GRANDMA OR THE WOLF?

A REAL OPTIONS FRAMEWORK FOR MANAGING

HUMAN-WILDLIFE CONFLICTS

Baishali Bakshi[1] and Jean-Daniel M. Saphores[2]

Revised 05/24/04

Abstract

As wildlife habitats shrink, some species are pushed into extinction, while conflicts with others increase and cause significant economic damages. This paper proposes a simple real options framework to analyze wildlife management policies that account for ecological uncertainty and the risk of extinction. Our application to wolves provides an economic justification for their reintroduction and highlights the importance of existence value. Our sensitivity analysis shows that the optimal management policy depends on the growth rate, the volatility, and the minimum viable density of the wolf population, but little on damages, existence value, and the discount rate for the parameters considered.

Keywords: endangered species; reintroduction; uncertainty; irreversibility; real options.

JEL Codes: D81, Q20, Q28

---

[1] Doctoral Candidate, Economics Department, University of California, Irvine, CA 92697. E-mail: bbakshi@uci.edu

[2] Corresponding author. Assistant Professor, School of Social Ecology and Economics Department, University of California, Irvine, CA 92697. Phone: (949) 824 7334. Fax: (949) 824 8566. E-mail: saphores@uci.edu.

"In those days we had never heard of passing up a chance to kill a wolf…Since I have lived to see state after state extirpate its wolves… I have seen every edible bush and seedling browned, first to anemic desuetude, and then to death. … I now suspect that just as a deer herd lives in mortal fear of its wolves, so does a mountain live in mortal fear of its deer…. Perhaps this is the hidden meaning in the howl of the wolf, long known among mountains, but seldom perceived among men." Aldo Leopold [23].

1. Introduction

Conflicts between wildlife and humans, fueled by population growth and the conversion of natural areas, are creating increasingly tough challenges for policy makers. All over the U.S., pet owners are concerned by coyotes visiting urban or suburban areas [36] [39] [32]; ranchers and hunters are complaining about wolves in Western [16] and Midwestern states [4]; and Californians are worried by mountain lion sightings [7] and the aggressive behavior of black bears with a taste for human cuisine [3]. At the same time, the eradication of predators is to blame for a surge in the deer population that dramatically increased the number of road collisions [13], and for an explosion in the goose population that led to significant pollution from goose droppings [42]. The situation is worse in many developing countries. In India, home to half of the world's remaining wild tigers, poor villagers are killed every year as they battle tigers for the use of protected areas [22]. In Zimbabwe, the government is erecting electric fences around villages to reduce the threat to human life and crops from elephants and other wild animals [51].

The task of wildlife managers is particularly difficult for the long-term management of potentially dangerous, endangered predators such as wolves, grizzly bears, or tigers, at a time where the preservation of these species appears ever more essential for the health of entire

ecosystems [41]. Yet the environmental economics literature does not offer simple management rules that account for biological uncertainty, the risk of extinction and the possibility of reintroduction (when available). This paper proposes a continuous time real options framework to derive such rules. Although our approach may be applied to any species that enters in conflict with humans, we focus here on large carnivores. Our application to wolves highlights the importance of existence value for the preservation of endangered species (a point also made in [1]), and it provides an economic justification for the reintroduction of wolves. Our sensitivity analysis shows that the optimal management policy depends on the growth rate, the volatility, and the minimum viable density of the wolf population, but not much on damages (for the parameters considered), existence value (provided it remains large), and the discount rate ($\geq 2\%$).

As emphasized by a number of conservation biologists, large carnivores are especially important among endangered predators for several reasons [2] [43]. First, they are good indicators of ecosystem health because the areas where they still live typically retain most native species and have a high potential for ecological integrity [30]. Second, they often provide a "protective umbrella" to many other wild species.[1] Indeed, recent evidence supports the view that many carnivorous predators have a strong positive influence on biodiversity by limiting the overexploitation of vegetation by herbivores, the so-called "top-down" theory [27] [41]. Third, as large carnivores often require large expanses of land to maintain viable populations, focusing on their preservation is a useful strategy to conserve large-scale ecosystems for the long term.[2] Finally, large predators have become a symbol for the ongoing public debate on conservation [19] because big animals tend to inspire people in ways that microorganisms probably never will.

Although the Endangered Species Act explicitly bans the use of benefit cost analysis, the tools of environmental economics are useful for informing decision makers about the tradeoffs

entailed in the management of recovering endangered species. Two features of this problem are particularly noteworthy: uncertainty and irreversibility. First, there is uncertainty in the benefits and costs associated with different wildlife population levels, and they tend to vary randomly over time: unfavorable conditions may cause a predator to prey more on livestock, or public interest for a species may increase following positive press coverage. Second, the ecology of a number of predators is still insufficiently understood [8]. Irreversibility stems from the risk of extinction, although it may be overcome with reintroduction. The importance of uncertainty and irreversibility is now well-known in the economic literature on investment [12] so these features should be incorporated in wildlife management models. This motivates our approach, which extends the harvesting framework proposed by Saphores [38] to include a flow of amenities between controls and the loss of existence value from extinction [21].

We analyze the problem of a wildlife manager who intends to maximize the stream of net benefits from a species by controlling at what density culling is necessary, and the size of culling. For a fixed territory, the wildlife density follows a diffusion process (see [18]) due to fluctuations in natural conditions that affect reproduction, migration, food availability, and losses from predators. In a first class of models, if the wildlife population thrives, its density increases to the point where the flow of net benefits is negative so culling becomes necessary. If, on the other hand the wildlife population becomes extinct, there is a loss of existence value. In a second class of models, we allow for reintroduction as soon as the wildlife density reaches a lower threshold K_0. Comparing both classes of models is useful for assessing the value of the option to reintroduce. We then apply our framework to the management of wolves, probably the most controversial and charismatic predators around [33].

Two recent papers study the inter-temporal trade-offs associated with the management of

valuable wildlife populations that cause damages at high densities. Using optimal control, Rondeau [34] analyzes the transitory dynamics following the reintroduction of a wildlife species. He investigates the timing of costly control measures and conditions under which reintroduction is not desirable. An application to deer management illustrates the losses resulting from delays in implementing the optimal control policy, and provides a very informative discussion of deer management issues. However, Rondeau's analysis deals neither with uncertainty nor with the risk of population extinction. In a follow-up paper, Conrad and Rondeau [35] build a discrete time dynamic model to study deer management. They show that pulsing controls are more efficient than steady state policies over a wide range of conditions, including biological uncertainty, and discuss the acceptability of various deer control measures. They do not, however, discuss the risk of extinction and the resulting loss in existence value.

In modeling wildlife population dynamics, it would clearly be useful to account for mutualistic and predator-prey relationships; a logical and promising approach would be to build a general equilibrium ecosystem as in [14]. However, our biological understanding of many species is still too poor at this point for this type of models. This is the case for many large carnivore species [48] because their secretive behavior, low densities, and the long intervals between successive generations make their study expensive [8]. While the approach we propose in this paper is not ideal, it provides resource managers with simple decision rules until biological complexities are better understood.

The paper is organized as follows. In the next section, we provide a brief overview of the management policies applied to wolves in the United States. Section 3 presents a general framework for managing species that enter in conflict with humans. Section 4 applies this framework to the management of wolves. Section 5 summarizes our conclusions.

2. The case of wolves

Wolves typically generate strong emotions; they are emblematic of our difficult relationship with nature [19]. Their qualities make them tough competitors: they are highly intelligent with an acute hearing and an exceptional sense of smell [9]. Like humans, they adapt remarkably well to very diverse climatic conditions: they are distributed between 20° N. latitude (mid-Mexico and India) and the North Pole [28]. North America has two main species of wolves: gray wolves (Canis lupus, or timber wolves), used to range from Canada to Mexico, while red wolves (Canis rufus) occupied mostly the Southeastern United States. Apart from the color of their pelage, red wolves differ from gray wolves by their slightly smaller size and their more slender head.

Wolves have been the object of both fascination and age-old fears.[3] According to the legend, a she-wolf fed Romulus, the founder of ancient Rome, and his brother Remus. In addition, most Native Americans revered gray wolves and tried to emulate their remarkable hunting skills [49]. On the other hand, most American settlers inherited the traditional European antipathy towards wolves (Remember the big bad wolf in Little Red Riding Hood or in Peter and the Wolf).[4] This attitude is partly based on religious beliefs that give people dominion over the Earth's creatures, and hold that the wilderness is godless and should be conquered [19]. It is therefore not surprising that the American colonies' first wildlife act targeted wolves with the creation in 1630 by the Plymouth Colony pilgrims of a one-cent bounty [26]. As settlers moved west, they decimated animal populations traditionally preyed upon by wolves (such as deer, elk, moose, or bison [9]), so wolves turned increasingly to sheep and cattle. Wolves became "an object of pathological hatred" [25] and ranchers created bounty programs to eliminate them, along with other predators. Eradication efforts were boosted by technological improvements in

traps, firearms, and poisons; they became systematic when the government created predator control programs at the beginning of the 20th century [8]. Some of these programs continued until 1965 [44]. By the middle of the 20th century, wolves had been wiped out of the continental United States, except for several hundred gray wolves in Minnesota, an isolated population on Michigan's Isle Royale, and a few red wolves in Louisiana and Texas. Wolves, however, continued to thrive in Canada and were not on the brink of extinction in Alaska.

In March 1967, following the Endangered Species Act (ESA) of 1966, wolves were listed as endangered in parts of the Southwest, and as threatened in the rest of the Southwest and in most of the Eastern United States. This first ESA did not protect species, however; it just called on federal agencies to pay attention to them and to foster their recovery. Wolves became effectively protected in August 1974, after the passage of the 1973 ESA [28].

In spite of illegal takings early on, the wolf population in the lower 48 states began to grow after 1974. Gray wolves from Minnesota migrated to Wisconsin, Michigan [29], and the Dakotas [24]. Canadian wolves spread into Montana, Washington State and Idaho [29]. In 1995 and 1996, 31 gray wolves from Canada were reintroduced in Yellowstone and 35 in Idaho [44]. By the end of 2002, there were approximately 664 wolves in Northern Montana, Central Idaho, and in the Yellowstone area, 295 in Michigan (including 17 on Isle Royale), 2,445 in Minnesota, and 323 in Wisconsin [44]. An additional 21 "Mexican" gray wolves lived in western Texas, central Arizona, and northern Mexico. Following this success, gray wolves were reclassified as "threatened" in the Rocky Mountain region; wolves in Yellowstone and Central Idaho were designated as "non-essential, experimental" populations [45].

The reintroduction of the red wolf was also successful: approximately 100 red wolves now live in the wild in North Carolina and another 156 are kept for breeding to insure genetic

diversity [45]. They are still endangered, except in parts of North Carolina and Tennessee [46].

The persecution of wolves was mostly based on misconceptions. Little scientific information on wolves was available until the late 1960s [28]; it is now clear that they play a very important biological role. First, they regulate the populations of large hoofed animals (such as bison or elk), removing the weakest individuals from their populations [45]. This enhances plant diversity [1], and it makes biological room for smaller herbivores (e.g., beavers and small rodents). Wolves also displace some coyotes, allowing smaller predators such as foxes to thrive [10]. In addition, they indirectly support a number of other species such as grizzly bears, foxes, wolverines, vultures and ravens that feed on the carcasses of preys killed by wolves [44]. Many insect species also benefit: nearly 450 beetle species use these carcasses in Yellowstone [15].

In spite of their much-improved reputation [20], the reintroduction of wolves has generated a lot of opposition. Contrary to what the title of this paper suggests, wolf attacks on humans are rare in North America, and they took place in areas where wolves were attracted to garbage or were hand-fed [44]. Western ranchers and pet owners were the most affected: between 1987 and 2002, wolves killed approximately 600 sheep, 200 heads of cattle, and 50 pet dogs [33].[5] Some elk hunters also blame wolves for a drop in the elk herd north of Yellowstone, although biologists point instead to severe droughts and other predators [33].

To calm public anger and limit further conflicts, lethal control has been exercised. Although this angers some conservationists [40], it still appears the most effective way of curbing wolf damage [28], so culling is the control we consider in our models. Indeed, traditional husbandry techniques relying on shepherds with guard dogs or electric fences are uneconomical; translocation does not work because wolves tend to return to their territories; and aversive conditioning is ineffective for wolves [28].

3. Models of Wildlife Management under Uncertainty

We consider an area jointly occupied by humans and a wildlife population, whose density is denoted by X. We suppose that there is a minimum viable wildlife density $K_0 > 0$, so as soon as X reaches K_0, the wildlife population becomes extinct with certainty, which results in the loss of existence value L. K_0 therefore acts as an absorbing barrier for X [38]. We assume that X follows the autonomous diffusion process:

$$dX = m(X)dt + v(X)dz, \tag{1}$$

where $m(.)$ and $v(.)>0$ are continuous functions defined on $[K_0,+\infty)$; dt is an infinitesimal time duration; and dz is an increment of a standard Wiener process [18].

The presence of wildlife creates benefits and costs for the local economy. Possible benefits include hunting, eco-tourism, and the control of species that could damage agriculture or forestry if left unchecked. Conversely, costs result from wildlife attacks on livestock, pets, or even humans. We denote by $f(x)$ the flow of net wildlife benefits when $X = x$.[6] If $f(x)$ is sufficiently negative, the wildlife density may be reduced by h at a cost $Q(h,x)$, where $0 \le h \le x$. $Q(h,x)$ is assumed to be increasing in h and decreasing in x. For simplicity, we suppose that culling is instantaneous, which is not unreasonable because it usually takes only a few days to find and remove radio-collared wolves.

Let us now consider the problem of a resource manager whose goal is to maximize the wildlife value function by choosing the sequences of wildlife densities at which culling should take place and the number of animals that should be culled each time. The wildlife value function is the sum of the discounted stream of net wildlife benefits plus the discounted culling costs and the possible loss of existence value or the discounted reintroduction costs, if

reintroduction is possible and optimum. This is a-priori a difficult task but it can be greatly simplified, as shown below. Let us first consider the case without reintroduction.

Managing the wildlife population without reintroduction

To motivate our approach, it is fruitful to emphasize the parallel between culling and investing. First, the decision to cull is made under uncertainty. Second, it is clearly irreversible because culling entails sunk costs that cannot be recovered if it turns out that too many animals were removed. More importantly, excessive culling may lead to extinction. Finally, the decision to cull is flexible. These characteristics call for using real options.

If we see the wildlife population as an asset whose productivity we can affect by culling (or by reintroduction, when it is possible), the manager of this resource is holding a perpetual compound option. This option gives the right but not the obligation to cull, and it never expires if the resource manager's time horizon is infinite, which we assume.

Moreover, just as in the Faustmann problem in forestry, the resource manager faces the same problem after each cull, albeit possibly with different starting points. As in [38], we rely on stochastic dynamic programming and invoke Bellman's optimality principle to infer that, if there is a unique solution, the optimal strategy is to cull by h^* as soon as X reaches $x^* > K_0$. x^* and h^* are chosen to maximize the present value of net benefits between culls, plus the present value of culling costs. Culling takes place at randomly spaced intervals (T_i, $i=1\ldots+\infty$), but since X is Markovian, the T_is are independent and identically distributed.[7] In addition, it is necessary to account for the risk of resource extinction.

Therefore, the wildlife value function $V_N(.)$ has three components: 1) the flow of net benefits until either culling or extinction; 2) the cost of culling plus the value function after

culling if X reaches x^* before K_0; and 3) the loss of existence value if instead X reaches K_0 before x^* and extinction occurs. Hence, for $x_0 \in (K_0, x^*)$,

$$V_N(x_0) = F(x_0; x^*) + D_{x^*|x_0; K_0}\{-Q(x^*, h^*) + V_N(x^* - h^*)\} - D_{K_0|x_0; x^*}L, \qquad (2)$$

where:

- x_0 is the current wildlife density;

- $F(x_0; x^*) = E\left(\displaystyle\int_0^{T_{K_0 \vee x^*|x_0}} f(X_t)e^{-\rho t}\, dt\right)$ is the present value of the flow of expected net benefits between the moment where X equals x_0 and the first time it hits either K_0 or x^*;

- E is the expectation operator with respect to X;

- $T_{K_0 \vee x^*|x_0}$ is the random duration between $X = x_0$ and the first time X hits either K_0 or x^*;

- $f(.)$ is the instantaneous flow of net wildlife benefits, including monitoring costs;

- ρ is the resource manager's discount rate.

- $D_{x^*|x_0; K_0} = E\left(e^{-\rho T_{x^*|x_0; K_0}}\right)$ and $D_{K_0|x_0; x^*} = E\left(e^{-\rho T_{K_0|x_0; x^*}}\right)$ are expected discount factors; $T_{x^*|x_0; K_0}$ is the first time X hits x^* starting from x_0, conditional on hitting x^* before K_0; conversely, $T_{K_0|x_0; x^*}$ is the first time X hits K_0 starting from x_0, conditional on hitting K_0 before x^*;

- $Q(x^*, h^*)$ is the cost of reducing the wildlife density by h^* when $X = x^*$; and

- L is the existence value of the wildlife population [21].

The method for deriving $D_{x^*|x_0; K_0}$, $D_{K_0|x_0; x^*}$, and $F(x_0; x^*)$ is outlined in the appendix.

Likewise, the value of the wildlife asset immediately after culling is

$$V_N(x*-h*) = F(x*-h*;x*) +$$
$$D_{x*|x*-h*;K_0}\{-Q(x*,h*) + V_N(x*-h*)\} - D_{K_0|x*-h*;x*}L. \tag{3}$$

Isolating $V(x*-h*)$ in (3) and plugging it into (2) allows us to reformulate the resource manager's problem as follows:

$$V_N(x_0) = \underset{\{h,x,\ 0\leq h\leq x-K_0\}}{Max}\left\{FA_N(x_0,x,h) + \frac{D_{x|x_0;K_0}}{1-D_{x|x-h;K_0}}FA_N(x-h,x,h)\right\}, \tag{4}$$

where

$$FA_N(x_0,x,h) = F(x_0;x) - D_{x|x_0;K_0}Q(x,h) - D_{K_0|x_0;x}L \tag{5}$$

is the present value of the expected flow of net benefits from wildlife, including control costs, monitoring costs, and the expected loss of existence value, between $X=x_0$ and the moment where the wildlife population is either culled or becomes extinct. Likewise, $FA_N(x-h,x,h)$ is the present value of the expected flow of net wildlife benefits during a management cycle.[8]

Let us now derive necessary first-order conditions for an interior solution. As noted in [38], both first order conditions need to be written for $x=x_0=x^*$ and $h=h^*$. First order necessary conditions for $x_0 \neq x^*$ merely indicate whether or not the maximum of (4) has been attained at x_0.

Both necessary first order conditions for (4) have the same structure, so let ζ be a dummy variable representing either h or x. Taking the derivative of Equation (4) with respect to ζ, equating it to zero, and rearranging terms gives

$$\frac{\zeta\frac{\partial FA_N(x_0,x,h)}{\partial\zeta}}{\frac{FA_N(x-h,x,h)}{1-D_{x|x-h;K_0}}} = \varepsilon_\zeta^{FA_N} + \varepsilon_\zeta^{D_N}, \text{ at } x=x_0=x^*, h=h^*, \tag{6}$$

where $\varepsilon_\zeta^{FA_N} = -\dfrac{\partial Ln\left[FA_N(x-h,x,h)\right]}{\partial Ln[\zeta]}$ is the elasticity with respect to ζ of the present value of

the expected flow of net wildlife benefits during a management cycle and

$\varepsilon_\zeta^{D_N} = -\dfrac{\partial Ln\left[D_{x|x_0;K_0}\left(1-D_{x|x-h;K_0}\right)^{-1}\right]}{\partial Ln[\zeta]}$ is the elasticity with respect to ζ of

$D_{x|x_0;K_0}\left(1-D_{x|x-h;K_0}\right)^{-1}$. On the left side of (6), the numerator is the average present value of

the flow of net wildlife benefits during a management cycle, and the denominator is the sum of

the expected flow of net wildlife benefits during all management cycles.

To gain some insight into the timing of culling, let us look at (6) for $\zeta=x$. First, consider

the elasticity of the modified discount factor. Waiting to cull delays incurring the culling costs

and increases the flow of net wildlife benefits; culling is justified only if this flow is negative at

x^*, which we now assume. If a small increase in x^* (i.e., waiting longer) causes the discount

factor to increase a lot, future culling costs are heavily discounted. The resource manager is then

more likely to cull now because the present value of future culling costs is small compared to the

value of the flow of wildlife damages. Conversely, if a small increase in x^* causes the discount

factor to increase very little, discounted future culling costs are now only slightly lower so the

manager is more likely to wait. Now consider the elasticity of the present value of the expected

flow of net wildlife benefits. If the flow of net benefits increases a lot while waiting, the

manager culls now. Conversely, if this flow changes little with a small increase in x^*, culling is

delayed. The same logic holds for the first order condition with respect to h.

Formulating the optimum management strategy in terms of elasticities provides wildlife

managers with a convenient rule of thumb if estimates of elasticity measures are available.

Managing the wildlife population with reintroduction

When reintroduction is feasible and optimum, key problem features are unchanged. If there is a unique solution, it is therefore still optimum to reduce the wildlife density by h^* as soon as it reaches x^*, although the value of x^* and h^* here usually differ from the case without reintroduction. For simplicity, we assume that reintroduction increases the wildlife density from K_0 up to $x^*- h^*$, and we denote reintroduction costs by $R(x^*- h^*)$. Given $x_0 \in (K_0, x^*)$, the value function of the wildlife asset, now denoted by $V_R(.)$, becomes

$$V_R(x_0) = F(x_0; x^*) - D_{x^*|x_0;K_0}Q(x^*, h^*) - D_{K_0|x_0;x^*}R(x^* - h^*) + \\ D_{x^* \vee K_0|x_0}V_R(x^* - h^*).$$
(7)

In (7), $D_{x^* \vee K_0|x_0} = E\left(e^{-\rho T_{x^* \vee K_0|x_0}}\right)$ is the expected discount factor for $T_{x^* \vee K_0|x_0}$, the first time X hits either x^* or K_0 starting from x_0 (see appendix); other terms are defined as above.

If we compare (7) to the value function for the no-reintroduction case (Equation (2)), the resource manager never loses the wildlife asset, although reintroduction costs may be incurred periodically. Writing (7) for $x_0 = x^*- h^*$, isolating $V(x^* - h^*)$, and plugging it back into (7) allows us to reformulate the resource manager's problem as follows:

$$V_R(x_0) = \underset{\{h,x, \; 0 \le h \le x-K_0\}}{Max}\left\{FA_R(x_0, x, h) + \frac{D_{x \vee K_0|x_0}}{1 - D_{x \vee K_0|x-h}}FA_R(x - h, x, h)\right\},$$
(8)

where

$$FA_R(x_0, x, h) = F(x_0; x) - D_{x|x_0;K_0}Q(x, h) - D_{K_0|x_0;x}R(x - h)$$
(9)

is the present value of the expected flow of net wildlife benefits, including control, monitoring and reintroduction costs, between $X=x_0$ and the moment where wildlife is either culled or reintroduced. Likewise, $FA_R(x - h, x, h)$ is the present value of the expected flow of net wildlife

benefits, including control and reintroduction costs, during a management cycle. Comparing (5) and (9), we see that $FA_N(.)$ and $FA_R(.)$ are similar except that L in the former replaces $R(x\text{-}h)$ in the latter.

Proceeding as above, if ζ represents either h or x, the two first order necessary conditions for an interior solution, which hold for $x=x^*$ and $h=h^*$, imposing $x_0=x^*$ can be written

$$\frac{\zeta \dfrac{\partial FA_R(x_0,x,h)}{\partial \zeta}}{\dfrac{FA_R(x-h,x,h)}{1-D_{x\vee K_0|x-h}}} = \varepsilon_\zeta^{FA_R} + \varepsilon_\zeta^{D_R}, \text{ at } x = x_0 = x^*, h = h^*, \tag{10}$$

where $\varepsilon_\zeta^{FA_R} = -\dfrac{\partial Ln\left[FA_R(x-h,x,h)\right]}{\partial Ln(\zeta)}$ is the elasticity with respect to ζ of $FA_R(x-h,x,h)$ and

$$\varepsilon_\zeta^{D_R} = -\frac{\partial Ln\left[D_{x\vee K_0|x_0}\left(1-D_{x\vee K_0|x-h}\right)^{-1}\right]}{\partial Ln\left[\zeta\right]} \text{ is the elasticity with respect to } \zeta \text{ of}$$

$D_{x\vee K_0|x_0}\left(1-D_{x\vee K_0|x-h}\right)^{-1}$. The right sides of Equations (6) and (10) have the same interpretation.

The difference between $V_R(x_0)$ and $V_N(x_0)$ gives the value of the option to reintroduce the wildlife population.

Comparing management alternatives

Reintroduction may be possible but it may be too expensive, or a species may cause very high damages and have a very low existence value. In this case, eradication may be optimum. With the notation defined above, the corresponding value function is then

$$V_E(x_0) = -L - Q(x_0,x_0), \tag{11}$$

where x_0 is the current wildlife density.

14

To choose between management with or without reintroduction and instant eradication, we simply select the management strategy corresponding to the highest value functions at the current density. From the Bellman optimality principle, if we follow now a management strategy, then it should remain optimum in the future unless the nature of our problem changes over time, which we ruled out by assumption.

4. An Application to the Management of Wolves

Let us now apply our framework to the management of wolves. For simplicity, we assume that X follows the geometric Brownian motion (GBM)

$$dX = \mu X dt + \sigma X dz, \tag{12}$$

where μ and $\sigma > 0$ are respectively the infinitesimal mean and volatility parameters, and dz is an increment of a standard Wiener process [18]. Equation (12) implies that the wildlife density tends to increase ($\mu > 0$) around an exponential trend, which is reasonable for fast growing wildlife populations not limited by food availability for example [37]. It appears adequate for wolves in many situations since the growth rate of the wolf populations in Michigan (1990-2001) and Yellowstone (1995-2002) exceeds 30% per year.

Assuming that X follows a GBM ignores the natural carrying capacity K_1 of the territory where the wolf population lives, so it leads to overestimating future wolf damages since they increase with wolf density; x^* with the GBM is thus likely to be lower than its optimum value. The GBM assumption is reasonable, however, if the wildlife population should be controlled significantly below K_1, which is often the case for wolves living in close proximity to humans. Assuming a GBM for X also yields much simpler analytical results than for a mean-reverting process (see [38] for some results), which explains its popularity in finance and economics.

When we solve (A.1) with boundary conditions (A.2) and (A.3), we obtain

$$D_{x|x_0;K_0} = \frac{K_0^{\theta^-} x_0^{\theta^+} - K_0^{\theta^+} x_0^{\theta^-}}{K_0^{\theta^-} x^{\theta^+} - K_0^{\theta^+} x^{\theta^-}}, \quad D_{K_0|x_0;x} = \frac{x^{\theta^-} x_0^{\theta^+} - x^{\theta^+} x_0^{\theta^-}}{x^{\theta^-} K_0^{\theta^+} - x^{\theta^+} K_0^{\theta^-}}, \tag{13}$$

and

$$D_{x \vee K_0|x_0} = \frac{(K_0^{\theta^-} - x^{\theta^-}) x_0^{\theta^+} + (x^{\theta^+} - K_0^{\theta^+}) x_0^{\theta^-}}{K_0^{\theta^-} x^{\theta^+} - K_0^{\theta^+} x^{\theta^-}}. \tag{14}$$

$\theta^+ > 0$ and $\theta^- < 0$ are defined by

$$\theta^{\pm} = \frac{0.5\sigma^2 - \mu \pm \sqrt{(0.5\sigma^2 - \mu)^2 + 2\rho\sigma^2}}{\sigma^2}. \tag{15}$$

For the net flow of amenities resulting from the presence of wolves, we suppose that

$$f(x) = -a_0 - a_1 x^2. \tag{16}$$

In (16), $a_0 > 0$ is the flow of monitoring costs for the wolf population and $a_1 x^2$ ($a_1 > 0$) is the flow of net damages; there is no flow of benefits because we could not find data linking wolf density to ecological benefits (see Section 2) or to increases in tourism, although there is some evidence that economic growth is stimulated by the conservation of wilderness habitats that sustain wolves [31]. Our decision rule thus generates a lower x^* than optimum. With (16), the derivation of $F(x_0; x)$ is straightforward (see appendix):

$$F(x_0; x) = \frac{-a_0}{\rho} \left\{ \frac{(x^{\theta^-} - K_0^{\theta^-}) x_0^{\theta^+} - (x^{\theta^+} - K_0^{\theta^+}) x_0^{\theta^-}}{x^{\theta^+} K_0^{\theta^-} - x^{\theta^-} K_0^{\theta^+}} + 1 \right\} +$$

$$\frac{-a_1}{\rho - 2\mu - \sigma^2} \left\{ \frac{(x^{\theta^-} K_0^2 - x^2 K_0^{\theta^-}) x_0^{\theta^+} - (x^{\theta^+} K_0^2 - x^2 K_0^{\theta^+}) x_0^{\theta^-}}{x^{\theta^+} K_0^{\theta^-} - x^{\theta^-} K_0^{\theta^+}} + x_0^2 \right\}. \tag{17}$$

With reintroduction, monitoring costs are always incurred so their total contribution to the objective function is simply $-a_0/\rho$; case 1 is more complex because of the risk of extinction.

In addition, we suppose that the cost of reducing the wolf density by h when the total wolf density is x can be adequately described by

$$Q(x,h) = q_0 + q_1 \frac{h}{x},$$ (18)

where $q_0 > 0$ and $q_1 > 0$ are respectively the fixed and variable culling costs. $Q(x,h)$ thus increases with the culling level and decreases with the wolf density.

Finally, we model reintroduction costs using the functional form

$$R(\xi) = r_0 + r_1 \xi.$$ (19)

Fixed costs (r_0), which are not directly dependent on the number of wolves translocated, have several components. The first one is the cost of hiring scientists to craft a reintroduction plan: select wolves, decide where they should be released, and assess their potential impacts on ranchers, hunters, other local residents, and wildlife. In addition, administrative costs are incurred to secure approval for the reintroduction process from local, state, and federal authorities, and in the case of Yellowstone and Idaho, from the Canadian government. Finally, information campaigns need to be organized to educate the public about the reintroduction process, its potential impacts, and how to manage potential conflicts with reintroduced wolves.

Variable costs (r_1) include the costs of capturing wolves, transporting them close to the release site, preparing temporary pens to acclimatize them (for a "soft release," as in Yellowstone), and releasing them. With a "hard-release," which was used in Idaho, wolves are released directly without prior acclimatization.

Before we proceed to find x^* and h^*, let us first analyze the deterministic case to get a benchmark for the impact of uncertainty.

The Deterministic Case

When $\sigma=0$ in (12), the wolf population is no longer at risk of becoming extinct and we have a one to one relationship between time and the density of the wolf population. If we proceed as above and formulate the problem in terms of x^* and h^*, the deterministic objective function is

$$
\underset{\substack{x-h\geq K_0,\\ h\geq 0}}{Max} \left\{ \frac{a_1 x^2}{2\mu-\rho}\left(\left(\frac{x_0}{x}\right)^2 - \left(\frac{x_0}{x}\right)^\delta + \frac{x_0^\delta}{x^\delta-(x-h)^\delta}\left[\left(1-\frac{h}{x}\right)^2 - \left(1-\frac{h}{x}\right)^\delta \right] \right) \right.
$$
$$
\left. -\frac{x_0^\delta}{x^\delta-(x-h)^\delta}\left[q_0 + q_1\frac{h}{x} \right] - \frac{a_0}{\rho} \right\},
\tag{20}
$$

where the constraint $x-h\geq K_0$ is added for consistency with the stochastic case. From our assumptions, the flow of net benefits from wolves is always negative. It is not optimum to cull continuously, however, because of fixed costs ($q_0>0$ in (18)). When culling takes place, it brings the wolf density all the way down to K_0. We will see that this is not the case for the stochastic case. The relevant objective function is therefore obtained by setting h^* to x^*-K_0 in (20).

Data

For our data, we searched the economic and the biological literatures, as well as sources in the U.S. (including the National Prairie Wolf Research Center, the International Wolf Center, the Wisconsin Department of Natural Resources, and the U. S. Fish and Wildlife Service), Canada, Sweden, Norway, and India. We rely mostly on information from the Yellowstone area, which we complement with data from other U.S. states.

Only relatively short annual time series are available to estimate μ and σ in (12). Using maximum likelihood, we find a low value for μ of 0.09 / year ($\sigma=0.24$ / $\sqrt{year}$) for Montana (1985-2001), and a high of 0.34 ($\sigma=0.33$) for Yellowstone (1995-2002), with Wisconsin

18

(μ=0.28, σ=0.20 for 1985-2001) and Michigan (μ=0.30, σ=0.29 for 1990-2001) in between. For our base case, we therefore select μ=0.34 / year and σ=0.33 / $\sqrt{year}$. These high growth rates are in line with wolf biology: after a gestation period of 65 days, wolves give birth once a year to 1 to 9 pups, with an average litter size of 5 to 6 pups. Mortality is fairly high, however: only 1 or 2 of every 10 pups born live, survives to leave the pack to find a mate [9].

The survival of a wolf population does not hinge only on a minimum density; it also depends on factors such as the availability of preys and land use (e.g., road density). In addition, a minimum number of wolves are needed for genetic diversity. The Yellowstone area is 25,000 square miles (mi^2) and it offers an abundant food supply, so a starting population as small as 25 wolves is likely to thrive, as shown by recent history. We therefore set K_0 to the fairly low value of one wolf per 1000 mi^2.

To estimate reintroduction costs for Yellowstone, we rely mostly on [47]. Fixed costs, which comprise staff/support, public education/law enforcement and administration/support, total approximately r_0 = \$320,000.[9] Variable costs, which include monitoring of donor pack, facility preparation/care, capture/transport, and confine/release, reach \$13,000 per wolf. This number is high: for Idaho, variable costs were only \$5,300 per wolf released because there was no acclimation period. Since our decision variable is wolf density per 1,000 mi^2, our variable cost coefficient is then r_1 =13,000*25=325,000, in \$ * 1,000 mi^2 per wolf (recall that the greater Yellowstone area is 25,000 mi^2).[10] These costs exclude monitoring, which we estimate at \$100,000 per year (a_0=\$100,000): \$50,000 for the salary of the wolf program coordinator, and \$50,000 for radio telemetry expenses and howl surveys (see also [50]).

Evaluating the stream of damages from wolves is challenging because relatively few data are available. To come up with an estimate for a_1 in (16), we use OLS with White

19

heteroskedasticity-consistent standard errors to link wolf density with compensation paid to ranchers for livestock depredation, and we find $a_1 = 524$ \$ * (1,000 mi^2 per wolf)^2. This approach underestimates true damages because carcasses sometimes disappear and cannot be attributed to the misdeeds of wolves. However, it compensates partly for the GBM hypothesis, which overestimates the growth potential of the wolf population and therefore its potential damages.

For control costs, we only found data from Minnesota because until April 1st, 2003, it was the only state with a sizeable wolf population where culling was permitted (wolves were listed as threatened instead of endangered).[11] When we use OLS with White heteroskedasticity-consistent standard errors to estimate (18), we find q_0=\$108 and q_1=\$974 * 1,000 mi^2 per wolf. Estimations for both damage costs and culling costs are detailed in Tables 1A and 1B.

Existence value, which is measured by applying contingent valuation, is one of the most difficult parameters to estimate. We found only a couple of studies that estimate existence value for wolves [5] [47]. We adopt the existence value (L = \$8.3 million) from [47], since it focuses on the reintroduction of wolves in Yellowstone National Park.

Finally, we choose a discount rate ρ of 7% per year, a value commonly used by the federal government for conducting cost-benefit analyses. Table 2 summarizes the meaning and estimated values of our model parameters.

Since uncertainty surrounding some parameters is substantial, we conduct an extensive sensitivity analysis using the following values: $\mu \in [0.10; 0.50]$, $\sigma \in [0.10; 0.60]$, $K_0 \in [1.0; 4.0]$, $a_1 \in [-1,600; -500]$, $L \in [\$4*10^6; \$12*10^6]$, and $\rho \in [1.25\%, 10.5\%]$. We also examine how lower reintroduction costs (r_0=220,000 and r_1=132,500) impact x^* and h^*. Results were generated using MathCad on a PC. They are discussed below and illustrated on Figures 1 through 4.

Results

Let us first start with our base parameters (see Table 2). For case 1 (no reintroduction), we find x^*=4.92 wolves per 1000 mi^2 and h^*=0.70 wolf per 1000 mi^2, while for case 2 (with reintroduction), x^*=3.93 wolves per 1000 mi^2 and h^*= 0.87 wolf per 1000 mi^2. The corresponding value functions at X=1.5 wolves per 1000 mi^2 (an arbitrary value) are -\$2,295,000 and -\$1,626,000 respectively; they are negative because the presence of wolves is assumed to generate a flow of costs. The possibility of reintroduction increases the value of wolves to society and the best policy is to manage them according to case 2. These results compare with x^*=1.29 wolves per 1000 mi^2, h^*=0.29 wolves per 1000 mi^2, and V=-\$1,444,000 for the deterministic case. Managing the wolf population with the deterministic policy parameters (x^* and h^*) would lead to a value function of –\$8,470,000 for case 1, and it simply would not make sense for case 2 since culling would induce the need for reintroduction, leading to very large and unnecessary management costs (due to constant culling and reintroduction). Accounting for volatility in the wolf population is therefore important here.

Contrasting case 2 with case 1, we see that culling takes place at a 20% lower density and it is 24% more severe for case 2: since reintroduction costs are much lower than existence value, the wildlife manager keeps a lower wolf density to reduce damages, even if it requires more frequent reintroductions. The difference between case 1 and case 2 increases with uncertainty, especially for h^* (Figure 1A) and for the value function (Figure 1B).

From Figures 1A and 1B, we also see that x^* and h^* increase monotonically with volatility for both cases: a higher volatility increases the risk of a population crash so the wildlife manager needs to maintain a higher wolf density (and thus tolerate a higher flow of damages) in order to reduce the risk of loss of existence value or of incurring reintroduction costs. When uncertainty goes to 0, however, both x^* and h^* tend towards their deterministic values.

A good estimate of the wolf population growth rate (μ) turns out to be important for x^* and h^*. From Figure 2, we observe that as μ increases, both x^* and h^* decrease monotonically, and the difference between the two cases is sharply reduced. Indeed, when the rate of growth of the wolf population increases, the risks of extinction or of incurring reintroduction expenses decrease, and so does the difference between cases 1 and 2: the value functions at X=1.5 for μ=0.50 are respectively -\$1.747 and -\$1.517 million for cases 1 and 2.

The value of K_0 also impacts x^* and h^* (Figure 3). As K_0 increases, the risks of extinction (case 1) or of incurring reintroduction costs increases, so x^* becomes larger (x^*=12.82 (11.74) when K_0=4 for case 1 (case 2)), but this also swells wolf damages and lowers the value function (V_N=-\$6.920 10^6 and V_R=-\$3.878 10^6 at K_0=4). To partly compensate, culling (h^*) increases, especially for case 2: at K_0=4, h^*=0.95 for case 1 and 2.42 for case 2.

By contrast, the damage coefficient (a_1), the existence value (L), and especially the discount rate (ρ) have a limited impact on x^* and h^*. This result is particularly important for a_1 since many ranchers are claiming that only a fraction of the damages caused by wolves is officially acknowledged; they are therefore calling for much more severe culling. Yet if we triple the rate of damages (a_1) to 1600, x^* decreases to only 3.25 (from 3.93) while h^* also decreases to 0.61 (down from 0.87) for case 2 (the relevant situation here; see Figure 4). The underlying reason is that wolf damages are relatively small compared to other costs (e.g., reintroduction or monitoring costs). From these results, we see that the optimal policy when damages increase is to cull more frequently (at a lower wolf density) but also less severely. This also holds for case 1.

Although existence value justifies the reintroduction of wolves in our models, our results indicate that it is not necessary to have a precise measure of it because it is a large cost to avoid. This is good news for the practical implementation of our approach. Indeed, when L increases

from $4 million to $12 million, $x^*=4.34$ and $h^*=0.67$ increase only to $x^*=5.20$ and $h^*=0.71$ in case 1 (it does not intervene directly in case 2). So when existence value increases, culling is less frequent but slightly more severe.

The discount rate ρ also has a fairly small impact on x^* and h^*, except for very small values. When ρ grows from 2% to 10.5%, x^* increases from 4.48 to 4.91 and h^* grows from 0.68 to 0.70 for case 1; for case 2, x^* inches down from 3.99 to 3.89 and h^* decreases from 0.90 to 0.86. When the value of future damages decreases, it is therefore optimal to cull less frequently but slightly more severely for case 1, whereas the reverse holds for case 2. As expected, the value functions decrease when ρ increases as future damages are discounted more heavily. When ρ is below 2%, x^* and the value function start decreasing sharply; around $\rho=1.25\%$, immediate extinction becomes the preferred solution.

Finally, decreasing reintroduction costs (for $r_0=220,000$ and $r_1=132,500$) increases the value function to -$1,554,000 by limiting wolf damages: the wildlife manager culls more frequently ($x^*=3.56$ instead of 3.90) although less severely ($h^*=0.78$ instead of 0.87).

5. Conclusions

We have presented a continuous time framework that offers simple rules for managing a valuable endangered wildlife population under uncertainty, which also accounts for the risk of extinction and the possibility of reintroduction. These two important features have been neglected so far in the environmental economics literature.

Our application to the management of wolves in the greater Yellowstone area indicates that the possibility of reintroduction adds approximately 30% to the social value of the wolf population. A comprehensive sensitivity analysis reveals that the wolf population growth rate

(μ), its volatility (σ), and the minimum wolf density (K_0) are important determinants of the policy variables x^* (when to cull) and h^* (by how much to cull). By contrast, the damage coefficient (a_1), the existence value (L), and especially the discount rate (ρ) have a limited impact on x^* and h^*. This matters because of the difficulty of measuring existence value or of selecting a discount rate for preserving species. In addition, a number of Western ranchers have been asking for a reduction in wolf density on the basis of underestimated damages (or for a complete removal of wolves). Our results therefore do not support their claims.

Our results also highlight the importance of accounting for existence value in wildlife management models. However, it is critical that a mechanism be in place so that the general public (for whom wolves are a public good) compensates ranchers and hunters (for whom wolves are instead a public bad [6]), who bear the brunt of wolf damages. This compensation mechanism can be private, as illustrated by the successful creation by Defenders of Wildlife of the Bailey Wildlife Foundation Wolf Compensation Trust, which pays livestock owners for losses from wolf predation.[12]

Although our illustration assumes that the wolf density follows a GBM, our framework can be easily applied to mean-reverting processes (such as a logistic Brownian motion [38]) in order to account for carrying capacity. Useful extensions could include modeling uncertainty in damages (or benefits) at a given population density; incorporating errors in measuring the size of the wolf population; quantifying ecosystem benefits from wolves; analyzing land use policies that minimize conflicts with endangered species; and modeling interactions with other species when bioeconomic data become available.

References

[1]   R.R. Alexander, Modeling Species Extinction: The Case For Non-consumptive Values, Ecol. Econ. 13, 259-269 (2000).

[2]   W.S. Alverson, D.M. Waller, S.L. Solheim, Forests Too Deer: Edge Effects in Northern Wisconsin, Cons. Bio 2, 348-358 (December 1988).

[3]   B. Becher, Tough to Bear; Attacks on Humans by Hungry Black Bears A Growing Problem in State, The Daily News of Los Angeles, Valley Edition, November 6 (2003).

[4]   L. Bergquist, Rebound of Wolves in State Increases Conflicts with Hunters, Farmers, Milwaukee Journal Sentinel (Wisconsin), Final Edition December 4 (2003).

[5]   M. Boman, Estimating Costs and Genetic Benefits of Various Sizes of Predator Populations: The Case of Bear, Wolf, Wolverine and Lynx in Sweden, J. Environ. Manage. 43, 349-357 (1995).

[6]   Bostedt, G., Threatened Species as Public Goods and Public Bads - An Application to Wild Predators in Sweden. Environ. & Res. Econ. 13, 59-73 (1999).

[7]   E. Braunstein, Seeking Security Against Big Cats, Press Enterprise (Riverside, CA), October 23 (2003).

[8]   T.W. Clark, A.P. Curlee, R.P. Reading, Crafting Effective Solutions to the Large Carnivore Conservation Problem, Conserv. Biol. 10, 940-948, 1996.

[9]   Defenders of Wildlife, Wolves – The Basics of Wolf Biology and Taxonomy, http://www.defenders.org/wildlife/wolf/wolfbio.html > 12/02/03.

[10]   Defenders of Wildlife, Wolves – Restoring Wolves to the Northern Rockies, http://www.defenders.org/wildlife/wolf/ynpfact.html > 12/02/03.

[11] M. Delibes, Status and Conservation Needs of the Wolf (Canis lupus) in the Council of Europe Member States, 37 pages, Report to the Council of Europe (1990).

[12] A. Dixit, R.S. Pindyck, Investment under Uncertainty, Princeton University Press, Princeton, New Jersey (1994).

[13] T. J. Dolan, Growing Roadway Risk: A Surge In Collisions Has Communities Again Weighing Lethal -- And Controversial -- Deer Control Options, Buffalo News, New York, November 19 (2003).

[14] D. Finnoff, J. Tschirhart, Harvesting in an Eight-Species Ecosystem, Journal of Environmental Economics and Management 45, 589-611 (2003).

[15] H. Fisher, The Wolf Settles in at Yellowstone – Reintroducing Top Predator is Influencing Everything From Bears To Beetles, Defenders of Wildlife Magazine, http://www.defenders.org/magazinenew/guest/varleymag.html.

[16] D. Hansen, As Wolf Packs Grow, So Does Resentment, Spokesman Review, Spokane, WA, July 20 (2003).

[17] R.T. Johnson, On the Spoor of the "Big Bad Wolf", J. Environ. Edu. 6, 37-39 (1974).

[18] S. Karlin and H. M. Taylor, A Second Course In Stochastic Processes, Academic Press, New York, New York (1981).

[19] S.R. Kellert, Public Perceptions of Predators, Particularly the Wolf and Coyote, Bio. Cons. 31, 167-189 (1985).

[20] S.R. Kellert, M. Black, C.R. Rush, A.J. Bath, Human Culture and Large Carnivore Conservation in North America, Conserv. Biol. 10, 977-990 (1996).

[21] J.V. Krutilla, Conservation Reconsidered, Am. Econ. Rev. 57, 777-786 (1967).

[22] R. Lakshmi, Indian Villagers Cope With Sudden Death; Forays Into Forest Often Prove Fatal, Leaving 'Tiger Widows' to Mourn, The Washington Post, July 7 (2003).

[23] A. Leopold, A Sand County Almanac, Oxford University Press, New York (1948).

[24] D.S. Licht, S.H. Fritts, Gray Wolf (Canis Lupus) Occurenes in the Dakotas, Amer. Midl. Naturalist 132 (1), 74-81 (1994).

[25] B. Lopez, Of Wolves and Men, Charles Scribner and Sons, New York (1978).

[26] P. Matthiessen, Wildlife in America, Viking Press, New York (1987).

[27] B.E. McLaren, R.O. Peterson, Wolves, Moose, and Tree Rings on Isle Royale, Science 266, 1555-1558 (1994).

[28] L.D. Mech, The Challenges and Opportunity of Recovering Wolf Populations, Cons. Biol. 9 (2), 270-278 (1995).

[29] L.D. Mech, S.H. Fritts, D. Wagner, Minnesota Wolf Dispersal to Wisconsin and Michigan, Amer. Midl. Naturalist 133 (2), 368-370 (1995).

[30] R.F. Noss, H.B. Quigley, M.G. Hornocker, T. Merrill, P.C. Paquet, Conservation Biology and Carnivore Conservation in the Rocky Mountains, Conserv. Biol. 10, 949-963 (1996).

[31] R. Rasker, A. Hackman, Economic Development and the Conservation of Large Carnivores, Conserv. Biol. 10, 991-1002 (1996).

[32] J. Read, Coyote Mauls Manlius Couple's Dog, The Post-Standard, Syracuse, New York, September 21 (2003).

[33] J. Robbins, To Kill and To Be Killed, Los Angeles Times Magazine,July 27, 2003.

[34] D. Rondeau, Along the Way Back From the Brink, J.Environ. Econom. Management 42, 156-182 (2001).

[35] D. Rondeau and J. Conrad, Managing Urban Deer, Amer. J. Ag. Econ. 85, 266-281 (2003).

[36] S. Rubenstein, Coyote Makes S.F. a Truly Wild Town; Furry Visitor to Bernal Heights Delights, Perturbs Hill's Dog Walkers, The San Francisco Chronicle, November 14 (2003).

[37] J.D. Saphores, The Economic Threshold with a Stochastic Pest Population: A Real Options Approach, Amer. J. of Ag. Econ. 82, 541-555 (2000).

[38] J. D. Saphores, Harvesting a Renewable Resource under Uncertainty 28, 509-529 (2003).

[39] M. Scott, As Coyotes Prowl Nightly, Cats Go Missing, St. Petersburg Times, Florida, September 25 (2003).

[40] E. Shogren, Killing Wolves in the Pursuit of Their Salvation, Los Angeles Times, Sunday June 14 (2002).

[41] J. Terborgh, L. Lopez, P. Nuñez V, M. Rao, G. Shahabuddin, G. Orihuela, M. Riveros, R. Ascanio, G.H. Adler, T.D. Lambert, L, Balbas, Ecological Meltdown in Predator-Free Forest Fragments, Science 294, 1923-1926 (30 November 2001).

[42] L. Thomas, What's Good for the Goose Firms Pull Out All Stops to Halt Resident Fowl's Population Explosion, Pittsburgh Post-Gazette, Pennsylvania, September 7 (2003).

[43] A. Treves, K.U. Karanth, Human-Carnivore Conflict and Perspectives on Carnivore Management Worldwide, Conserv. Biol. 17 (6) 1491-1499 (2003).

[44] U.S. Fish and Wildlife Service, Gray Wolf, http://midwest.fws.gov/wolf/learn/biologue.pdf > 12/02/03.

[45] U.S. Fish and Wildlife Service, Wolf Recovery in North America, http://midwest.fws.gov/wolf/learn/grwo-na.pdf > 12/02/03.

[46] U.S. Fish and Wildlife Service, Species Profile for Red Wolf, https://ecos.fws.gov/species_profile/SpeciesProfile>spcode=A00F > 12/02/03.

[47] U.S. Fish and Wildlife Service, Federal Register 59 FR 60252 11/22/94, https://ecos.fws.gov/docs/frdocs/1994/94-28746.html > 12/02/03.

[48] J.L. Weaver, P.C. Paquet, L.F. Rugiero, Resiliency and Conservation of Large Carnivores in the Rocky Mountains, Conserv. Biol. 10, 964-976 (1996).

[49] L. White Jr., The historical root of our environmental crisis, in I. Barbour (Ed.), Western Man and Environmental Ethics, Addison-Wesley Publishing, Reading, Massachusetts, (1973).

[50] Wisconsin Department of Natural Resources, Wolf Management Budget, http://www.dnr.state.wi.us/org/land/er/publications/wolfplan/plan6.htm > 12/02/03.

[51] Xinhua General News Service, Zimbabwe Launches Project to Reduce Human-Animal Conflict, September 1 (2003).

Table 1A: Culling Costs Estimation

Model: $Q(h/x)=q_0+q_1*h/x$
Method: Least Squares with White Heteroskedasticity-Consistent Standard
Errors & Covariance
Sample: 1989-1998; 10 observations.

	Coefficient	Std. Error	t-Statistic	Prob.
q_0	107.51	24.346	4.416	0.002
q_1	973.57	426.701	2.282	0.052

R^2	0.226	Mean dependent variable	170.26
$\bar{R}^2$	0.130	S.D. dependent variable	41.14
S.E. of regression	38.38	Akaike info criterion	10.31
Sum squared residuals	11784.32	Log likelihood	-49.55

Table 1B: Damage Costs Estimation

Model: $f(x)=-a_1 x^2$
Method: Least Squares with White Heteroskedasticity-Consistent Standard
Errors & Covariance
Sample: 1995-2000; 6 observations.

	Coefficient	Std. Error	t-Statistic	Prob.
a_1	523.90	157.139	3.334	0.021

R^2	0.073	Mean dependent variable	-6950.09
$\bar{R}^2$	0.073	S.D. dependent variable	7144.67
S.E. of regression	6877.72	Akaike info criterion	20.66
Sum squared residuals	2.37E+08	Log likelihood	-60.98

Table 2: Base Parameter Values

Parameters	Meaning	Parameter estimate
μ	Rate of growth	0.34 per year.
σ	Volatility	0.33 per year$^{0.5}$.
K_0	Minimum viable density	1 per 1000 mi^2.
$Q(x,h) = q_0 + q_1 \dfrac{h}{x}$	Culling costs	\$108+974*(h/x).
L	Existence value	\$8.3 million.
$f(x) = -a_0 - a_1 x^2$	Stream of monitoring and damage costs.	-\$100,000-\$524 x^2 per year.
$R(\xi) = r_0 + r_1 \xi$	Reintroduction costs	\$320,000+325,000 ξ
ρ	Discount rate	0.07 per year
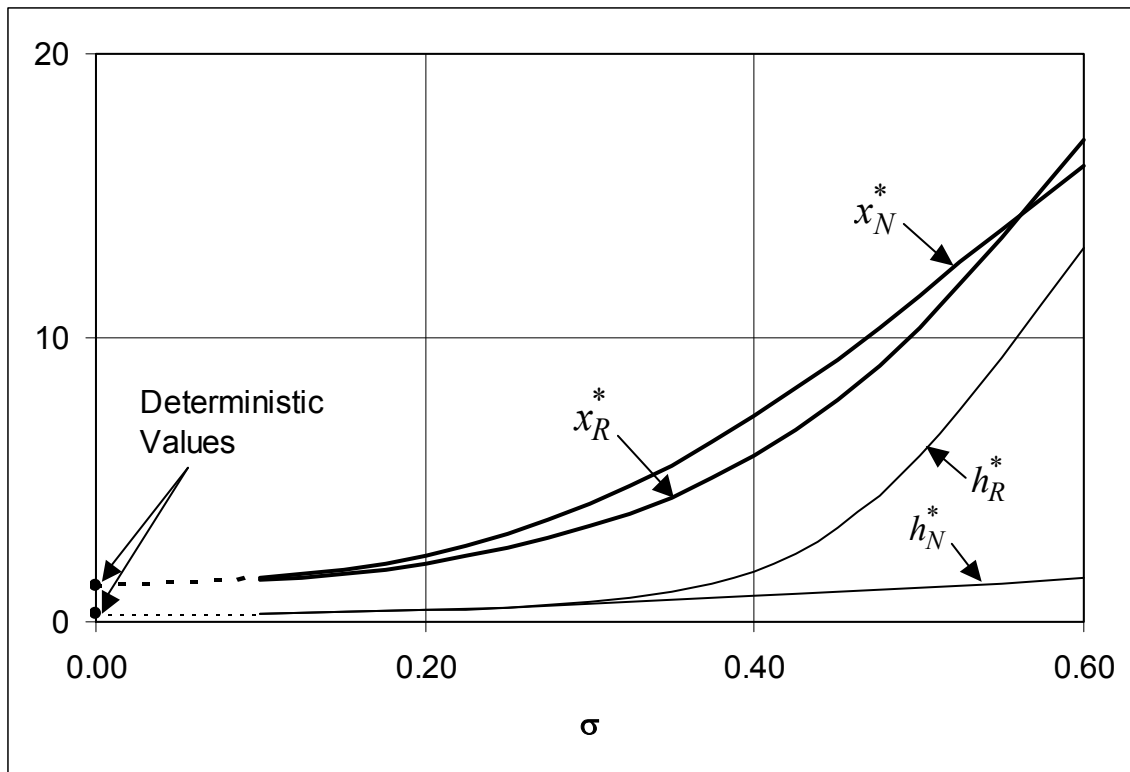
Figure 1A: x^* and h^* versus σ, with and without reintroduction.

Notes: This figure was generated with our base parameters, except for σ, the volatility coefficient of the wolf population. x_N^* and h_N^* are respectively the density of the wolf population at which culling should take place and by how much the wolf density should be reduced without reintroduction. x_R^* and h_R^* have the same meaning with reintroduction.
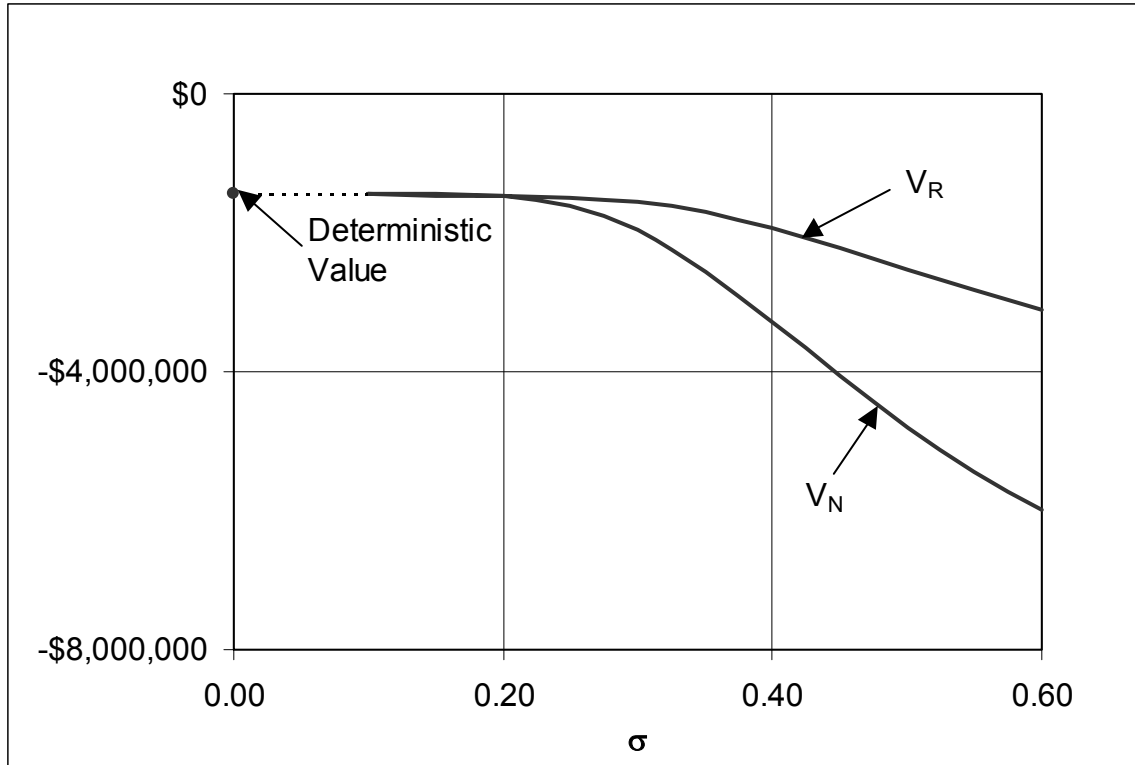
Figure 1B: V_N **and** V_R **versus** σ.

Notes: This figure was generated with our base parameters, except for σ, the volatility coefficient of the wolf population. V_N and V_R are the value functions for case 1 (no reintroduction) and case 2 (reintroduction) respectively. They are calculated at 1.5, an arbitrary value of the wolf density.
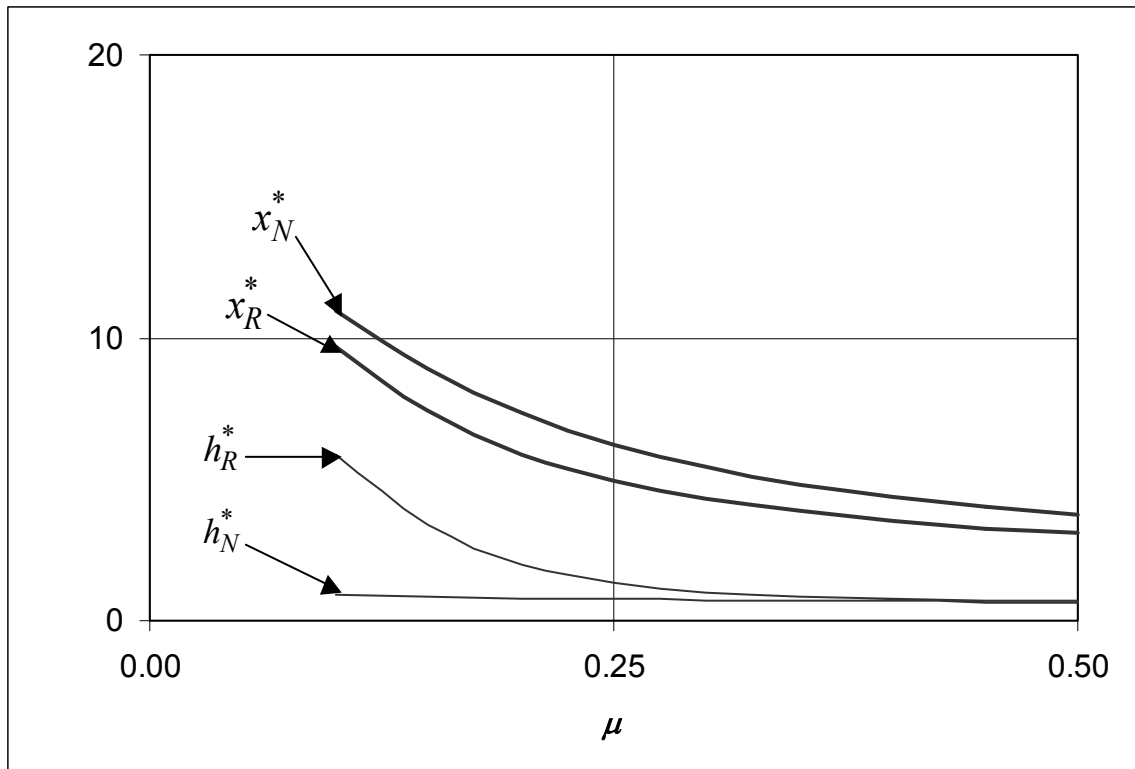
Figure 2: x^* and h^* versus μ, with and without reintroduction.

Notes: This figure was generated with our base parameters, except for μ, the growth rate of the wolf population. x_N^* and h_N^* are respectively the density of the wolf population at which culling should take place and by how much the wolf density should be reduced without reintroduction. x_R^* and h_R^* have the same meaning with reintroduction.
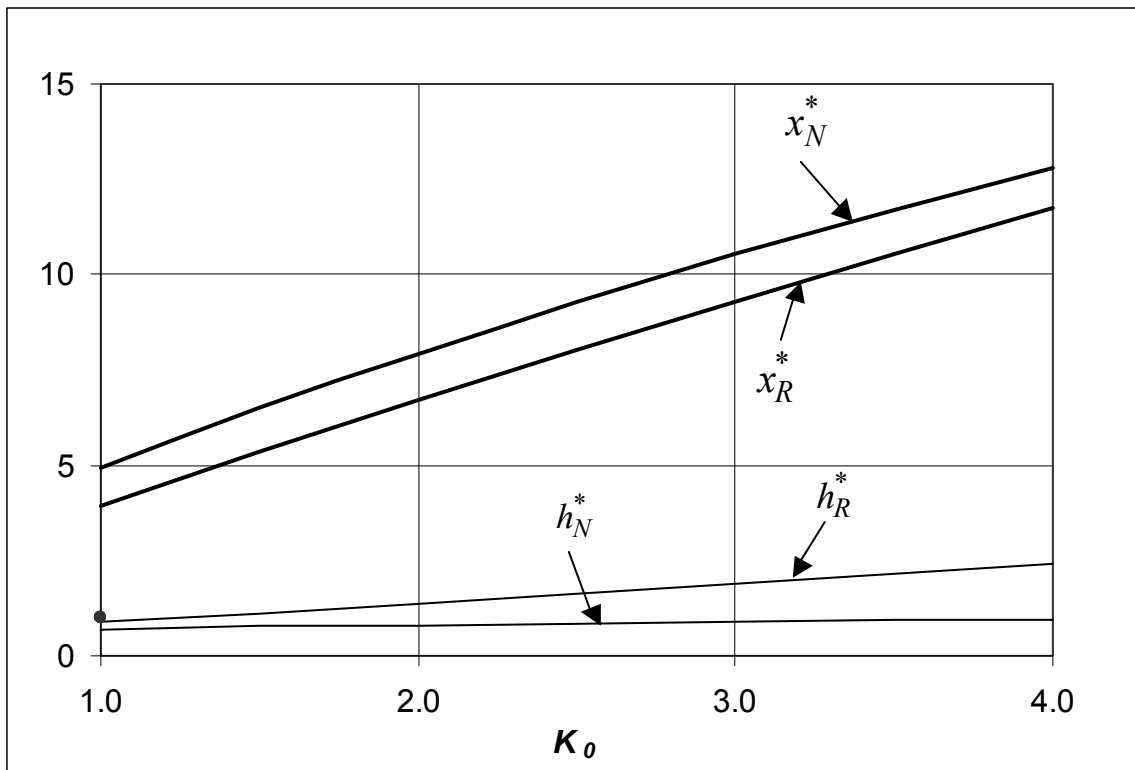
Figure 3: Impact of K_0 on x^* and h^*.

Notes: This figure was generated with our base parameters, except for K_0, the minimum density of the wolf population. x_N^* and h_N^* are respectively the density of the wolf population at which culling should take place and by how much the wolf density should be reduced without reintroduction. x_R^* and h_R^* have the same meaning with reintroduction.
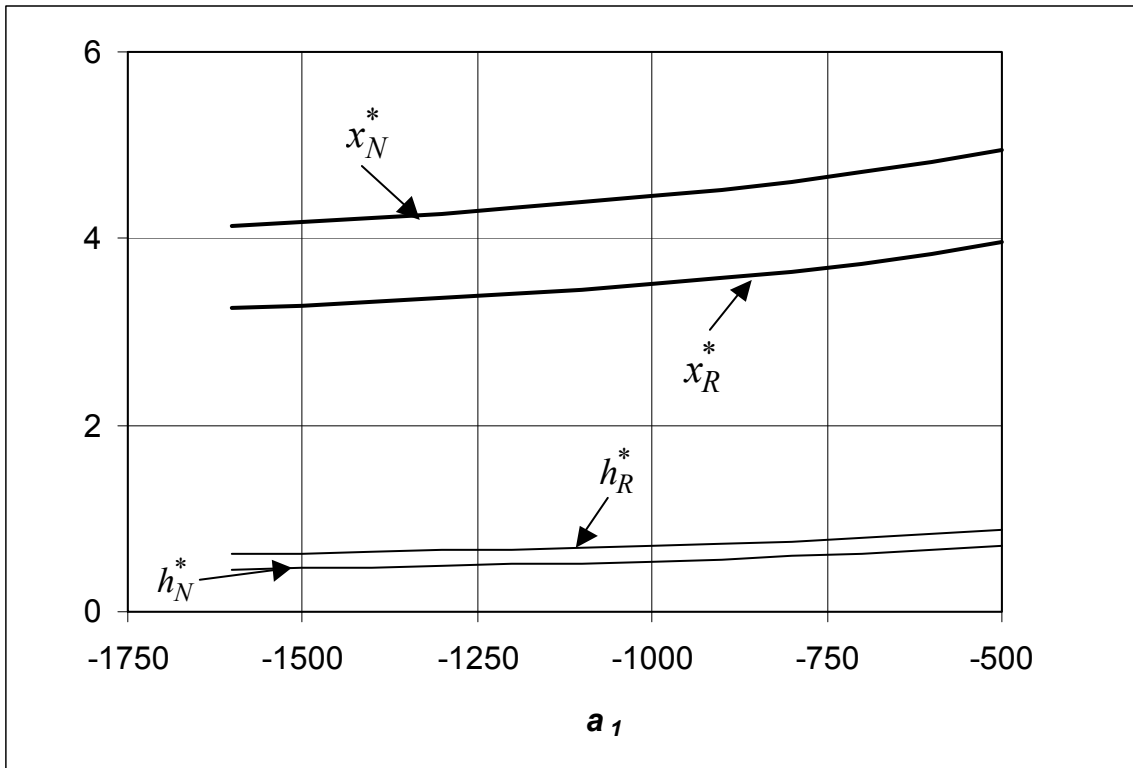
Figure 4: Impact of a_1 on x^* and h^*.

Notes: This figure was generated with our base parameters, except for a_1, the damage coefficient. x_N^* and h_N^* are respectively the density of the wolf population at which culling should take place and by how much the wolf density should be reduced without reintroduction. x_R^* and h_R^* have the same meaning with reintroduction.

Appendix

Let a, b, and x_0 be three real numbers such that $0 < a < x_0 < b$.

Expression of $D_{a|x_0;b}$ and $D_{a \vee b|x_0}$

We suppose that b is absorbing. Let $W(x_0) \equiv D_{a|x_0;b} = E\left(e^{-\rho T_{a|x_0;b}}\right)$, where X follows (1).

Using a Taylor expansion, the law of total probabilities and the Markov property, Karlin and Taylor (1981) show that $W(.)$ solves

$$\frac{v^2(\xi)}{2} \frac{d^2 W(\xi)}{d\xi^2} + m(\xi) \frac{dW(\xi)}{d\xi} - \rho W(\xi) = 0. \tag{A.1}$$

By construction,

$$W(a) = 1. \tag{A.2}$$

In addition, since $T_{a|x_0;b}$ is the first time X hits a starting from x_0, conditional on hitting a before b and b is assumed absorbing,

$$W(b) = 0. \tag{A.3}$$

$$V(x_0) \equiv D_{a \vee b|x_0} = E\left(e^{-\rho T_{a \vee b|x_0}}\right) \text{ also verifies (A.1); based on the definition of } D_{a \vee b|x_0},$$

the boundary conditions are simply

$$V(a) = V(b) = 1. \tag{A.4}$$

Expression of $F(x_0; x^*)$

Let $R(x_0) = E\left(\int_0^{T_{a \vee b|x_0}} f(X_t)e^{-\rho t}dt \mid X(0) = x_0\right)$. Using the same approach as for $D_{a|x_0;b}$, Karlin

and Taylor [18] show that $R(.)$ solves

$$\frac{v^2(\xi)}{2}\frac{d^2 R(\xi)}{d\xi^2} + m(\xi)\frac{dR(\xi)}{d\xi} - \rho R(\xi) + f(\xi) = 0, \tag{A.5}$$

with boundary conditions

$$R(a) = R(b) = 0. \tag{A.6}$$

If X follows the GBM described by (12), $m(\xi) = \mu\xi$ and $v(\xi) = \sigma\xi$. A general solution of

(A.5) with boundary conditions $R(K_0) = R(x) = 0$ is then

$$R(x_0) = \frac{x^{\theta^-}R_0(K_0) - R_0(x)K_0^{\theta^-}}{x^{\theta^+}K_0^{\theta^-} - x^{\theta^-}K_0^{\theta^+}}x_0^{\theta^+} - \frac{x^{\theta^+}R_0(K_0) - R_0(x)K_0^{\theta^+}}{x^{\theta^+}K_0^{\theta^-} - x^{\theta^-}K_0^{\theta^+}}x_0^{\theta^-} + R_0(x_0), \tag{A.7}$$

where $R_0(.)$ is a particular solution of (A.5). If $f(\xi) = -a_1\xi^2$, a particular solution of (A.5)

obtained by substitution is $R_0(\xi) = \dfrac{-a_1\xi^2}{\rho - 2\mu - \sigma^2}$. To model monitoring costs we simply solve

(A.5) again with $f(\xi) = -a_0$, where $a_0 > 0$. A particular solution of (A.5) obtained by

substitution is $R_0 = \dfrac{-a_0}{\rho}$. Since $F(x_0;x) \equiv R(x_0)$, Equation (17) follows.

[1] Umbrella species require habitats that encompass the habitats of many other species [30].

[2] In the Rocky Mountains, habitat requirements exceed 400 km^2 for mountain lions, reach 900 km^2 for grizzly bears, and range between 250 and 2000 km^2 for wolf packs [30].

[3] For a history of the interactions between wolves and humans in Europe and a synthesis of its current status there, see [11].

[4] Variations on the tale of Little Red Riding Hood are heard around the world. Apart from Western Europe, basic plot elements can also be found in tales from Japan, China and Korea (see http://mld.ursinus.edu/Maerchen/rrhintro.html). Such stories probably contribute to the bad image of wolves among young children [17].

[5] To this list, we need to add 9 llamas, and one horse [33]. These are documented attacks, so actual damages may be larger.

[6] Throughout this paper, X denotes a random variable and x is one of its realizations.

[7] Indeed, since X follows a diffusion process, it is Markovian, i.e. all the relevant information about X is contained in its current state [18].

[8] A management cycle goes from $x^*\text{-}h^*$ to either x^* or K_0. If X reaches K_0, the wildlife population becomes extinct if reintroduction is not possible; otherwise, reintroduction costs are incurred.

[9] \$50,000 of staff expenses are assumed to be allocated to monitoring (see footnotes in [50]).

[10] This classification is not absolute. According to Ed Bangs (U.S. Fish and Wildlife Service reintroduction coordinator for the Northern Rockies), reintroduction costs could in fact be much lower for the reintroduction of only a few wolves (personal communication, 12/08/03).

[11] See http://www.npwrc.usgs.gov/resource/1999/wpop/intro.htm#table1.

[12] See http://www.defenders.org/wolfcomp.html.