Evaluation of Two Approaches. PhD Dissertation, Department of Agricultural Economics, Michigan State University.

Liu, X., M.E. Lovely and J. Ondrich (2011). The Location Decisions of Foreign Investors in China: Untangling the Effect of Wages Using a Control Function Approach. *The Review of Economics and Statistics* 92:160-166.

Moeltner, K. and R. von Haefen (2011). Microeconometric Strategies for Dealing with Unobservables and Endogenous Variables in Recreation Demand Models. *American Review of Resource Economics* 3: 375-296.

Morey, E. and D.M. Waldman (1998). Measurement Error in Recreation Demand Models: The Joint Estimation of Participation, Site Choice, and Site Characteristics. *Journal of Environmental Economics and Management* 35: 262-276.

Murdock, J. (2006). Handling Unobserved Site Characteristics in Random Utility Models of Recreation Demand. *Journal of Environmental Economics and Management* 51: 1-25.

Parsons, G.R. (1991). A Note on Choice of Residential Location in Travel Cost Demand Models. *Land Economics* 67: 360-364.

Petrin, A. and K. Train (2006). Control Function Corrections for Unobserved Factors in Differentiated Product Models. Paper presented at Society for Economic Dynamics 2006 Meeting.

Petrin, A. and K. Train (2010). A Control Function Approach to Endogeneity in Consumer Choice Models. *Journal of Marketing Research* 47: 3-13.

Phaneuf, D.J., J.C. Carbone and J.A. Herriges (2009). Non-Price Equilibria for Non-Marketed Goods. *Journal of Environmental Economics and Management* 57: 45-64.

Phaneuf, D.J., V.K. Smith, R.B. Palmquist and J.C. Pope (2008). Integrating Property Value and Local Recreation Models to Value Ecosystem Services in Urban Watersheds. *Land Economics* 84: 361-381.

Provencher, B., and R.C. Bishop (1997). An Estimable Dynamic Model of Recreation Behavior with an Application to Great Lakes Angling. *Journal of Environmental Economics and Management* 33: 107-127.

Randall, A. (1994). A Difficulty with the Travel Cost Method. *Land Economics* 70: 88-96.

Ricker-Gilbert, J., T.S. Jayne and E. Chirwa (2011). Subsidies and Crowding Out: A Double-Hurdle Model of Fertilizer Demand in Malawi. *American Journal of Agricultural Economics* 93: 26-42.

Rivers, D. and Q.H. Vuong (1988). Limited Information Estimators and Exogeneity Tests for Simultaneous Probit Models. *Journal of Econometrics* 39: 347-366.

Simoes, J.C. (2009). Recreational angler surveys: Their Role and Importance Nationally and the 2008 Michigan Angler Survey. Master's thesis, Michigan State University.

Smith, R.J. and R.W. Blundell (1986). An Exogeneity Test for a Simultaneous Equation Tobit Model with an Application to Labor Supply. *Econometrica* 54: 679-685.

Train, K., D. McFadden and R. Johnson (2000). Discussion of Morey and Waldman's "Measurement Error in Recreation Demand Models". *Journal of Environmental Economics and Management* 40: 76-81.

Timar, L. and D.J. Phaneuf (2009). Modeling the Human-Induced Spread of an Aquatic Invader: The Case of the Zebra Mussel. *Ecological Economics* 68: 3060-3071.

Von Haefen, R.H. (2003). Incorporating Observed Choice into the Construction of Welfare Measures from Random Utility Models. *Journal of Environmental and Economic Management* 45: 145-165.

Von Haefen, R.H. and D.J. Phaneuf (2008). Identifying Demand Parameters in the Presence of Unobservables: A Combined Revealed and Stated Preference Approach. *Journal of Environmental Economics and Management* 56: 19-32.

Weber, M.A., P. Mozumder and R.P. Berrens (2012). Accounting for Unobserved Time-Varying Quality in Recreation Demand: An Application to a Sonoran Desert Wilderness. *Water Resources Research* 48.

Weithman, A.S. and M.A. Haas (1982). Socioeconomic Value of the Trout Fishers in Lake Taneycomo, Missouri. *Transactions of the American Fisheries Society* 111: 223-230.

Wooldridge, J.M. (2012). Control Function Methods in Econometrics. Unpublished notes, Michigan State University Department of Economics.