

# Managing Mortality of Multi-Use Megafauna

Aaron J. Enriquez<sup>a</sup>, David C. Finnoff<sup>a</sup>

<sup>a</sup>*Department of Economics, University of Wyoming, 1000 E University Ave, Laramie, WY 82071, United States*

---

## Abstract

Grizzly bears in the Greater Yellowstone Ecosystem, which are currently listed under the Endangered Species Act, are a multi-use species that cause both benefits and damages. As the grizzly bear population has increased over time, there has been an increase in the number of grizzly bear-human conflicts and non-harvest human-caused grizzly bear mortalities. Federal protections prevent active management (i.e., direct population control). Instead, wildlife managers rely on reactive management (i.e., indirect population control through conflict management). To shed light on when a recovery program ought to transition to active management, a bioeconomic model is constructed and parameterized. A representative wildlife agency decides whether or not to enact active management by taking into account how stock-dependent benefits and damages adjust along a recovery path. Given the assumptions in the base case, the grizzly bear population has surpassed the size at which protections ought to have been removed. When a program of active management is a contentious and negotiated settlement with little flexibility, the natural capital value of a live animal may well be negative for an interval of time, during which it is optimal for society to continue conserving the species by developing a buffer.

*Keywords:* multi-use species, endangered species, bioeconomic analysis, optimal control, recovery path, natural capital value

---

## 1. Introduction

“Multi-use” wildlife species cause both benefits and damages to people in society (Zivin et al., 2000; Rondeau, 2001). There are many examples of multi-use species, including livestock (Huffaker et al., 1989, 1990), wild horses (Huffaker et al., 1990; Fonner and Bohara, 2017), beavers (Huffaker et al., 1992), feral pigs (Zivin et al., 2000), white-tailed deer (Rondeau, 2001; Rondeau and Conrad, 2003), African elephants (Horan and Bulte, 2004), moose (Skonhøft and Olaussen, 2005; Skonhøft, 2006; Olaussen and Skonhøft, 2011), wolves (Skonhøft, 2006), and alewives (Fenichel et al., 2010). The abundance of a multi-use species at a particular point in time determines whether benefits or damages dominate, which makes the management of such species complicated (Rondeau, 2001). Multi-use species can be classified as either assets or liabilities, depending on whether their *in*

---

*Email addresses:* [aenriqu1@uwyo.edu](mailto:aenriqu1@uwyo.edu) (Aaron J. Enriquez), [finnoff@uwyo.edu](mailto:finnoff@uwyo.edu) (David C. Finnoff)

*situ* marginal benefits are positive or negative (Horan and Bulte, 2004). In other words, a multi-use species’ classification at a specific population size depends on whether the value of holding an additional animal in the wild (i.e, its shadow value or natural capital value) is positive or negative at that population size.<sup>1</sup> Optimal management calls for attaining and maintaining the optimal natural capital value of a multi-use species, in most cases by controlling its population size. Wildlife managers utilize a diverse array of control actions to achieve this goal, including trapping (Huffaker et al., 1992; Zivin et al., 2000), hunting (Zivin et al., 2000; Skonhofs and Olausen, 2005; Skonhofs, 2006; Olausen and Skonhofs, 2011), culling (Rondeau and Conrad, 2003), stocking (Huffaker et al., 1989, 1990; Rondeau, 2001; Fenichel et al., 2010), and non-lethal controls (Fonner and Bohara, 2017).

It is often assumed that wildlife managers are able to employ any control action in their arsenal, as long as it pushes society to a better outcome. However, this assumption does not reflect the reality facing certain multi-use species, including those that are currently protected by the Endangered Species Act (ESA). The rub with ESA protections is that they limit the types of control actions that managers can use to influence the natural capital values of the species. Further, if a protected multi-use species is high-profile enough, possible control actions may come under intense scrutiny from the general public. The contentiousness of management programs for such species can lead to negotiated settlements that offer very little in terms of flexibility.

A bioeconomic model is constructed, in which a representative wildlife agency is limited by federal protections when maximizing the social net benefits from a high-profile multi-use species. The agency is bound to using a “reactive” management strategy, meaning it can only indirectly influence the species’ population size in cases of severe economic damage. The outcomes under the reactive management strategy are compared to the outcomes under an alternative “active” management strategy, through which the agency chooses if and when to use a new control action (e.g., optimal harvest) to directly influence the species’ population size. The structure of linear optimal control reflects the rigidity faced by the agency in its management of the high-profile species. The agency’s decision of whether or not to enact active management is akin to pushing an “on/off” switch, after which the new management strategy persists indefinitely. The analysis yields a threshold population size and natural capital value at which the agency should switch from reactive to active management. A key result is that it may be optimal for society to continue conserving a species even after its natural capital value turns negative. In such cases, it is optimal for society to first develop a buffer of the species before enacting a new control action. The framework is applied to one of the most contentious wildlife species in recent American history: grizzly bears (*Ursus arctos horribilis*) in the Greater Yellowstone Ecosystem (GYE).<sup>2</sup>

GYE grizzly bears, which are currently listed as a “threatened” species under the ESA, are a high-profile species that pose unique management challenges. Grizzly bears are large charismatic mammals at the top of the food chain. They are powerful predators that symbolize wildness and represent the natural history of the American West (White et al., 2017). The area in which grizzly

---

<sup>1</sup>The natural capital literature, e.g., Fenichel and Abbott (2014), focuses on these values.

<sup>2</sup>The GYE is made up of federal, state, private, and tribal lands in three states: Wyoming, Montana, and Idaho. Estimates of the size of the GYE range from 18,750 to 34,375 square miles. The GYE encompasses two national parks, three national wildlife refuges, and parts of five national forests. There are myriad hydrothermal features in the area, including geysers and hot springs. The region also contains the greatest concentration of wildlife in the lower-48 states (National Park Service, 2020).

bears have recovered includes Yellowstone National Park (YNP) and Grand Teton National Park (GTNP), two of the most visited national parks in the United States.<sup>3</sup> The region is also home to roughly half a million locals. Grizzly bears are a prime example of a multi-use species in that they affect both GYE visitors and locals in myriad different ways. Grizzly bears can simultaneously inspire wonder, awe, and terror, and they are capable of causing considerable damage to economic activity (White et al., 2017).

On one hand, people clearly value grizzly bears. They are the animals that evoke the most emotion in YNP visitors (White et al., 2017). It has been well documented that people value grizzly bear viewing. Richardson et al. (2014) asked respondents the top five animals they would like to see on their trips to YNP from out of a list of 21 different mammals and birds. Bears were the animal that ranked the highest, with 81% of respondents listing bears in the top five. Skibins et al. (2012) found that grizzly bears were the species from the “big five” in Denali National Park (grizzly bears, wolves, caribou, Dall sheep, and moose) that had the most influence in terms of predicting wildlife viewing satisfaction. Anderson et al. (2010) found that grizzly bears were the species of primary interest for most Denali National Park visitors. Miller et al. (1998) estimated Alaskan voters’ total social benefit from trips for which the primary purpose was wildlife viewing. They found that voters had higher social benefit from trips during which bears were seen than from trips during which any other species were seen.

On the other hand, grizzly bears in the GYE pose risks to local economic activity, local individuals, and GYE visitors. As the grizzly bear population has recovered, there has been an increase in the number of grizzly bear-human conflicts, which include incidents such as livestock depredations, property damages, and human injuries. Such conflicts may be eroding public support for conservation (Gunther et al., 2004). Conservation is typically more challenging when the recovering species is a predator that causes economic damages, as seen for wolves (Skonhøft, 2006) and tigers (Zabel et al., 2011). When grizzly bear-human conflicts are severe enough, agencies may respond by removing offending grizzly bears. This gives wildlife agencies a control action through which they can indirectly influence the size of the grizzly bear population. ESA protections currently prevent other, more direct, control actions. These characteristics make grizzly bears particularly interesting, especially because they already incur high rates of human-caused mortality even while being federally protected.

Section 2 contains background information about grizzly bear management in the GYE. Grizzly bear recovery has coincided with an increase in both non-harvest human-caused grizzly bear mortalities and grizzly bear-human conflicts, which have in turn increased total grizzly bear program expenditures by a state wildlife agency. Data indicates that mortalities, conflicts, and expenditures are stock-dependent. In Section 3, a bioeconomic model is constructed, in which a representative state agency maximizes net social benefits by managing the mortality of grizzly bears given institutional limitations. The agency takes into account that population size influences both benefits and damages. In Section 4, all functions in the model are parameterized using existing grizzly bear data. Negative binomial (NB) and ordinary least squares (OLS) regressions are used to parameterize the stock-dependent relationships from Section 2. Important variables are controlled for, including expansion of grizzly bear occupied range on private lands, climate, and food resource

---

<sup>3</sup>Since 2015, over four million visitors have visited YNP every year while over three million visitors have visited GTNP every year.

quality. Section 5 includes the results of a base case that acts as the best representation of reality. A recovery path is constructed that tracks the recovery of grizzly bears and shows how a switch from reactive to active management would increase the natural capital value of the population. The results indicate that preventing wildlife agencies from undertaking active management may be having unintended effects on social net benefits, the natural capital value of a grizzly bear in the wild, and total grizzly bear mortality. Sensitivity analysis shows that the results are sensitive to key human and ecological parameters. Section 6 concludes with a discussion of management implications. The take-home message is that when constructing efficient conservation policies directed at charismatic multi-use megafauna, it may be optimal for society to endure an interval of time with negative natural capital values by creating a buffer of the species. Over these intervals, it is optimal to let the population recover enough to withstand future management changes that switch the resource from a liability to a long-run asset.

## 2. Management Background

Grizzly bears in the lower-48 states became protected by law in 1975 when they were listed as a threatened species under the ESA. Around that time, biologists estimated that there were between 136 and 312 grizzly bears in the GYE. Since then, conservation efforts have led to a significant increase in the number of grizzly bears in the GYE. Today, biologists estimate that there are over 700 grizzly bears in the Demographic Monitoring Area (DMA), which is a monitored region of the GYE. The U.S. Fish and Wildlife Service (USFWS) currently has three demographic recovery criteria in place, the third of which bans discretionary mortality (except if necessary for human safety) if the population ever falls below 600 grizzly bears in the DMA ([U.S. Fish and Wildlife Service, 2016](#)).<sup>4</sup> Data from the Interagency Grizzly Bear Study Team shows that the grizzly bear population in the GYE has been increasing over time (Figure 1a). The population reached 600 grizzly bears between 2001 and 2002. Since then, the population has consistently maintained a high level - it has never fallen below the threshold of 600 grizzly bears from the third USFWS recovery criterion. GYE grizzly bears are widely considered a conservation success story, and the USFWS has twice used evidence of recovery to justify removing GYE grizzly bears from the ESA: once in 2007, once in 2017. Both times, ensuing litigation led to grizzly bears being relisted.<sup>5</sup> Prior to grizzly bears being relisted in 2018, Wyoming and Idaho had planned to enact limited grizzly bear hunting seasons.

Recovery has led to a corresponding increase in the number of grizzly bear-human conflicts and human-caused grizzly bear mortalities (Figure 1b). Grizzly bear-human conflicts are delineated into several categories (Table 1). Livestock depredations occur when a bear injures or kills livestock. “Unnatural foods” conflicts occur when a bear obtains human foods or other edible human-related attractants, such as livestock feed. Property damages occur when a bear damages personal property, including homes, fences, orchards, beehives, and gardens. Human injuries occur when a bear injures

---

<sup>4</sup>The first criterion calls for a minimum population size of 500 grizzly bears and at least 48 female grizzly bears with cubs-of-the-year in the DMA. The second criterion calls for at least 16 of 18 bear management units to be occupied by females with young, with no two adjacent units unoccupied.

<sup>5</sup>In 2009, the first delisting attempt was blocked over concerns about the decline of whitebark pines at that time (whitebark pine nuts are an important source of food for GYE grizzly bears). In 2018, the second delisting attempt was blocked over concerns of a lack of genetic diversity between GYE grizzly bears and other grizzly bear populations in the lower-48 states.

or kills a person. Table 1 shows the average annual count of each type of conflict from 2010 to 2019. There have been an average of roughly 266 conflicts per year over the past decade. The average annual counts are used to calculate the average annual share of each type of conflict.

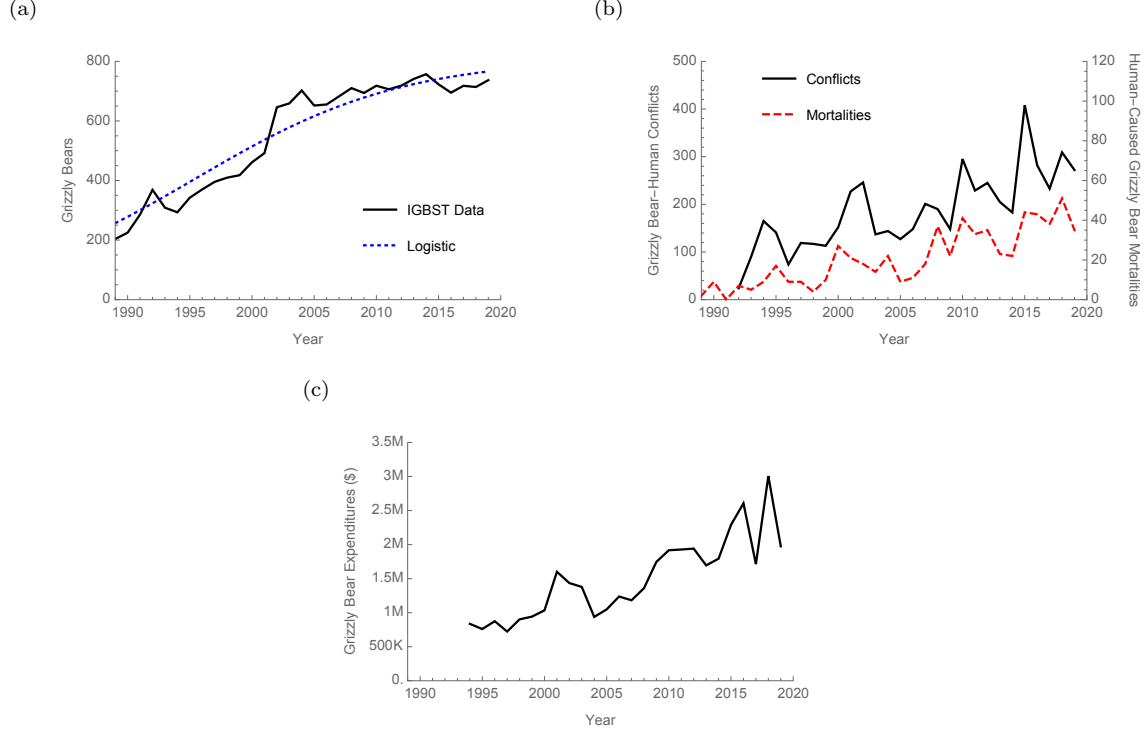


Figure 1: (a) Grizzly bear population in the DMA (data provided by the Interagency Grizzly Bear Study Team). (b) Grizzly bear-human conflicts and human-caused grizzly bear mortalities in the GYE (data provided by the Interagency Grizzly Bear Study Team). (c) Total annual grizzly bear program expenditures (data provided by the Wyoming Game and Fish Department).

As might be expected, conflicts and mortalities track each other closely (Gunther et al., 2004). There are three major categories of documented grizzly bear mortalities: unknown, natural, and human-caused. Harvest has been illegal given the federal protections of grizzly bears, so human-caused mortalities have been driven by non-harvest mortalities. Human-caused mortalities have included the following types of mortalities: management removals (e.g., when an agency culls a conflict grizzly bear), self-defense (e.g., when a hunter kills a charging grizzly bear), hunter-mistake kills (e.g., when a black bear hunter kills a grizzly bear), vehicular (e.g., when a grizzly bear is hit with a vehicle), and others (e.g., poaching, research-related, unknown but human-caused, etc.).<sup>6</sup> Table 2 shows the average annual count of each type of human-caused mortality from 2009 to 2018.<sup>7</sup> There were an average of roughly 36 human-caused grizzly bear mortalities per year in that 10-year

<sup>6</sup>A full breakdown of total documented mortality is shown in Supplementary Appendix A.

<sup>7</sup>Human-caused grizzly bear mortality data for each year from 2009 to 2018 was obtained from the Interagency Grizzly Bear Study Team’s annual reports for those years (e.g., van Manen et al. (2019)). Known and probable

time-period. The average annual counts are used to calculate the average annual share of each type of human-caused mortality.

Conflict Type	Average Annual Count	Average Annual Share
Livestock depredations	157	59%
Unnatural foods	61	23%
Property damages	41	15%
Human injuries	7	2%
<b>Total annual</b>	<b>266</b>	<b>100%</b>

Table 1: Types of grizzly bear-human conflicts in the GYE (data provided by the Interagency Grizzly Bear Study Team). Average annual counts are calculated over the past decade (2010-2019). Average annual shares are calculated using the counts. The shares do not add up to exactly 100% due to rounding.

Mortality Type	Average Annual Count	Average Annual Share
Management	19	54%
Self-Defense	10	28%
Vehicular	3	8%
Hunter-Mistake	2	4%
Other	2	6%
<b>Total annual</b>	<b>36</b>	<b>100%</b>

Table 2: Types of human-caused grizzly bear mortalities in the GYE (Interagency Grizzly Bear Study Team, Annual Reports). Average annual counts are calculated over the years from 2009 to 2018. Average annual shares are calculated using the counts. For hunter-mistake and other human-caused mortalities, average annual counts are identical but the average annual shares differ; this occurs due to the rounding of average annual counts to the nearest whole number.

Under the current strategy of reactive management, wildlife agencies (both state and federal) respond to grizzly bear-human conflicts. Possible actions include removing or securing human foods and other edible human-related attractants, capturing and relocating bears involved in conflicts, and, in severe cases, killing an offending bear to eliminate the risk of future conflicts. Such actions are costly. Total annual grizzly bear program expenditures, which include both direct expenditures and overhead, have been increasing over time (Figure 1c).

Annual counts of conflicts (Figure 2a) and annual counts of human-caused grizzly bear mortalities (Figure 2b) in relation to annual grizzly bear population estimates indicate that conflicts and mortalities are stock-dependent: higher grizzly bear population sizes correspond to higher numbers of conflicts and grizzly bear mortalities. Figure 2c indicates that expenditures are higher in years with more conflicts.<sup>8</sup>

---

mortalities were classified by type given the description of loss in the “Documented known and probable grizzly bear mortalities” tables of the annual reports.

<sup>8</sup>In Section 4, the lines of best fit shown in Figure 2 are constructed. The counts of conflicts and mortalities are estimated as functions of stock size using NB regressions with the following control variables: expansion of grizzly

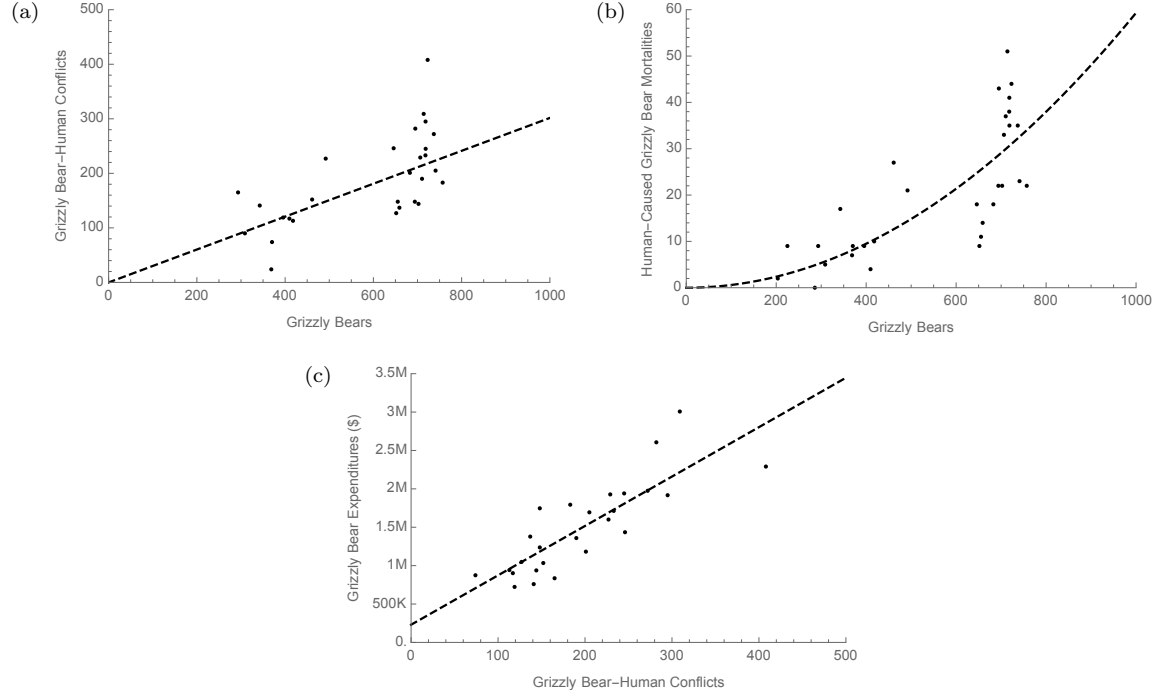


Figure 2: (a) Annual counts of grizzly bear-human conflicts over annual estimates of the grizzly bear population. (b) Annual counts of human-caused grizzly bear mortalities over annual estimates of the grizzly bear population. (c) Annual grizzly bear program expenditures over annual counts of grizzly bear-human conflicts.

### 3. Bioeconomic Model

To identify the circumstances that warrant active management, a representative wildlife agency chooses whether or not to implement an optimal harvest program, and if so, how to set the size of the program. The agency chooses the number of grizzly bears harvested,  $h$ , in each time-period,  $t$ , so as to maximize discounted social net benefits from grizzly bears,  $SNB$ . Benefits in each time-period,  $B(t)$ , are modeled as a function of both the grizzly bear stock in each time-period,  $G(t)$ , and the optimal harvest in each time-period,  $h(t)$ , while damages in each time-period,  $D(t)$ , are modeled as a function of the grizzly bear stock in each time-period. Suppressing time notation, the expression is

$$SNB = \max_h \int_0^\infty (B(G, h) - D(G))e^{-\delta t} dt, \quad (1)$$

in which  $\delta$  is the discount rate. When choosing a harvest program to maximize Expression (1), the agency accounts for constraints on the initial conditions ( $G(0) = G_0$ ,  $G \geq 0$ ), harvests ( $0 \leq h \leq h^{\max}$ ), and the ecological dynamics (as developed in the next section). It is assumed that a maximum limit on harvests,  $h^{\max}$ , would be exogenously set such that there could be no danger

---

bear occupied range on private lands, climate, and quality of whitebark pine. Total grizzly bear expenditures are estimated as a function of the count of conflicts using OLS regression.

of the grizzly bear population failing any of the three demographic recovery criteria used by the USFWS (U.S. Fish and Wildlife Service, 2016).<sup>9</sup>

### 3.1. Ecology

The *in situ* stock of grizzly bears grows according to

$$\dot{G} = L(G) - M(G) - h. \quad (2)$$

$L(G)$  represents density-dependent logistic growth of the stock ( $L(0) = L(K) = 0$ ,  $L'(0) = r$ ,  $L''(G) < 0$ ), in which  $K$  is the “logistic limit” and  $r$  is the intrinsic rate of growth:

$$L(G) = rG - \frac{r}{K}G^2. \quad (3)$$

$M(G)$  represents non-harvest human-caused grizzly bear mortality. Adding non-harvest human-caused mortality as a separable term in the dynamic constraint (rather than lumping mortality into the density dependent growth equation as is standard) allows for additional richness in the analysis. This is especially important when considering a multi-use species such as grizzly bears, which incur high non-harvest human-caused mortality. Mortality is modeled as a convex function of the grizzly bear population, such that  $M(0) = 0$ ,  $M'(G) > 0$ , and  $M''(G) > 0$ .<sup>10</sup> The specification is shown in Figure 2b. Given the presence of the mortality equation in Equation (2), the logistic limit is not the same as the “true” carrying capacity,  $K_T$ , which occurs at the positive population size where net growth is zero (i.e., at the population size where  $L(G) = M(G)$  given the current ban on harvest). In the remainder of the paper, the term “carrying capacity” refers to the true carrying capacity.

### 3.2. Benefits

Benefits from grizzly bears are the sum of nonuse value,  $N(G)$ , and use value (National Research Council, 2005). Use value in turn consists of consumptive use value,  $C(h)$ , and non-consumptive use value,  $S(G)$ . It is assumed that nonuse value simultaneously includes existence, bequest, and option value, as in US Fish and Wildlife Service (2000). Nonuse value is modeled as a concave function of the grizzly bear population, such that  $N(0) = 0$ ,  $N'(G) > 0$ , and  $N''(G) < 0$ . At a population of zero grizzly bears (i.e., extinction), there can be no nonuse value. As the number of grizzly bears in the wild increases, nonuse value increases at a decreasing rate. The specification reflects that extinction is less of a concern at higher population sizes and matches previous literature. For example, Loomis and White (1996) and Richardson and Loomis (2009) found that willingness-to-pay (WTP) for rare, threatened, and endangered species increases with increases in proposed percentage changes in species populations, but at a decreasing rate.

Under certain conditions, it may be socially optimal for the agency to enact a harvest program, which would generate aggregate consumptive use value,  $C(h)$ , for the private agents who harvest

---

<sup>9</sup>Given the existence of mortality limits, there is a natural cap on optimal harvest (because mortality limits would be exceeded if optimal harvest were too high). A maximum limit could also make an active management program less controversial: it is more feasible to imagine smaller harvest numbers, especially in the early years of an active management program, during which the full effects of the active management program are still being researched. When a grizzly bear hunting season was scheduled in the GYE for the fall of 2018, the maximum allowable harvest was set at 25 grizzly bears: up to 24 in Wyoming and one in Idaho. 25 is therefore used as the estimate of  $h^{\max}$ .

<sup>10</sup>The decision to use a convex specification is grounded in data and is discussed in Section 4.



a grizzly bear. It is assumed that each private agent would only be allowed to harvest one grizzly bear, meaning that the optimal number of grizzly bears harvested,  $h$ , would directly translate into the number of private agents who harvested a grizzly bear. Consumptive use value is modeled such that marginal consumptive use value is constant ( $C(0) = 0$ ,  $C'(h) = c > 0$ , and  $C''(h) = 0$ ), which makes the problem linear in the control. The equation is

$$C(h) = ch, \quad (4)$$

in which  $c$  is the consumptive use value obtained by each private agent who harvests a grizzly bear (reflecting hunting, trophy, and meat value net of hunting expenditures). The resulting linear specification reflects the realities of a harvesting program directed at a charismatic species such as grizzly bears. In the past, proposed harvest programs for grizzly bears have been highly contentious and have come under great public scrutiny. Typically, such programs are also limited by statute in their flexibility (e.g., mortality limits in the species' recovery plan). The "on/off" management decision and equilibrium principles that follow from a linear control structure provide a reasonable approximation of such realities. Further, if grizzly bear hunting seasons were established, the number of grizzly bears harvested would be low. With low harvest numbers, it seems unlikely for there to be diminishing returns from factors such as hunter congestion.<sup>11</sup>

Non-consumptive use value,  $S(G)$ , is assumed to consist entirely of direct non-consumptive use value, in the form of aggregate value from grizzly bear sightings. It is modeled as a concave function of the grizzly bear population, such that  $S(0) = 0$ ,  $S'(G) > 0$ , and  $S''(G) < 0$ . When there are no grizzly bears in the wild, no grizzly bears are seen, and there is no aggregate value from grizzly bear sightings. As the number of grizzly bears in the wild increases, non-consumptive use value increases at a decreasing rate. This matches the assumptions used by [Rondeau \(2001\)](#) for non-consumptive benefits.

### 3.3. Damages

Damages are represented by the agency's damage-related expenditures,  $E(I)$ , which are a function of the number of grizzly bear-human conflicts,  $I$ . A linear specification, shown in Figure 2c, is used, such that  $E'(I) > 0$  and  $E''(I) = 0$ . In turn, conflicts are a linear function of the grizzly bear population, such that  $I(0) = 0$ ,  $I'(G) > 0$ , and  $I''(G) = 0$ . The linear specification is shown in Figure 2a.<sup>12</sup> It reflects that there can be no grizzly bear-human conflicts when there are no grizzly bears. The implicit equation for damages is

$$D(G) = E(I(G)). \quad (5)$$

### 3.4. Socially Optimal Management

The optimization program follows from the associated current-value Hamiltonian,  $\mathcal{H}$ , given by

$$\mathcal{H} = N(G) + C(h) + S(G) - E(I(G)) + \mu [L(G) - M(G) - h], \quad (6)$$

---

<sup>11</sup>As a robustness check, the model is estimated with a nonlinear specification of  $C(h)$ . The results are included in Supplementary Appendix B.2.

<sup>12</sup>The decision to use linear specifications for both expenditures and conflicts is grounded in data and is discussed in Section 4.

in which  $\mu$  is the natural capital value of a live grizzly bear in the wild (i.e., the shadow value) at each point in time. Using the maximum principle (Conrad, 2010), the optimal harvest program follows from a simultaneous solution of three conditions:<sup>13</sup>

$$\frac{\partial \mathcal{H}}{\partial h} = C'(h) - \mu, \quad (7)$$

$$\dot{G} = L(G) - M(G) - h, \quad (8)$$

and

$$\dot{\mu} = \delta\mu - \frac{\partial \mathcal{H}}{\partial G} = \mu[\delta - L'(G) + M'(G)] - N'(G) - S'(G) + E'(I(G))I'(G). \quad (9)$$

The first condition, Condition (7), provides a rule to determine whether or not an active management program should be employed, and if so, how many grizzly bears should be harvested. The decision rule follows from a comparison of what a grizzly bear in the wild is worth,  $\mu$ , to the consumptive use value,  $C'(h) = c$ . If  $\frac{\partial \mathcal{H}}{\partial h} < 0$  (“Case i”), then harvesting grizzly bears always decreases the value of the Hamiltonian, and there ought to be no active management program (i.e., it is optimal to harvest zero grizzly bears). If the value of a grizzly bear in the wild just equals the consumptive use value ( $\mu = c$ ), then  $\frac{\partial \mathcal{H}}{\partial h} = 0$  (“Case ii”). In this case,  $h$  should be set at the singular value,  $h^*$ , which falls between zero and  $h^{\max}$ . If  $\frac{\partial \mathcal{H}}{\partial h} > 0$  (“Case iii”), then harvesting grizzly bears always increases the value of the Hamiltonian, and an active management program ought to be established. In this case, optimal harvest should be set at the maximum limit,  $h^{\max}$ . The three cases, which are discussed in more detail in the following subsections, can be summarized by the control rule

$$h(t) = \begin{cases} h^* = 0 & \text{if } \mu(t) > c, \\ h^* \in [0, h^{\max}] & \text{if } \mu(t) = c, \\ h^* = h^{\max} & \text{if } \mu(t) < c. \end{cases} \quad (10)$$

The second condition, Condition (8), is the dynamic constraint that dictates the evolution of the stock over time. The third condition, Condition (9), is a differential equation that governs the evolution of the natural capital value of grizzly bears in the wild. Without an active management program, the condition simply tracks the relative contributions of the stock throughout the bioeconomic system. If an active management program is optimal, then the condition must be optimally satisfied by adjustments in the stock (e.g., using optimal harvest) such that there can be no gains in social net benefits from reallocating harvest across time. Rearranging the condition yields the “golden rule” (Clark and Munro, 1975; Conrad and Clark, 1987):

$$\delta = L'(G) - M'(G) + \frac{\dot{\mu}}{\mu} + \frac{1}{\mu} \left[ N'(G) + S'(G) - E'(I(G))I'(G) \right]. \quad (11)$$

---

<sup>13</sup>As in most multi-use problems, the shadow value can be negative, which makes the problem non-convex. As such, the standard sufficiency conditions (i.e., the Mangasarian sufficiency conditions) can be violated. Furthermore, Spence and Starrett (1975)’s most rapid approach path optimality requirements are specified for convex problems. Given the challenges associated with analytically demonstrating sufficiency in our numerical application, the discounted social net benefits of alternative solutions are evaluated and compared in Section 5.

The golden rule requires a balance of the rate of return from the exploitation of grizzly bears with the rate of return from conservation. Exploitation generates dollars that could be invested in an alternative investment and earn  $\delta$ , society's discount rate. The discount rate represents the opportunity cost of investing in the grizzly bear stock. Conservation is an investment in the grizzly bear stock, which produces a rate of return given by the right-hand side (RHS) of Equation (11). The first two terms on the RHS,  $L'(G) - M'(G)$ , are the marginal net growth rate of the stock. The second term,  $\frac{\dot{\mu}}{\mu}$ , represents the capital gains from holding an additional unit of stock (or capital losses, if damages overshadow benefits at that particular stock level). The capital gains capture the change in the value of the grizzly bear stock over time. The remaining terms on the RHS are the marginal stock effects, which can either be marginal stock benefits (e.g., nonuse value, consumptive use value, and sighting value) or marginal stock damages.

#### 3.4.1. Case i - No Harvest

If the value of a grizzly bear in the wild exceeds the consumptive use value ( $\mu(t) > c$ ), it is optimal to have no harvest program. In this case, Condition (8), with  $h = 0$ , provides a differential equation for the grizzly bear population. Condition (9) provides a differential equation for the shadow value over time, which in turn depends on the stock over time. The magnitude of the grizzly bear population in the steady state,  $G_{h^*=0}$ , depends on growth and mortality parameters and determines the associated steady state grizzly bear shadow value,  $\mu_{h^*=0}$ .

#### 3.4.2. Case ii - Singular Harvest

The threshold case occurs when the value of a grizzly bear in the wild just matches the consumptive use value ( $\mu(t) = c$ ). In this case, because  $c$  is a constant, the natural capital value of a grizzly bear in the wild is also constant over time ( $\dot{\mu} = 0$ ). Substitutions and manipulation lead to a steady state golden rule:

$$\delta = L'(G) - M'(G) + \frac{1}{c} \left[ N'(G) + S'(G) - E'(I(G))I'(G) \right]. \quad (12)$$

The intuition of the rule is simple: grizzly bears ought to be managed as natural assets that are competitive with other assets in the economy. They are competitive when the rate of return from holding an additional grizzly bear in the wild (conserving) just equals the rate of return the agency could earn from harvesting (exploiting) the grizzly bear and investing the proceeds. Equation (12) implicitly defines the singular solution,  $G^*$ , as a function of exogenous parameters.

#### 3.4.3. Case iii - Maximum Harvest

If the consumptive use value exceeds the value of a grizzly bear in the wild ( $c > \mu(t)$ ), then it is optimal for there to be an active management program with harvest set at the maximum limit,  $h^{\max}$ . In this case, Condition (8), with  $h = h^{\max}$ , provides a differential equation for the grizzly bear population. Condition (9), evaluated as the grizzly bear stock changes over time, is the differential equation of the grizzly bear natural capital value over time. If  $h^{\max}$  were to result in a steady state, the resulting grizzly bear stock,  $G_{h^*=h^{\max}}$ , would depend on growth parameters and mortality parameters, as well as  $h^{\max}$ . The stock would in turn influence the steady state natural capital value,  $\mu_{h^*=h^{\max}}(G_{h^*=h^{\max}})$ .

The complete solution becomes

$$h(t) = \begin{cases} h^* = 0 & \text{if } G(t) < G^*, \\ h^* \in [0, h^{\max}] & \text{if } G(t) = G^*, \\ h^* = h^{\max} & \text{if } G(t) > G^*, \end{cases} \quad (13)$$

which is an example of a most rapid approach path (Spence and Starrett, 1975).

## 4. Numerical Specification

The numerical application of the model to grizzly bear management in the GYE requires an explicit specification of each functional form shown in Equation (6). The following subsections document each specification employed.

### 4.1. Ecology

#### 4.1.1. Mortality

Non-harvest human-caused grizzly bear mortality,  $M(G)$ , can reasonably be expected to be a function of the number of grizzly bears. In addition, as grizzly bear occupied range in the GYE has expanded, grizzly bears have reached more human-dominated landscapes. This has led to a significant increase in the amount of private land contained within grizzly bear occupied range. In 1990, a little over 600 km<sup>2</sup> of grizzly bear occupied range consisted of private land. By 2018, that number had grown to nearly 12,000 km<sup>2</sup> (Bjornlie and Haroldson, 2019). As grizzly bear range expands to more private lands, the potential for grizzly bear-human conflicts increases. Expansion of grizzly bear range on private lands is thus included as a control when estimating mortality. The expansion of grizzly bear range on private land in a given year (PVT) is obtained by subtracting the total area of privately-owned grizzly bear range in that year from the total area of privately-owned grizzly bear range in the previous year (Bjornlie and Haroldson, 2019).

In GTNP, it has been observed that abundant berry crops may entice bears into roadside meadows, where they are more visible to visitors (Gunther et al., 2018). It is plausible that the number of grizzly bear-human conflicts is higher in years during which bears are more visible. Abundance of berries, as well as abundance of myriad other food sources, likely depends on the climate in a given year. The Palmer Drought Severity Index (PDSI) is a convenient aggregation of weather conditions, so PDSI is included as a control when estimating mortality.<sup>14</sup>

Haroldson and Gunther (2013) found that during years with good whitebark pine cone production, there are fewer “bear-jams” (i.e., traffic stoppages due to bear viewing) in YNP. This occurs because whitebark pine feeding sites are typically located away from roads at higher elevations. Gunther et al. (2018) show the number of bear-jams over time in YNP and GTNP. They conjecture that abundance of whitebark pine seed production causes bears to transition from roadside meadows to forested areas, where it is harder for visitors to spot them. It is plausible that the number of grizzly bear-human conflicts is lower in years during which there are less bear-jams. For this

---

<sup>14</sup>Raw PDSI data was obtained from NOAA’s National Climatic Data Center. Average annual PDSI for growing season months (April through July) was calculated across the NOAA climate divisions that make up the GYE.

reason, whitebark pine quality, in terms of mortality-adjusted median whitebark pine cones per tree (WBP), is included as a control when estimating mortality (Haroldson, 2019).<sup>15</sup>

In sum, the estimation equation used is

$$M(G) = M(G, PVT, PDSI, WBP), \quad (14)$$

which is tested using a count model.<sup>16</sup> The NB is employed to account for overdispersion in the mortality data.<sup>17</sup> The general NB regression equation is

$$M(G) = e^{\alpha_0 + \alpha_1 \ln(G) + \alpha_2 PVT + \alpha_3 PDSI + \alpha_4 WBP + \varepsilon}, \quad (15)$$

in which  $\ln(G)$  is the exposure variable. The exposure variable accounts for the correlation between grizzly bear population size and mortality. Three variants of the mortality NB model are tested, the results of which are shown in Table 3.

In MNB (1), the  $\alpha_1$  coefficient is constrained to equal one. When the  $\alpha_1$  coefficient is equal to one, a doubling in the number of grizzly bears leads to a doubling of mortality. In the regression equation, this occurs because the exponential of the log of  $G$  simplifies to  $G$ . In other words, MNB (1) captures a linear relationship between grizzly bear population and non-harvest human-caused mortality. The coefficient on PVT is positive and significant, which implies that expansion of private range is correlated with an increase in mortality. The coefficient on PDSI is insignificant, which indicates that the climate index is not correlated with mortality. The coefficient on WBP is negative and significant, which implies that mortality is lower in years with higher whitebark pine quality. In MNB (2), the  $\alpha_1$  coefficient is constrained to equal two. In the regression equation, the exponential of two times the log of  $G$  simplifies to  $G^2$ . In other words, this variant builds in a convex relationship between population size and mortality. The coefficient on WBP is negative and (weakly) significant. No other regressors are found to be significant in this specification. In MNB (3), the  $\alpha_1$  coefficient is unconstrained. It is significant and greater than one, which implies that doubling the grizzly bear population more than doubles the rate of non-harvest human-caused mortality. This provides evidence that there is a nonlinear relationship between population size and mortality. The coefficient on WBP is negative and significant. No other controls are found to be significant, but all coefficients are similar in magnitude to those from MNB (2).

The squared coefficient of correlation between fitted and actual values of the dependent variable, which is shown in the  $Corr^2$  row of Table 3, measures the fit of the three variants (Cameron and Trivedi, 2009). The squared correlation is highest for MNB (3), the unconstrained nonlinear model, and lowest for MNB (1), the linear model. The coefficient estimates from MNB (2), as well as historic averages of private land expansion, PDSI, and whitebark pine quality,<sup>18</sup> are used to

---

<sup>15</sup>The mortality adjustment accounts for the proportion of whitebark pine trees monitored since 2002 that are still alive.

<sup>16</sup>The research objective is not to determine causation of non-harvest human-caused grizzly bear mortality. Instead, the objective is to incorporate non-harvest human-caused grizzly bear mortality as accurately as possible in the dynamic constraint of a bioeconomic model of grizzly bear management. The metric of success is being able to use the parameterized regression equation to accurately predict historical mortality given historical data on private land expansion, PDSI, and whitebark pine quality.

<sup>17</sup>In Table 3, the  $\alpha$  parameter is significantly different from zero for all three variants of the NB model, which indicates overdispersion. The NB is thus preferred over the alternative Poisson.

<sup>18</sup>Average values over the entire time-period are  $\overline{PVT} = 392.48$ ,  $\overline{PDSI} = -0.11$ , and  $\overline{WBP} = 5.80$ .

parameterize and reduce Equation (15):

$$\begin{aligned} M(G) &= \gamma_M(\overline{PVT}, \overline{PDSI}, \overline{WBP})G^2 \\ &= 0.00006 \cdot G^2. \end{aligned} \tag{16}$$

The full derivation is included in Supplementary Appendix C.1. As shown in the next subsection, MNB (2) is chosen for mathematical convenience because it is possible to explicitly solve the net growth differential equation when the population is squared.<sup>19</sup>

	MNB (1)	MNB (2)	MNB (3)
lnG	1 (.)	2 (.)	1.61*** (5.33)
PVT	0.00050** (2.19)	0.00024 (0.88)	0.00034 (1.51)
PDSI	0.0093 (0.28)	0.025 (0.60)	0.019 (0.51)
WBP	-0.035*** (-3.33)	-0.025* (-1.71)	-0.029** (-2.21)
Constant	-3.38*** (-21.59)	-9.68*** (-51.75)	-7.20*** (-3.67)
$\alpha$	0.154	0.141	0.127
Rb. Std. Err.	0.064	0.050	0.049
$N$	28	28	28
Corr <sup>2</sup>	0.596	0.612	0.618

*z*-scores in parentheses

\*  $p < .10$ , \*\*  $p < .05$ , \*\*\*  $p < .01$

Table 3: Mortality negative binomial regressions

<sup>19</sup>The existence of nonlinearity in the MNB (3) regression provides justification for constraining the coefficient on  $\ln(G)$  to two in the MNB (2) regression.

#### 4.1.2. Modified Logistic Growth

Combining the standard logistic growth equation,  $L(G(t))$ , with the mortality equation,  $M(G(t))$ , yields a modified net growth equation,  $F(G(t))$ :

$$\begin{aligned} F(G(t)) &= L(G(t)) - M(G(t)) \\ &= rG(t) \left(1 - \frac{G(t)}{K}\right) - \gamma_M G(t)^2. \end{aligned} \quad (17)$$

Solving the differential equation and expanding  $\gamma_M$  yields

$$G(t) = \frac{e^{rt} G_0 K r}{\gamma_M(PVT(t), PDSI(t), WBP(t))(e^{rt} - 1)G_0 K + ((e^{rt} - 1)G_0 + K)r}. \quad (18)$$

Equation (18) defines the grizzly bear population at each instant in time as a function of time, the initial population size, the intrinsic rate of growth, the logistic limit, private land expansion, the climate index, and whitebark pine quality in that time period. The equation is used to numerically fit  $r$  and  $K$  by minimizing the sum of squared errors between observed and predicted grizzly bear population estimates. The process is explained in Supplementary Appendix C.2. The parameter values for  $r$ ,  $K$ , and  $K_T$  are shown in Table 4. The population never actually reaches the logistic limit because of the self-limiting nature of human-caused mortality. This is apparent in Figure 3, which shows the logistic growth and mortality curves in the top plot. At the point where the curves intersect, net growth (in the bottom plot) is equal to zero. Net growth has the standard logistic appearance.

Parameter	Value
$r$	0.11
$K$	1,646
$K_T$	876

Table 4: Ecological parameters.

## 4.2. Benefits

### 4.2.1. Nonuse Value

The concave functional form used for nonuse value is

$$N(G) = n_A R(G), \quad (19)$$

in which  $n_A$  is aggregate nonuse value from grizzly bears and  $R(G)$  is a function that scales aggregate nonuse value depending on the population size.<sup>20</sup>

---

<sup>20</sup>Aggregate nonuse value is defined as the nonuse value from all domestic households. This results in a conservative estimate of nonuse value.

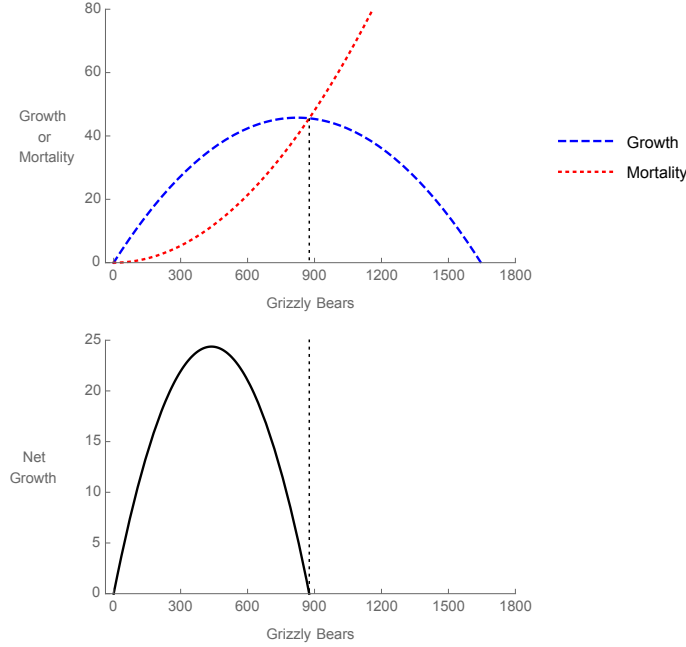


Figure 3: Growth, mortality, and net growth curves.

$n_A$  can be decomposed following [US Fish and Wildlife Service \(2000\)](#):

$$n_A = r(fu - (1 - f)u)\delta z, \quad (20)$$

in which  $r$  is mean lifetime nonuse value (in terms of WTP by households that support grizzly bear reintroduction),  $f$  is the proportion of households in favor of grizzly bear reintroduction,  $(1 - f)$  is the proportion of households not in favor of reintroduction,  $u$  is the number of households,  $\delta$  is the discount rate, and  $z$  is a scaler. [US Fish and Wildlife Service \(2000\)](#) estimated mean household lifetime nonuse value for the reintroduction of grizzly bears in the Bitterroot Ecosystem (BE) among local, regional, and national sample residents. They assumed that the mean lifetime nonuse value for grizzly bear recovery (by households in favor of reintroduction) was identical to mean lifetime value for the continued absence of grizzly bears (by households not in favor of reintroduction). Equation (20) therefore calculates the nonuse value of people who favor reintroduction net of the nonuse value of people who do not.

The mean lifetime nonuse value for grizzly bears in the BE, as reported by [US Fish and Wildlife Service \(2000\)](#) for national sample respondents, is used as the best guess of the mean lifetime nonuse value for grizzly bears in the GYE:  $r = \$65.45$ .<sup>21</sup> The BE covers portions of Idaho and Montana while the GYE covers portions of Idaho, Montana, and Wyoming. In that sense, the two ecosystems are close geographically. In the national sample, [US Fish and Wildlife Service](#)

<sup>21</sup>Data on the historical consumer price index from the U.S. Bureau of Labor Statistics was used to convert their estimate from 1996 dollars to 2019 dollars.



(2000) estimated that 66,671,516 households supported grizzly bear reintroduction in the BE while 7,439,739 households opposed reintroduction. The two estimates are used to obtain a predicted ratio of domestic households who support grizzly bear existence in the GYE,  $f$ :

$$f = \frac{\text{Population supporting reintroduction}}{\text{Total population}} = \frac{66,671,516}{66,671,516 + 7,439,739} = 0.90. \quad (21)$$

For  $u$ , the total number of US households in 2019 is used: 128.58 million ([Statista.com](#)). The discount rate reflects a “medium future subperiod” of 26 to 75 years hence:  $\delta = 0.02$  ([Weitzman, 2001](#)). The scalar from [US Fish and Wildlife Service \(2000\)](#) is used:  $z = 0.286$ . Multiplying all terms yields

$$n_A = \$65.45 \cdot (0.90 \cdot 128,580,000 - (1 - 0.90) \cdot 128,580,000) \cdot 0.02 \cdot 0.286 = \$38,474,822. \quad (22)$$

It is assumed that full nonuse value is obtained at the population size at which the species has fully recovered. [US Fish and Wildlife Service \(2000\)](#) estimated nonuse value for an alternative that entailed a recovery goal of about 280 grizzly bears in the BE. For an alternative with a higher recovery goal (about 400 grizzly bears in the BE), they estimated identical nonuse values. This supports the idea that nonuse value stops increasing after a threshold population size is reached. The threshold is defined as the carrying capacity,  $K_T$ . Once the population reaches the carrying capacity, further increases in population do not increase people’s nonuse value for the species. The scaling function,  $R(G)$ , builds in these assumptions. A concave specification is used, such that  $R'(G) > 0$  for  $G < K_T$  and  $R''(G) < 0$  for  $G < K_T$ . A Gompertz function, as used by [Rondeau \(2001\)](#) for non-consumptive benefits, is parameterized using two pieces of information: there is zero nonuse value if the grizzly bear population is zero ( $R(0) = 0$ ), and full nonuse value is realized at carrying capacity ( $R(K_T) = 1$ ). The function is

$$R(G) = (1 - e^{-qG}), \quad (23)$$

in which parameter  $q$  dictates the size of  $G$  needed to attain the maximum value of the function.  $q$  is calibrated such that the function is exactly equal to one at carrying capacity:

$$q = 0.03744. \quad (24)$$

#### 4.2.2. Consumptive Use Value

There have been no grizzly bear hunts in the GYE since federal protections were established. However, brown bear hunting has been legal in Alaska. [Miller et al. \(1998\)](#) found that the net economic value of a hunting trip in which a brown bear is targeted is \$390.43 for Alaskan resident hunters and \$1,137.50 for non-resident hunters (as converted from 1991 dollars to 2019 dollars). For the proposed GYE grizzly bear hunting season in the fall of 2018, residents received three quarters of the available tags while nonresidents received the remaining quarter. This information is used to create a weighted consumptive use value estimate for grizzly bear harvests in the GYE:

$$c = 0.75 \cdot \$390.43 + 0.25 \cdot \$1,137.50 = \$577.20. \quad (25)$$

#### 4.2.3. Non-Consumptive Use Value

Aggregate non-consumptive use value is a concave function found as the product of a national park visitor’s average WTP for a grizzly bear sighting,  $s$ , the percent of visitors who see a grizzly bear,

$P(G)$ , and the number of visitors who value seeing a grizzly bear and actively view a grizzly bear if they see one,  $v$ :

$$S(G) = sP(G)v. \quad (26)$$

Richardson et al. (2017) found that visitors have a WTP of \$287 (in 2014 dollars) per day of bear viewing in the Brooks Camp area of Katmai National Park and Preserve. For the sake of comparison, they list that Miller et al. (1998) found a WTP of \$113 per day of bear viewing in Alaska and that Clayton and Mendelsohn (1993) found WTP values of \$165 and \$200 per day of bear viewing at McNeil River (all in 2014 dollars). The average of the four values is \$206.54 (converted to 2019 dollars), which is used as the estimate of WTP per day of bear viewing in the national parks of the GYE.<sup>22</sup> The average WTP value is further adjusted to account for the length of time that YNP visitors spend viewing grizzly bears. Given the multiple possible reasons for visiting the national parks in the GYE (e.g., viewing thermal features, viewing natural scenery, viewing other wildlife species, etc.), it is unlikely that a typical visitor would devote an entire day in the GYE national parks solely to bear viewing. Haroldson and Gunther (2013) found the median duration of a bear-jam in YNP to be 45 minutes. Their estimate is used as the best guess of the duration of time spent viewing a bear by someone who sees a bear and actively views the bear in the national parks of the GYE. 45 minutes is 3.125% of the total time in a 24-hour day; the percent is multiplied by the average WTP per day of bear viewing from the literature to obtain the average WTP per bear sighting in the GYE:

$$s = \$6.45. \quad (27)$$

Richardson et al. (2014, 2015) found that trip visitors reported a median of one bear seen per trip, so  $s$  is not scaled up to account for multiple possible sightings.

The percent of visitors who see a grizzly bear,  $P(G)$ , is modeled as a concave function of the number of grizzly bears in the wild, such that  $P(0) = 0$ ,  $P'(G) > 0$ , and  $P''(G) < 0$ .<sup>23</sup> Haroldson and Gunther (2013) present evidence that more bears are seen in YNP when bear population numbers are higher. Rondeau (2001) uses a Gompertz functional form for non-consumptive benefits, which include wildlife viewing. A Gompertz function is used to estimate the probability of visitors who see a grizzly bear:

$$P(G) = p_1(1 - e^{-p_2G}), \quad (28)$$

---

<sup>22</sup>Richardson et al. (2017) explain that their WTP estimate may be higher than those of the other authors because there is a known premium associated with activities in national parks. For that reason, it is possible that WTP per day of bear viewing in the national parks of the GYE may be closer to the higher estimates. On the other hand, the aforementioned studies provide estimates of WTP for viewing bears in Alaska. Visitors to the Brooks Camp area of Katmai National Park and Preserve often travel to the remote destination between June and September primarily to watch bears feed on salmon in the Brooks River; there are bear viewing platforms along the river where visitors have great views of the spectacle (Richardson et al., 2017). The bear-viewing situation in the national parks of the GYE is quite different. First, bears in the GYE do not feed on salmon. Second, there are no structures designed specifically for bear viewing in the national parks of the GYE. Third, it is common for visitors to observe bears from the roads (i.e., from easily accessible locations). Fourth, a significant proportion of visitors might never see a grizzly bear during their trip. All of these factors could cause WTP per day of bear viewing in the national parks of the GYE to be closer to the lower estimates. Given the opposing arguments for whether WTP per day of bear viewing lies on the low or the high end, it is natural to use the average WTP value.

<sup>23</sup>There is not enough existing data to confidently determine whether the relationship is linear, concave, or convex. However, using a concave specification here later translates into a concave specification of aggregate non-consumptive use value (Supplementary Appendix Figure C.3).

in which  $p_1$  and  $p_2$  are parameters.  $p_1$  is the maximum possible value of the function, taken to be the probability of one.

$p_2$  is parameterized using three data points. For the first data point, it is straightforward to assume that the percent of visitors who see a grizzly bear is zero when the grizzly bear population is zero. The remaining two data points are obtained from the literature. Duffield et al. (2006) list the percent of YNP visitors who reported seeing a grizzly bear by season in 2005. 15.2% of visitors reported seeing a grizzly bear in the summer, which is the season in which the majority of visitors visit YNP.<sup>24</sup> The population of grizzly bears in 2005 was 652 (provided by the Interagency Grizzly Bear Study Team). This yields the second data point used to parameterize the function: 15.2% of visitors see a grizzly bear when the population size is 652. Richardson et al. (2014, 2015) conducted a mail-back survey of YNP visitors from May to September of 2009. They found that 67% of visitors saw a bear on their most recent visit. The statistic does not differentiate between grizzly bears and black bears. Although there is data on grizzly bear population in the GYE over time, there is not yet equivalent data for black bears (Haroldson and Gunther, 2013). The percent of total bears seen that are grizzly bears is estimated using percents reported by Duffield et al. (2006): 31.6% of summer visitors reported seeing a black bear in 2005 while 15.2% of summer visitors reported seeing a grizzly bear in the same year. This sums to 46.8% of summer visitors who reported seeing any bear in 2005. Dividing the percent of visitors who reported seeing a grizzly bear by the percent of visitors who reported seeing any bear and multiplying by 100 yields the percent of bear-seeing visitors who saw a grizzly bear: 32.48%. This percent is used as the best guess of the percent of bear-seeing visitors who saw a grizzly bear in that time-period. The percent is multiplied by Richardson et al. (2014, 2015)’s percent of visitors who saw a bear on their most recent trip (67%) to obtain the percent of visitors who saw a grizzly bear in the summer of 2009: 21.76%.<sup>25</sup> The population of grizzly bears in 2009 was 694 (provided by the Interagency Grizzly Bear Study Team). This yields the third data point used to parameterize the function: 21.76% of visitors see a grizzly bear when the population size is 694. Excel’s nonlinear solver is used to obtain the value of  $p_2$  at which the sum of squared error between predicted and actual probabilities is minimized:

$$p_2 = 0.0003. \quad (29)$$

The estimates of  $p_1$  and  $p_2$  are used to parameterize Equation (28).

The final step is to calculate the number of visitors who would value seeing a grizzly bear and who would actively view a grizzly bear if they saw one. In the summer months of 2019, 2,537,921

---

<sup>24</sup>This was confirmed using the YNP “Recreation Visits By Month (1979 - Current Calendar Year)” data set from National Park Service Integrated Resource Management Applications at <https://irma.nps.gov/STATS/>. Observations were kept for the years 1989-2019. Total visitation was summed over the entire time-period by month. The percent of visitors by month was then calculated by dividing each monthly sum by the total visitation over the entire time-period and multiplying by 100. It was found that 18.86% of visitors visited in June, 25.78% of visitors visited in July, and 22.86% of visitors visited in August. Summing the three percents yields 67.49%, which verifies that the majority of visitors visit YNP in the summer months.

<sup>25</sup>For the sake of comparison, the percent of total bears seen that are grizzly bears is also calculated using a different method. Gunther et al. (2018) find the percent of total bear-jams in YNP that are caused by grizzly bears. They report that, from 1990 to 2017, there were 14,251 documented bear-jams, of which 5,272 were grizzly bear-jams (i.e., 36.99% were grizzly bear-jams). Richardson et al. (2014, 2015)’s percent of visitors who saw a bear on their most recent trip (67%) is multiplied by the percent of bear-jams that are grizzly bear-jams (36.99%) to obtain the percent of visitors who saw a grizzly bear in the summer of 2009: 24.79%. The estimate is close to the one found using the Duffield et al. (2006) data (21.76%).

people visited YNP (<https://irma.nps.gov/stats>).<sup>26</sup> Richardson et al. (2015) asked respondents whether viewing bears is important in terms of their decision to take trips to YNP throughout the year. 62% of respondents said it was “very important.” The number of summer visitors is multiplied by 62% to narrow the number down to those that say viewing bears is very important (leading to conservative estimates). Richardson et al. (2015) found that some respondents prefer to see a grizzly bear (35%), some respondents prefer to see a black bear (5%), and some respondents have no preference between the two (60%). It is possible that people who prefer to see black bears may not value grizzly bear sightings (e.g., they may be afraid of seeing grizzly bears, which are more aggressive than black bears). To eliminate the 5% of visitors who would prefer to see a black bear, 95% is multiplied by the 62% of summer visitors who consider viewing bears to be very important. Richardson et al. (2014, 2015) found that 67% of respondents saw a bear on their most recent YNP trip and 55% of respondents participated specifically in bear viewing. Dividing 55% by 67% and multiplying by 100 yields the percent of people who actively viewed a bear given that they saw a bear (82%), which is multiplied by the previous product. The result is the number of summer visitors who find viewing a bear to be very important, do not prefer to see a black bear, and actively view a bear given that they see a bear:

$$v = 2,537,921 \cdot 0.62 \cdot 0.95 \cdot 0.82 = 1,227,104. \quad (30)$$

The estimates of  $s$ ,  $p_1$ ,  $p_2$ , and  $q$  are used to parameterize Equation (26).

### 4.3. Damages

The number of grizzly bear-human conflicts,  $I(G)$ , is estimated as a function of the same control variables used in the mortality NB regressions in Section 4.1.1: PVT, PDSI, and WBP. As in that subsection, the relationship between the dependent variable and grizzly bear population size is tested using NB regressions that build in a linear specification, CNB (1), a convex specification, CNB (2), and an unconstrained nonlinear specification, CNB (3). The general NB regression equation is

$$I(G) = e^{\beta_0 + \beta_1 \ln(G) + \beta_2 PVT + \beta_3 PDSI + \beta_4 WBP + \varepsilon}, \quad (31)$$

in which  $\ln(G)$  is the exposure variable. The results are shown in Table 5.

In CNB (1), the coefficient on PVT is positive and significant, which implies that expansion of grizzly bear range on private lands is correlated with an increase in grizzly bear-human conflicts. The coefficient on PDSI is insignificant, which indicates that the climate index is not correlated with conflicts. The coefficient on WBP is negative and significant, which implies that conflicts are lower in years with higher whitebark pine quality. In CNB (2), none of the control variables are significant. In CNB (3), The coefficient on  $\ln(G)$  is significant and equal to 0.78, which implies that a doubling of the grizzly bear population less than doubles grizzly bear-human conflicts. The number is close to one (i.e., close to a linear relationship between population size and grizzly bear-human conflicts). The coefficient on PVT is positive and significant, the coefficient on PDSI is insignificant, and the coefficient on WBP is negative and significant. The coefficients are similar in magnitude to those from CNB (1).

---

<sup>26</sup>There are two national parks in the GYE: YNP and GTNP. Summing the number of summer visitors to both national parks would lead to an overestimate of the total number of unique visitors to the national parks in the GYE because a substantial proportion of visitors may visit both national parks on the same trip. Only YNP visitors are considered, which leads to conservative estimates of aggregate non-consumptive use value.

The squared coefficient of correlation between fitted and actual values of the dependent variable is highest for CNB (3), the unconstrained nonlinear model, and lowest for CNB (2), the convex model. The coefficient estimates from CNB (1), as well as historical averages of private land expansion, PDSI, and whitebark pine quality, are used to parameterize and reduce Equation (31):

$$\begin{aligned} I(G) &= \gamma_I(\overline{PVT}, \overline{PDSI}, \overline{WBP})G \\ &= 0.30161 \cdot G. \end{aligned} \tag{32}$$

The full derivation is included in Supplementary Appendix C.5. CNB (1) is chosen to keep the numerical application tractable. The coefficient on  $\ln(G)$  in CNB (3) is significant and close to one, which provides justification for constraining the coefficient on  $\ln(G)$  to one in the CNB (1) regression.

	CNB (1)	CNB (2)	CNB (3)
$\ln G$	1	2	0.78***
	(.)	(.)	(3.93)
PVT	0.00057***	0.00042	0.00061***
	(2.70)	(1.34)	(3.18)
PDSI	0.0091	0.044	0.0027
	(0.40)	(1.20)	(0.11)
WBP	-0.018**	-0.015	-0.020***
	(-2.21)	(-1.08)	(-2.61)
Constant	-1.32***	-7.53***	0.040
	(-11.49)	(-41.56)	(0.03)
$\alpha$	0.088	0.216	0.084
Rb. Std. Err.	0.036	0.063	0.039
$N$	27	27	27
Corr <sup>2</sup>	0.641	0.544	0.654

*z*-scores in parentheses

\*  $p < .10$ , \*\*  $p < .05$ , \*\*\*  $p < .01$

Table 5: Conflict negative binomial regressions

As shown in Figure 2c, total annual grizzly bear program expenditures,  $E(I)$ , are higher in years

with more grizzly bear-human conflicts. The relationship is tested using OLS regressions.<sup>27</sup> The general OLS regression equation is

$$E(I(G)) = \zeta_0 + \zeta_1 I(G) + \zeta_2 I(G)^2 + \varepsilon, \quad (33)$$

in which  $I(G)$  and  $I(G)^2$  are the number of grizzly bear-human conflicts and the number of grizzly bear-human conflicts squared, respectively. The results, which are estimated with robust standard errors, are shown in Table 6.

	OLS (1)	OLS (2)
I	6,432.6*** (5.34)	10,644.6*** (2.87)
I <sup>2</sup>		-9.30 (-1.14)
Constant	230,397.8 (1.08)	-186,208.8 (-0.48)
$N$	26	26
$R^2$	0.686	0.700

*t* statistics in parentheses

\*  $p < .10$ , \*\*  $p < .05$ , \*\*\*  $p < .01$

Table 6: Total expenditure OLS regressions

In OLS (1),  $\zeta_2$  is constrained to zero, which amounts to a linear specification of total expenditures as a function of conflicts. The  $\zeta_1$  coefficient is positive and significant. It represents the marginal damages (in terms of increased expenditures) per conflict. In OLS (2),  $\zeta_2$  is unconstrained, which amounts to a quadratic specification of total expenditures as a function of conflicts. The  $\zeta_1$  coefficient is positive and significant while the  $\zeta_2$  coefficient is negative and insignificant. OLS (2) has a higher  $R^2$  value, but it is eliminated on account of intuition: the regression predicts negative expenditures in years with low numbers of conflicts, which is infeasible. OLS (1) is thus used to parameterize Equation (33):

$$E(I(G)) = 230,397.8 + 6,432.6 \cdot I(G). \quad (34)$$

## 5. Results

### 5.1. Base Case

Numerical results, generated in Mathematica and presented in  $(G, \mu)$  space, are summarized in Figure 4. Combinations of grizzly bear population sizes and natural capital values along the recovery

<sup>27</sup>The dependent variable, total annual grizzly bear program expenditures, is continuous, in contrast to the count data employed in the other parameterizations.

path are depicted, along with the current point and singular solution. Under the protections of the ESA and the associated reactive management regime, and given the benchmark parameterization, the grizzly bear natural capital value has decreased and become negative as grizzly bears have recovered. If the species were to remain listed, society would reach a reactive management equilibrium,  $(G_{h^*=0}, \mu_{h^*=0})$ , with a negative natural capital value. In contrast, the singular solution of active management has fewer grizzly bears but a positive natural capital value. Numerical solutions for all possible cases are shown in Table 7.

Point	Analytical Solution	Grizzly Bears	Natural Capital Value	Harvest
Reactive equilibrium	$(G_{h^*=0}, \mu_{h^*=0})$	876	-\$694.72	0
Active equilibrium	$(G^*, h^*)$	648	\$577.20	19
Switching point	$(G^*, \mu^{RP}(G^*))$	648	-\$192.38	0
Current point	$(G_0, \mu^{RP}(G_0))$	737	-\$426.80	0

Table 7: Solutions for points of interest.

The reactive equilibrium occurs at the intersection of the  $\dot{G}_i$  and  $\dot{\mu}_i$  isoclines in Figure 4.<sup>28</sup> The recovery path under reactive management is given by a separatrix (dashed line) that leads to the equilibrium. The path is unique, as grizzly bear growth depends only on its own stock, growth, and mortality parameters. The equilibrium is stable, whether the initial stock is below or above the equilibrium. From a low initial condition as observed, the grizzly bear population grows towards the reactive management equilibrium. As the population grows, the natural capital value changes according to the co-state equation, with no influence on the evolution of the grizzly bear population.<sup>29</sup> At low population sizes, the value of a live grizzly bear is close to infinite. As the population size increases, so too do the number of conflicts (which are a function of the stock size), and value drops. By the time the recovery path reaches the reactive equilibrium, the natural capital value is negative, meaning that grizzly bears would be a liability into perpetuity.

The equilibrium singular solution,  $(G^*, h^*)$ , defines the long-run outcome under an active management program. The singular solution has fewer grizzly bears at a higher natural capital value than the reactive equilibrium. Reaching the active equilibrium is possible via a most rapid approach path (Spence and Starrett, 1975) that depends on the current grizzly bear population. If the current population is less than  $G(t) < G^*$ , then the fastest approach is simply to follow the recovery path (reactive management and  $h^* = 0$ ) up until the singular stock is attained ( $G(t) = G^*$ ) and then immediately change the management regime (e.g., implement optimal harvest,  $h^*$ , into perpetuity). When the population is at the optimal population size,  $G^*$ , reaching the active equilibrium is as simple as immediately implementing singular harvests from that moment on. Evaluating the recovery path at  $G^*$  yields the point, termed the “switching point” in Figure 4, at which an active management program should be established. Interestingly, the switching point occurs at a population associated with a negative natural capital value. It is optimal to let the population recover

<sup>28</sup>The  $i$  subscripts reflect that the isoclines correspond to Case i.

<sup>29</sup>The directional arrows refer to the Case i solutions, which are uniquely determined for  $G(t)$  by Equation (18) and for  $\mu(t)$  by the solution to Equation (9) given  $G(t)$ . The feedback control diagram for Case ii reflects Equation (13) and is omitted given the detailed description in Section 3.

past the level where the shadow value falls to zero. The recovery path includes a buffer where extreme conservation is maintained to allow the grizzly bear population to grow to such an extent that the species can absorb the switch to active management. In the base case, the buffer calls for the stock to grow by 58 additional bears before the switch to active management. The switch to active management then transforms the species back into a commodity with positive natural capital value.

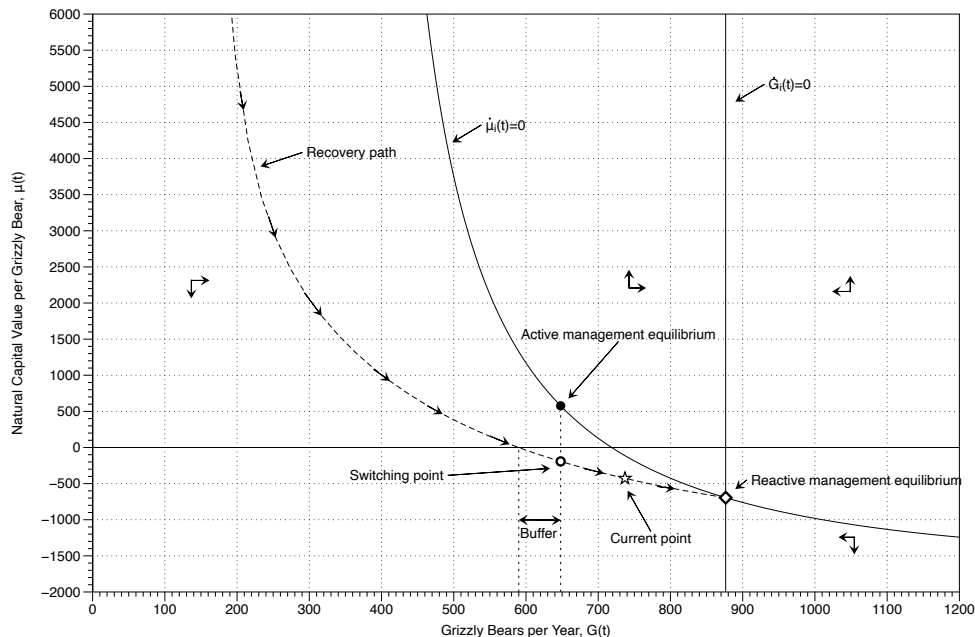


Figure 4: Phase plane with potential long-run outcomes.

Reaching the active equilibrium is more complicated if the population has surpassed the optimal size ( $G(t) > G^*$ ). In the GYE, the most recent population estimate was 737 grizzly bears, which is already larger than  $G^*$ . As shown in Figure 4, the natural capital value of a live grizzly bear at this “current point” is negative. In this situation, a most rapid approach would call for a harvesting at the maximum rate,  $h^{\max}$ , to reach  $G^*$  as fast as possible.<sup>30</sup> Once the optimal population size is reached, the agency allows harvests of the singular amount into perpetuity.

Whether continuing along the recovery path to the reactive equilibrium or transitioning to the active equilibrium is socially preferred can be determined by evaluating the discounted social net benefits of the strategies (Tahvonen and Salo, 1996). Results are shown in Table 8. Active management yields substantially higher discounted social net benefits. In the long-run, the different management strategies lead to different equilibrium numbers of conflicts and human-caused mortalities in each time-period, as shown in Figure 5. By assumption, conflicts are stock-dependent, so it is no surprise

<sup>30</sup>It is assumed that the agency would never undershoot the optimal population size. In the time-period in which establishing maximum harvest would undershoot the optimal population size, the agency instead sets a one-time intermediate harvest that brings the population to  $G^*$ .



that there are less conflicts when the agency establishes active management and reduces the long-run steady state population size. Analyzing the total number of human-caused mortalities, which includes both harvest and non-harvest mortalities, is more interesting. There are (two) less human-caused mortalities under the active management strategy, which implies that the number of grizzly bears lost to harvest is made up for by the number of grizzly bears gained from there being less non-harvest human-caused mortalities at the lower population size. The ratio of the grizzly bear population that is lost to human-caused mortality at the long run equilibrium is similar across the two management strategies: 0.052 under reactive management and 0.067 under active management.

Management Strategy	Discounted Social Net Benefits (\$)
Reactive	1,920,168,145
Active	1,928,482,102
<b>Difference</b>	<b>8,313,957</b>

Table 8: Discounted social net benefits for both management strategies.

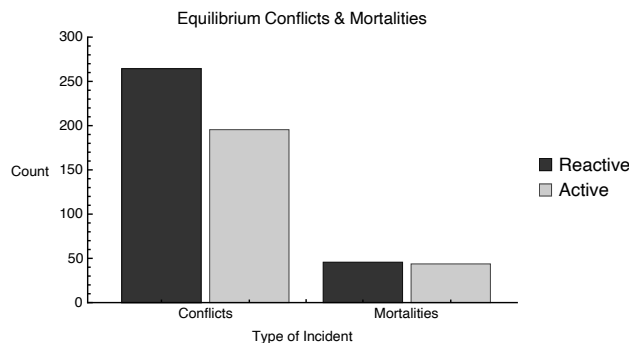


Figure 5: Comparison of long-run equilibrium conflict and mortality counts across management strategies.

## 5.2. Sensitivity Analysis

The sensitivity of the results to changes in key parameters characterizes the boundaries of active versus reactive management. Figure 6 shows how the active management equilibrium solutions ( $G^*$  on the left vertical axis,  $h^*$  on the right vertical axis) change as the parameter values (horizontal axis) change. The size of the population at the active management equilibrium is the size of the population at the switching point. In other words, the active management equilibrium stock level dictates when the agency should “turn on” active management. All of the parameters in Figure 6 have threshold values that significantly influence the optimality of active management.

Panel (a) of Figure 6 shows the Case ii equilibrium solutions as nonuse value parameter  $q$  changes. The parameter dictates the grizzly bear population size at which full aggregate nonuse value is obtained: the larger the parameter, the smaller the population size associated with full recovery of the species. At small values of  $q$ , it takes a large population size for society to obtain full nonuse value. This creates pressure to allow the grizzly bear population to reach the size needed for full nonuse value, which is easier if no optimal harvest occurs. Active management only becomes

optimal at values of  $q$  larger than 0.008. At the value of  $q$  used in the base case ( $q = 0.03744$ ), changes in aggregate nonuse value from grizzly bears ( $n_A$ ) do not affect the optimality of active management. This is shown in Supplementary Appendix Figure D.1. The current population is far from the low population size at which full nonuse value is obtained, meaning that changes in the current population have negligible effects on nonuse value.

Panel (b) of Figure 6 shows the Case ii equilibrium solutions as average WTP for a grizzly bear sighting ( $s$ ) changes. At high enough WTP values ( $s \geq 7.50$ ), the benefits from aggregate non-consumptive use value overshadow the damages from grizzly bear expenditures, and it becomes optimal to keep a reactive management strategy (i.e., maintain zero harvest). A similar pattern occurs as sighting probability parameter  $p_2$  changes, as shown in panel (c) of Figure 6. At high enough  $p_2$  parameter values ( $p_2 \geq 0.00036$ ), it becomes optimal to keep a reactive management strategy with zero harvest. Of note, there is a peak in optimal harvest that occurs in both panels (b) and (c). The spike occurs where net growth of grizzly bears (shown in the bottom plot of Figure 3) is at its peak. At lower population sizes, the agency cannot sustainably keep increasing harvests, so it adjusts optimal harvest downward.

Panel (d) of Figure 6 shows the Case ii equilibrium solutions as marginal expenditures per conflict ( $\zeta_1$ ) change. When marginal expenditures are low enough, damages are overshadowed by benefits, and the agency should not switch to active management. At high enough marginal expenditures ( $\zeta_1 > \$5,800$ ), it becomes optimal to turn on active management by establishing optimal harvest. As in plots (b) and (c), there is a peak in optimal harvest at the population size where net growth is at its maximum.

Panel (e) of Figure 6 shows the Case ii equilibrium solutions as average whitebark pine quality (measured in terms of mortality-adjusted median cones per tree) changes. Whitebark pine quality affects damages from grizzly bears because the parameter appears in the conflicts equation, which in turn appears in the expenditures equation. The sensitivity analysis for whitebark pine quality is different from that of all parameters discussed previously because the parameter enters the growth equation through the mortality term. Both the Case i and Case ii solutions adjust as the parameter changes (for the parameters discussed previously, only the Case ii solutions adjust). Higher whitebark pine quality leads to less non-harvest human-caused grizzly bear mortalities, which increases the equilibrium population attained under a reactive management strategy. It seems intuitive to expect optimal harvest to increase as grizzly bear mortalities decrease due to higher whitebark pine quality. However, the simultaneous reduction in conflicts from increased whitebark pine quality causes grizzly bear expenditures to go down. As grizzly bear damages decrease because of less conflicts, there is less pressure for the agency to reduce the number of grizzly bears in the wild. At high enough WBP quality ( $\overline{WBP} \geq 12$ ), it becomes optimal to keep a reactive management strategy with no harvest.

Other parameters influence the results but do not affect the optimality of active management. The plots for such variables are included in Supplementary Appendix D.1. As consumptive use value increases, optimal harvest increases while the equilibrium population size decreases. Active management is optimal over the entire domain; even at a consumptive use value of zero, there is pressure to turn on active management due to the damages from grizzly bears. Similar results hold for the discount rate: as the discount rate increases, optimal harvest increases while the equilibrium population size decreases. This reflects that higher discount rates typically call for lower stock levels of renewable resources (Conrad, 2010).

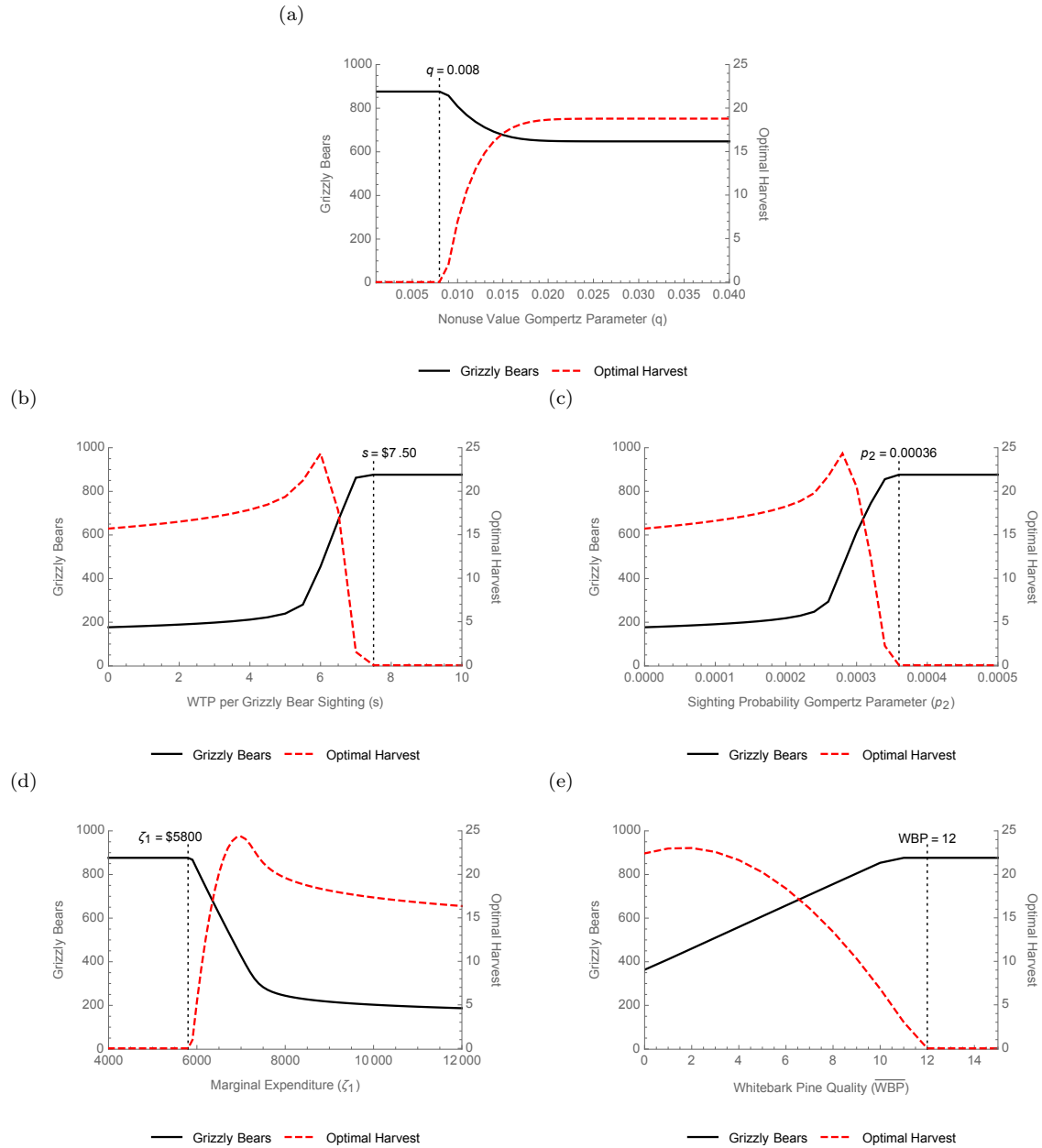


Figure 6: Active management equilibrium solutions across changes in parameters: (a) nonuse value Gompertz parameter  $q$ , (b) average WTP per grizzly bear sighting,  $s$ , (c) sighting probability Gompertz parameter  $p_2$ , (d) marginal expenditures per conflict,  $\zeta_1$ , (e) average whitebark pine quality,  $\overline{WBP}$ .

## 6. Conclusion

Management of high-profile multi-use species under conservation measures such as the ESA is challenging and often beyond the insights of standard bioeconomic analysis. Grizzly bears are a unique multi-use resource because, even with federal protections in place that restrict active management, they incur high non-harvest human-caused mortality under reactive management. As the species has recovered, higher population sizes have led to higher numbers of grizzly bear-human conflicts and non-harvest mortalities. For the parameterization used in the base case, grizzly bears are on a recovery path to a long-run reactive management equilibrium at which the natural capital value of a live grizzly bear is negative. By establishing an active management program (e.g., optimal harvest), an agency could instead reach an equilibrium at which the value of a live grizzly bear is positive. An active management program would decrease the long-run grizzly bear population size, which would in turn lead to fewer conflicts and mortalities. Total mortality, which is the sum of harvest and non-harvest mortality, is actually lower at the active equilibrium: the gain in grizzly bears from less non-harvest mortality overshadows the number of grizzly bears lost to harvest. The agency can use active management to reduce the risks from grizzly bears, increase the natural capital value of the resource such that it becomes an asset, and increase social net benefits from grizzly bears.

Given the base case parameterization, the results indicate that the grizzly bear population has surpassed the size at which an active management strategy should have been enacted. At the switching point, the natural capital value of a live grizzly bear was already negative. The implications are that when constructing conservation policies directed at charismatic multi-use megafauna, it may be optimal for society to endure intervals of time with negative natural capital values. In other words, it may be optimal for managers to develop buffers of the species. Over these intervals, it is optimal to let the population recover enough to withstand future changes in management that transform the resource from a liability to a long-run asset. The optimality of active management depends on key parameters, including nonuse values, average WTP for a grizzly bear sighting, the probability of sighting a grizzly bear, marginal expenditures per conflict, and the quality of whitebark pine. These dependencies demonstrate the key role of not only understanding ecological complexities but also understanding how humans value and interact with a species such as grizzly bears. This literature is still being developed and is definitely needed for future policy development.

Grizzly bear recovery likely would not have been possible without ESA protections, which served the grizzly bear population well over low population sizes (when the natural capital value per live grizzly bear was large). However, as the grizzly bear population has followed the recovery path to the long-run reactive management equilibrium, the natural capital value per grizzly bear has diminished significantly. The benchmark parameterization indicates that ESA protections may no longer be ideal, as they are binding grizzly bears to a recovery path that leads to a long-run equilibrium with negative natural capital value.

## References

- Anderson, L.E., Manning, R.E., Valliere, W.A., Hallo, J.C., 2010. Normative standards for wildlife viewing in parks and protected areas. *Human Dimensions of Wildlife* 15, 1–15.
- Bjornlie, D.D., Haroldson, M.A., 2019. Yellowstone grizzly bear investigations: 2018 annual re-

- port of the Interagency Grizzly Bear Study Team. U.S. Geological Survey. chapter Grizzly bear occupied range in the Greater Yellowstone Ecosystem, 1990-2018. pp. 25–28.
- Cameron, A.C., Trivedi, P.K., 2009. Microeconometrics using Stata. volume 5. Stata Press College Station, TX.
- Clark, C.W., Munro, G.R., 1975. The economics of fishing and modern capital theory: a simplified approach. *Journal of Environmental Economics and Management* 2, 92–106.
- Clayton, C., Mendelsohn, R., 1993. The value of watchable wildlife: A case study of McNeil River. *Journal of Environmental Management* 39, 101–106.
- Conrad, J.M., 2010. Resource economics. 2 ed., Cambridge University Press.
- Conrad, J.M., Clark, C.W., 1987. Natural resource economics: notes and problems. Cambridge University Press.
- Duffield, J., Patterson, D., Neher, C.J., 2006. Wolves and people in Yellowstone: Impacts on the regional economy. University of Montana, Department of Mathematical Sciences.
- Fenichel, E.P., Abbott, J.K., 2014. Natural capital: from metaphor to measurement. *Journal of the Association of Environmental and Resource Economists* 1, 1–27.
- Fenichel, E.P., Horan, R.D., Bence, J.R., 2010. Indirect management of invasive species through bio-controls: a bioeconomic model of salmon and alewife in Lake Michigan. *Resource and Energy Economics* 32, 500–518.
- Fonner, R., Bohara, A.K., 2017. Optimal control of wild horse populations with nonlethal methods. *Land Economics* 93, 390–412.
- Gunther, K.A., Haroldson, M.A., Frey, K., Cain, S.L., Copeland, J., Schwartz, C.C., 2004. Grizzly bear–human conflicts in the Greater Yellowstone Ecosystem, 1992–2000. *Ursus* 15, 10–23.
- Gunther, K.A., Wilmot, K.R., Cain, S.L., Wyman, T.C., Reinertson, E.G., Bramblett, A.M., 2018. Managing human-habituated bears to enhance survival, habitat effectiveness, and public viewing. *Human–Wildlife Interactions* 12, 7.
- Haroldson, M.A., 2019. Yellowstone grizzly bear investigations: 2018 annual report of the Interagency Grizzly Bear Study Team. U.S. Geological Survey. chapter Whitebark pine cone production. pp. 54–56.
- Haroldson, M.A., Gunther, K.A., 2013. Roadside bear viewing opportunities in Yellowstone National Park: characteristics, trends, and influence of whitebark pine. *Ursus* 24, 27–41.
- Horan, R.D., Bulte, E.H., 2004. Optimal and open access harvesting of multi-use species in a second-best world. *Environmental and Resource Economics* 28, 251–272.
- Huffaker, R.G., Bhat, M.G., Lenhart, S.M., 1992. Optimal trapping strategies for diffusing nuisance-beaver populations. *Natural Resource Modeling* 6, 71–97.
- Huffaker, R.G., Wilen, J.E., Gardner, B.D., 1989. Multiple use benefits on public rangelands: an incentive-based fee system. *American Journal of Agricultural Economics* 71, 670–678.

- Huffaker, R.G., Wilen, J.E., Gardner, B.D., 1990. A bioeconomic livestock/wild horse trade-off mechanism for conserving public rangeland vegetation. *Western Journal of Agricultural Economics* 15, 73–82.
- Loomis, J.B., White, D.S., 1996. Economic benefits of rare and endangered species: summary and meta-analysis. *Ecological Economics* 18, 197–206.
- van Manen, F.T., Haroldson, M.A., Karabensh, B.E. (Eds.), 2019. Yellowstone grizzly bear investigations: 2018 annual report of the Interagency Grizzly Bear Study Team. U.S. Geological Survey.
- Miller, S.M., Miller, S.D., McCollum, D.W., 1998. Attitudes toward and relative value of Alaskan brown and black bears to resident voters, resident hunters, and nonresident hunters. *Ursus* , 357–376.
- National Park Service, 2020. Greater Yellowstone Ecosystem. URL: <https://www.nps.gov/yell/learn/nature/greater-yellowstone-ecosystem.htm>.
- National Research Council, 2005. Valuing ecosystem services: toward better environmental decision-making. National Academies Press.
- Olaussen, J.O., Skonhoft, A., 2011. A cost-benefit analysis of moose harvesting in Scandinavia. a stage structured modelling approach. *Resource and Energy Economics* 33, 589–611.
- Richardson, L., Gunther, K., Rosen, T., Schwartz, C., 2015. Visitor perceptions of roadside bear viewing and management in Yellowstone National Park, in: *The George Wright Forum*, JSTOR. pp. 299–307.
- Richardson, L., Huber, C., Loomis, J., 2017. Challenges and solutions for applying the travel cost demand model to geographically remote visitor destinations: A case study of bear viewing at Katmai National Park and Preserve. *Human Dimensions of Wildlife* 22, 550–563. doi:[10.1080/10871209.2017.1369196](https://doi.org/10.1080/10871209.2017.1369196).
- Richardson, L., Loomis, J., 2009. The total economic value of threatened, endangered and rare species: an updated meta-analysis. *Ecological Economics* 68, 1535–1548.
- Richardson, L., Rosen, T., Gunther, K., Schwartz, C., 2014. The economics of roadside bear viewing. *Journal of Environmental Management* 140, 102–110.
- Rondeau, D., 2001. Along the way back from the brink. *Journal of Environmental Economics and Management* 42, 156–182.
- Rondeau, D., Conrad, J.M., 2003. Managing urban deer. *American Journal of Agricultural Economics* 85, 266–281.
- Skibins, J.C., Hallo, J.C., Sharp, J.L., Manning, R.E., 2012. Quantifying the role of viewing the Denali “big 5” in visitor satisfaction and awareness: conservation implications for flagship recognition and resource management. *Human Dimensions of Wildlife* 17, 112–128.
- Skonhoft, A., 2006. The costs and benefits of animal predation: an analysis of Scandinavian wolf re-colonization. *Ecological Economics* 58, 830–841.

- Skonhofs, A., Olaussen, J.O., 2005. Managing a migratory species that is both a value and a pest. *Land Economics* 81, 34–50.
- Spence, M., Starrett, D., 1975. Most rapid approach paths in accumulation problems. *International Economic Review* 16, 388–403.
- Tahvonen, O., Salo, S., 1996. Nonconvexities in optimal pollution accumulation. *Journal of Environmental Economics and Management* 31, 160–177.
- US Fish and Wildlife Service, 2000. Grizzly bear recovery in the Bitterroot Ecosystem. Final Environmental Impact Statement .
- U.S. Fish and Wildlife Service, 2016. Federal Register / Vol. 81 , No. 48 / Friday, March 11, 2016 / Proposed Rules. URL: <https://www.fws.gov/policy/library/2016/2016-05167.html>.
- Weitzman, M.L., 2001. Gamma discounting. *American Economic Review* 91, 260–271.
- White, P.J., Gunther, K.A., van Manen, F.T. (Eds.), 2017. Yellowstone grizzly bears: Ecology and conservation of an icon of wildness. *Yellowstone Forever*.
- Zabel, A., Pittel, K., Bostedt, G., Engel, S., 2011. Comparing conventional and new policy approaches for carnivore conservation: theoretical results and application to tiger conservation. *Environmental and Resource Economics* 48, 287–301.
- Zivin, J., Hueth, B.M., Zilberman, D., 2000. Managing a multiple-use resource: the case of feral pig management in California rangeland. *Journal of Environmental Economics and Management* 39, 189–204.

## Supplementary Appendices

### A. Management Background

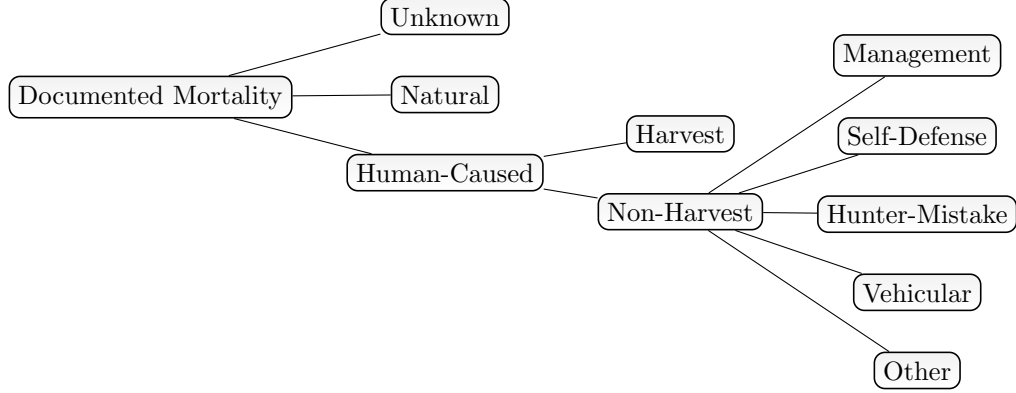


Figure A.1: Types of documented grizzly bear mortalities.

### B. Bioeconomic Model

#### B.1. Total Economic Value

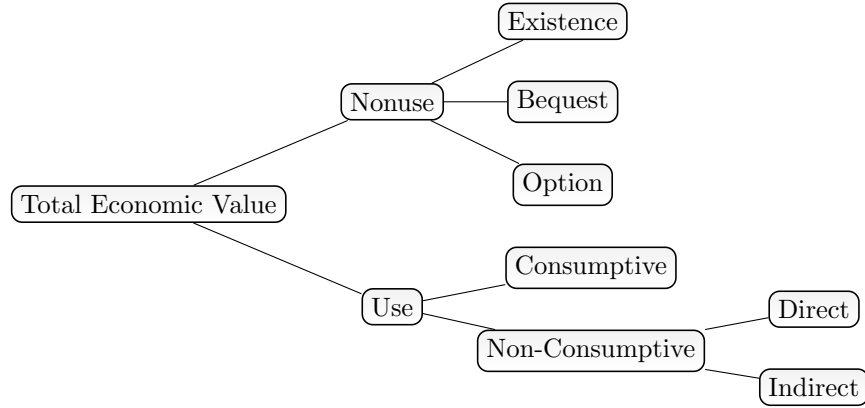


Figure B.1: Types of values in the TEV framework (National Research Council, 2005; Loomis and White, 1996).

#### B.2. Nonlinear Harvest Program

A nonlinear model version of the model was simulated with a concave consumptive use value, in which Equation (4) becomes

$$C_c(h) = c_c \ln h. \quad (\text{B.1})$$

$c_c$  is parameterized such that

$$\begin{aligned} C'(h^*) &= C'_c(h^*) \\ \Rightarrow \quad c &= \frac{c_c}{h^*}. \end{aligned}$$



The concave specification reflects that consumptive use value may exhibit diminishing returns as more harvest occurs (i.e., as harvesting a grizzly bear becomes less rare). The resulting phase plane, in grizzly bear and harvest space, is shown in Figure B.2. The saddle path into the active management equilibrium indicates that it would be optimal to start harvesting grizzly bears at a low population size (i.e., at a population size of roughly 100 grizzly bears). The corresponding recovery path, which tracks how the natural capital value per grizzly bear adjusts given the nonlinear harvest program, is shown in Figure B.3 and compared to the recovery path under the linear harvest program.

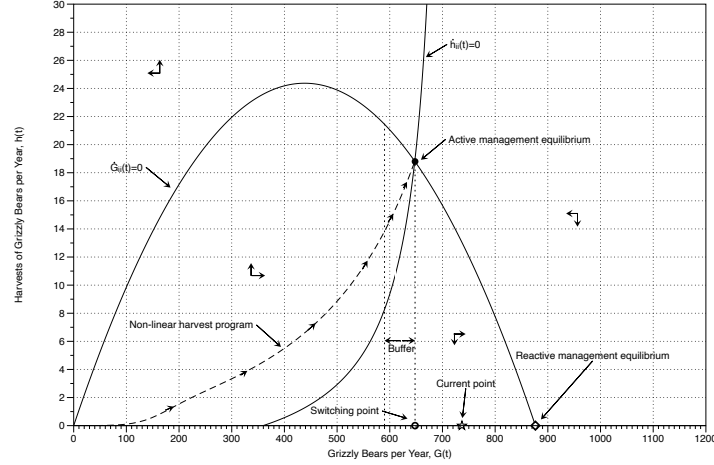


Figure B.2: Phase plane under a nonlinear harvest program, shown in  $(G, h)$  space.

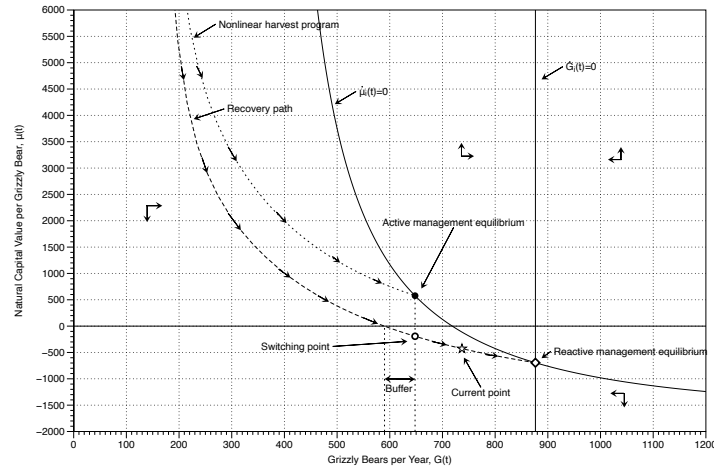


Figure B.3: Phase plane under a linear harvest program, shown in  $(G, \mu)$  space with recovery paths from both linear and nonlinear harvest programs.

The nonlinear results reflect [Rondeau \(2001\)](#)'s description of a recovery path, but omit some important realities of grizzly bear management. Grizzly bears have been federally protected for over

40 years, and their protections have been designed to limit any form of active management at low population sizes.

### C. Numerical Specification

#### C.1. Mortality Negative Binomial Equation

The full derivation of the parameterized mortality equation begins with the NB regression equation:

$$M(G) = \exp(\alpha_0 + \alpha_1 \ln(G) + \alpha_2 PVT + \alpha_3 PDSI + \alpha_4 WBP).$$

The coefficient estimates from MNB (2) in Table 3 are substituted in:

$$\begin{aligned} M(G) &= \exp(\hat{\alpha}_0 + \hat{\alpha}_1 \ln(G) + \hat{\alpha}_2 PVT + \hat{\alpha}_3 PDSI + \hat{\alpha}_4 WBP) \\ &= \exp(-9.68 + 2 \ln(G) + 0.00024 \cdot PVT + 0.025 \cdot PDSI - 0.025 \cdot WBP) \\ &= \exp(-9.68 + 0.00024 \cdot PVT + 0.025 \cdot PDSI - 0.025 \cdot WBP) \cdot \exp(2 \ln(G)) \\ &= \underbrace{\exp(-9.68 + 0.00024 \cdot PVT + 0.025 \cdot PDSI - 0.025 \cdot WBP)}_{\gamma_M = \gamma_M(PVT, PDSI, WBP)} \cdot G^2. \end{aligned}$$

Average values over the entire time-period are substituted in for private land expansion ( $\overline{PVT} = 392.48$ ), Palmer Drought Severity Index ( $\overline{PDSI} = -0.10671$ ), and whitebark pine quality ( $\overline{WBP} = 5.8001$ ). This yields

$$\begin{aligned} M(G) &= \underbrace{\exp(-9.68 + 0.00024 \cdot \overline{PVT} + 0.025 \cdot \overline{PDSI} - 0.025 \cdot \overline{WBP})}_{\bar{\gamma}_M = \gamma_M(\overline{PVT}, \overline{PDSI}, \overline{WBP})} \cdot G^2 \\ &= \exp(-9.68 + 0.00024 \cdot 392.48 + 0.025 \cdot -0.10671 - 0.025 \cdot 5.8001) \cdot G^2 \\ &\approx \underbrace{0.00006}_{\bar{\gamma}_M} \cdot G^2. \end{aligned}$$

#### C.2. Net Growth Differential Equation

The modified logistic growth function that accounts for the nonlinear relationship between grizzly bear population and mortality is

$$\begin{aligned} F(G(t)) &= L(G(t)) - M(G(t)) \\ &= rG(t) \left(1 - \frac{G(t)}{K}\right) - \gamma_M G(t)^2. \end{aligned}$$

Solving the differential equation yields

$$G(t) = \frac{e^{rt} G_0 K r}{\gamma_M (e^{rt} - 1) G_0 K + ((e^{rt} - 1) G_0 + K) r}.$$

$\gamma_M$  is substituted in:

$$G(t) = \frac{e^{rt} G_0 K r}{\gamma_M(PVT(t), PDSI(t), WBP(t)) (e^{rt} - 1) G_0 K + ((e^{rt} - 1) G_0 + K) r}.$$

Using the explicit functional form for  $\gamma_M$  yields

$$G(t) = \frac{e^{rt} G_0 K r}{\exp(\alpha_0 + \alpha_2 PVT(t) + \alpha_3 PDSI(t) + \alpha_4 WBP(t))(e^{rt} - 1)G_0 K + ((e^{rt} - 1)G_0 + K)r}$$

As a last step, the coefficient estimates from MNB (2) in Table 3 are substituted in:

$$G(t) = \frac{e^{rt} G_0 K r}{\exp(\hat{\alpha}_0 + \hat{\alpha}_2 PVT(t) + \hat{\alpha}_3 PDSI(t) + \hat{\alpha}_4 WBP(t))(e^{rt} - 1)G_0 K + ((e^{rt} - 1)G_0 + K)r} \quad (C.1)$$

Initial values are set for  $G_0$ ,  $r$ , and  $K$ . For  $G_0$ , the size of the population from the first year of the data (1987) is used. The initial guesses are 0.05 for  $r$  and 1000 for  $K$ . The population in each time period from  $t = 0$  (1987) to  $t = 32$  (2019) is used as  $G(0)$  to  $G(32)$ . The predicted population for each year,  $\hat{G}(t)$ , is calculated by plugging the initial values of  $G_0$ ,  $r$ , and  $K$ , as well as the actual annual data on private land expansion, Palmer Drought Severity Index, and whitebark pine quality, into Equation (C.1).

For time-periods zero through three and 32, there is no data on private land expansion, so average private land expansion ( $\overline{PVT} = 392.48$ ) is substituted in. For time-periods zero and one, there is no data on Palmer Drought Severity Index and whitebark pine quality, so averages ( $\overline{PDSI} = -0.10671$  and  $\overline{WBP} = 5.8001$ ) are substituted in. For each year, the actual population estimate,  $G(t)$ , is subtracted from the predicted population estimate,  $\hat{G}(t)$ , and then the difference is squared. Summing across all years yields the sum of squared errors. As a final step, Microsoft Excel's nonlinear solver is used to minimize the sum of squared errors by optimizing  $r$  and  $K$ .

### C.3. Nonuse Value

Figure C.1 shows aggregate nonuse value over different grizzly bear population sizes, given the estimate of  $q$  from Equation (24). The figure also shows aggregate nonuse value with high and low estimates of  $q$ :  $q^H = 5q$  and  $q^L = \frac{1}{5}q$ .

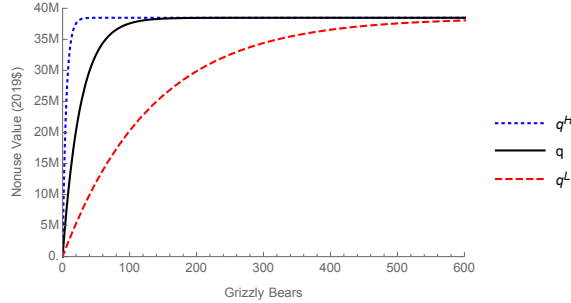


Figure C.1: Aggregate nonuse value as a function of population size using a Gompertz function with three specifications of parameter  $q$ .

### C.4. Non-Consumptive Use Value

Figure C.1 shows the probability of visitors who see a grizzly bear over different grizzly bear population sizes, given the estimate  $p_2$  from Equation (29). The figure also shows total aggregate nonuse value with high and low values of  $p_2$ :  $p_2^H = 5p_2$  and  $p_2^L = \frac{1}{5}p_2$ .

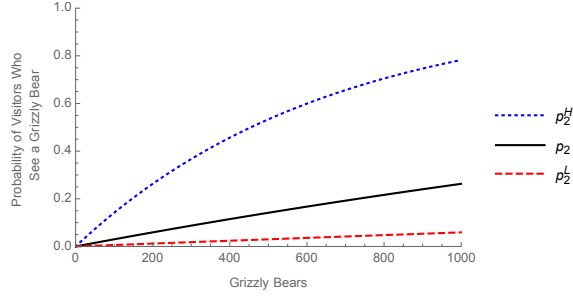


Figure C.2: The probability of visitors who see a grizzly bear as a function of population size using a Gompertz function with three specifications of parameter  $p_2$ .

Figure C.3 shows aggregate non-consumptive use value over different grizzly bear population sizes, given the estimates of  $s$ ,  $p_1$ ,  $p_2$ , and  $q$  from Section 4.2.3. The figure also shows aggregate non-consumptive use value with high and low values of  $p_2$ .

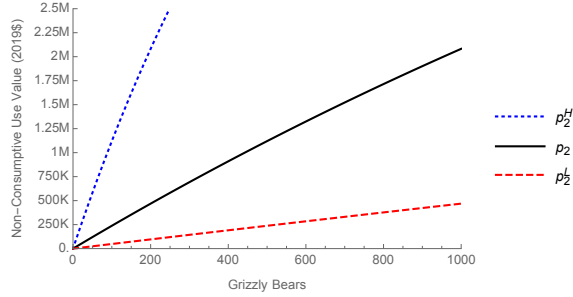


Figure C.3: Aggregate non-consumptive use value as a function of population size given three specifications of parameter  $p_2$ .

### C.5. Conflict Negative Binomial Equation

The full derivation of the parameterized conflict equation begins with the NB regression equation:

$$I(G) = \exp(\beta_0 + \beta_1 \ln(G) + \beta_2 PVT + \beta_3 PDSI + \beta_4 WBP).$$

The coefficient estimates from CNB (1) in Table 5 are substituted in:

$$\begin{aligned} I(G) &= \exp(\hat{\beta}_0 + \hat{\beta}_1 \ln(G) + \hat{\beta}_2 PVT + \hat{\beta}_3 PDSI + \hat{\beta}_4 WBP) \\ &= \exp(-1.32 + \ln(G) + 0.00057 \cdot PVT + 0.0091 \cdot PDSI - 0.018 \cdot WBP) \\ &= \exp(-1.32 + 0.00057 \cdot PVT + 0.0091 \cdot PDSI - 0.018 \cdot WBP) \cdot \exp(\ln(G)) \\ &= \underbrace{\exp(-1.32 + 0.00057 \cdot PVT + 0.0091 \cdot PDSI - 0.018 \cdot WBP)}_{\gamma_I = \gamma_I(PVT, PDSI, WBP)} \cdot G. \end{aligned}$$

Average values over the entire time-period are substituted in for private land expansion ( $\overline{PVT} = 392.48$ ), Palmer Drought Severity Index ( $\overline{PDSI} = -0.10671$ ), and whitebark pine quality ( $\overline{WBP} =$

5.8001). This yields

$$\begin{aligned}
I(G) &= \underbrace{\exp(-1.32 + 0.00057 \cdot \overline{PVT} + 0.0091 \cdot \overline{PDSI} - 0.018 \cdot \overline{WBP})}_{\bar{\gamma}_I = \gamma_I(\overline{PVT}, \overline{PDSI}, \overline{WBP})} \cdot G \\
&= \exp(-1.32 + 0.00057 \cdot 392.48 + 0.0091 \cdot -0.10671 - 0.018 \cdot 5.8001) \cdot G \\
&\approx \underbrace{0.30161}_{\bar{\gamma}_I} \cdot G.
\end{aligned}$$

## D. Results

### D.1. Sensitivity Analysis

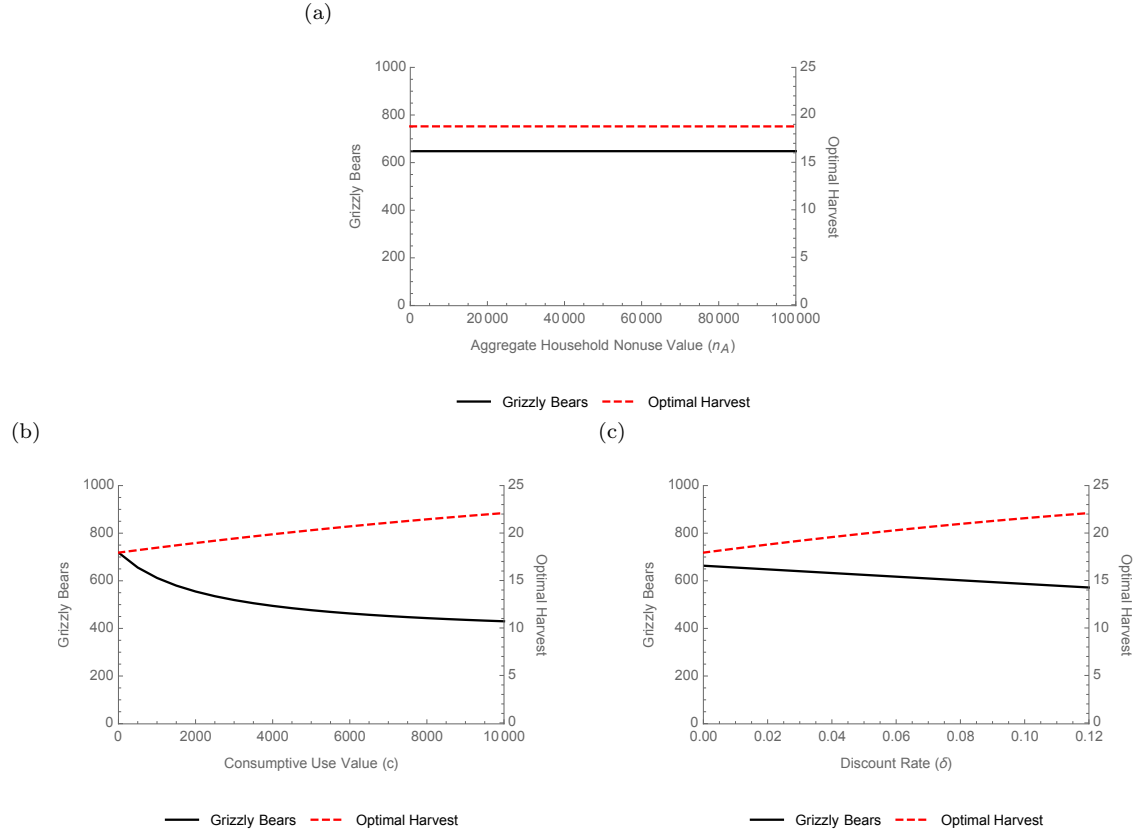


Figure D.1: Active management equilibrium solutions across changes in parameters: (a) aggregate household nonuse value,  $n_A$ , (b) consumptive use value,  $c$ , (c) the discount rate,  $\delta$ .