Estimating the Economic Benefits of a Mt St Helens Wolf Population: a Cost Benefit Analysis Using Benefit Transfer

Matthew Schoellhamer

A thesis submitted in partial fulfillment of the requirements for the degree of

Master of Science

University of Washington

2013

Committee:

Sergey Rabotyagov, chair

Joe Cook

David Layton

Program Authorized to Offer Degree:

School of Environmental and Forest Sciences

Abstract: It has been almost 20 years since the US Fish and Wildlife Service reintroduced wolves to Yellowstone and central Idaho, yet the controversy surrounding wolves in the western US remains as fierce as ever. Meanwhile, wolf populations across MT, WY, ID, and even WA have grown to the point where many states are shifting their goals from a focus on recovery to one on ongoing management. With this shift comes new questions about how to balance the various competing interests involved. In order to address some of these questions, this study looks at SW Washington, an area of the state so far unpopulated by wolves, and uses value estimates from previous research in an attempt to quantify the economic costs and benefits associated with the return of wolves to the Mt St Helens area.

Table of Contents

Background

Wolves in Washington State

Before European-American settlement, The Gray Wolf (*Canis lupus*) was common throughout most of Washington. However, a government-sponsored period of hunting, trapping and poisoning caused their numbers to decline rapidly between 1850-1920. By 1930, wolves were considered extirpated (locally extinct) in Washington. Then, after 2005, reports of wolf sightings in the state became increasingly common as wolves moved into the state from Canada, and west from Idaho. Then, in 2008, the first resident wolf pack in the state was confirmed. As of 2013, nine wolf packs have been confirmed in Washington.

As a result, in 2007 the state began developing a plan to manage wolves in Washington. Adopted in 2011, the plan establishes a recovery plan for wolves in the state. In order to be delisted, the state's wolf population needs to consist of a minimum of 15 breeding pairs present for a minimum of five years, with the following criteria:

- At least 4 pairs in Eastern Washington
- At least 4 pairs in the North Cascades
- At least 4 pairs in the southern Cascades/Northwest coast
- Three more pairs present anywhere in the state

As yet, none of the nine Washington packs have been located in the southern Cascades/northwest coast region, the state's only recovery region that lacks a single confirmed pack. However, excellent unused wolf habitat exists in this portion of Washington. Some of the most suitable habitat exists in a triangle that extends from Mt St.

Helens (MSH), north to Mt Rainier, and southeast to Mt Adams. Of particular interest are the forested slopes north and west of MSH, which support an abundance of elk.

Measuring Environmental Value

Classical microeconomics has a number of tools designed for estimating the value of goods exchanged in markets. However, it becomes more difficult to measure the value of many environmental goods and recreational activities, such as wildlife viewing, which do not meet this criterion (Freeman 2003). Economists have developed several methods for estimating the benefits associated with such goods. These methods fall into two primary categories: revealed preference models, and stated preference models (Freeman 2003). Revealed preferences use an aspect of individual behavior that does have a measurable price as a proxy for the value to be estimated. The most common of these are travel cost models, which use individuals' costs of travel to a destination to estimate the value of the location or activity they are engaging in. In stated preference approaches, the sample population tells the researcher the amount that they value the particular good. In the most common approach, called contingent valuation, respondents are surveyed to elicit their value estimates. Contingent valuation studies are the sources for all value estimates for non-market traded goods in this analysis, specifically the use and non use value of wolves, and the value of hunting.

The CV method involves using survey questions to establish a hypothetical market for the nonmarket good. The process includes three components:

1. A description of the good whose value is being measured.
2. The manner in which the respondent will be asked to pay for the good, called the payment vehicle. Typical options include new or increased taxes, and payments into a dedicated trust fund. Payments can be one time only, or ongoing.

3. The structure of the question that asks for the respondent's WTP. Options include open-ended (e.g. simply asking how much the person would pay), asking the respondent to choose from a written list of dollar amounts, and dichotomous choice, in which a person is asked to respond yes or no to a given price that the survey varies for each respondent.

The methods used for each of these components affect the quality of the overall analysis.

Perhaps the most intuitive issue is that the description of the good must be articulated to the respondent such that the respondent and the researcher share a common understanding of the good in question. For example, is the respondent valuing a population of wolves, or individual wolves? If the respondent is being asked to value wolf viewing, the researcher must convey the characteristics of that experience. Will participants be able to see wolves? See wolf sign? Only hear them? How likely will they be to experience any of these? Will wolves be easily viewable, or will doing so require journeying into the backcountry?

The type of payment vehicle used can also bias the result. Describing the payment as being a tax can make respondents provide lower WTP values than they would with an alternative payment vehicle (Loomis and White 1996). Also, respondents tend to better understand the value of the payment if it is offered one-time, rather than periodically, and therefore offer more accurate WTP estimates (Loomis and White 1996).

The earliest forms of CV surveys tended to use open-ended WTP questions. However, dichotomous choice tends to yield more accurate responses, as it more closely mimics the way most people purchase goods and services. That is, you wish to purchase something, an individual or firm names a price for the good, and you make a yes or no decision as to whether or not to purchase the item.

In addition to CV specific methodological concerns, all of the traditional questions associated with surveys apply to CV analyses. These include issues of sampling, and question design and order.

Over the years, CV has been met with its fair share of skepticism, particularly regarding whether the results of such a survey are a reliable and valid measure of what people would actually pay for a particular good. However, research has fairly consistently found CV results to be reliable and valid (Loomis 1989, Kling et al 2012), and it has been deemed appropriate and recommended by various federal agencies (US Water Resources Council 1983).

Benefit Transfer

In an ideal world, any economic analysis would use data obtained directly from the system under study. However, for a variety of pragmatic reasons, namely cost and time, this is not always possible. The alternative is referred to as benefit transfer: taking results from studies of a different location, activity, population, or time period, and applying those results to the research question. The process takes two primary forms: 1) value transfer, and 2) function transfer. Value transfer involves transferring a single value (usually a measure of central tendency) to the study site from a site (often called the transfer site) or sites. Functional transfer involves transferring a full function from a site to the study site, sometimes with some adjustment to fit the study site. The type of function can vary depending on the nature of the analysis. The functional transfer category includes meta analysis transfers, which combine the findings from multiple studies into a single function. More recently, researchers have increasingly turned to various Bayesian approaches for addressing some of the concerns with function transfers (Leon-Gonzalez 2008, Moeltner 2007).

In general, the benefits transfer literature is long on theory, but relatively short on specific recommendations and empirical support. Various sources recommend different takes on the following characteristics for more accurate benefits transfer (Rosenberger & Loomis 2001):

1. The original studies should use adequate data, sound econometric methods, and correct empirical techniques.
2. The relevant populations at the study site and the transfer site should have similar characteristics, such as ranges of income, age, and preferences for the goods in question.
3. The nature of the environmental good being measured, and the expected change in provision level should be similar at both sites
4. The characteristics of the transfer site and the study site should be similar.

These criteria seem reasonable at first glance, but they were developed with little empirical support. Furthermore, there still has been relatively little evidence to support how important they are, either in aggregate or relative to each other (Brouwer 2000). This makes practical application difficult. For instance, what if you have the option of using values that come from either a site with more similar user population characteristics to your study site, or from a site where the nature of the environmental good is a better match? How does one know which of those factors will lead to a better fit, or whether either will achieve an appropriate level of accuracy? The closest thing to direction one tends to find in the literature is that the analyst must determine if a transfer has the needed level of accuracy based on the specific situation and local context of the analysis (Rosenberg and Loomis, 2001). While true, this is not entirely helpful.

What the available literature does show is that the level of accuracy of benefit transfer studies can be quite low. Studies that have tested the validity of benefit transfer estimates have found the errors to be quite large (Rosenberger & Stanley 2006). Additionally, the research cautions that while function transfer is preferable to point transfer in a large number of cases, the benefits often rely on the ability to adapt the function in question to the particular characteristics of the policy site (Chattopadhyay 2003).

After reviewing the body of literature on benefits transfer, one is left with the distinct impression that its application remains at least as much art as science. The lesson for policy analysts is not to attempt to eliminate uncertainty, but to approximate its level and explain its impact on the question at hand.

Analysis

Estimating the Total Net Benefits of reintroducing wolves to the Mt St Helens (MSH) area entails a multi step approach. The steps, in sequence, are:

1. Identify the relevant benefit and cost categories that would result from a new wolf population in the MSH area (Table 1).
2. Identify the populations that have standing.
3. Identify primary studies of the economic costs and benefits of wolves to quantify the relevant values, specifying them to the MSH area, as applicable.

Table 1: Benefit and Cost Categories of Reintroducing Wolves to the MSH Area[1]

	Type	Impacted Population
Benefits		
	Non-use value of wolves	Park visitors, Washington residents, US residents
	Option value of wolves	Washington residents, US residents
	Use value of wolves	Park visitors
	Reduced cost of elk management	Washington residents
Costs		
	Reduced hunting opportunities	WA residents hunting in the study area, US residents hunting in the study area
	Reintroduction and management costs	Washington residents
	Reduced elk and deer viewing	Washington residents
	Livestock losses	Commercial ranchers in likely wolf habitat

Standing

As noted in the background section, while wolves in Western Washington are federally protected under the ESA, recovery in Washington State is not subject to any federal requirements (Wiles et al 2011). The only legal recovery requirements for wolves in the state come from their listing under as an endangered species in Washington, and the subsequent State Management Plan. Furthermore, USFWS currently is considering

[1] Underlined components were not quantified in this analysis.

delisting wolves across the lower 48.[2] Therefore, this study considers only those costs and benefits that accrue to Washington and its residents, although allowances are made for out of state visitors to MSH.

Some may consider this decision controversial. In particular, pro-wolf advocates may argue that wolves are a charismatic species of national interest, and that many other studies of the net benefits of wolves have taken a regional or national perspective (USFWS 1994, Rosen 1997). However, such studies dealt with wolf populations that were subject to Federal recovery requirements. They also analyzed areas that were major national and international tourist draws[3], and where a local wolf population would be the only such population within a very wide geographic area. Yellowstone receives roughly six times more viewers than does MSH. In addition, a MSH population would be just one of many in Washington State. While conceptually it makes sense to suppose that someone in Florida might place a significant value on having wolves in Yellowstone, it is much more tenuous to argue that such a person would place a significant and distinct value on wolves in MSH above and beyond their value for wolves in northern or eastern Washington. Given these concerns and the relevant legal and policy framework, this analysis errs on the side of caution and considers only the costs and benefits that accrue to Washington State.

On the benefits side, the analysis further restricts standing to only those current visitors to MSH. One of the chief methodological difficulties in estimating the benefits of wolves in MSH is that Monument visitors will also have access to a number of other wolf populations in the state. In contrast, other studies of wolf value (especially Duffield 1992) have been

---

[2] The small population of Mexican wolves located in the southwestern US would remain listed.
[3] Yellowstone and Smokey Mountain National Parks, respectively.

conducted in areas with few, if any, reasonable wolf viewing substitutes. Therefore, one would expect the value of wolves to be much higher in these locations than it would be in MSH. This analysis addresses this issue by considering only the value that would accrue to individuals already visiting the park. This approach likely understates the total value of MSH wolves to Washington. However, it provides strong support for the validity of the results should wolves pass the benefit cost test.

Discount Rate

MSH itself is one of five national monuments administered exclusively by the US Forest Service (USFS). In addition, MSH sits on the western edge of the Gifford Pinchot National Forest, making USFS an important partner in managing wolves is MSH, and across SW Washington. In order to allow the Forest Service to make ready comparisons with their other projects, this study uses the 4% real discount rate mandated for USFS projects in calculating Net Present Value. I also tested a 7% discount rate because it has been used in other studies of wolf value (EIS 1994, Duffield 1991), thus allowing for direct comparison with those results. 7% also is the discount rate used by USFWS, the agency that manages species under the Endangered Species Act, and therefore another important partner in Washington wolf recovery.

Time Scale

There is little theoretical basis for choosing an appropriate time scale for measuring the total net benefits of wolves. Certainly any study should consider the absolute lower bound to include the time it would take for wolves to reach a minimum viable population. Based on a discussion with a wolf biologist (Eisenberg 2011), I assumed that MSH recovery would involve a natural recolonization by wolves, as opposed to the federal reintroduction of wolves in Yellowstone and Central Idaho. Such a recolonization would be slower than a reintroduction, and would begin with a smaller initial population. I modeled this population growth using estimates obtained from prior wolf and ungulate research (see the section in this thesis: Wolf Population Growth and Impact on Prey Numbers). In order to include the entire period from the first returning wolves to a final, stable population, this thesis measures the net value of wolves over a 30-year period. As the costs and benefits of wolf are expected to accrue at different rates, it seemed prudent to evaluate at least one additional time period as well. I opted for a 50-year period, both because it is commonly used in other similar studies (Duffield 1991), and beyond 50 years any reasonable discount rate reduces the annual values to insignificant quantities.

WTP vs WTA

This study uses Willingness to Pay (WTP) to measure costs and benefits that result from a MSH wolf population. WTP is the amount one would be willing to pay to obtain a certain quantity of a particular good. However, studies have demonstrated that people will pay more to retain ownership over something than they will to obtain it in the first place (Kling et al 2012, Hanemann 1999). The amount one would pay to forego a loss is referred to as Willingness to Accept (WTA), and in theory it should be used in cases such as these, where people would have to relinquish some good that they hold a right to own or use. However, for practical, methodological reasons, economic valuation studies rarely measure value using WTA, so there is no pool of data to draw from. Simply put, it is difficult to measure, and most people are not used to being offered compensation, as opposed to having to pay for goods and services (Loomis 1996). This lack of WTA estimates can be problematic in cost benefit analyses where one side can be viewed as having to accept a loss while the other is paying for a gain. Fortunately, it is reasonable to assume that such concerns would not unduly impact the final conclusions in this analysis, as both the individuals receiving benefits and those receiving costs have a reasonable claim to have their value measured by WTA. In the case of wolf advocates, they likely see wolves in MSH not as a gain, but as a previous loss rightfully restored. Likewise, hunters and ranchers see the return of wolves as imposing losses on them, in the form of reduced hunting opportunities and livestock depredation.

Study Area

With the exception of existence value, both the positive and negative values of wolves stem largely from their location. Wolves in more easily accessible areas will tend to attract more visitors who want to see them. Those populations in areas with larger livestock populations will have more opportunities to predate on them. To identify the geographic boundaries of the study area, three factors were considered critical:

- **Habitat and population viability**. The study area must be sufficient to support a viable population of wolves.
- **Habitat quality**. Given an area capable of supporting wolves, wolves will prefer higher quality habitats to lower quality ones.
- **Economic trade offs**. As the purpose of this study is to compare the costs and benefits of wolf reintroduction to the MSH area, it need only consider those potential areas that would impact the answer to that question.

To determine the study area, I considered these factors in sequential steps: first identifying viable wolf habitat in the MSH area and the minimum number of wolves needed to support a viable population (viability). Then, within that pool of suitable habitat, I identified those areas of greater quality, assuming that wolves would prefer those areas. Finally, once I identified those areas of high habitat quality, I eliminated those regions where economic costs were likely to be much smaller than benefits, and therefore would not impact the final conclusion. These areas could then be reconsidered if the initial study area (effectively a worst-case scenario for MSH wolves) produced negative Total Net Benefits.

Viable Population Size

There is little agreement in the biological community as to what number of wolves constitutes a minimum viable population, even assuming that one already knew the characteristics of local prey, the likelihood of genetic exchange with other wolf populations, the local climate, the likelihood of disease, and the rate of human-related mortality, whether by poaching or vehicle accidents. Even then, many of these factors are impossible to estimate meaningfully for MSH without conducting primary research in the area. What little consensus exists in the literature suggests that 100 wolves is a reasonable estimate for a minimum viable wolf population in an isolated location (Fritts and Carbyn 1995). MSH, however, is not isolated, standing as it does on the edge of the Gifford Pinchot National Forest. Therefore, this study does not consider minimum population size for viability to be a concern in this case, and no necessary minimum population threshold was set for the analysis.

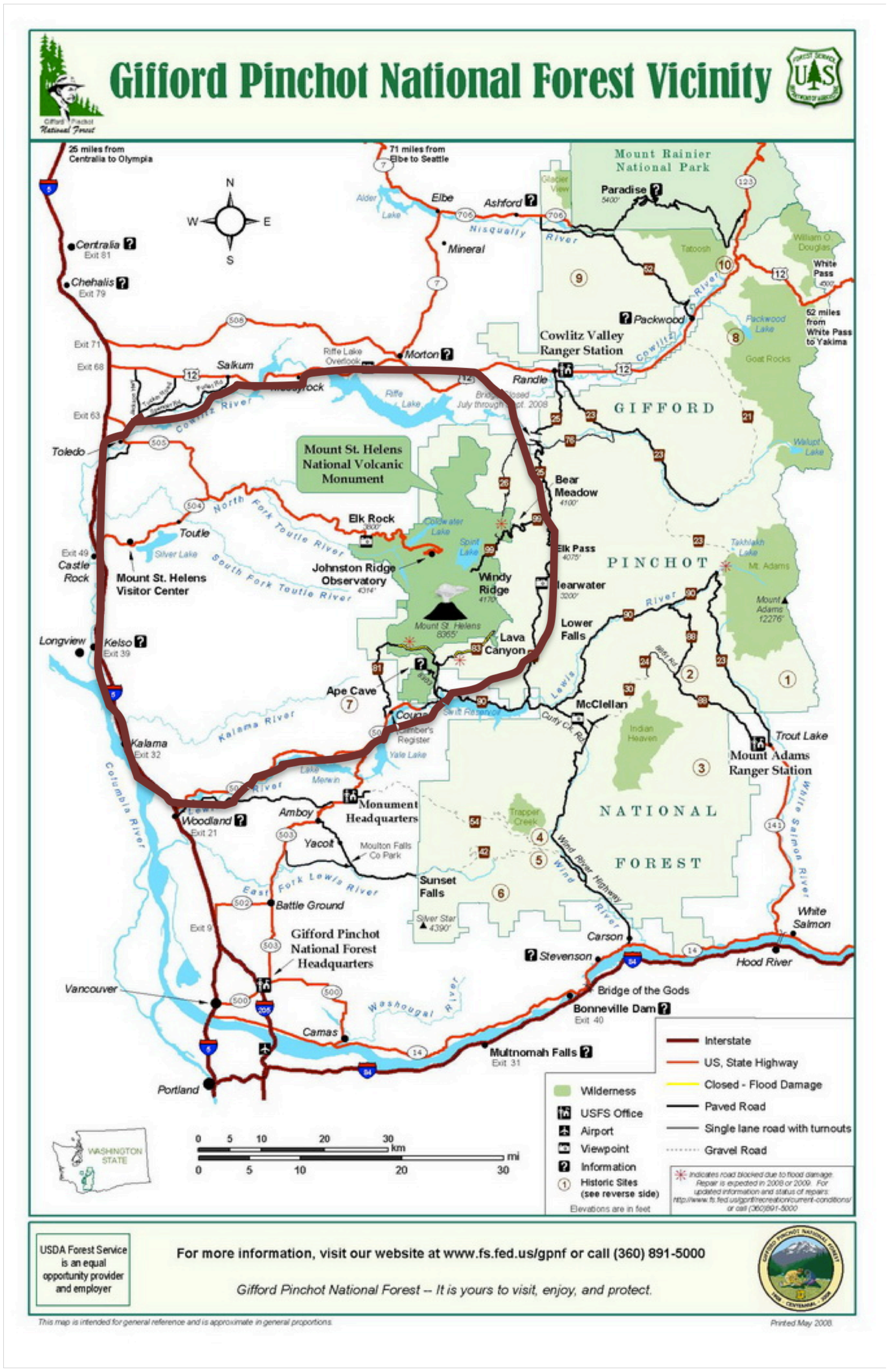
Suitable Habitat

Wolves are habitat generalists. They are found in almost every type of habitat available in North America, provided they have a sufficient prey base (Mech 1970). Other than prey availability, human activity seems to be one of the only factors limiting wolf habitat. In particular, researchers have found a correlation between lower road densities and a higher probability of land use by wolves (Mech et al, 1988, Mladenoff et al 2009).

Additional studies have used these factors to analyze wolf habitat viability across Washington and the Pacific Northwest (Larsen and Ripple 2006). Being state-wide and regional in scope, their methodologies are fairly coarse, but they do indicate that much of the area surrounding MSH on all sides is at least capable of supporting wolves.

Given the preference of wolves to avoid larger roads, this analysis somewhat arbitrarily bounds the study area using the major roads surrounding the Monument: US 12 to the north, US I-5 to the west, and WA 503 to the south (Figure 1).

Figure 1: Map of MSH Area, with Habitat Range Circled



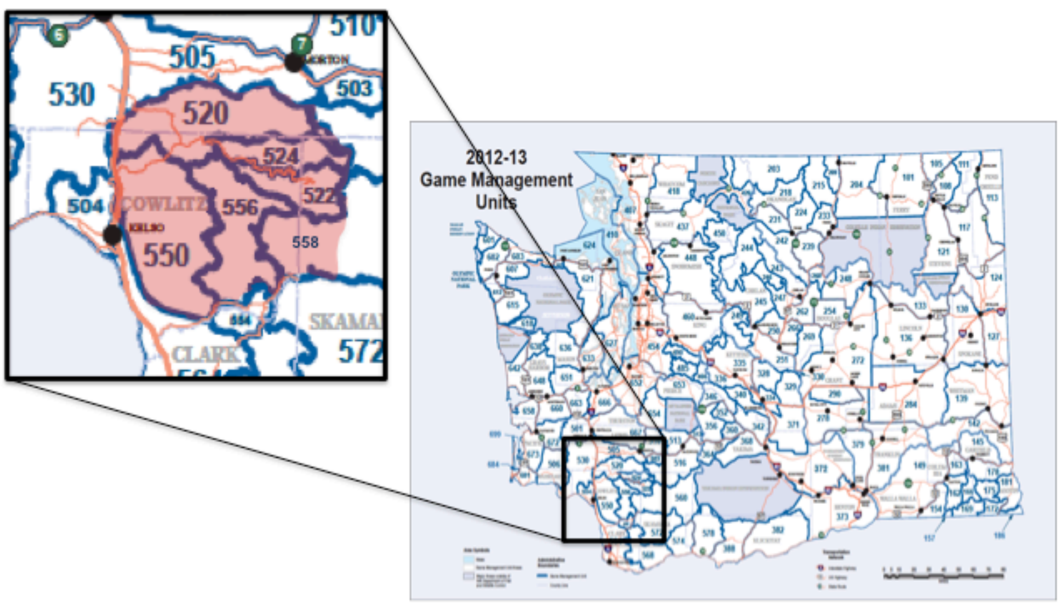
Prey Availability

As noted above, wolves rely on ungulates as their primary prey. In MSH, that means deer and elk, with elk being particularly important. The MSH elk herd is one of the largest in

Washington (WDFW 2006), and its presence is a key reason MSH is considered prime wolf habitat. Due to the likely dietary importance of elk, I considered elk availability to be the key predictive factor for future wolf population density.

WDFW tracks data for elk at the Game Management Unit (GMU) level. The MSH Elk herd is concentrated in the western portion of its range, in GMUs 520, 522, 524, 550, and 556. Therefore, these GMUs were used as the study area for this analysis (Figure 2), with the addition of GMU 558, which was incorporated into GMU 560 in 2006. The area encompasses approximately 1,300 mi^2, of which 172 mi^2 is the Monument itself.

Figure 2: Study Area



Ecology

The first step in estimating the total net benefits of wolves in MSH is to estimate the characteristics of the wolf population as it establishes, and the extent of that population's impacts on any aspects of other species that would impact overall economic value. The key elements for such a scenario are the size of a viable wolf population, how quickly it would grow to reach that size, and the location and extent of its range. Such questions are extraordinarily complex to answer precisely, with outcomes often varying dramatically under different local conditions, and what follows should by no means be considered a definitive estimate of wolf ecology in the area. It is a rough estimate designed to be a reasonable starting point for estimating the resulting economic values. The methodology in this section draws heavily from a feasibility study on the reintroduction of wolves to Olympic National Park conducted by USFWS (1999).

Background: Predator Prey Interaction

A large segment of wolf research has focused on wolf-prey interactions. Numerous studies have found that the presence of prey is a primary driver of the relative size of different wolf populations (Fuller 1989). However, the specifics of that relationship have proven to be much more difficult to untangle when one attempts to model wolf population dynamics.

Prey Availability

Given that this study models wolf numbers as a function of prey availability, the first step in the analysis was to estimate ungulate prey numbers for the MSH area. In the MSH area, elk and deer are the only available ungulate prey of any meaningful quantity.[4] Data on local elk and deer population numbers and spatial variability are derived from WDFW aerial surveys and surveys of hunter success rates.

Elk

Game species in Washington are managed at the Game Management Unit (GMU) level (Appendix 1: Map of Washington State Game Management Areas)

I applied the Sex Age Kill (SAK) methodology, used by both Bendner and Spencer (1999) and WDFW, to estimate the size of the MSH elk population. Data on elk bull/cow/calf

[4] While a population of mountain goats exists in the nearby goat rocks area, it is relatively small, and its elevation makes frequent contact with wolves unlikely.

ratios come from WDFW pre hunting season aerial surveys. Hunter harvest data come from WDFW surveys of elk hunters.

Following Bendner and Spencer (1999), I modeled the total elk population as a function of bull mortality and the age and sex distribution within the elk population. The model is defined as follows:

Equation 1

$$N_T = \left(\frac{K}{M_B * P_H}\right) * \left(1 + R_{C/B} + R_{C/B} * R_{C/C}\right)$$

where:

NT = Total number of elk.
K = number of bulls harvested.
M_B = the bull elk harvest rate.
P_H = the proportion of MB due to hunter harvest.
$R_{C/B}$, and $R_{C/C}$ = Preseason ratios of cows to bulls, and calves to cows.

WDFW conducts annual preseason surveys of elk to determine bull/cow/calf ratios, which must be maintained at certain minimum levels to meet hunting management goals. In order to use data for GMU 568 and ensure comparable data, I used survey data through 2004 for all available GMUs (WDFW). Such data are available from 1997, although not every studied GMU was surveyed in each year. In those cases where data were unavailable for any GMU for a particular year, I assumed the data for the available areas was representative for the whole study area. To determine the bull/cow/calf ratio for the study area, I summed the observed ratios for each GMU by year, and then averaged that total across the period 1997-2004. Per Bendner and Spencer (1999), I assumed a 50% annual mortality rate for elk bulls, and that 100% of bull mortality was due to hunter harvest.

Hunter harvest rates also come from WDFW data. The department requires all elk hunters to report the number of days spent hunting, whether or not they were successful in killing an elk, and the animal's gender. Data were available for all relevant years. The MSH area includes both special permit and general season hunting, and seasons for archery, muzzleloaders, and modern rifles. I aggregated harvest data for all permit and weapon types for the final analysis. I also assumed that bull mortality rate was equal to the bull harvest rate.

With these inputs, the estimated population varied significantly from year to year, with a high of 6,737 and a low of 2,062 (Table 2). The estimated average annual elk population of this area is 4,903 elk.

Table 2: Annual Elk Population and Hunting Data for MSH

	Sample Population from WDFW Aerial Surveys			Elk Harvest Data and Estimates			
Year	Bulls	Cows	Calves	Total Bulls Harvested	Annual Bull Mortality Rate	Proportion of Bull Mortality Rate Due to Hunter Harvest	Total Study Area Elk Population
1997	202	556	253	349	50%	100%	3,493
1998	210	507	174	243	50%	100%	2,062
1999	150	416	155	648	50%	100%	6,229
2000	228	507	243	546	50%	100%	4,684
2001	216	504	260	523	50%	100%	4,746
2002	307	562	262	627	50%	100%	4,620
2003	230	470	221	831	50%	100%	6,655
2004	215	583	201	725	50%	100%	6,737

Average Elk Population in Study Area	4,903
Standard Deviation	1,622

Deer

Data for deer population in the study area were obtained directly from WDFW (WDFW, unpublished data). Data were unavailable for GMU 522, although the population there is quite low, and was assumed to be insignificant (conversation with Eric Holman, WDFW Region 5 Biologist). WDFW estimates these populations using the SAK method. As for elk, I summed the total population for the GMUs in the study area by year, and then averaged the population across all years. For GMU 558, the available range was 1991-2005. For all other GMUs the available range was 1991-2011. The average deer population of the study area is 14, 954 (SD=4,793).

Wolf Population Growth and Impact on Prey Numbers

The relationship between wolf populations and prey populations is one of the most studied aspects of wolf ecology. However, local variability has made broadly applicable conclusions difficult to draw. While research has demonstrated quite conclusively that ungulate numbers and behavior are extremely important in determining wolf numbers, and vice versa, the exact nature of the relationship has been elusive. Key questions to consider are:

- Do wolves limit prey numbers, or regulate them?
- Are wolf kills of ungulates primarily additive (occur in addition to other forms of mortality), or compensatory (wolves tend to kill animals who would have died anyway from some other cause)?
- How many ungulates will wolves kill, on average?
- In systems with multiple ungulate species, such as MSH, how do wolves exhibit preference for certain ungulate species?

Answering each of these questions is complicated by the enormous range of factors that influence the answer in a given location, including:

- Prey population characteristics.
- Wolf population characteristics.

- Population characteristics of other predators.
- Nature of interaction between carnivore populations.
- Impact of human activities on all predators and prey.
- Habitat characteristics.
- Weather characteristics, including variability.
- Predator and prey behavior, including breeding, feeding, and any migratory behavior.

It is especially important to consider the range of variation in these factors across ecosystems, because of the differences between MSH and the locations for previous wolf studies. Due in large part to extirpation efforts, there is a shortage of wolf research from areas of the US and Canada where elk are a primary prey source. Simply put, for much of the last century, healthy and relatively unmanaged wolf populations generally were found in systems that rely either on deer, caribou, or moose. Furthermore, most of that work that has been done in elk systems has been done in the northern Rockies since the 1994 federal wolf reintroduction. While those areas have a similar, though far from identical, prey base to MSH, they possess dramatically different predator populations. In particular, these ecosystems include grizzly bear populations, which themselves can exert tremendous pressure on elk and deer populations, primarily through calf mortality.

Given these concerns, and the fact that the goal of this report is to understand the broad-scale economic impacts of wolves, rather than fine-scale ecological changes, this analysis employs a variation of the Lotka-Volterra predator-prey model. This model provides reasonable enough levels of accuracy, while also remaining simple enough to be easily interpreted, and requiring fewer assumptions in identifying inputs.

Eberhardt (1998) modeled wolf-prey interaction using a difference equation variation of the Lotka-Volterra model. This model uses two equations: a "functional response"

(Equation 2) that describes the impact of predators on the prey population, and a "numerical response" (Equation 3) that describes the effects of prey numbers on the predator population.

Equation 2

$$V_t = V_{t-1} + r_1 V_{t-1}\left[1 - \left(\frac{V_{t-1}}{K}\right)^Z\right] - cH_{t-1}$$

Equation 3

$$H_1 = H_{t-1} + r_2 H_{t-1}\left(1 - \frac{H_{t-1}}{aV_{t-1}}\right)$$

Table 3: Variable Definitions and for Elk, Deer and Wolf Population Model

Variable	Definition	Value used	Source
a	The number of prey required to support one wolf under equilibrium conditions	272 deer-equivalent units[5]	Fuller et al 2003
c	Estimated number of ungulate kills per wolf	8 each for elk and deer	Various
H_t	Wolf population size at time t	1 at t=0	-
K	The study area's carrying capacity for ungulates	29,663 deer-equivalent units[4]	WDFW
r_1	The maximum finite growth rate of the ungulate species	Elk: 1.35 Deer: 1.213	USFWS 1999
r_2	The wolf population growth rate	1.48	USFWS 1999
t	Time period, in years from beginning of analysis	Varies	-
V_t	Ungulate population at time t	K at t=0	WDFW
Z	A scaling parameter that regulates the shape of the ungulate density dependent response to predation	5	Eberhardt 1998

Assigning an elk population growth rate and a value for Z was relatively straightforward. Following the ONP study, I used a value of 5 for Z.

[5] 1 elk = 3 deer

Raedeke et. al (1986) found a maximum population growth rate of 1.35 for elk in Mt St Helens. In addition to being from the study area, the value is also within the range of other studies of elk from the Rockies and the Pacific Coast (Bomar 1999), although it is noticeably higher for values from elk populations on the Olympic Peninsula. Given that it falls within the range of accepted values, and it comes from the study area, it was used unmodified. For deer, I used the growth rate of 1.213 that was used in the ONP study.

To determine the number of ungulate kills per wolf, I averaged kill rates across numerous North American wolf studies (Mech and Peterson 2003).[6] Wolf kill rates very widely across studies, and there is little in the way of evidence identifying the underlying factors that are responsible. Researchers have noted that kill rates are lower for wolves that prey primarily on deer than they are for those that prey primarily on elk, deer, caribou, or moose (Schmidt and Mech 1997). A leading theory for this difference is that deer are smaller, and therefore wolves are more likely to have fully consumed a kill before researchers can locate it (Kolenosky 1972). However, studies that track wolves on the ground report similarly lower values. Another consideration is that the wolf populations that feed on elk tend to live amongst very different predator populations. Most importantly, they tend to coexist with grizzly bears. Grizzlies can move wolves off a kill (Mech and Peterson 2003), so perhaps wolves in these areas are forced to kill more elk in order to consume a similar amount. As an elk-rich system with no grizzly bears, there was no obvious cause to choose either a lower or a higher kill rate, so I averaged across all

[6] Note that kill rate is different from consumption rate. As noted, it is the amount killed, usually expressed in kg/wolf/day. Consumption rate is the amount consumed, which is much lower than kill rate due to scavenging by other species, and the fact that not every part of the prey is consumable.

studies, for an estimated kill rate of 5.43 kg/wolf/day (SD = 2.64), or a yearly rate of 1,981 kg/wolf/day.

The averaged values, as is almost always true in wolf research, are winter values, when the biomass of ungulates killed by wolves is highest. Therefore, many wolf advocates have argued that these estimates likely overestimate the number of ungulates killed when they are used to determine the number of annual kills. While that is certainly possible, studies have founded that even in the summer ungulates often represent 75% or more of the biomass killed by wolves (as compared to roughly 100% in the winter). Furthermore, wolves tend to prioritize ungulate fawns during the warmer months relative to winter. So even though the proportion of ungulates in the wolf's diet might be lower in summer than in winter, the number of individuals killed in summer should decline little, if at all. It may even increase.

Translating the estimated annual kill rate into ungulate population reductions requires estimating two additional factors. The first is the rate of selectivity of different age classes within both ungulate populations, and the average weight of animals in those classes. The second is to estimate the relative level of selectivity wolves will show between elk and deer.

To answer the first question, I adapted the methodology of the ONP report (1997), in large part because of similarities in the local elk populations. North America contains two subspecies of elk: Roosevelt Elk and Rocky Mountain Elk. Roosevelt Elk are larger than Rocky Mountain Elk, and are the only species found on the Olympic Peninsula. The Rocky Mountains contain only Rocky Mountain Elk. However, the MSH Elk Herd contains genetic stock from both Rocky Mountain and Roosevelt Elk (WDFW), making estimation of elk size relatively complicated. While no one knows precisely how Roosevelt and Rocky Mountain Elk have occupied the area and/or interbred, WDFW (2006) estimates that Roosevelt Elk predominate in the western portion of the herd, while Rocky Mountain tend to occur in the eastern portion of the range. As the study area primarily covers the western portion of the herd's range, I assumed that the area elk population of elk is comprised of Roosevelt Elk. Therefore I used the average elk weights from ONP of 385 kg for bulls, 249 kg for does, and 91 kg for calves. I also applied the estimated weights for deer used by the ONP study: 69 kg for bucks, 50 kg for does, and 26 kg for fawns. Using a selection ratio of 1.3:1:2 for bucks/does/fawns, I calculated a weight of 217 kg for the average elk killed by wolves, and 45 kg for the average deer killed by wolves.

In calculating the ratio of the total kill rate that would be composed of elk and deer, I assumed a 1:1 ratio of selectivity, per relative biomass available.

These inputs resulted in an estimate of roughly eight elk and eight deer killed per year per wolf in MSH.

The final step was to estimate the number of prey (in deer equivalent units) per wolf under equilibrium conditions. Calculated in deer equivalent units, with one elk assumed equivalent to 3 deer, previous studies (Fuller, USFWS) have used Fuller's average of 209. However, that average has crept up as biologists have begun to research the recovering wolf population in the northern Rocky Mountains, with its higher kill rates and lower wolf densities. I used an updated average of 272 deer equivalent units per wolf (Fuller et al 2003).

Given an assumption that the MSH wolf population would colonize the area naturally, I assumed an initial population of one wolf. I then used Microsoft Excel to run the model until the deer, elk, and wolf populations stabilized, or until any of the three populations declined to zero. The wolf population stabilized in year 20, with all three populations having stabilized by year 30. The final populations are 3,901 elk, 14,160 deer, and 95 wolves (Table 4). Those values represent a14% decline in the elk population, and a 5% decline in the deer population.

Table 4: Modeled Changes in MSH Wolf, Elk, and Deer Populations

Year	Wolves	Deer	Elk
1	1	14,954	4,532
5	5	14,930	4,517
10	28	14,806	4,428
15	82	14,388	4,115
20	95	14,165	3,910
30	95	14,160	3,901
50	95	14,160	3,901
100	95	14,160	3,901

Economic Value

Some individuals value the existence of wolves or the opportunity to see and/or hear wolves in the wild. However, wolves also impose costs on society, typically in the form of reduced hunting opportunities, killing livestock, and the costs of management. This section attempts to identify the relevant costs and benefits of wolves recolonizing MSH, and to assign those costs and benefits appropriate values. The overall goal is to determine whether the total benefits exceed the total costs.

Benefits

A wolf population in MSH has the potential to provide many benefits to Washington State. Because of a lack of information on how state residents might value MSH wolves separately from the rest of the state's wolf population, this study measures only the value of wolves to MSH visitors. Two types of visitor values are relevant: use values, and non-use, or existence, values. This study does not subdivide existence value into existence and bequest value. I used a contingent valuation study of visitor value conducted prior to wolf reintroduction to Yellowstone (Duffield 1992) to estimate value quantities.

Data

One of the most critical and difficult aspects for estimating benefits and costs in any study is finding data for an appropriately comparable situation. This is a particular problem in determining the value of wolves, as very few studies have been conducted, with only Duffield (1991) measuring value for site visitors independently. Of additional concern is that there are reasons to believe that Yellowstone is not especially comparable to MSH. Yellowstone is far more popular than MSH, attracting approximately 6x as many annual visitors. Furthermore, there are reasons to think that the two visitor populations would have very different Willingness to Pay (WTP) functions for wolves. Wildlife viewing is a primary draw for visitors of Yellowstone, whereas MSH tends to attract visitors for its scenery, especially the volcano itself. In addition, the surrounding populations are very different. MSH is relatively close to Seattle and Portland, two large cities with populations that are far more pro-wolf than are the communities within several hours of Yellowstone, and which have much higher average incomes. The two landscapes are very different, in a way that could have a significant impact on the value of wolves at each location. While Yellowstone has extensive forests, those forests are relatively open, and much of the rest is broad grassy valleys that allow visitors to see wildlife at a great distance. MSH has relatively open terrain in the blast area along Johnson Ridge Rd on the north side of the park that might be similarly ideal for viewing wolves. However, much of the likely wolf habitat stretches west from MSH itself, and is thickly forested, which could make wolves very difficult to see, and lower their use value.

Despite these shortcomings, Duffield has strengths as well. In particular, the study methodology is sound. Surveyors were able to work with the Park to stop drivers at the park entrance for questioning, helping the collection of a representative sample of summer visitors. The sample size of completed surveys was relatively large (N=762), with a high return rate of 86%. The payment vehicle is a trust fund, which biases the results less than when participants are asked to pay a hypothetical tax. Finally, the survey asks participants to make a one-time payment into the trust fund. This is preferable to an annual payment, as respondents may have difficulty properly estimating their value for a good for each year of a long-term study such as this one.

There also is a more fundamental concern regarding the validity of transferring Duffield's value estimates to MSH. At the time of that study, the only wolf population in the US existed in the upper Midwest.[7] It is possible that Yellowstone visitors in the early 1990s derived more value from a wolf population than would MSH visitors in the 2010s because they considered wolves critically endangered, and viewed recovery efforts in the PArkas

[7] A small, resident population was established in NW MT at the time, but it is likely that very few respondents in Duffield's study would have known about it.

critical for the species' very survival, both in the Park and in the country.[8] Therefore, it is possible that Duffield's values substantially overestimate the likely values of MSH visitors.

Despite the validity of these concerns, there are sound reasons to consider that Duffield's estimates are appropriate to use in this case. For one, my analysis transfers the median WTP estimates from Duffield, which reduces the importance of large, positive WTP outliers compared to using the mean estimate. In addition, Chambers and Whitehead (2003) calculated WTP for a wolf population in MN that offers some findings for comparison. Average WTP for residents outside of the study area, the group most likely to be similar to Yellowstone visitors who support wolf recovery, was $25.12 per person (2012 dollars). That is smaller than the $39 found by Duffield, but one would expect the value to be larger for visitors than for the broader sample of state residents used by Chambers (2003). While there is no clear method for quantifying what that difference should be, Chambers and Whitehead's estimate is quite close to Duffield's estimate for the existence value of wolf supporters ($27). Duffield also conducted a follow up study in Yellowstone in 2006, after wolves had become well established in the Park and across the northern Rockies. Despite the difference in the regional wolf population, he nonetheless found very similar WTP values among park visitors to those in 1992. The value ranged from $22-40 per visitor depending on the season, with only two of the six values lower than $29 (Duffield et al 2006).

Given the needs of this study, there is no question that Duffield (1992) provides the best available estimates for visitor value of wolves. However, the comparability concerns

[8] Certainly, wolf populations worldwide were quite healthy at the time. Nonetheless, Yellowstone visitors likely still would have perceived restoration as critical for a threatened species' survival, due to the extirpation of wolves from the US portions of their historical range.

between the study site and the policy site are real and potentially significant. In an attempt to combat this problem, this study calculates three separate Net Present Value estimates: one using total benefits from Duffield, and one each using only use and non-use values. The theory is that by considering the results using existence-value benefits only, it is possible to eliminate all of the problems associated with possible differences in the value of viewing wolves in MSH and Yellowstone.

MSH visitor data comes from the National Visitor Use Monitoring survey (Graefe 2002), a program of the US Forest Service and from The Monument's own visitor data. NVUM program data for MSH come from 2002. The monument estimates number of visitors using vehicle cameras located on multiple roads leading to the park. The cameras track number of vehicles, and the park assumes 2.4 visitors per vehicle (MSH National Monument, based on Graefe 2002). Some of the camera locations are outside park boundaries, but all are within the study area boundaries.

Methodology

There are approximately 500,000 visitors to the Monument area annually (MSH unpublished data). Since Duffield uses one-time payments, visitor value in future years is a function of the number of visitors that year who had never been to the area before. Graefe (2002) used NVUM data to determine percentage of first time ever visitors for the Monument (45%), the Mt St Helens Ranger District (32%) and the Gifford Pinchot National Forest as a whole (19%). I used the estimate for the Ranger District, as it includes the Monument and portions of the surrounding area, similar to the study area. Finally, I used Duffield's (1992) findings that 68% of Yellowstone visitors would support wolf reintroduction, and transferred the result to MSH unmodified.

Upon calculating the relevant visitor population at MSH, I estimated the use and non-use values associated with wolf reintroduction by transferring values from Duffield (1992) without modification. These values are derived for a wolf population, unrelated to the number of individual wolves. I transferred the median value estimates, as they were the most conservative, and considered by the original study to be the most appropriate treatment for the outlier values (Duffield 1992), and converted the total to 2012 dollars. For existence value, this produced a median total WTP estimate of $26.93 per person per year for MSH visitors, and a median use value of $12.09 per person per year.

However, Duffield estimated the value of a reintroduced population, with a breeding pair and an estimated initial population of 10 wolves. In contrast, this study assumes a natural recolonization by MSH wolves, with a more gradual buildup to the establishment of the first breeding pair. In contrast to Yellowstone, NW Montana was colonized naturally, in a manner likely similar to MSH (Eisenberg, personal communication). Therefore, I assumed that a MSH wolf population would establish its first breeding pair in year seven, as happened in NW MT, and I assumed that benefits would begin accruing at that point.

Benefits not Included

This study does not include measurements for several possible benefit sources. These are: option values, additional value from changes in Monument visitiation, reduced elk management costs resulting from wolf recovery at MSH, and increased biodiversity. This section details why each benefit was left out.

Increased MSH Visitation

In order to be conservative in determining whether a MSH wolf population would pass a benefit cost test, this analysis assumes no additional visitation for the park. The reason for this is that wolf populations likely will already be well established in many other portions of the state by the time they recolonize MSH, and therefore current and potential future visitors would have numerous additional locations for viewing wolves. In contrast, visitors to Yellowstone had many fewer alternative locations for viewing wolves, at least in the initial years of wolf recovery. Therefore they likely placed a much greater value on the opportunity than a potential new visitor to MSH would. Because of the paucity of evidence for how to adjust MSH visitation estimates due to this difference, I elected to remove this value category entirely. However, note that only the extent of this value is uncertain, not its sign. The addition of wolves would very likely lead to some increase in MSH visitation, however small, and therefore some additional increase in economic value over the estimate in this thesis.

Option Values

Option value is the value an individual derives from having the option to realize some benefit in the future. In this case, individuals might not be willing to pay for the opportunity to see wolves at MSH now, but might value maintaining the option to observe them at some future date. There was a practical reason for leaving out option values, as no relevant study estimated values for them for wolf viewing. However, they also are difficult to measure, and their use is particularly controversial.

Reduced Elk Management Costs

WDFW estimates that the MSH elk herd is too large, and harms the local ecosystem and agriculture through overgrazing (WDFW 2006). As a result, the agency employs various control measures to reduce the overall population. However, the bulk of these harmful impacts happen well outside of the study area, in areas with higher levels of development.

In addition, while no research has emerged on this topic yet, some managers have theorized that elk might adapt to the presence of wolves by spending more time in more heavily developed areas as a safety measure. If true, introducing wolves might actually increase the cost of limiting elk damage. Since both the sign and magnitude of this value are highly uncertain, and that value is likely to be relatively small, it was excluded from the analysis.

Increased Biodiversity

Elk and deer can have a negative impact on an area's biodiversity by over browsing, which can reduce habitat availability for certain species in certain cases. Reducing elk herbivory therefore could have a huge economic benefit, especially if it improves the value of birding in an area, which studies have found to be a relatively high value activity. In the last decade, some researchers have found that wolves can achieve this benefit not only be reducing ungulate numbers, but by changing their behavior (Ripple and Beschta 2004), causing them to move around more and not damage any one area as much. However, this research is new, and fairly controversial. Given that the scientific community is still attempting to determine whether this effect even occurs, let alone estimating its value reliably, I left it out of the analysis. However, if the theory turns out to be valid, it could form a large new category of economic value associated with wolves. This is a research area that all wildlife policymakers and analysts should monitor closely, but for now there remains sufficient uncertainty to exclude it.

Reduced Livestock Predation by Coyotes

The reintroduction of wolves to Yellowstone has corresponded with a significant decline in the area's coyote population. Coyotes, in turn, are a significant livestock predator in

Washington, responsible for 600 confirmed cattle deaths and 500 confirmed sheep deaths in 2004-2005 (WDFW 2012). For comparison, that represents the same number of cattle, and more sheep, than were killed by cougars and bobcats over the same period. It is possible that wolf reintroduction to MSH could cause a similar decline in area coyote populations, which might in turn have some economic benefit for local ranchers in the form of reduced livestock losses. However, no evidence exists for the connection between wolf predation on coyotes and reduced livestock predation, so this potential benefit was not included in the analysis.

Costs

In addition to benefits, wolves impose costs on society. The costs of wolves included in the analysis are negative existence value, lost value of hunting, losses from livestock predation, and administrative costs associated with managing a wolf population.

Negative Existence Value

Negative existence value exists as a result of the reduced utility people experience from knowing that there are wolves in the MSH area.

Data

Duffield (1992) estimated negative existence value for visitors to Yellowstone. After asking respondents whether or not they supported wolf recovery, those who answered no were asked if they would pay to block reintroduction. Payment vehicle characteristics and survey design are otherwise identical to those asked of supporters.

Methods

I applied Duffield's (1992) estimates to MSH in the same manner as I applied the study's benefit estimates. The same visitation estimates were used, only this transfer required estimating the number of visitors to MSH who would oppose wolf recovery. In Duffield

(1992), that estimate is 32% of visitors, and the study estimates that for those visitors the total median WTP to prevent wolf reintroduction is $2.87 per visitor, in 2012 dollars. As in the case of benefits, that is a one-time payment. Therefore, costs accrue in future years only for first time visitors. Also as in the case of benefits, costs are assumed to begin accruing when the first breeding pair establishes, rather than when wolves first move into the area.

Lost Hunting Value

Hunting is an extremely valuable industry in Washington, and SW Washington is one of the State's premier locations to hunt, especially for elk. The introduction of wolves to the area should decrease the economic value of its hunting. Estimating the extent of this impact involves a two-part process:

1) Estimate the impact wolves will have on hunting opportunities in Washington.
2) Estimate the total value of those impacts.

As noted in the Biology section, to date there has been no evidence that wolves have caused reduce deer hunting levels in any part of their US range, including areas where one would reasonably expect wolves to cause greater deer population declines than would be likely in the MSH area. Therefore, this analysis considers it very likely that wolves will have a negligible impact on deer hunting in the study area. Only potential impacts on elk hunting are included in the analysis.

The impact of wolves on elk hunting opportunities has been extremely contentious ever since advocacy groups and USFWS proposed reintroducing wolves to Central Idaho and Yellowstone. However, it has become even more so in the last decade, as the population of the Northern Yellowstone elk herd has declined precipitously. Of course, wolves are not the only variable that has changed in that time, with a series of above average winter snow packs likely playing an important role in the population decline (Barber-Meyer et al 2008).

As noted in the Biology section, there already is significant uncertainty about the impact wolves have had on elk populations themselves. Attempting to take the next step, and model how those impacts go on to impact hunter value, is even more complex. After all, wolves may impact elk hunter values in many different ways, beyond the obvious impact imposed by changes in elk populations. Wolves may alter the behavior of elk in a way that impacts hunter value, for instance pushing them into terrain that is more difficult for hunters for access. It is possible that wolves have a psychological impact on hunters. This impact could go in either direction: hunters could derive more value from recreating in a "wilder" environment populated by wolves, or hunters might fear hunting in wolf habitat or resent a wolf presence, leading to reduced value.

All of these theories are possible, but other than a link between elk population numbers and hunter value, no meaningful evidence exists either in favor of or against any of them. Therefore, this study considers only the impact that changes in elk population will have on hunter experience.

Estimates for the relationship between hunting and elk numbers have been varied. Miller (1982) regressed Washington hunter survey demographic data on geographically weighted elk population numbers. The study estimated that a 10% reduction in elk populations statewide would lead to an 11% reduction in hunter days. The ONP study assumed a 20% reduction in revenue for a 16-17% reduction in the elk population (USFWS 2006), albeit in an area where elk populations are lower relative to management goals than they are in the

MSH area. However, in contrast to such modeled estimates, wolf recovery in the Northern Rockies has had very little impact on the amount of elk hunting in the region. In Montana, the only losses of hunter opportunities have occurred in three management areas (MFWP 2007). Even then, those reductions were for antlerless elk permits only. Wyoming has experienced no loss in elk hunting. Idaho has experienced similar results to Montana, with 23 of 29 management zones at or above elk population goals as of 2010 (IDFG 2010).

Applying these numbers to MSH only increases the level of uncertainty. As noted in the Biology section, the ecosystems of the Northern Rockies are very different from Western Washington. Summer and winter climate, rainfall, number of predator species, and ungulate and predator population numbers all differ substantially from the MSH area. Elk hunter participation in Washington is also quite different than in the three northern Rockies states, with lower levels overall, and much lower levels of participation by out of state residents. Therefore, I concluded that it was more important to use value estimates for Washington than more recent, out of state values. I used Miller's findings to estimate the relationship between elk population losses and reduced hunting. I assumed the relationship between percentage decline in elk herds and decline in hunter days was linear, and therefore conducted the analysis using a 1.1 percentage point drop in hunter days for each percentage point decline in elk population.

Using this estimate, total elk hunter days in the study area would be expected to decline from 49,300 to 40,608, a decline of 17.6%.

Economic Value of Hunting

While less popular than in the surrounding states, elk hunting nonetheless remains a popular activity in Washington. Several reports, including the WA wolf plan (2012) and ONP analysis (1999), have considered only potential declines in this revenue in their analyses of economic impacts of wolves on the state. This approach substantially underestimates the economic value of hunting, as additional value accrues to the hunters above what they pay the State to hunt.

Studies of elk hunter value have used the range of methods typically found in studies of the value of recreation, including contingent valuation, and travel cost methods. This field of research also spans a much greater range of geographic areas than research on the economic value of wolves, allowing far more transfer site choices than for the value of wolves. In selecting an estimate for the loss of hunter value, it is important to note that hunter participation in Washington is very different than it is in other elk hunting states. In Washington, out of state hunters represent less than 2% of total hunters in the state and 0.1% of the hunter days. In contrast, the other Western States experience much higher participation rates by out of state hunters.

These facts suggest that using a value estimate from Washington State would be preferable to an out of state estimate. USFWS conducted contingent valuation surveys of economic value of wildlife-related recreation, including elk hunting, in 1980, 1985, 1991, 1996, 2001, and 2008. Unfortunately, USFWS dropped Washington from the Elk Hunter survey following the 1985 report. In addition, the 1985 survey used open-ended valuation questions, as opposed to the more accurate dichotomous choice questions adopted in the 1991 survey. Despite these concerns, given the strong likelihood that elk hunting in

Washington has very different characteristics than in the other western states, this study uses the 1985 estimate of $102.77 (2012 dollars)[9] per hunter day. To move from this estimate to total value, I used the following equation (Duffield 1992):

*(number of original hunter days in MSH area) * (percent reduction in hunter days) *($ willingness to pay for each hunter day).*

Between 1997 and 2005, the study area averaged 8,934 hunters and 49,300 hunter days per year. The estimated loss of 9,131 elk hunter days (18.5%) over the first 30 years calculated above represents a loss of over $938,000 per year in hunting value once the estimated MSH wolf population stabilizes.

Losses from Livestock Predation

While livestock losses from wolf predation have been an extremely contentious political topic, all available evidence suggests that the economic impact of wolves is negligible at any scale larger than the individual ranch level. The economic impact in the study area likely would be small even by this standard, as the area is a minor producer of livestock compared to the rest of the state. The USDA Census (2007) reports only 5,088 cattle and sheep combined in all of Cowlitz County, the county comprising the private land areas of the study area. Livestock impacts, like prey impacts, are difficult to transfer. The problems in the two categories are similar, with the high variability of important local conditions hindering easy comparison. However, data on confirmed livestock predation cases in the northern Rockies suggest that kill rates are highly uncorrelated to such seemingly important variables as the number of wolves. Due to the political importance of livestock

[9] $48 annually in 1985 dollars in the original analysis

predation, this study attempts to quantify its costs despite the likely low value, and the complexities involved.

Data are available for livestock predation rates by wolves in six locations in the Western US and Canada: Alberta, British Columbia, NW Montana, Minnesota, Yellowstone National Park area, and Idaho. None of these sites is a very good comparison for MSH. All either have different predator or prey species, and they all offer more opportunities for wolves to encounter livestock. As a result of the fact that no one study is demonstrably preferable, I averaged predation rates across all of the studies.

Value estimates for lost livestock come from estimates of the animals' market value. Since wolves often kill younger animals, before they would be sold, it is important to take into account the future value of the animal. This analysis uses the same value estimates (converted to 2012 dollars) as WDFW's Wolf Management Plan (2012). While other livestock are important in other areas of the state, cattle and sheep are the only livestock of any significant population present in the study area. Cattle are assumed to be worth $743 per head, which is the average 2004-2007 fall value of 600 lb steer calves (WDFW 2012, from Livestock Market Information Center). The value of sheep was set at $152 per head, based on the average price for sheep sold in all size and weight classes in Washington in 2007 (WDFW 2012, from USDA 2007).

Management Costs

The primary difficulty in estimating management costs is that while some level of expense is essential, every possible spending category has a discretionary component. Often it is difficult to separate the essential from the discretionary. For this analysis, management was taken to include the following categories:

- Monitoring the wolf population
- Protecting the wolf population
- Managing wolf-livestock conflicts, including proactive deterrents[10]
- Outreach and education

While many states maintain a record of the annual management costs of wolf reintroduction, data for specific sub populations are less readily available. Therefore it seemed preferable to adapt data from a state whose growing population was restricted to one region. I elected to use the data from Wisconsin's wolf management program (WDNR 1999), as the state's wolves have been relatively restricted to one portion of the state since reintroduction.

I developed a simple, univariate linear model to estimate the relationship between Wisconsin's annual management costs, and that state's wolf population, estimating the parameters using OLS regression in the R software package. In addition to a model where the wolf variable was unmodified, I ran two additional variations where I took the natural log of the wolf variable, and cubed the wolf variable, respectively.

[10] This does not include compensating ranchers. That value is already included in this analysis as livestock losses.

Table 5: OLS regression of management costs (in 2012 $) on number of wolves in Wisconsin

$C2012 = \ln(Wolves) + \varepsilon$

	Coefficients			
	Estimate	SE	t value	Pr (>|t|)
Intercept	20103	54300	0.370	0.7161
Ln(Wolves)	33850	14709	2.301	0.0352 *

(*) = significant at 0.05

Residual standard error: 46390 on 16 degrees of freedom

Adjusted $R^2 = 0.2017$

Due to the large degree of variance in the data, none of these models proved a much better fit than any of the others, with adjusted R^2 values between .2 and .22. Therefore, I elected to employ the model that took the natural log of the wolf variable (Table 5), as the assumption that each wolf's impact on management costs would get smaller as the population grew is theoretically sound.

I calculated the cost of managing a MSH wolf population by plugging the wolf population estimates into the final model.

Costs not Included

This study does not quantify all benefits and costs associated with introducing wolves to MSH (Table 1). Costs left out of the study include: reduced utility from reduced viewing of prey species, reintroduction costs, and negative existence value for all WA residents.

Reduced Elk Viewing Opportunities

Many people enjoy viewing elk and deer in the wild, and it is possible that any reduction in their population caused by wolves would impose a cost on society. However, research has concluded that it is extremely unlikely that anyone would notice a difference from the degree of reduction caused by wolves (Duffield 1992). Therefore the analysis does not include a cost estimate in our analysis

Reintroduction Costs

It is quite possible that WDFW would expend some amount of resources to facilitate wolves relocating to MSH. However, such efforts likely would have extremely low marginal costs. The most likely approach would be to improve connectivity measures that would facilitate wolves crossing I-90. However, such measures are being constructed already for other species of concerns, meaning the cost imposed by wolves would be minimal to nonexistent.

Negative Existence Value of WA Residents

As with existence value, this report estimates values for visitors only, despite the fact that some Washington residents likely do have negative existence value for wolves at MSH. However, quantifying this value faces the same concern as positive existence value, and therefore it has been left out of the final analysis. Given that statewide existence values have also been excluded, and likely exceed statewide negative existence values, the choice to ignore both still constitutes a conservative approach to valuing wolves.

Results

This analysis reveals substantial positive net benefits to Washington State from reintroducing wolves to MSH (over both time scales at a 4% discount rate (Table 6). There also is a large, positive value for total net benefits for all other tested discount rates. Benefits even exceed costs if one were to include only use, or only existence benefits on the benefit side of the calculus.

Table 6: Total Value of Costs and Benefits, 4% Discount Rate, 30 and 50 Year Time Scales

	30 Years	50 Years
Benefits		
Existence Value	$45,124,899	$54,903,912
Use Value	$19,821,357	$24,116,841
Total Value	$64,946,256	$79,020,656
Costs		
Negative Existence Value	$452,149	$550,134
Hunting Cost	$5,920,170	$10,009,028
Livestock Cost	$132,867	$197,288
Management Cost	$2,227,200	$2,992,632
Total Net Benefits	**$56,213,870**	**65,271,670**
Total Net Benefits, Existence Value Only	$36,392,513	41,154,829
Total Net Benefits, Use Value Only	$11,088,673	10,367,759

Sensitivity Analysis

Estimating an average value tells only part of the story. Especially in a case like this with multiple estimates relying on each other, and large uncertainty around many of those estimates, decision makers should also consider the range of likely values around the point estimate. As in the body of the analysis, the first step in the sensitivity analysis must be to estimate the range of the populations of deer, elk, and wolves within the study area.

To estimate the range of possible population growth rates, I first took into account that several of the key variables appeared to be linked. Especially as more research has been conducted on the rapidly growing wolf population in the northern Rocky Mountains, it has been noted that these and other wolf populations that feed in large part on elk and other large ungulates tend to have much larger kill rates than do populations that feed primarily on deer. In addition, those populations tend to have lower wolf/ungulate biomass ratios than the deer dependent ones. Therefore, rather than a continuous range of likely outcomes, I considered three distinct scenarios that varied in their estimated kill rates and ungulate biomass/wolf ratios. For both factors, high and low estimates are plus/minus one standard deviation above or below the analysis values for kill rate (estimate = 5.43 kg/wolf/day, SD: 2.64) and prey biomass (estimate: 272 deer equivalent units/wolf, SD = 133). For each scenario, I tested low and high elk population growth rates. Varying the deer population growth rate had a minimal impact on the results, and therefore was not included in the final analysis.

Table 7: Inputs changed for three sensitivity analysis scenarios

Scenario		Kill Rate	Ungulate Biomass per Wolf	Elk Population Maximum Growth Rate
1	Low	2.79	139	1.25
	High	2.79	139	1.42
2	Low	5.43	272	1.25
	High	5.43	272	1.42
3	Low	8.06	405	1.25
	High	8.06	405	1.42

Table 8: Wolf population estimates and TNB for sensitivity analysis scenarios

Scenario		% Decline in Elk Population	Final Wolf Population	TNB (millions)	
				30 yrs	50 yrs
1	Low	34%	181	$54.1	$61.5
	High	11%	197	$57.5	$67.0
2	Low	34%	93	$53.0	$60.0
	High	11%	101	$57.3	$66.9
3	Low	33%	62	$52.4	$59.3
	High	11%	68	$52.4	$59.4

Adjusting these parameters produced wide variations in the wolf population estimates, ranging from a final population of 62-197, and more modest variations in the ungulate populations. However, variations in TNB are almost nonexistent, ranging from $52.4-$57.5 million over 30 years, and $59.3-67.0 million over 50 years.

Given that the primary goal of this analysis is to test whether or not Total Net Benefits are positive, I selected the scenario with the lowest TNB for additional testing. The scenario tested was a high kill rate, and low elk population maximum growth rate. Using Crystal Ball, I set probability distributions for multiple model inputs (see Appendix 2: Sensitivity Analysis Parameters for a complete list), and ran a 10,000 trial Monte Carlo simulation (Table 9).

Table 9: 95% confidence intervals for TNB, 4% discount rate (in millions of dollars)

	30 Year		**50 Year**	
	Lower Bound	Upper Bound	Lower Bound	Upper Bound
TNB, Use Value Only	-$1.4	$12.4	-$8.2	$10.7
TNB, Existence Value Only	$18.5	$44.2	$16.4	$49.3
TNB	**$33.3**	**$69.6**	**$34.7**	**$80.2**

At the 95% level and a 4% discount rate, TNB are positive when all benefits are considered and when only existence value is considered. When only use benefits are considered, the confidence interval does dip below zero. TNB in this case will be positive 94% of the time at 30 years, and 64% of the time at 50 years.

As a final test, the present value of the total costs considered in this scenario is $12.5 million over 30 years, and $19.7 million over 50 years. Given a Washington State population of 6.9 million (US Census Bureau 2013), Washington residents would only need to place an average value of $2.9 each on the value of a MSH wolf population. That is barely more than half of the average value Chambers (2003) found residents in MN wolf country placed on wolves, a WTP of $5.53. It is dwarfed by the value residents outside of wolf country placed on wolves, an average WTP of $25.12. Unlike Duffield's estimates, these values were calculated for a pre-existing population, and at a time when wolf recovery in the US was much further along. Therefore, these values likely are more conservative, as respondents are less likely to interpret their choice as being necessary to preserve wolves in the country.

Discussion

Based on this analysis, there is considerable evidence that Washington State would benefit from a Gray Wolf population in the MSH area. Total net benefits are both very large on average, and likely to be positive. The latter should speak volumes to decision makers concerned with the uncertainty surrounding some of the estimates used. While that uncertainty is real, and important, the benefits involved so vastly outweigh the costs that considering the full range of possible outcomes does not impact the conclusion. Under the range of possible outcomes tested in this thesis, the only way to find negative TNB for wolves in MSH is to consider only benefits from visitor use value, an approach not supported by theory or empirical evidence.

This large gap between benefits and costs is in spite of many attempts to err on the side of caution when considering the value of the benefits of wolves. For instance, it is very likely that Washington residents who do not visit MSH will accrue additional value with the presence of wolves. Certainly they will have other wolf populations in the state available to them, but they also are likely to place a much higher value on wolves than residents in Duffield's study. Montana, Wyoming, and Idaho have nothing close to the number of large urban centers that Washington has; places that tend to support wolf conservation. Furthermore, this analysis assumes that MSH visitation rates will remain constant. However, given that the park is not seen as a prime destination for wildlife, it is likely that the arrival of wolves would attract a relatively new class of visitor to the park, and that visitation would increase at least somewhat.

In addition, the population dynamics estimates, while rough, are comparable to other findings. The ONP feasibility study estimated a stable population of 56 wolves for an area roughly 30% larger and more lightly-roaded, albeit with lower prey densities than MSH. The estimated final, stable population of 95 wolves represents a wolf density of approximately 14 km^2/wolf in the study area[11]. That is slightly outside the range (15-219 km2/wolf) of North American wolf densities reported by Fuller et al (2003) for all prey types. However, as a higher wolf density overestimates costs relative to benefits, it is preferable for the purposes of this analysis to air on the side of overestimating the wolf population.

Despite these findings, it is important to exercise some caution in applying them. The most immediate concern is that the values used in this study assume that all individuals of standing hold the same value of a dollar. In reality, this is not true: all else equal, the more money you have, the less you value each individual dollar. This fact is particularly relevant to the study question, because money is not generally evenly distributed between those

[11] This does not account for the fact that portions of MSH, namely the flanks of the volcano itself, are largely uninhabitable to wolves.

who pay the costs of wolf recovery and those who reap the benefits. The study area likely follows this pattern, with visitors who support wolf recovery likely coming from areas of the state with higher incomes than affected hunters and ranchers. Given the large positive values associated with wolf recovery, it is unlikely that this would affect the conclusion, but any analyst relying on these results would be wise to consider its relevance to their particular problem.

There also is a larger question of applicability. Given the lack of economic data on the costs and benefits of wolves, it may be tempting to apply the broadest conclusions of this analysis (that society values wolves highly) directly to other scenarios. As hopefully has been made clear throughout this report, this would be unwise. Wolves, like most species, exhibit enormous variability in their behavior, even over relatively small geographic distances. That wolf value appears to be net positive in MSH should not be taken as conclusive evidence that wolves would be a net benefit across their potential range. Even within Washington, the likely costs and benefits might vary greatly in different parts of the state. For instance, NE Washington has a much larger livestock population, and total costs of livestock losses are likely much higher there than they would be in MSH.

Furthermore, total net benefits are only one piece of evidence in a larger policy question. The range of other aspects that should be considered in a topic so complex as wolf policy is large, and a complete handling them is well beyond the scope of this analysis. However, a few topics are of particular relevance: the geographic distribution of the costs and benefits, the nature of the risk to livestock owners, and the different impacts the various wolf population scenarios have on the specific cost benefit categories.

Duffield (1992), and Chambers and Whitehead (2003) have both found that those who benefit from wolves tend to live outside of wolf habitat, while those who pay the costs of wolves tend to live in wolf habitat, or at least closer to it. This divide might be even greater in SE Washington, as it is relatively close to Seattle and other large urban centers that are likely very supportive of wolf recovery. While irrelevant from the point of view of this study, such divides might be of interest to advocates and decision makers.

Also worth considering is the specific nature of the impact wolves have on ranchers. Despite frequent political battles on the subject, wolves in North America actually kill very few livestock. However, when they do, they often continue targeting a particular ranch, rather then spreading out to surrounding operations. Therefore, while the total societal cost is quite low, and generally irrelevant for studies such as these, the cost to an individual rancher can be enormous. Wolves thus offer a low probability- potentially catastrophic risk to all of the farmers in their habitat.

Finally, the three scenarios tested in the sensitivity analysis had little to no impact on the final summative analysis, but they have very different impacts on the underlying sources of value. The largest cost categories are hunting losses and management costs. The changes made in the scenarios tend to offset each other primarily because of their impact on hunting value: larger wolf numbers are able to exist because they are eating fewer deer and elk. This, in turn, keeps hunting costs from rising with the wolf population. However, there is no such control in place on livestock predation. Therefore, the scenarios that predict larger wolf populations may impose a much larger cost on ranchers. This outcome was not sufficient to impact the conclusions of this report, but someone interested in outcomes for ranchers or outreach involving that community would be wise to consider it.

Conclusion

Always tempestuous, wolf policy has exploded in recent years, making headlines nationwide. USFWS has removed the northern Rockies population of wolves from endangered species protection, and wolves across the rest of the West may follow shortly. Montana and Idaho have already held wolf-hunting seasons for several years. Populations across the Rockies are stable. The population in Washington has been growing steadily, and packs have established in NE Oregon. One famous individual wolf was tracked all the way to California. California and other states that don't yet have wolves are starting to develop state wolf management plans. Those states with wolves, including Washington, are in various stages of moving from recovery to management. It may surprise many Americans, who are used to wolves as a formerly extirpated species, but wolves are resilient, versatile animals. Short of widespread, intensive, government sponsored eradication programs, wolves are here to stay, and in an increasingly large and diverse range of habitats and locations. And as wolves become more ubiquitous, there is the possibility that the heated political debates surrounding the species will only intensify. Coupled with the large variability in wolf behavior and impacts, these factors place real pressure on policy and management agencies to begin to consider both the ecological and social ramifications of wolves on the increasingly fine scale required of wolf management. While there is plenty of biological research to be done, we must remember the importance of the human side of wolf management. As in the recent case of the elimination of Washington's Wedge Pack, the policy problems posed by wolves are not necessarily getting easier in the wake of delisting. It would behoove managing institutions to be prepared for these new challenges in the years ahead.

Works Cited

Barber-Meyer, SM, LD Mech, and PJ White. 2008. Elk calf survival and mortality following wolf restoration to Yellowstone National Park. *Wildlife Monographs*. 169:1-30.

Bendner, LC, and RD Spencer. 1999. Estimating elk population size by reconstruction from harvest data and herd ratios. *Wildlife Society Bulletin*. 27(3): 636-645.

Bomar. et al. 1999. Final draft: feasibility study on the reintroduction of gray wolves to the Olympic Peninsula. Dept. of Fish and Wildlife Resources and the Idaho Coop. Research Unit, University of Idaho.

Bolon, NA. 1994. Estimates of the values of elk in the Blue Mountains of Oregon and Washington: evidence from existing literature. Gen. Tech. Rep. PNW-GTR-316. Portland, OR: US Department of Agriculture, Forest Service, Pacific Northwest Research Station. 38 p.

Brouwer, Roy. 2000. Environmental value transfer: state of the art and future prospects. *Ecological Economics*. 32:137-152.

Carbyn, LN. 1980. Ecology and management of wolves in Riding Mountain National Park, Manitoba. Canadian Wildlife Service. Edmonton, Alberta.

Carbyn, LN. 1983. Wolf predation on elk in Riding Mountain National Park, Manitoba. *Journal of Wildlife Management.* 47:963-976.

Chambers, Catherine M, and John C Whitehead. 2003. A contingent valuation estimate of the benefits of wolves in Minnesota. *Environmental and Resource Economics* 26:249-67.

Chattopadhyay, Sudip. (2003). A repeated sampling technique in assessing the validity of benefit transfer in valuing non-market goods. *Land Economics.* 79(4): 576-596.

Drovers Cattle Network. Spring 2012 cattle price outlook. Online at http://www.cattlenetwork.com/cattle-news/Spring-2012-cattle-price-outlook-139465678.html. Checked on 6/4/12

Duffield, John W. 1992. An economic analysis of wolf recovery in Yellowstone: Park visitor attitudes and values. In *Wolves for Yellowstone? A report to the United States Congress. Vol. 4, research and analysis,* eds. John D. Varley and Wayne G Brewster, 2-87. Yellowstone National Park, WY: National Park Service, Yellowstone National Park.

Duffield, J., Neher, C., & Patterson, D. 2006. Wolves and people in Yellowstone: impacts on the regional economy. University of Montana, Final Report for Yellowstone Park Foundation.

Eberhardt, LL. Applying difference equations to wolf predation. *Canadian Journal of Zoology*. 76: 380-386.

Eisenberg, Cristina. 2011. Personal conversation. 1, May.

Freeman, M. 2003. *The Measure of Environmental and Resource Values: Theory and Methods*. Washington, DC. Resources for the Future.

Fritts, SH, and LN Carbyn. 1995. Population viability, nature reserves, and the outlook for gray wolf conservation in North America. *Restoration Ecology*, 3(1): 26-38.

Fuller, Todd K. 1989. Population dynamics of wolves in north-central Minnesota. *Wildlife Monographs*. 105: 3-41.

Fuller, Todd K, David Mech, and Jean Fitts Cochrane. 2003. Wolf Population Dynamics. In *Wolves: Behavior, Ecology and Conservation*. Eds. L. David Mech and Luigi Boitani. Chicago, IL: University of Chicago Press

Graefe, AR, et al. 2002. Recreationists on the Gifford Pinchot National Forest: a survey of user characteristics, behaviors, and attitudes. Final report submitted to the Gifford Pinchot National Forest, USDA Forest Service, Region 6, Portland, OR. November 11, 2002.

Hanemann, M. 1991. Willingness to pay and willingness to accept: how much can they differ? *The American Economic Review*. 81(3): 635-647.

IDFG (Idaho Department of Fish and Game). 2010. Study shows effects of predators on Idaho elk. *Idaho Fish and Game News*. 22(2):1-4

Kling, Catherine L, Daniel J Phaneuf, and Jinhua Zhao. 2012. From Exxon to BP: has some number become better than no number? Journal of Economic Perspectives. 26(4).

Kolenosky, GB. 1972. Wolf predation on wintering deer in east-central Ontario. Journal of Wildlife Management. 36:337-69.

Leon-Gonzalez, R and R Scarpa. 2008. Improving multi-site benefit functions via Bayesian model averaging: a new approach to benefits transfer. *Journal of Environmental Economics and Management*. 56: 50-68.

Larsen, T and WJ Ripple. 2006. Modeling gray wolf (Canis lupus) habitat in the Pacific Northwest, USA. *Journal of Conservation Planning*. 2:17-33.

Loomis, JB and DS White. 1996. Economic benefits of rare and endangered species: summary and meta-analysis. *Ecological Economics*. 18(1996): 197-206

Mech, LD. 1970. *The Wolf: the Ecology and Behavior of an Endangered Species*. Natural History Press, Garden City, NY.

Mech, LD, SH Fritts, and D Wagner. 1988. Wolf distribution and road density in Minnesota. *Wildland Society Bulletin*. 16: 85-87.

Mech, LD, and Rolf O Peterson. 2003. Wolf Prey Relations. In *Wolves: Behavior, Ecology and Conservation*. Eds. L. David Mech and Luigi Boitani. Chicago, IL: University of Chicago Press

Miller, JR. 1982. Game availability and hunter participation: a study of Washington elk hunting. *Annals of Regional Science.* 16(1): 79-94.

Mitchell, R.C., and R.T. Carson. 1989. *Using Surveys to Value Public Goods: the Contingent Valuation Method*. Resources for the Future, Washington, D.C.

Mladenoff, DJ, MK Clayton, SD Pratt, TA Sickley, and AP Wydeven. 2009. Change in occupied wolf habitat in the northern Great Lakes region. In *Recovery of Gray Wolves in the Great Lakes Region of the United States: an Endangered Species Success Story*. Wydeven, TR, Van Deelen, and EJ Heske, editors. Springer, New York, NY.

Moeltner, Klaus, Kevin J Boyle and Robert W Paterson. 2007. Meta-analysis and benefit transfer for resource valuation: addressing classical challenges with Bayesian modeling. Journal of Environmental Economics and Management. 53: 250-269.

Raedeke, KJ, EH Merrill, and SM McCorquodale. 1986. Estimates of intrinsic growth rates in three elk populations in Washington. In *Proceedings of the Western States and Provinces Elk Workshop*. Ed. Eastman, DL. Oregon Department of Fish and Wildlife, Oregon, USA.

Ripple, WJ, and RL Beschta. 2004. Wolves and the ecology of fear: Can predation risk structure ecosystems? *Bioscience*. 54(8): 755-65.

Rosen, W. 1997. Recovery of the red wolf in northeastern North Carolina and the Great Smokey Mountains National Park: an analysis of the social and economic impacts, report submitted to the US Fish and Wildlife Service and the Point Defiance Zoo and Aquarium, 31-6. *Wolves of America Conference Proceeding*.

Rosenberger, Randall S, and John B Loomis. 2001. Benefit Transfer of Outdoor Recreation Use Values.

Rosenberger, Randall S, and Tom D Staley. 2006. Measurement, generalization, and publication: sources of error in benefit transfer and their management. *Ecological Economics*. 60:372-378.

Schmidt, PA, and LD Mech. 1997. Wolf pack size and food acquisition. *American Naturalist*. 150:513-517.

US Census Bureau. 2013. State and County QuickFacts. Online at http://quickfacts.census.gov/qfd/states/53000.html as of 6/12/13.

US Department of Agriculture (USDA). 2007. Census of Agriculture. Washington, DC.

US Department of Interior, Water Resources Council. 1983. Economic and environmental principles for water and related land resources implementation studies. US Government Printing Office, Washington, DC.

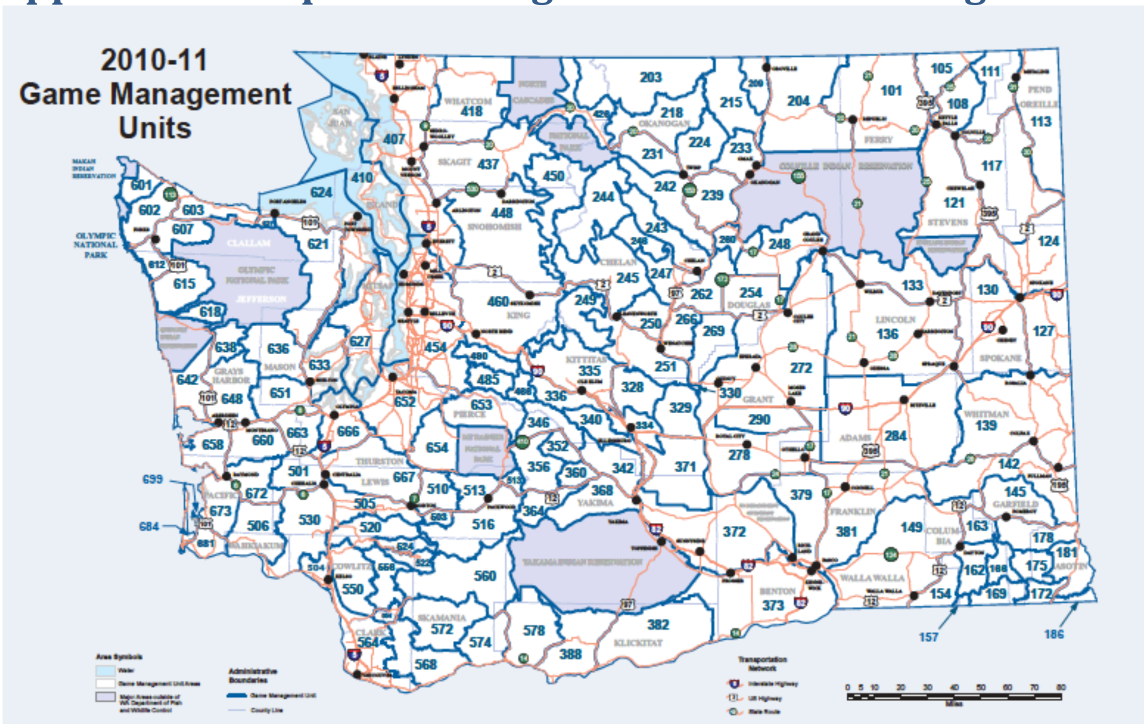
US Fish and Wildlife Service (USFWS). 1994. *The reintroduction of gray wolves to Yellowstone National Park and central Idaho, final environmental impact statement*. Helena, MT: US Fish and Wildlife Service.

Washington Department of Fish and Wildlife (WDFW). 2006. *Washington State elk herd plan: Mt St Helens elk herd*. Washington Department of Fish and Wildlife, Olympia.

WDNR (Wisconsin Department of Natural Resources). 1999. *Wisconsin Wolf Management Plan*. Madison, WI.

Wiles, GJ, HL Allen, an GE Hayes 2011. *Wolf conservation and management plan for Washington*. Washington Department of Fish and Wildlife, Olympia. 297 pp.

Appendix 1: Map of Washington State Game Management Areas



Appendix 2: Sensitivity Analysis Parameters

Parameter Type	Parameter Name	Point Estimate	Distribution Type	Distribution Parameters
Park Visitor Benefits and costs	**Annual Visitors**	500,000	triangular	Min: 450,000 Max: 550,000
Park Visitor Benefits and costs	**% Respondents Favoring Wolf Recovery**	68%	normal	St Dev: 7%
Park Visitor Benefits and costs	**% First Time Visitors**	32%	normal	St Dev: 3%
Park Visitor Benefit	**Willingness to Pay (Supporters)**	$39.02	normal	St Dev: $3.3
Park Visitor Cost	**Willingness to Pay (Opponents)**	$2.87	triangular	Min: $1.39 Max: $4.35
Management Cost	**Annual Government Management Costs (year 1)**	Varies by number of wolves	triangular	Calculated from model
Hunting Cost	**Hunter Success Rate**	8.2%	triangular	Min: 5.2% Max: 12.6%
Hunting Cost	**Elk Harvest**	716	triangular	Min: 356 Max: 988
Hunting Cost	**Total Elk Hunter Days/ Year**	49,300	triangular	Min: 42,829 Max: 80,647
Hunting Cost	**Value of Hunter Days Lost**	$103	normal	St Dev: $10
Lost Livestock	**Cattle Killed per Wolf per Year**	0.07	triangular	Min: 0 Max: 0.157
Lost Livestock	**Sheep Killed per Wolf Per Year**	0.15	triangular	Min: 0 Max: 0.32
Lost Livestock	**Value of Cattle**	$743	normal	St Dev: $74
Lost Livestock	**Value of Sheep**	$152	normal	St Dev: $15