Optimal Fisheries Management in the Presence of An Endangered Predator and Harvestable Prey

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Abstract: This paper analyzes optimal fishery management in the presence of an endangered predator that competes with humans for a commercially viable prey. Because traditional predator controls are not possible when the predator is endangered, we focus on harvest effort controls over the prey's habitat as a means to maintain the predator-prey relationship and sustain the economic viability of the fishery. The management model is based on optimizing fishery rents subject to maintaining a growing predator population. We derive optimal management decisions, along a singular path, with and without a predator constraint and demonstrate the need to consider the predator-prey relationship explicitly. In addition, we derive an expression for the shadow value of the endangered predator, along the singular path, in terms of forgone fishery rents. To illustrate these results, we provide an application to the contentious California sea otter-urchin system and the related urchin fishery.

Keywords: predator-prey, singular control, fixed endpoint, endangered species

I. INTRODUCTION

This paper analyzes optimal fishery management in the presence of an endangered predator that competes with humans for a commercially viable prey. In the past, natural predators were implicit in the fishery model. We, however, explicitly model the predator-prey relationship given that endangered predators can be found in many fisheries and the importance of allowing these predator populations to expand in conjunction with maximizing rents in the fishery. Because traditional predator controls are not possible when the predator is endangered, we focus on harvest effort controls over the prey's habitat as a means to maintain the predator-prey relationship and sustain the economic viability of the fishery.

The analysis consists of a predator-prey model that allows for density-dependent growth. The management model is based on optimizing fishery rents subject to maintaining an expanding predator population. We derive optimal management decisions, along a singular path, with and without the predator constraint and demonstrate the need to consider the predator-prey relationship explicitly. In addition, we derive an expression for the shadow value of the endangered predator, along the singular path, in terms of forgone fishery rents.

To illustrate these results, we analyze the contentious California sea otter-urchin system and the related urchin fishery. We simulate the growth of the otter population with explicit consideration of the predator-prey relationship. Next, the net present value of alternative harvest policies are compared with the net present value for the singular path approach to the steady state predator-prey system. In addition, the shadow value (cost) of the allowing the predator population to grow is calculated for each scenario to illustrate the cost to the fishery for protecting the endangered predator.

We extend the literature in two directions. First, we explicitly consider the role of the predator-prey relationship in optimal fisheries management when one of the species is endangered. Although there is prior research on predator-prey models (Goh, Leitman and Vincent; Tu and Wilman; Wilen and Brown; Ragozin and Brown; Solow), our analysis compares optimal fishery management with and without a constraint on an endangered predator population. Several authors have noted how the failure of a number of fisheries can be attributed to a lack of predator-prey relationships in fisheries management (Ragozin and Brown; Tsoa; Wilen and Brown; Holmlund and Hammer). Furthermore, several authors derive expressions for the shadow value of a non-market prey that is consumed by a harvested predator (Tu and Wilman; Wilen and Brown; Ragozin and Brown). This indirect valuation method is similar to the approach we take in deriving the shadow cost of protecting the endangered predator indirectly by controlling the harvest of their prey.

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Second, by incorporating recovery targets for a non-harvested predator, our approach is comparable to a multiple-criteria decision making (MCDM) model in a fisheries management framework. First introduced by Cohon and Marks, MCDM modeling is a recent phenomenon in fisheries management (see Sylvia and Cai; and Mardle and Pascoe for surveys of the literature). The many objectives of fishery management include resource conservation, maximizing rents, and maintaining employment opportunities. Mardle and Pascoe stress the importance of the MCDM approach by pointing out that considerable effort has been expended in developing bioeconomic models, yet these models are not adopted by fishery managers because they do not incorporate multiple objectives. Our approach is a natural extension for the fishery management model given that the manager wishes to increase the endangered predator population while maximizing fishery rents. We adopt a constrained model in which rents are maximized subject to an inequality constraint on the predator population. We are able to directly evaluate the trade-off between an endangered predator and the fishery rents from harvesting the prey. The trade-off between maintaining the predator population and maximizing rents in the fishery allows us to value the endangered predator based upon the forgone rents necessary to maintain the predator population, which enters the problem through the multiplier on the predator constraint.

In the next section we provide background and context for the urchin-otter issue. We develop the predator-prey model in Section III. We incorporate spatial and density dependent components into a Lotka-Volterra predator prey model and characterize the optimal management decision with and without explicit consideration of the endangered predator. Section IV details the application to California Sea Otter-Urchin Fishery, describes the data used in calibrating the simulation model, and presents the results from simulating the predator-prey model based on the California problem. Section V concludes and discusses future research.

II. BACKGROUND AND MOTIVATION

Concern for marine mammals has long been a part of the environmental movement. Early actions taken to preserve marine mammals focused on the unsustainable harvest of animals such as the International Whaling Commission in 1937 (Gulland). The early 1970s saw international protests of all commercial whaling activity, and most countries abandoned whaling altogether. The Stockholm Conference on the Human Environment in 1972 adopted a resolution that put a ten-year moratorium on commercial whaling, and the United States adopted the Marine Mammal Protection Act in 1972. In the 1980s, many environmentalists focused on incidental deaths of marine mammals due to commercial fishing gear that targets other species. Bycatch of dolphins in the tuna fishery, for example, drew considerable attention. This led to gear restrictions and a 1990 international trade dispute over dolphin-safe tuna (Musgrave and Garland).

More recently, endangered species conflicts have centered around threats to a specie's habitat. Perhaps nowhere is this issue more poignant than in the case of the Northern spotted owl in the Pacific Northwest. Here the interests of environmentalists and renewable resource harvesters come into conflict because the threatened species relies on the harvestable resource. Similar issues have arisen in ocean fisheries. For instance, threatened Steller sea lions in Alaska arguably compete for food sources with the commercial groundfish fisheries that target walleye pollock, Pacific cod, Atka mackerel, and other species (Pascual and Adkison; Fritz, Ferrero, and Berg; Merrick and Loughlin). The potential for a similar conflict of interest exists between sea otter preservation and shellfish fisheries in California, since the favorite foods of sea otters are high value shellfish that are harvested commercially in much of California.

The Southern sea otter was once abundant in California's coastal waters but was hunted to near extinction in California by the early twentieth century. A current estimate suggests that more than 15,000 otters once lived along the California coast (CDFG). Russian traders with Aleutian crews began intensive harvesting of otters in 1807 (Ogden). Large scale hunting began to decline by 1830. By 1850, commercial hunting had all but ceased, and upwards of 200,0000 pelts had been taken (Macdonald). Otters were protected by the California legislature in 1913. A group of otters escaped extinction, apparently migrated northward, and returned to California in 1938. Since then, the Southern sea otter has made a considerable comeback. Current estimates of the otter population in California are between 2,300 and 2,500 otters (USGS, CDGG). However, there are still far fewer animals today than there were before large-scale commercial hunting began in the early nineteenth century.

Although the conflict between sea otters and shellfish fishermen is not entirely new, recent trends have reinforced the conflict. The University of California sponsored a conference in 1981 to address potential shellfish fishery and sea otter conflicts. Since then, the commercial red sea urchin fishery has grown to become one of California's largest revenue producing fisheries. The fishery, which exports millions of pounds of red sea urchin each year to Japan, includes both northern and southern California. During the same time, sea otters have continued their comeback and have been spotted outside their protected zone in central California. Such sitings have upset sea urchin fishermen (Pleschner). Numerous studies have documented sea otters foraging on sea urchins (Kvitek, Shull, Canestro, Bowlby, and

Troutman; Estes; Stewart, Foster, Carson, and Breen; Doroff and DeGange). Some have even found that when sea otters enter a new territory, they forage heavily on red sea urchins until they have harvested down the stocks and then move on to other prey (Ostfeld; Hardy, Wendell, and DeMartini). This suggests that otters have lexicographic preferences, with sea urchins being at the top of the list. Table 1 presents some stylized calculations that highlight potential conflict between otter preservation and the urchin fishery. At current population levels, otters consume about 7.3 million pounds of food each year in California. The total annual harvest of sea urchins is only approximately 12 million pounds. The current overlap between the otter range and the urchin fishery is minimal, but clearly a major expansion of otter range would affect the urchin industry signicantly. The potential conflict is fueled further by concerns about otter vulnerability to oil spills. Because of their limited geographic range in California, the otters could be catastrophically affected by a major oil spill (Ralls, Demaster, and Estes; Brody, Ralls, and Siniff). This vulnerability creates pressure to allow the sea otter range to expand and even to force such an expansion through relocation experiments.

III. THE ENDANGERED PREDATOR-HARVESTABLE PREY MODEL

The predator-prey system is not simply the standard Lotka-Volterra system because sea otters are not the only limit on sea urchin abundance, and similarly, sea urchins are not the only source of food for sea otters. Thus, we employ logistic growth for each species with the addition of species interaction terms. Denoting urchins by x, otters by y, and harvest as h, the following differential equations summarize the predator-prey system dynamics:

$$\dot{x} = (a_1 - a_2 x)x - a_3 y - h \qquad (1)$$

$$\dot{y} = (b_1 + b_2 x - b_3 y)y \qquad (2)$$

where a's and b's are biological growth and interaction parameters. Since the otters (predators) are not harvested (and we assume that for political reasons they cannot be harvested) and the urchins (prey) are harvested, the goal of a policy is to find a harvest path that maximizes fishery rents subject to the system dynamics. Denoting price as p, cost as c(x), time as t, and the discount rate as r, this problem can be characterized in an optimal control framework as follows:

$$\underset{h}{\text{Max}} \int_0^\infty e^{-rt}[p - c(x)]h \, dt \qquad (3)$$

subject to (1), (2), and initial conditions on the urchin and otter populations (denoted as x_0 and y_0). Denoting the co-state variable of x as λ and the co-state for y as μ, the Current Value Hamiltonian is:

$$\tilde{H} = [p - c(x)]h + \lambda[(a_1 - a_2 x)x - a_3 y - h] + \mu[(b_1 + b_2 x - b_3 y)y] \qquad (4)$$

By applying Pontryagin's Maximum Principle, the following co-state equations combined with (1) and (2) are first order necessary conditions:

$$\dot{\lambda} = r\lambda + c'(x)h - \lambda(a_1 - 2a_2 x) - \mu b_2 y \qquad (5)$$

$$\dot{\mu} = r\mu + -\mu(b_1 + b_2 x - 2b_3 y) + \lambda a_3 \qquad (6)$$

Because the problem is linear in the control, the usual $\tilde{H}_h = 0$ condition is replaced by a bang-bang control that is characterized by a switching function. As such, $\lambda = p - c$ in the steady state. For simplicity let us assume that $c(x) = c$ and hence $c'(x) = 0$. Therefore we can write the steady state for this system as

$$\dot{x} = (a_1 - a_2 x)x - a_3 y - h = 0 \qquad (1')$$

$$\dot{y} = (b_1 + b_2 x - b_3 y)y = 0 \qquad (2')$$

$$\dot{\lambda} = r\lambda - \lambda(a_1 - 2a_2 x) - \mu b_2 y = 0 \qquad (5')$$

$$\dot{\mu} = r\mu + -\mu(b_1 + b_2 x - 2b_3 y) + \lambda a_3 = 0 \qquad (6')$$

From the optimality conditions we know that along the singular path $\lambda = p - c$. Substituting for λ in (5') and (6') and solving for μ yields

$$\mu = \frac{p - c}{b_2 y}[r - a_1 + 2a_2 x] \qquad (7)$$

and

$$\mu = -a_3(p - c)[r - b_1 - b_2 x + 2b_3 y]^{-1} \qquad (8)$$

From (2') we obtain

$$y = (1/b_3)(b_2 x + b_1) \qquad (9)$$

Now, substituting (9) into (7) and (8) and equating yields

$$0 = [r^2 - ra_1 + rb_1 - a_1 b_1 + (\frac{1}{b_3})(a_3 b_2 b_1)] + [rb_2 - a_1 b_2 + 2ra_2 + 2a_2 b_1 + (\frac{1}{b_3})(a_3 b_2^2)]x + 2a_2 b_2 x^2 \qquad (10)$$

The positive root for this quadratic defines the steady state value for x in terms of the parameters of the model. With the steady state solution for x we can then define the steady state levels for y, h, λ and μ such that

$$y^{ss} = (1/b_3)(b_2 x^{ss} + b_1) \qquad (11)$$

$$h^{ss} = (a_1 - a_2 x^{ss})x^{ss} - a_3 y^{ss} \qquad (12)$$

$$\lambda^{ss} = p - c \qquad (13)$$

and

$$\mu^{ss} = -a_3(p - c)[r - b_1 - b_2 x^{ss} + 2b_3 y^{ss}]^{-1} \qquad (14)$$

The non-harvested steady state for this system is easily derived and thus we only present the steady state expression for x, which is defined by the following quadratic

$$0 = a_3 b_1 + [a_3 b_2 - a_1 b_3]x + a_2 b_3 x^2 \qquad (15)$$

Derivation of the Approach Path

The approach we employ can be found in Ragozin and Brown as well as Wilen and Brown. This approach was originally presented in Spence and Starrett. To obtain the approach path to the harvested steady state we construct $\sigma = \lambda - (p - c)$, which defines the switching condition on harvest. When we are on the singular path $\sigma = 0$ and $\dot{\sigma} = 0$, where

$$\dot{\sigma} = (p - c)(r - a_1 + 2a_2 x) - \mu b_2 y = 0 \quad (16)$$

Rearranging terms we obtain

$$\mu b_2 y = (p - c)(r - a_1 + 2a_2 x) \quad (17)$$

now multiply (6') by $b_2 y$ and substituting (17) and its time derivative into this new expression for (6') we obtain

$$\dot{x} = (1/2a_2)[r^2 - ra_1 + rb_3 y - a_1 b_3 y + 2ra_2 x + 2a_2 b_3 xy + a_3 b_2 y] \quad (18)$$

To find the singular value for x, the level of the prey at which the singular control is employed we set (18) equal to zero and find

$$x^{sv} = \frac{[-r^2 + ra_1 - a_3 b_2 y + a_1 b_3 y - rb_3 y]}{2ra_2 + 2a_2 b_3 y} \quad (19)$$

and

$$h^{sv} = (a_1 - a_2 x^{sv})x^{sv} - a_3 y \quad (20)$$

and now we can define the policy control

$$h = \begin{cases} h_{max} & x > x^{sv} \\ h = h^{sv} & x = x^{sv} \\ h = 0 & x < x^{sv} \end{cases} \quad (21)$$

where h_{max} is the maximum feasible harvest for the industry. When x exceeds its singular value, we follow a most rapid approach path by harvesting the maximum possible. When x is below its singular value, we harvest none.

The set-up and solution to this problem are essentially the same as those in Ragozin and Brown. The key differences are the roles that the predator and prey play in the economic system and the interpretation of the shadow values of the two species. In Ragozin and Brown, the predator is harvested and the prey is not. The non-harvested prey takes on economic value through its contribution to predator biomass, and the shadow value of the prey in the predator-prey optimal control problem reflects this value.

In our setting, the prey are harvested and the predators are not. The shadow value of the predator reflects lost fishery rents due to predator-prey interactions. Of course, sea otters (the predator) have considerable nonmarket values as indicated by the political involvement of such groups as Friends of the Sea Otter. We do not attempt to estimate these nonmarket values, but we note that the above control problem provides a framework that can estimate the minimum costs (in terms of fishery rents) of otter preservation. In this sense, we derive a bound for nonmarket values below which restrictions on the fishery would not be socially optimal.

It is difficult to compare the predator-prey model above and a fishery management model that ignores predator-prey dynamics. On the surface, one might simply replace the Hamiltonian in (4) with one that sets a_3=0 and ignores (2), and then solve for the optimal control. This Hamiltonian would be:

$$\tilde{\tilde{H}} = [p - c(x)]h + \lambda[(a_1 - a_2 x)x - h] \quad (22)$$

One could proceed to compare the stock evolution in this system to the one described above, but such an exercise ignores a more fundamental problem with this type of comparison. The parameters used in (22) likely would not be the same as the ones used in (4). If one modeled the fishery without predator-prey dynamics, it is difficult to believe that the parameter estimates used would have come from a statistical model of the predator-prey system. Thus, the parameters would be biased, and one possible consequence is the understatement of the intrinsic growth rate. So, it is difficult to gain qualitative insight on a fishery manager's mistake due to ignoring the predator-prey dynamics.

Recovery Constraints on the Predator Population

Suppose that the regulator determines a recovered level for the sea otter population and sets a time horizon over which this recovery must be achieved. This problem complicates the singular control problem by including a fixed endpoint at time τ. The functional to maximize is divided into two pieces and the otter population constraint must hold throughout the post-recovery phase. Denoting the recovered otter population as y_R, this can be characterized as the following combined with (1) and (2):

$$\underset{h}{\text{Max}} \int_0^{\tau} e^{-rt}[p - c(x)]h\,dt + \int_{\tau}^{\infty} e^{-rt}[p - c(x)]h\,dt \quad (23)$$

$$\text{subject to} \quad y_t \geq y_R \text{ for } t \geq \tau$$

Our conjecture is that the solution to this problem involves the same singular path as the one we derived above with a departure from the singular path that allows the otter population to grow to y_R by τ. The reasoning is simple. The singular path coupled with a most rapid approach path is optimal in the fishery maximization problem. So, with the added constraint, the singular path with a most rapid approach is also optimal. The departure from the singular path is a most rapid approach to the constraint.

IV. SIMULATIONS AND WELFARE ANALYSIS

Table 2 presents parameter values used in the simulations and both harvested and non-harvested steady states for urchin and otter populations. These results are meant to be illustrative. They have not been calibrated to match the population dynamics of either species. However, we have chosen the parameters to be reasonably consistent with historical urchin and otter populations that may have existed in pre-harvested states. Moreover, the parameter a_3 reflects possible urchin consumption in one year by one adult otter that consumes urchin as the primary ingredient in its diet (1.6 tons = 3,200 pounds). All simulations begin with initial populations of 2,000 otters and 180,000 tons of sea urchin. Figure 1 depicts optimal trajectories for the predator and prey stocks under fishery rent maximization with a maximum annual harvest of 50,000 tons. The urchin population starts above the steady state, so it is harvested at the maximum rate until the singular path is reached. With such a high h_{max}, the urchin stock reaches this singular path within 2.5 years. On the other hand, the otter stock grows slowly towards its steady state and reaches it only after 70 years. Note that both stocks approach the harvested steady states recorded in Table 2.

Figure 2 depicts the same situation but with a lower maximum harvest. Here h_{max}=19,000, which only slightly exceeds the maximum of the singular path harvest. As a result, the urchin population takes longer to reach the singular path (15 years as compared to 2.5 years). This, in turn, allows the otter population to grow larger and overshoot its steady state before it turns down and approaches the steady state from above. This simple example illustrates the difficulty of managing a predator-prey system with a single control.

In Figure 3, we simulate the recovery policy and compare it to fishery rent maximization. Under the recovery policy, the urchin population follows the singular path until harvesting must stop, thereby growing the urchin population and allowing the otter population to reach 5,500 by τ. This harvest rule is simply implementing our conjecture that the optimal way to reach the fixed endpoint on otters is to follow a most rapid approach policy. We approach the singular path as quickly as possible, as in the rent maximization without an otter constraint. Then we stay on the singular path as long as possible, getting off of the path with the minimum possible amount of time left before τ. Because of the predator-prey dynamics, the growth of the urchin population up to τ causes the otter population to overshoot 5,000 after τ before it turns back towards the target level. After the target otter population is reached, the urchin population returns to a new singular path. This path is above the old one in order to sustain a larger otter population.

The overshooting of the otter population led us to question our conjecture about the optimal way to achieve the target otter population. We tried an alternative policy that uses a most rapid approach to the post-target singular path, stays on this singular path as long as possible, and gets off the path just in time to achieve the target otter population. The present value cost of this policy, in fact, was lower than that of the policy depicted in Figure 3. We depict this improved policy along with the first recovery policy in Figure 4. Though it requires further research to determine whether this improved policy is the optimal one, we believe that its improvement stems from a smaller overshooting of the target otter population.

Figure 5 plots the trajectory of the current value shadow price of otters, μ. This shadow value reflects the present value of the stream of lost fishery rents due to an additional otter. Table 3 compares rent maximization and the improved recovery policy under four different scenarios. Because we are not confident that our parameters have been calibrated adequately, we avoid dollar figures and report the results in percentages only. Nevertheless, Table 3 suggests that short recovery horizons would lead to substantial losses for the fishery. In contrast, an increase in the otter population to 5,500 (just 735 more than the harvested steady state) over a long time horizon creates a small impact on fishery rents.

V. CONCLUSIONS AND FUTURE RESEARCH

Although these results are preliminary, we have reached several conclusions. First, depending on the maximum possible effort in the fishery, the stock of a non-harvested predator can grow beyond its bioeconomic steady state value. This overshooting is due to a relatively slow approach to the singular path harvest of the prey. Second, the shadow cost of a non-harvested predator can be derived in an optimal control setting in which the planner maximizes fishery rents subject to predator-prey dynamics. While this result is analogous to deriving the nonmarket value of non-harvested prey in Ragozin and Brown and Wilen and Brown, it is a different interpretation, using predator-prey dynamics to calculate how predator expansion impinges on fishery rents. Third, recovery policies can be characterized by splitting the control problem into two pieces: a fixed endpoint problem with a terminal condition at τ and an infinite horizon problem that begins at τ and includes an inequality constraint on the predator.

We did not discuss infeasible policies for which the recovery target cannot be met within the specified time. But, a properly calibrated bioeconomic framework would be useful for determining what recovery paths are feasible. Ignoring the commercial harvest of sea urchins, for instance, could lead to overestimated otter growth on new territory in the otter range. We also did not address recovery policies that do not bind. Again, a well calibrated model would identify recovery targets for which the regulator need not stray from fishery rent maximization.

There are many questions left unanswered by this work. We conjectured that the optimal recovery policy involved the original singular path with a most rapid approach to the predator constraint, but we showed that an alternative harvest policy other generates greater fishery rents. Though we believe that we have cast this problem correctly as fixed end-point problem, we have not yet found an analytical solution to it. Finding this solution may lead to more interesting insights in both the ability to protect endangered predators and in deriving the shadow value on the protected specie. An important next step for empirical analysis of the urchin-otter system is to calibrate the model to the population dynamics of the two species. This will allow us to obtain reliable dollar costs to the fishery for different otter recovery plans and range expansions.

Finally, we did not address the spatial characteristics of the predator-prey system. If the system is panmictic this does not create a bias in the results. A population is defined as panmictic when there are high rates of migration and larval mixing (Fogarty et al.). However, if the population is dispersed with limited interaction between the distant patches (i.e., a meta-population) then we must carefully consider the spatial aspects of harvesting over multiple patches. This type of extension follows the single species fishery models by incorporating spatial aspects in order to study the general problem of dispersal mechanisms (Sanchirico and Wilen; Low et al.) and to compare meta-population and panmictic populations (Wilson et al.; and Horan and Shortle). Only management of the aggregate population is relevant when the population is panmictic, since any depletion from a single area will quickly be replenished (Wilson et al.). In the case of the urchin-otter system, the sea otter range currently only overlaps with the commercial fishery at its edges. Further expansion of otters into commercial fishing areas is a natural situation to analyze with a spatially explicit model.

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Figure 1

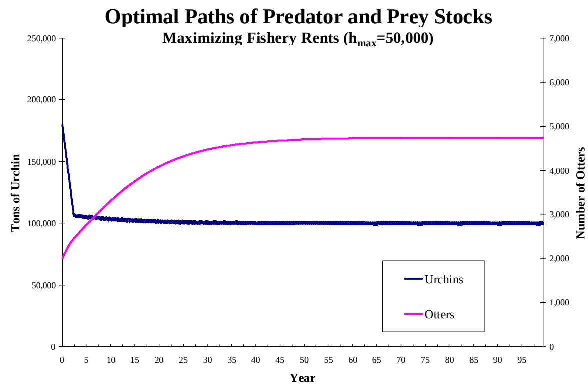


Figure 2

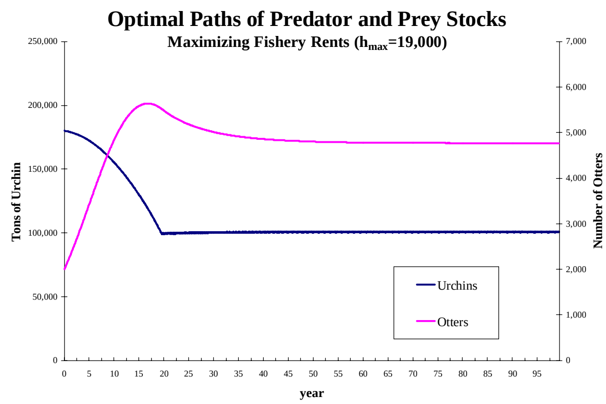


Figure 3

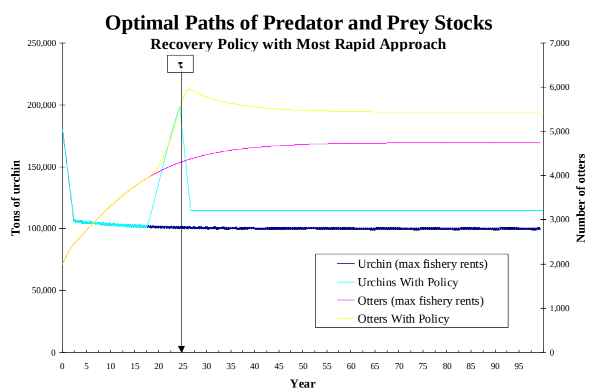
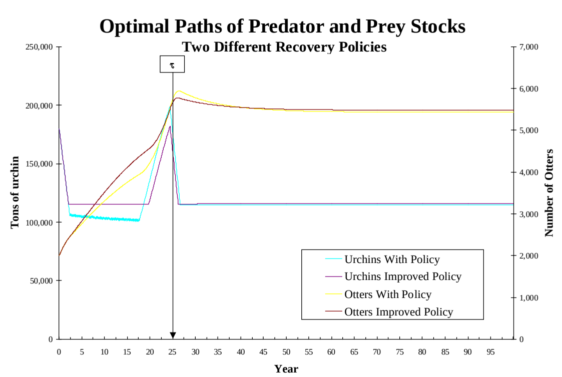


Figure 4



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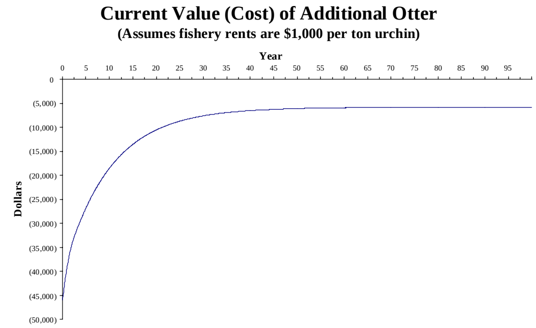

Table 1
Potential For Sea Otter and Sea Urchin Conflict

Otters in California	2,000
Average weight (pounds)	40
Food Eaten Per Day Per Otter (pounds)	10
Total Food Consumed Each Day (pounds)	20,000
Total Food Consumed Each Year (pounds)	7,300,000
Annual Harvest of California Sea Urchin (pounds)	12,000,000
Based on 1997-99 average	

Table 2
Parameters and Steady States for Simulations

a1	0.3
a2	0.000001
a3	1.6
b1	5E-07
b2	9E-07
b3	0.000019
r	0.05

	Unharvested Steady State	Harvested Steady State
Urchins (Tons)	224,210	100,588
Otters (Number)	10,621	4,765

Table 3
Comparision of Present Value of Policy

Percent Difference Between Maximizing Fishery Rents and Recovery Policy
Initial Otter Population = 2,000 and Initial Urchin Population = 180,000 tons

τ (years)	Minimum Otter Population 5,500	6,500
10	18.76%	33.28%
24.4	2.42%	8.11%