University of Nevada, Reno

The Effect of Invasive Annual Plants on Wildfire Suppression Costs

A thesis submitted in partial fulfillment of the requirements for the degree Master of Science in Resource and Applied Economics

By

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May, 2010

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University of Nevada, Reno
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We recommend that the thesis
prepared under our supervision by

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entitled

The Effect of Invasive Annual Plants on Wildfire Suppression Costs

be accepted in partial fulfillment of the
requirements for the degree of

MASTER OF SCIENCE

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May, 2010

ABSTRACT

The problems posed by the role of invasive plants on changing wildfire regimes is considered to be the greatest threat to the ecological integrity of the Great Basin sagebrush steppe of the western U.S. Ecological changes that occur because of these problems threaten the economic and social fabric of rural communities throughout the region. The rest of the nation is affected by these changes through the increasing costs of wildfire suppression that is covered by federal land management agencies. The cost of wildfire suppression has increased at an increasing rate over the last several decades; to the point where funds that would have been used for preventative fuel management programs are being diverted to cover expenditures of wildfire fighting. Preventative fuel treatment is an effort such as thinning of brush and young trees, herbicidal treatment of invasive grasses, controlled fires to reduce fuel loads, and other actions to reduce the probability that a wildland fire will be a catastrophic wildfire that results in further expansion of invasive grasses, losses to ecological integrity, and more frequent wildfires. The trade-off between preventative fuel treatment efforts and wildfire fighting expenditures is generally well understood in principle. However, in practice, the feedback that continues to result in increasing wildfire costs continues. At this time there is little information about the benefits of preventative fuel treatment efforts. One category of benefits is in reducing the expected value of wildfire suppression expenditures. The purpose of this research is to develop a model to estimate the wildfire suppression cost savings from management actions intended to reduce fuel loadings from invasive plants in the Great Basin.

Wildfire suppression cost is estimated using historical data from the US Forest Service regarding the costs of large wildfires on Great Basin lands managed by the Forest Service and the Bureau of Land Management between 1995 and 2007. These data include vegetation type and condition. This information along with an index of fire danger rating is used to construct an index of weighted fuel types. This is used in a regression model to estimate wildfire suppression costs as a function of vegetation type, resulting in the marginal wildfire suppression cost contribution for each given vegetation type. The regression coefficients on vegetation type are used with treatment success rates and wildfire event probabilities to conduct a cost savings analysis for a 200 year period. Public land managers can utilize processes of this type to determine how limited resources can be allocated to maximize cost savings. The framework developed is highly adaptable and can be used to analyze impacts of other activities such as ranching, recreation, and development.

DEDICATION

This thesis is dedicated to my grandmothers, Oma Gayle Gross and Oma Marty Landis. I'm blessed to have had such strong, smart and loving women in my life. You both continue to inspire me every day. I love you both more than words can express.

This thesis is also dedicated to my father Jim Landis. I'm glad we got the chance to celebrate this accomplishment together, Daddy. To borrow a phrase from Che, you have lived magnificent days. I love you.

ACKNOWLEDGEMENTS

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Chapter 1.INTRODUCTION

The sagebrush biome of the western United States is threatened by the interrelated problems of invasive plants and an acceleration of the wildfire cycle (Pellant 2004; Knapp 1996). This arid high elevation desert is sparsely populated yet the past and current actions of people who interact with the land have a profound impact on the future of this fragile landscape. Ranching, development, and recreation have introduced and spread invasive plant species that out-compete native plants, provide a fuel source that has shortened wildfire cycles and increased the severity of wildfire events (Pimentel 2005; Knapp 1996). Changes in fire regimes further the proliferation of invasive exotic weeds by reducing resiliency of native plants (RMRScience 2009; Pendleton 2007). Despite the negative impacts resulting from human interaction (Mack 2000; Zouhar 2003), appropriate land management decisions are the best hope in fighting the proliferation of invasives and regaining a healthy, productive ecology (Dale 2009).

The consequences of invasive plants surfaced slowly over time. The most ubiquitous invasive plant in the region is cheatgrass (*Bromus tectorum*), a native of the Eurasian Steppe that is now found throughout Great Basin Rangelands. The first sighting of cheatgrass in the Great Basin occurred in Utah in 1894 and was directly linked to human activity associated with railroad lines (Knapp 1996). During the same era federal wildfire management began with the United States Calvary fighting the 1886 Yellowstone fires (Pyne 1996). Just fourteen years earlier Yellowstone was established as a national park and a wildfire management strategy of aggressive suppression was adopted (Rothman 2007). Two decades later in 1905 the United States Forest Service

(USFS) was formed (USFS 2010) soon found wildfire suppression to be their key objective. The convergence in time of cheatgrass detection and wildfire management foreshadows the impact of land use and land management choices, particularly in terms of invasive species and wildfire.

Though a variety of invasive plants threaten the Great Basin biome, cheatgrass is the most ubiquitous.[1] In 2004 an estimated 56 million acres across seventeen western states were infested with cheatgrass, which is spreading at a rate of fourteen percent annually (Duncan 2004). Cheatgrass is an annual grass that germinates, goes to seed and dies earlier than native plants, out competing native plants for water and nutrients. Native rangeland plants cannot thrive in areas that have become dominated by invasive weeds. Because cheatgrass typically dies off by mid June, the remaining dense stands provide a source of rangeland wildfire fuel earlier than would occur otherwise. These early season cheatgrass-fueled fires further weaken native plants, spread quickly to form larger wildfires than would occur otherwise, and contribute to soil disturbances encouraging spread of cheatgrass post wildfire (Rice 2008). Cheatgrass dominated areas are prone to erosion and disrupted hydrological activity (Spaeth 2007; Pellant 2004). Cheatgrass roots are much smaller than those of native perennials which contributes to soil erosion and minimal decomposition of plant material in the arid environment, further diminishing soil fertility. In areas affected by cheatgrass, overgrazing further compromises native plant vigor by increasing soil disturbance and over consumption of plant matter (McKnight 2008). Indeed, cheatgrass initially entered the Great Basin

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[1] This is currently true. *Taeniatherum caput-medusae*, otherwise known as "medusahead", has already invaded the Great Basin. While currently at lower population numbers than cheatgrass, medusahead out-competes cheatgrass under specific soil conditions and offers virtually no nutritional value to grazers and wildlife (Archer, 2001).

ecosystem by filling a niche vacated by natives damaged due to unregulated grazing practices.

The aggressive nature and biological structure of cheatgrass contribute to fire regime changes resulting in more frequent and intense wildfires (Zouhar 2003). As early as 1965 scientists noted that invasives were 10 percent to 500 percent more likely to burn in a given year than native grasses and that the overall wildfire season had increased by three months (Knapp 1996). Areas infested with cheatgrass eventually burn often enough so that the 30-110 year rangeland wildfire cycle typical of native plant communities (Rice 2008) is shortened to 3 to 5 years (Pimentel 2005). Cheatgrass is a flash fuel, meaning that it catches fire easily and burns readily. As cheatgrass multiplies it builds a dense ground cover providing more flash fuels thus increasing wildfire probability as well as crowding out struggling native plants (Brooks 2008). This cycle repeats, accelerating fire and furthering cheatgrass proliferation.

Historically, cheatgrass has been found primarily on flatter, lower elevation acreage but is now adapting and encroaching on steeper slopes and higher elevations (Devine 1993). Higher elevations, home to piñon-juniper vegetation, contain greater quantities of fuels that once infested by cheatgrass create an environment with more severe fire potential (Zouhar 2009; Keane 2008). Both piñon pine and juniper have large, dense plant structures that provide longer burning wildfire fuel compared to grasses, forbs, or shrubs. A healthy piñon-juniper environment also contains native shrubs, forbs, and grasses. Smaller grasses and forbs are flash fuels. The risk of severe wildfire increases when larger fuel types are interspersed as flash fuels can serve as tinder for the

larger fuels. Once cheatgrass encroaches piñon-juniper stands it out competes native grasses and forbs and the cycle of dominance takes effect (Zouhar 2009). This is particularly troublesome given the presence of large fuels and the potential for severe wildfire.

In addition to the problems of cheatgrass invasion, past wildfire policies emphasized zero tolerance of wildfire even in regions where vegetation evolved with naturally occurring wildfire. This has resulted in over crowded stands of piñon pine and juniper at higher elevations, encroachment of piñon-juniper into lower elevation rangelands, and large accumulations of dead wood from overgrown shrubs and trees – all of which contribute to wildfires with increased burning severity when they eventually occur (GAO 2007). Invasive species have permanently altered the lifecycle of wildfire in the Great Basin (Pellant 2004; Pimentel 1999). This implies that land management in the Great Basin now includes wildfire and invasive species management in order to control the escalation of wildfire suppression costs and to preserve the sagebrush rangeland ecosystem type for current and future generations.

The number of acres burned on public lands each year has been steadily increasing since the 1980's (NASF 2004). In 2005 a federal land management Inter-agency report[2] estimated that forty percent of all federal lands were at a high risk for catastrophic fire events due to deteriorated ecosystems exacerbated by invasive plants. Table 1.1 summarizes the number of wildfires and acres burned on Great Basin public lands from 2004 – 2008 (NICC 2007). The annual average of total acres burned over the

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[2] The Quadrennial Fire Review (QFR 2005) is an assessment and strategizing process conducted every four years to address wildfire issues that will impact management in the future. All five federal land management agencies participate as do Native American, state, and local agencies participating in wildfire management.

years from 2003-2007 is 1.675 million. This represents only slightly less than half of the

3.4 million acres that burned in all fifty states during 1990 to1994 (QFR 2005).

Table 1.1

	Number of Wildfires and Acres 2004 - 2008					
	2004	**2005**	**2006**	**2007**	**2008***	**5 yr average (2003-2007)**
Eastern GB	2,286	2,158	3,202	2,482	1,654	2,615
	89,187	953,362	1,244,452	2,411,428	140,705	1,010,861
Western GB	954	798	1,274	888	451	942
	39,791	1,032,114	1,348,871	890,171	71,930	665,705

*as of December 5

Viewed independently, the impacts of cheatgrass infestation on rangeland

ecology, rangeland productivity, and wildfire costs to society are cause for great concern.

However, the positive feedback loop occurring between wildfire and invasive plants

complicates the land manager's problem of determining when and where to expend

limited resources to control invasive plant growth. Ecologists are finding that after

cheatgrass dominated areas have repeatedly burned, the damage can be irreversible and

expensive restoration efforts are ineffective at restoring the highly eroded lands

(Stringham 2003; Knapp 1996). Unless management effort can be applied to limit the

wildfire-annual weed cycle, ecologists predict that the majority of the Great Basin

rangelands will undergo an irreversible transition to an invasive weed dominated, highly

eroded landscape that cannot support livestock grazing nor other wildlife (Knapp 1996).

Wildfires will be frequent (every 3 to 5 years in most areas) and increasing in size as

more fire adapted native plant communities dwindle and flash fuel invasives like

cheatgrass dominate. Societal costs from this outcome would be great. Alternatively,

land managers can treat lands that have not been irreversibly lost to annual weed

domination by controlling the spread of invasive grasses and reducing fuel loads on lands with accumulated dead woody materials and heavy grass build-up (Hjerpe 2008).

The United States Forest Service (USFS) and Bureau of Land Management (BLM) are responsible for approximately 80 percent of the 135 million acres of public land in the Great Basin (USFS 2010). Both agencies share the objective of managing lands for multiple use and sustainable yield. The most common uses on federal lands in the Great Basin are ranching, mining, forestry, and recreation. Each of these user groups impact and are impacted by the health, type, and quantity of vegetation. Public land managers are also responsible for wildfire management. Wildfire management objectives of federal lands are to first protect public and firefighter safety, then to minimize wildfire cost given the natural and constructed resources at risk. Considering USFS wildfire suppression costs alone averaged $900 million per year for the years 2000-2005 (Brown 2006) analysis of wildfire suppression cost particularly in terms of the cost minimization objective has become pressing land management issue.

Invasive species and accumulated dead vegetation create hazardous conditions that drive the frequency, severity, and size of wildfires and thus wildfire cost. Choices that land managers make about vegetation management can fulfill both the land management and fire management objectives (LANDFIRE 2010). Protecting native plant populations is more effective biologically and financially than attempting to repair or restore an area strongly impacted by invasive species (RMRS 2009; Devine 1993). Moreover, areas at the lowest risk of wildfire have the highest number of natives (RMRS 2009). The interagency fire program analysis (FPA) provides federal land managers with

standardized wildfire planning processes with the objectives of reducing wildfire losses, improving ecology, and minimizing wildfire suppression cost (FPA 2010). Considering the interaction of each of these objectives is essential to maximizing social value of public lands.

1.1 Introduction to the Economic Problem

Rangeland managers are expected to protect rangeland productivity while also addressing the increasing costs of wildfire suppression through targeted vegetation management efforts aimed at reducing fuel loads and limit the spread of invasive plants (Mercer 2008; GAO 2007). Land managers faced with limited resources and budgets require decision criteria to help ensure that they are choosing the most cost-effective allocation of vegetation management effort among various sites within a given management unit. Ideally, such decision-support criteria would be based on empirical estimates of the induced expected incremental reduction in the costs of wildfires, as well as the expected incremental change in ecological goods and services that would have been lost if the management effort had not been expended. Wildfire suppression costs tend to vary with the type and amount of vegetation ("Cost of Wildfire Suppression", 2007). A land manager responsible for a highly heterogeneous landscape may at any time have the option to apply treatment effort to an area with an invasive weed problem, dead woody fuel accumulation, and sufficient healthy natives; another area may be invaded by dense piñon pine and juniper trees from decades of fire suppression; still another may be dominated by invasive weeds but still has a chance to respond to restoration efforts. How does the manager decide which to invest in if funds are

sufficient for just one of the three scenarios? Economic theory would suggest that the optimal choice would be the one that generates the greatest net present valued expected return. However, to date, there is no such information that is readily available to rangeland managers (Gebert 2008; Mercer 2008; Kline 2004). The induced incremental benefits from both vegetation and fuels management to Great Basin rangelands have not been computed in a way that can support these decisions (Gebert 2008; Forbis 2006). As a result, it is likely that resources are being inefficiently allocated to vegetation and fuels management efforts. The goal of this research is to develop the methods to estimate expected returns to fuel management efforts on Great Basin rangelands, and to apply these methods to generate empirical estimates that can be used to support vegetation and fuels management activities. As a starting point, this research will focus on the costs of fire suppression avoided as a result of fuels and vegetation management. Estimates of the value of ecosystem goods and services protected will be left for future work.

1.2 Introduction to the Research Problem

A direct but impractical approach to such a problem would be to conduct a long-term experiment in which numerous plots of varying ecological condition, fuel loading, and invasive plant levels are treated with vegetation and fuels management methods, while control plots are left untreated. Over the following years, all plots would be exposed to fire ignition events and the difference in the wildfire costs between treated and control plots could then be directly measured. Given a sufficient number of plots of varying plot size and vegetation characteristics, it might be possible to estimate expected savings in wildfire costs associated with the various treatments and vegetation/fuel types

An alternative, and more practical approach proposed in this research, is to develop a model to estimate the effect of specific categories of vegetation and fuel types on fire suppression costs for wildfires in the Great Basin using data from past wildfire suppression expenditures. Such a model would predict fire suppression costs as a function of vegetation and fuel type. Assuming the data included sufficient variation in vegetation and fuel type over a representative sample of rangeland wildfires, the marginal effects of each vegetation type on wildfire suppression cost would indicate the contribution of that vegetation type. The differences in marginal effects between two vegetation types that represent 'before' and 'after' vegetation management conditions could serve as a proxy for the change in wildfire suppression costs resulting from a given management action. Finally, the expected benefit of a management action could be computed as the product of the probability of a wildfire and the predicted cost of a wildfire in the 'before' state minus the product of the probability of a wildfire and the predicted cost of a wildfire in the 'after' state.

This approach requires development of an estimable model that predicts wildfire costs as a function of vegetation type and data that exhibit the necessary variation and types of vegetation characteristics applicable to Great Basin rangeland wildfires. The approach also assumes that data were generated using similar criteria that include minimizing costs of wildfire suppression. Finally, it is assumed that the difference in wildfire suppression costs given two distinct and separate locations each with a different vegetation type can be used to proxy a change in wildfire suppression costs for a single location given a shift in vegetation type resulting from land management actions.

This approach would provide estimates of the change in wildfire suppression costs induced by management actions that shift vegetation and fuel type from one category to another. However, in order to use these estimates to calculate the value of a particular management action, additional information would be needed, such as the probability of wildfire on a given location, and whether the probability of a wildfire would change on a given location between 'before' and 'after' conditions.

1.3 Research Objectives

Objectives of this research are to:

(1) Develop an estimable model that predicts wildfire suppression costs as a function of rangeland vegetation and fuel categories, which characterize landscape vegetation "before" and "after" land management actions.

(2) Empirically estimate this model using data available through the US Forest Service and other sources.

(3) Use parameter estimates on vegetation and fuel types to calculate the induced predicted changes in wildfire suppression costs that would be a result of shifting from one vegetation type to another

(4) Demonstrate how these results can be used by land managers to predict the benefits of a given land management activity.

1.4 Outline of Thesis

Chapter two develops the economic model that corresponds with objective one above. Chapter three describes the data used and limitations, and summarizes the results of the econometric model. Chapter four presents the application of the econometric

model to perform a cost savings analysis. Chapter five reviews contributions of this study, offers conclusions, and suggests areas for further study related to the findings.

Chapter 2.CHANGES IN RANGELAND VEGETATION AND INCREASING COST OF WILDFIRE SUPPRESSION

The public lands of the arid west supply a variety of goods, services and inputs to production processes that are valued by society. These include forage for wildlife and livestock operations, recreational opportunities, ecological services that maintain water and air quality, scenic vistas, and others. Virtually all of the goods, services and ecological processes on a given land area rely directly or indirectly on the type and quality of the vegetation on the land. Plant density and vigor, the mixture of species, the age of woody shrubs and other characteristics of the local vegetation affect these values. On Great Basin lands rangeland fire has historically been a natural force that contributed to maintaining healthy vegetation (Chambers 2008). However, the combined effects of overgrazing of domestic livestock, the introduction of exotic annual grasses, such as cheatgrass, and decades of fire suppression have created a situation that threatens the ecological integrity of these landscapes, and the flows of costs and benefits to society (Forbis 2006; Devine 1993).

Piñon pine and juniper trees historically grew in higher elevations, where periodic rangeland fires allowed for patchworks of different types of plants, including perennial grasses and shrubs to grow interspersed with scattered groups of these small evergreen trees. Decades of fire suppression have encouraged dense stands of piñon pine and juniper (Zouhar 2008). These stands have crowded out other native plants and have generated large amounts of dead woody material that fuels wildfires that burn hotter than would have otherwise. In addition, and aided by heavy grazing pressure, these dense

piñon pine and juniper stands have encroached into lower elevation landscapes, which had previously been characterized as sagebrush and perennial grass plant communities (Rice 2008; Miller 1999). Fire suppression also affects sagebrush systems, which historically would burn in cycles of 10-70 years as small rangeland fires that were checked by natural firebreaks from patchworks of perennial bunch grasses and forbs and bare ground between plants (Rice 2008). Livestock grazing reduced the quantity of grasses that would carry rangeland fire from shrub to shrub, combined with fire suppression contributed to a landscape with older and larger shrubs with large amounts of dead and woody material (Chambers 2008; Rice 2008). Thus rangeland fires that ultimately occur on such landscapes burn hotter, with longer flames that spread from shrub to shrub, covering more area than the previous rangeland fires. These fires, like fires fueled by dense piñon pine and juniper stands, are dangerous, larger and can spread quickly, triggering wildfire suppression efforts to prevent them from becoming large uncontained wildfires. More and more frequently, these fires are not contained with regional resources, and become large wildland fires requiring mobilization of fire suppression resources from other management regions across the U.S. Gebert (2008) analyzed wildfire suppression cost data by region for years 1995-2006 to reveal the proportion of within region and out of region expenses. The findings show that for all USFS regions, region four (the Great Basin) accounted for the highest percentage of out-of-region expense.[3]

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[3] 91.4% of region four's expenditures were within region – region five reflected the largest within region expenditures at 94.8% and region one had within expenditures of 91.5%.

Cheatgrass, an exotic annual grass, has had a profound effect on these altered vegetation growth and rangeland fire patterns (Epanchin-Niell 2009; Getz 2008; Chambers 2008; RMRS 2009; Knapp 1996). Cheatgrass seeds now can be found throughout the Great Basin. The seeds germinate in soils that have been disturbed by grazing, construction, vehicular use, and recent fire. They germinate long before any native perennial plants do, thus drawing limited water and nutrients from the desert, over time reducing the vigor of native perennial plants. Cheatgrass sets seed and dies off in mid may to June, leaving a thick layer of highly flammable dead material filling in what would have been bare ground between native plants, long before what would ordinarily have been the start of the rangeland fire season. In this way, cheatgrass has accelerated the onset of the fire season and lead to a general tendency for more frequent fires occurring over larger areas (Brooks 2008). After a fire, cheatgrass can immediately emerge in the next season as the dominant plant type. This is particularly likely on areas which dense piñon pine and juniper stands and areas where sagebrush and other rangeland shrubs have overgrown large amounts of dead woody fuels which increase wildfire intensity. Once cheatgrass dominates an area, wildfires can occur as often as every 2 to 5 years.

2.1 Rangeland management and fire suppression costs

Public land managers are responsible for maintaining rangeland health to ensure that current and future flows of rangeland values are sustained. Fire Management Plan's (FMP's) and FPA's are federal requirements for all public land managers (FWFMP 2001)

and establish strategies for wildfire and vegetation management keeping land and fire objectives in mind. Land management objectives are multiple use and sustainable yield, while fire objectives seek to first protect human life then minimize cost given resources at risk. While the land manager may not take an active role at the time of a wildfire event, they are responsible for providing the incident management teams[4] (IMT) with framework for optimal decision making for a specific location.

An important tool available to achieve the FMP and FPA goals is the management of vegetation (GAO 2009; LANDFIRE 2010). The land manager can influence vegetation with treatment to maintain the current health of vegetation or to shift vegetation to a healthier state. Rangeland vegetation treatment consists of grazing management, mechanical or manual removal of dead woody fuel accumulation, herbicidal treatment of exotic plants, promoting native plant vigor, controlled burning to remove accumulated annual grasses and woody fuels, rehabilitation by planting and reseeding native and non-native plants that can out-compete exotic annuals, and increasingly wildfire use (WFU)[5]. All of these treatment methods accumulate costs. Land managers may also choose to forego vegetation treatment and allow the current vegetation to convert towards an unhealthier state. The land manager is assumed to have a fixed budget and a given area of landscape with a variety of patches over which the condition of vegetation and fuel accumulation varies. The land manager's problem is to

[4] Incident management teams are comprised of the wildfire specialist commander and staff members that coordinate wildfire fighting tactics at the time of the event.
[5] Wildfire use is a management decision to allow a naturally ignited fire to burn in a specific area as a means of accomplishing a pre-determined objective stated in the area's wildfire plan.

choose which patches to treat or not treat in order to maximize the net value of the landscape.

Starting for the moment with a land manager whose problem is to maximize the net expected value of the landscape for one season, we can write:

$$\max_{m} L(v(m)) - \pi(v(m))\ C^{f}(v(m)) - pm, \qquad where\ pm \leq B \qquad (2.1)$$

Where L is value of public lands as a function of a vegetation condition index v which in turn is a function of m representing a land management action chosen by the land manager to change the values of v, π is the probability of wildfire as a function of v(m), C^{f} is the cost of wildfire suppression as a function of v(m), p is the marginal cost of land management, and B is a budget constraint. This simplified model also ignores other land management actions affecting vegetation such as ranching, recreation, and development. Further, the science suggests that stochastic processes influence the effect of management effort on vegetation outcomes and that much more still needs to be known about these before ecologists can provide the probability of a given management action's effectiveness on a specific site (sageSTEP 2009; Perrings 2005; Richards 1999). The full social value impact of the wildfire-cheatgrass cycle is sufficiently complex (Gebert 2008) and is beyond the scope of this paper.

An empirical solution to this problem would maximum value of public lands by setting the marginal cost of management choices equal to the marginal benefit of management choices as they impact vegetation health. The cost of the marginal cost of management choices can be explicitly represented by the suppression cost of wildfire. Estimating the marginal benefit requires information about marginal changes in

ecological goods and services induced by wildfire suppression. To date, much more research needs to be done to better understand and estimate these components of the rangeland manager's problem (Gebert 2008; Kaiser 2006).

This research will focus on one part of the problem, specifically on how wildfire suppression costs vary with vegetation type and condition, given that a wildfire has occurred. This work will not explicitly consider the costs of management in terms of vegetation treatment type, nor the overall relationship between tradeoffs related to managing land for fire suppression cost minimization in conjunction with land management for rangeland productivity maximization. In addition, the relationship between management action and resulting vegetation condition in terms of treatment effectiveness given a specific vegetation type will not be considered, or how the probability that a fire ignited will become a large wildfire may change with vegetation condition and fuel type. The research problem is to develop an empirical model that determines how wildfire suppression costs vary with vegetation type and condition. The resulting estimates can then be used to support decision-making by land managers who must allocate rangeland management resources to maintain public rangelands.

2.2 Characterizing wildfire suppression costs as a function of vegetation

Characterizing wildfire suppression costs as a function of vegetation can be examined from the perspective of a decision maker, who, given that a wildfire has occurred, seeks the optimal combination of fire suppression inputs to minimize the expected costs of fire suppression and ultimately maximize the value of public lands. This specification is a simplification of the problem faced by those individuals who

manage wildfire suppression efforts for large rangeland fires that occur every year throughout the western U.S.

The land manager's total cost of wildfire suppression can be described:

$$TC \quad = \sum_{i=1}^{I} w_i x_i + C \qquad (2.2)$$

where: $\qquad C \quad = g(\boldsymbol{x_i};R,\alpha) \qquad (2.3)$

Where w_i = price of firefighting asset type i, x_i = quantity of firefighting asset type i, C = cost of damage to resources, R = value of threatened resources such as housing, infrastructure, timber stands, and recreation sites and α = environmental attributes such as weather, topography, and vegetation. However, not all factors can be influenced by managerial decisions. Firefighting assets are chosen, but selection is based on topography, weather, and availability which are exogenous to the wildfire manager. While resource values and vegetation are also exogenous to the wildfire manager at the time of the event, the land manager can influence vegetation through management practices in prior periods.

Vegetation types vary significantly in their contribution to wildfire cost and size. This variability depends on the quantity of fuel provided by a given type of vegetation as well as the characteristic manner in which it burns. For example, a grass fires tend to ignite easily and spread quickly, but may not require significant asset allocation because grass fires consume the available fuel very quickly and completely. Conversely, a dense piñon-juniper stand with large quantities of dead fall may not ignite as readily or spread as quickly, but the large quantity of available fuels from the large vegetation may require a significant number of suppression assets to minimize wildfire severity. Not only does

vegetation impact the current fire, but future fires as well. Historic suppression of wildfires and spread of invasive species have contributed to changes in vegetation type, distribution, and density resulting in altered fire regimes (Dombeck 2004). In a sense, as wildfire activity increases, the probability of future wildfire increases (Knapp 1996). Analyzing marginal effects of vegetation is important because the unique biological properties of each vegetation type impact wildfire behavior and thus wildfire suppression cost.

Land managers make vegetation management decisions based on vegetation feasibility given the land ecology. For example, the Great Basin can support a variety of drought tolerant species such as sagebrush and piñon pine but can't support blue spruce or sawgrass. Some vegetation types, like cheatgrass, are feasible yet undesirable. This study will assume five vegetations types[6] are feasible in the Great Basin – annual grass (A), intermediate brush (F), mature chaparral (B), open timber (C), and sagebrush (T) (FEIS 2010; USFS 1978). Annual grass includes invasive species such as cheatgrass and medusahead with very sparse inclusion of trees or shrubs. Vegetation type A is considered the least healthy and least desirable type (Zouhar 2003). Conversion to a healthier type using any treatment method is difficult if not impossible. Intermediate brush (F) contains closed stands of piñon-juniper susceptible to conversion to the less desirable type A after a fire event. Mature chaparral (B) contains mature, dense brush where at least one-fourth of vegetation is decadent. While type B is primarily native species, the high quantity of dead ground fuel is a high fire risk potential. Open timber

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[6] Vegetation classification based on the 1978 National Fire Danger Rating System fuel model definitions as well as consultation with Dr. Tamzen Stringham.

(C) contains open stands of piñon-juniper mixed with native grasses and shrubs. Type C tends to occur at higher elevations. While there is some ground fuels that increase wildfire risk, type C is considered healthy due to the mix of native plants and lack of invasive species. Sagebrush (T) is also considered a healthy vegetation type native to the Great Basin. Type T contains young healthy sagebrush intermixed with native grasses covering at least one-third of the site. While type T burns easily, there is relatively little ground fuel as the overwhelming majority of plants are live. Like type C, type T does not contain invasive species.

2.3 Research Problem

Revealing the marginal effects of vegetation on wildfire suppression costs is a multi-step process. This section will discuss each stage of the process as well as the data requirements. First, the land manager's total wildfire suppression cost represented in Eq.(2.2) will be used to derive a wildfire suppression total cost function. Second, I discuss application of wildfire suppression total cost function factors to the empirical model estimated in Ch.3. Finally, I present an overview of the cost savings analysis estimated in Ch.4 that estimates the marginal effects of treatment decisions on wildfire suppression cost.

2.3.1 Cost Minimization

The cost minimizing point is derived from the total cost equation Eq.(2.2)

using the Lagrangian function :

$$Min\ \{x_i\} \qquad TC \quad = \quad \sum_{j=1}^{J}\sum_{i=1}^{I} w_{ij}x_{ij} + C_j$$

$$s.t.: \qquad C_j \quad = \quad g(x_j, R_j, \alpha_j)$$
$$A_j \quad = \quad f(x_j; \alpha_j) \leq A_{0j}$$
$$x_j \quad \leq \overline{x}_j$$

$$\varPi_j \quad = \quad \sum_{j=1}^{J}\sum_{i=1}^{I} w_{ij}x_{ij} + g(x_j; R_j, \alpha_j) + \lambda[f(\overline{x}_j; \alpha_j) - A_{0j}]$$

$$TC_j^* \quad = \quad TC_j^*(w, A_{0j}, R_j, \alpha_j, \overline{x}_j) \qquad\qquad (2.4)$$

The minimization of total cost is subject to the total area burned A_j (which is constrained

by the targeted fire size A_{0j}), the cost of damage to resources C_j, and, and a vector of

availability of wildfire fighting resources $\overline{x}_j$. TC_j^* represents the total wildfire

suppression cost minimizing function for an individual fire j given A_{0j} and is a function

of a vector of fire suppression resource prices (**w**) for all asset types, value of resources

threatened by fire j (R_j), fire j characteristics given environmental attributes (α_j), and a

vector of available wildfire fighting assets ($\overline{x}_j$) for wildfire j.

Implicit in the total cost function Eq.(2.4) is the cost minimizing combination of

wildfire fighting assets x_i given the target size for fire j. Types of fire fighting assets x_i

include wildfire strategy management and wildfire suppression assets such as ground

crews, engines, and air attack to control and extinguish the fire. Also included in Eq.(2.4)

is the cost of damage to resources resulting form a given fire j such as damage to

residential and commercial property, public infrastructure, and natural resources. However, the true total cost of wildfire is not limited to the costs of wildfire suppression and resource damage reflected in Eq.(2.4), but also includes other economic, social, and environmental losses. Economic and social losses may include decreased employment, health impacts, or loss of recreation lands. Environmental losses such as diminished health of native vegetation have the potential to permanently decrease the production potential of the land (Pellant 2004). Estimation of these losses is challenging and beyond the scope of this study.

2.4 Theoretical Model

2.4.1 Asset Price w and Asset Constraints $\overline{x}_j$

Fire fighting asset availability constraints and prices are factors influencing the asset mix employed for a given fire. Determination of the cost minimizing asset combination depends on the target fire size, asset price ratios, and asset availability. Though federal policy requires wildfires to be suppressed at minimum cost, asset price is not necessarily the primary influence on asset demand. Asset quantity and prices are set at the beginning of each wildfire season (May – October) for the entire U.S. Assets are then allocated to each region[7]. Simultaneous wildfire occurrence, either within or outside of a region, creates competing demand which often exerts a stronger impact on asset selection than prices. During the height of the wildfire season a wildfire manager may be

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[7] There are nine USFS regions across the U.S. These regions are management divisions responsible for not only oversight of their own public lands, resources, and assets but also for coordinating with all other regions to ensure that federal forest plan goals are met.

forced to accept a sub-optimal mix of wildfire assets due to competition among wildfire events.

Asset types include ground crews, engine crews, dozers, helicopters, and fixed wing tanker planes. For each type there are a range of prices based on level of specialization. In general, higher skilled, higher risk-taking assets command higher prices. Large suppression efforts also incur support service and overhead costs. Examples of support services include water transport trucks and camp crews that provide meals and camp maintenance. Overhead costs include management (IMT's) and strategy specialists that coordinate and oversee the attack plan. The total number of ready assets available for all wildfires is determined at the outset of each wildfire season. The number of each asset type required for each individual fire is determined daily.

Ideally the exact number, type, and price of each asset deployed to each fire are known when empirically estimating wildfire suppression cost. Moreover, proportionate costs of asset sharing across wildfire events, especially common with aircraft, would be explicit to avoid attributing of expense to wildfire events incorrectly. Detail of asset quantity and asset type available for each wildfire date is necessary to capture asset selection strategy effects on cost. Throughout the wildfire season assets are staged at strategic points within each region to minimize travel time and suppression cost to the highest risk areas. Asset staging changes along with wildfire conditions and may result in asset reallocation to an outside region, in turn decreasing asset availability. Without regional asset availability data, cost impacts due to sub-optimal asset selection could potentially distort marginal cost effects of other model variables. From the wildfire

manager's perspective, selection of fire fighting assets is the only means of influencing event size, severity, and cost. The expected impact of asset availability and prices on wildfire suppression cost in the empirical model is positive since prices must be greater than zero and all assets have a corresponding price.

2.4.2 Targeted Fire Size A_{0j}

In this model target fire size acts as a production constraint so that the value TC_j^* represents the cost minimizing combination of fire fighting assets given a target acreage, A_{0j}. Both acres available to burn and management strategy influence the targeted fire size. Acres available to burn are delineated by topographic limitations on total acres. Natural fire breaks such as large bodies of water or large naturally un-vegetated areas (e.g. sand dunes, salt flats) will limit fire potential in a given direction. Additionally, previously burned areas devoid of vegetation will also act as a fire beak, limiting fire size potential. Management strategy also determines fire size though is exogenous to the land manager as it is determined by available fire fighting assets and environmental variables. The target fire size for a given fire along with the asset price ratio will determine the cost minimizing combination of fire fighting assets. Fire breaks can be created using hand crews or heavy equipment to limit a fire by removing vegetation to create an artificial fire break. Instead of limiting size, wildfire managers will sometimes allow a fire to burn either as wildfire use (WFU) or because the cost of asset deployment is not justifiable given the wildfire characteristics. This last scenario can occur if fire characteristics are sufficiently risky that the event cannot be battled without certain harm to firefighters or the public. However, the decision to let burn can occur with areas pre-approved in

FMP's and FPA's for WFU. Wildfire use is a management decision to allow a naturally ignited fire to burn in a specific area as a means of accomplishing a pre-determined objective (HFI 2010).

GIS data would provide ideal detail to accurately estimate target fire size. Image mapping of fuel type and distribution, topographic measures, access roads, and FMP's approved WFU areas could describe non-strategic factors of area available to burn. Strategic factors affected by wildfire managers include using constructed wildfire breaks or other fire fighting techniques to orchestrate wildfire direction and shape. Daily wildfire reports revealing management strategy and techniques would best represent these factors. The target fire size is made by the manager at the time of fire but the probability of reaching the targeted size is influenced by variables (i.e. weather) exogenous to managerial decisions.

Endogeneity is a potential issue since strategy and allocation of resources impact total cost and total size simultaneously. In general, deciding to let a fire burn decreases total suppression cost and increase fire size while a decision to aggressively fight a wildfire increases costs and decreases size. Regardless, the expected sign is positive since fire suppression cost generally increases with fire size.

2.4.3 Threatened Resources R_j

Fire fighting effort, and thus suppression cost, is influenced to a great extent by the type and value of resources threatened. Resources include human lives, residential and commercial property, infrastructure (e.g. energy, water, transportation, etc.), rangelands, and protected sites (e.g. national parks, historic sites, endangered species

habitat). Until 1995 federal policies resource protection priority was life, property, and natural resources (FWFMP 2001). Since then the priority is to protect human lives, then minimize cost given the resources at risk. However, increased development in the wildland urban interface (WUI) over the past decades has resulted in a distortion of the cost minimizing imperative. A 2006 USFS wildfire reports 85 percent of WUI acreage existing on non-federal lands, yet wildfire management and staff unofficially estimate 50 percent of suppression expense is attributable to protecting WUI property. Public expectations strongly influence this reprioritization; however, it may result in damage to social value maximizing resources.

Consideration of threatened resources is particularly important given the scarcity of fire fighting assets. During the height of fire season when competition for assets peaks, social and economic valuation of existing resources determines allocation of assets based on the FMP. In fact, the November 2008 National Fire Plan[8] (NFP) calls for development of a matrix that will allow consistent weighting of threatened resources across regions to facilitate allocation decisions.

GIS data is also ideal for representing resource distribution and value. Location, description and value of structures threatened by each fire aids in explaining much of fire fighting asset allocation. Additionally, data representing natural resource and special designation areas (i.e. endangered species habitat) is necessary to understand fire fighting strategy and the resulting suppression cost. While lands can be managed prior to the fire to minimize risk of damage or destruction to valued resources, it may be for naught due

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[8] The National Fire Plan is a nationwide interagency effort to develop strategies to more efficiently manage wildfire events and impacts.

to the unpredictable nature of wildfire. The wildfire and land managers cannot impact this cost factor at the time of fire – resources either are or are not threatened and have a fixed value at the time of a given wildfire event. Regardless, as the value of threatened resource increases, total wildfire cost increases as more efforts are directed towards protection.

2.4.4 Fire/Environmental Characteristics α_j

This group of characteristics is the most influential in determining wildfire cost. Fire behavior is determined by weather, topography, and vegetation. Fire behavior thus determines fire fighting strategy, assets employed, resources damaged or destroyed and ultimately the total cost of rangeland fire fighting. Moreover, environmental characteristics impact one another. For example, high wind causes burning vegetation to increase in intensity and area of spread, especially on steeper slopes.

Wind, moisture, and lightning are the primary meteorological influences on wildfire. Wind encourages fire intensity and spread. Moisture content decreases wildfire activity at the time of the wildfire event. However, exceptionally precipitous years yield greater vegetation growth providing more wildfire fuel prior to the fire event. Lightning is one of the primary causes of wildland fire ignitions. While fire managers can anticipate and plan for weather events in advance of ignition, they are unable to influence these factors at the time of fire. Ideal variables to represent this factor would be weather measures specific to each wildfire such as wind speed, humidity levels, precipitation measures, and lightning activity. Both Wind speed and lightning activity will increase

total suppression cost. Humidity levels and precipitation will decrease total cost in the current year, but lagged values for these factors would likely increase total wildfire cost.

Topographic variables influence the rate of spread, potential fire size, and what type and quantities of vegetation will thrive (Liang 2008). Topography also determines what type of fire fighting assets can access the event site. Slope of the terrain has a strong influence on wildfire direction of travel as well as rate and breadth of spread. To a lesser extent, slope affects vegetation potential as steeper slopes exhibit a shading effect and only specific vegetation can thrive on steep slopes with shallow soil. Lastly, steeper slopes are difficult for some fire fighting assets to access. Often only air attacks are possible, which are the most costly fire fighting assets. Slope is expected to have a positive effect on total wildfire cost as steeper slopes encourage greater fire intensity and spread.

Aspect primarily affects potential vegetation type (Gebert 2007). South and west aspects receive more sun than north and east aspects. Western aspects also tend to receive more wind and more rain while the opposite is true for eastern aspects. In general, southern and western aspects have denser, drier vegetation populations and receive stronger winds, contributing to increased suppression cost. Southern and western aspects have a positive impact on cost, while northern and eastern aspects are expected to exhibit a negative impact on cost.

In general, elevation positively influences precipitation and thus potential vegetation (Liang 2008; Gebert 2007). Higher elevations experience greater amounts moisture, thus contributing to increased vegetation quantities. Higher elevations often

experience stronger winds as well. Like slope, elevation also limits asset access. Lastly, sizeable naturally occurring fire breaks (i.e. large lakes, rivers) contribute to natural wildfire limitations. Elevation is expected to have a positive impact on cost.

GIS data is ideal for reporting topographic factors. Mapping of natural fire breaks as well as measurements of slope, aspect, and elevation for the wildfire area can aid understanding topographic impact on suppression costs. Wildfire managers can prepare to fight fires on many different topographic sites or attempt control of fire direction, but once the event begins the manager has little control over topography and must manage it as given.

Vegetation literally fuels the fire. Combined with topographic and weather features vegetation determines the size and severity of a wildfire event. It is the ladder that allows fire to spread over hundreds and thousands of acres. The plant habit, life cycle stage, and proximity to other fuels influence wildfire size and severity. Vegetation habit determines the size, shape, and density of a given plant. Finer fuels with shallow root systems (e.g. cheatgrass) may combust easily but provide little fuel. Larger, denser fuels such as piñon pines not only provide more fuel above ground, but have deeper root systems that smolder creating risk of potential flare-ups. Also, branching vegetation types like sagebrush are more likely to spread fire to nearby vegetation. Plant lifecycle is important as decadent, dead, and downed vegetation burns more readily than live healthy vegetation. Decadent vegetation is also susceptible to infringement by other plant types (Chambers 2008), especially if the plant is an invasive species such as cheatgrass which easily out-competes deeper rooted plants for water and nutrients. As native plants decline

they provide more fire fuel and increase the probability of wildfire. Increased wildfire cycles further compromise native plants ability to fight off invasives. Additionally, wildfire risk increases when native and invasive communities become overgrown and densely packed together. Fine invasive fuels more easily ignite the larger denser native fuels, contributing to increased probability of ignition as well as severity and spread. Topography and weather factors further influence vegetation impacts.

Vegetation is the primary factor of interest in this study. Vegetation management choices can reduce the expected cost of wildfire suppression by reducing the probability and severity of large wildfires. Ideally, wildfire data provides information about the variety and distribution of vegetation type for the entire fire area. Capturing differences in wildfire effects due to type, size, distribution, and lifecycle can reveal the marginal effects of vegetation on wildfire suppression cost. Vegetation types with biological characteristics that contribute to wildfire severity and frequency proportionately more than others will also have a greater impact on wildfire cost suppression.

2.5 Cost Savings Analysis

Factors discussed in the theoretical model present decision variables for the wildfire and land managers. In particular, vegetation marginal effects capture suppression cost differences attributable to biological characteristics unique to each vegetation type. Estimating suppression cost for each vegetation type provides a basis to perform a cost savings comparison of treatment benefits with costs of foregoing treatment. Full cost-benefit analyses incorporating treatment choices by method could be

conducted, but would require treatment specific data which is beyond the scope of this study.

Vegetation treatment choices and the resulting impacts on suppression cost reach across years and even decades (Gebert 2008). A net present value of treatment choices is discounted over 200 years to account for these temporal effects. Values for future periods should be discounted to a net present value (NPV) using a reasonable discount rate: 4 percent is commonly used for government project analysis.

Prior to temporal estimation, the basis for each vegetation type should be amended to reflect biological characteristics that may impact future period values. In particular probability of treatment success, vegetation fire cycles, probability of succession due to fire, and lifecycle length are relevant measures that should be interacted with the basis value prior to cost savings calculation (Currie 2009; Forbis 2006; Richards 1999). Once incorporated, separate NPV calculations under treatment and no treatment assumptions by vegetation can be calculated. Netting the treatment NPV and no treatment NPV reveals the cost savings of wildfire suppression per vegetation type. Land managers can use this information to determine treatment priorities by vegetation type. Allocation of resources to treat vegetation that minimizes wildfire cost contributes to the land manager's goal of maximizing social value of land by managing for healthy vegetation.

Chapter 3.EMPIRICAL ESTIMATION OF WILDFIRE SUPPRESSION COSTS

This chapter describes the steps to empirically estimate the cost of fire suppression as a function of vegetation type for wildfires in the Eastern and Western Great Basin, based on the conceptual model developed in Ch.2. The first section describes the data and its limitations. The second section presents the empirical model, variables, and steps to address data limitations. The third section presents and evaluates the estimation results.

3.1 The Empirical Model

The cost function for wildfire suppression derived in chapter two is:

$$TC_j^* = TC_j^*(\mathbf{w}, A_{0j}, R_j, \alpha_j, \overline{x_j}) \tag{3.1}$$

for an individual fire j, where TC is total wildfire suppression cost, $\mathbf{w}$ is a vector of unit prices for fire suppression assets used, A_{0j} is the size of the fire, R_j, is the value of resources threatened by the fire, α_j denotes environmental attributes (including vegetation type) that affect wildfire characteristics, R_j is the value of threatened property, and $\overline{x_j}$ is a vector of wildfire fighting assets available to use on wildfire j, if needed. The log of the cost function provides the reduced form model used in the estimation:

$$ln(TC) = \beta_0 + \beta_1 ln(\mathbf{w}) + \beta_2 ln(A_0) + \beta_3 ln(\alpha) + \beta_4 ln(R) + \beta_5 ln(\overline{x}) + \mu \tag{3.2}$$

3.2 The Data and Variables Used in the Estimation

The data used in this study were provided by Krista Gebert an economist with the U.S. Forest Service Rocky Mountain Research Station (RMRS) in 2008. The data include types of equipment, crews and other resources used by public agencies on

individual wildfire suppression operations on public lands. Also included are fuel types (i.e. vegetation conditions), number of threatened and damaged structures, weather conditions, topography and other details relevant to firefighting decisions. These data represent fire suppression operations for wildfire events defined as "large" fires (Gebert 2007). Up through 2002, the definition of a "large" wildfire is one that escapes initial suppression efforts at the local level and expands to 100 acres or more. This definition was modified in 2003 to fires that expand to 300 acres and over (Gebert 2007). Included in the database are a sample of wildfires that ignited on public lands for which suppression efforts were under the jurisdiction of USFS, BLM, Bureau of Indian Affairs (BIA), Fish and Wildlife Service (FWS), National Park Service (NPS).

Gebert (2007) included actual suppression expenditures in the RMRS database. The entire RMRS database was compiled from multiple federal agency databases that did not share a unique identifier for each wildfire event (Gebert 2008). Thus, only fires that could be definitively tied to both systems were included in the RMRS dataset, which is then represents subset of total large wildland fires during 1995 – 2007. In particular, there is reason to expect that there may be some systematic bias regarding fires known as 'complexes' for which actual expenditures could and could not be matched. A fire complex is comprised of two or more fires in the same general vicinity that are managed simultaneously by the same wildfire manager. Often the fires will merge into a single fire. When this occurs, one of the fires will retain their original name and incident ID, while the others will adopt this new name. For inclusion in the RMRS database, complex fires required reconciliation of multiple fire names and ID's across databases. This

proved more challenging than reconciliation of individual fires (Gebert 2007). Large fires that merged into fire complexes may therefore be underrepresented.

The expenses captured in the RMRS database include financial costs of fire suppression incurred by both the USFS as well as the Department of the Interior (DOI) for each included event. Distribution of wildfire events within the dataset are reported in Table 3.2. While USFS reports a disproportionately large share of events, in an average year they are responsible for 73 percent of all U.S. wildfire suppression costs (Gebert 2008). All recorded expenses are directly related to a specific wildland fire. Not included in this tracking method are other economic costs of wildfires such as impacts on recreation, businesses, and personal property losses. External costs from wildfires, such as changes in water and air quality are not included. Ex-ante fire mitigation expenses related to prevention are not included, nor are ex-post expenses of vegetation rehabilitation applied to burned lands. Finally the costs included in the data are only for multi-day fires. The expenses associated with the "initial attack" of the fire before it reaches the size of a "large" fire are classified, budgeted, and tracked separately from the costs of suppression expenses incurred by federal agencies and are not included. The RMRS database include 400 observations for the Eastern and Western Great Basin regions over 1995-2007. Three of the 400 RMRS observations were dropped due to missing data – the final number of observations used in this study is 397. Table 3.1

shows the wildfire distribution by year, state, forest unit, and primary wildfire protection agency[9]. Table 3.2 shows summary statistics of variables relevant for the estimation.

Table 3.1

Year		State		Forest Unit		Agency	
1995	8	CA	12	401	5	BIA	2
1996	45	CO	1	402	36	BLM	14
1997	6	ID	186	403	26	ID	6
1998	14	NV	67	405	6	NV	2
1999	19	UT	103	407	21	PVT	5
2000	67	WY	28	408	28	USFS	358
2001	23			409	34	UT	8
2002	40			410	10	WY	1
2003	27			412	52	Other	1
2004	19			413	66		
2005	21			414	21		
2006	56			415	11		
2007	52			417	45		
				418	21		
				419	15		

3.3 Independent Variables

3.3.1 Target Fire Size (A_{0j})

No explicit data were available to represent target fire size. Rather, variables representative of this value were used. First, total acres burned for each fire is included in the model to proxy target fire size. The natural log of total acres burned in a given fire is *lntot_acre*. Total acres burned were obtained from the RMRS dataset. Acreage as a measure of fire size was chosen instead of fire perimeter measures based on availability of data. Additionally, when considering vegetation treatment options, viewing each fire as a total area rather than a "fence" is more useful in terms of decision making. Effects associated with constraints on fire size discussed in Sec.2.4.2 are indirectly controlled for

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[9] The primary protection agency reflects the wildfire agency that engaged in wildfire suppression activity. Agencies sometimes fight fires outside of their land jurisdiction. When this occurs with a non-federal agency suppression expenses are later billed to the appropriate federal agency. All wildfires in dataset are federal expenditures.

with year dummy variables. These variables are expected to proxy annual meteorological trends, price levels, fire fighting asset availability, and annual simultaneous wildfire competition that could influenced asset constraint variation and thus strategy choices in the given year. Using 1998 as the base year, dummy variables are included for all other years 1995 – 2007. 1998 was chosen as the base year as it reflects the lowest *lntot_exp* mean value of all years included in dataset. Therefore, it would be expected that coefficients on the year dummies would be positive because the year with the lowest mean total wildfire suppression cost is used as the base year.

3.3.2 Threatened Resources (R_j)

Federal mandate for wildfire suppression on public lands is to first protect human safety, and then minimize fire cost given the resources at risk. As discussed previously, suppression efforts often prioritize private property over other non-human assets regardless of wildfire suppression cost minimization objectives due to public expectations. Therefore, wildfire suppression effort and cost would be expected to increase with population density (Lankoande 2006). The RMRS data includes the log of total housing value, based on the 2000 census*,* within a 20-mile radius of the ignition point (*lntot_20*), to indicate proximity and monetary value of private property. This variable was developed by Gebert (2007). Because of it's usefulness in estimating suppression costs similar variables have since been used in other studies (Liang 2008). Because the log of the values is used (and ln(0) is undefined), observations with zero values were recoded to 0.0001.

3.3.3 Environmental Attributes (α_j)

Environmental attributes include topographic, meteorological, and vegetation influence on wildfire suppression strategy. The variable *lnnfdrs_wgt_fl* indicates the type of vegetation fuel for each observed wildfire in the data. *lnnfdrs_wgt_fl* is an ordinal variable based on the National Fire Danger Rating System (NFDRS) fuel type. The fire suppression manager assigns a vegetation type from the NFDRS fuel classification system to describe the vegetation at the point of the ignition. Because of the current lack of a more accurate information recording system for wildfire information (Gebert 2008) and for purposes of this study, the vegetation type given at the ignition point is assumed to be homogenous across all acres for the given fire.

Using the recorded NFDRS description, *lnnfdrs_wgt_fl* was constructed for this study by incorporating the NFDRS fuel load definitions (Andrews 1997). The fuel load definitions are used in calculating wildfire behavior indices as well as reporting wildfire event vegetation types. Fuels are defined by several characteristics that together express the quantity and potential severity for a given vegetation type. Fuels are first classified as either live or dead. Dead fuels are further categorized by fuel moisture hours (FMH). FMH are proportional to the vegetation's approximate diameter and describe the amount of time necessary for a fuel of that size to reach 2/3 of the moisture present in its environment. There are four categories of fuel moisture hours used by the NFDRS – 1, 10, 100, and 1000. Live fuels are categorized as either herbaceous or woody plants. Both live and dead fuel loads are further described in tons per acre typical for a location specific to a given fuel type. Fuel depth is the final characteristic applied to both live and

dead fuels and measures in feet the amount of ground material. The final calculation for each NFDRS fuel type is a weighted index reflecting the quantity of fuel available at each point of ignition. The variable is constructed as follows (where X_i = *NFDRS vegetation type)*:

$$\beta_{i(nfdrs\ weighted\ scale)} = [(X_{i(Dead\ Fuel\ Density)} * X_{i(Fuel\ MoistureHours)}) + X_{i(Live\ Woody\ Fuel\ Density)} + X_{i(Live\ Herbaceous\ Fuel\ Density)}] * X_{i(Fuel\ Depth\ (ft.))} \tag{3.3}$$

Moisture content and vegetation density are considered to be the most significant factors that determine how fuels influence wildfire behavior (Miller 1999). Vegetation impact on wildfire behavior heavily influences wildfire severity and size, and thus cost. For this reason the NFDRS weighted fuel variable is the factor of most interest within the model. Weather and topography are the other environmental attributes that influence size and severity of wildfire. Measurements of attributes such as wind, humidity, slope or elevation are less meaningful when viewed independently than when they are combined into an index (NWS 2010). Wildfire behavior indices are used by wildfire experts to better express the combined effect of weather, topography, and slope on wildfire severity (Cohen 1985). Fire Intensity Level (FIL) is one of the indices commonly employed, and is used as a categorical variable in this model. FIL (*lnfil)* is valued 1-6 expressing flame length at the wildfire front where 1 = 0-2 feet, 2=2-4 feet, 3=4-6 feet, 4=6-8 feet, 5=8-12 feet, and 6=12 or more feet. Flame length is a function of weather, topography, and fuels. Suppression efforts and resource impacts are greater at higher FIL values.

3.3.4 Asset Price w and Asset Constraints $\overline{x}_j$

Explicit data for asset prices and constraints were not available for the wildfires included in this study. Data reflecting the number of suppression assets used for each fire is available from the National Interagency Fire Management Integrated Database (NIFMID), but a few issues prevent inclusion of the asset data as a desirable proxy. First, fixed wing aircraft use data is not included in the NIFMID data. Fixed wing aircraft are both the most effective and most expense large wildfire fighting asset available to a manager. Helicopter and ground assets employment may depend heavily on fixed wing aircraft availability; failure to include fixed wing tankers could contribute to a biased estimation because of an omitted variable. Secondly, asset availability strongly influences management asset selection. Theoretically, use of a sub-optimal asset mix results in increased total cost. Analysis of actual assets employed for a given event in no way infers whether or not that particular mix approached the suppression cost minimizing combination for that particular event. Effects associated with asset prices and constraints are controlled for with year dummy variables. These annual variables are expected to pick up yearly asset prices as well as yearly meteorological and wildfire location patterns. The construction of these annual dummy variables was discussed previously in Sec.3.3.1.

Table 3.2

Variable	Mean	Std. Dev	Min	Max
			Number of obs	397
lntot_exp	12.48117	2.12071	4.76328	17.77370
lntot_acre	6.99892	1.78599	4.60517	12.17820
lnnfdrs_wgt_fl	5.27185	3.16965	-0.91629	10.17595
lntot_20	13.85677	11.95368	-11.51290	24.57390
lnfil	1.24219	0.53648	0	1.79176
tot_exp	1,468,990.00	4,033,857.00	117.13	52,000,000.00
tot_acre	6,850.718	20,238.340	100	194,496.00
nfdrs_wgt_fl	3,171.993	5,333.709	0.4	26,264.00
tot_20	2,230,000,000.00	6,260,000,000.00	315,000.00	47,000,000,000.00 **
fil	3.91184	1.69067	1	6
d1995	0.02015	0.14069	0	1
d1996	0.11335	0.31742	0	1
d1997	0.01511	0.12216	0	1
d1999	0.04786	0.21374	0	1
d2000	0.16877	0.37502	0	1
d2001	0.05793	0.23391	0	1
d2002	0.10076	0.30138	0	1
d2003	0.06801	0.25208	0	1
d2004	0.04786	0.21374	0	1
d2005	0.05290	0.22411	0	1
d2006	0.14106	0.34852	0	1
d2007	0.13098	0.33781	0	1

**Actual minimum value = 0.0001, reflects minimum value for valued observations.

3.4 Results and Analysis

The reduced form model used in the estimation using the log transformation of all non-dummy variables is:

$$lnTC = \beta_0 + \beta_1 lntot\_acre + \beta_2 lnnfdrs\_wgt\_fl + \beta_3 lntot\_20 + \beta_4 lnfil + \beta_5 d1995 + \beta_6 d1996 + \beta_7 d1997 + \beta_8 d1999 + \beta_9 2000 + \beta_{10} 2001 + \beta_{11} 2002 + \beta_{12} 2003 + \beta_{13} 2004 + \beta_{14} 2005 + \beta_{15} 2006 + \beta_{16} 2007 + \mu. \quad (3.4)$$

The model was initially estimated using ordinary least squares (OLS), and tested for omitted variable bias, heteroskedasticity, multicolinearity, and normality of distribution. The Ramsey RESET test (Ramsey 1969) fails to reject the null hypothesis that the logged model has no omitted variables (p>0.1527). Testing for heteroskedasticity, the Breusch-Pagan test (Breusch 1979) rejects the null hypothesis of constant variance (p>0.0052),

while the White's test (White 1980) fails to reject homoskedasticity (p>0.3501). Impure heteroskedasticity can be ruled out since tests reject omitted variable or misspecification bias. The link test reports $\hat{y}^2$ significant (p>0.013) and $\hat{y}^2$ insignificant (p>0.445) suggesting no omitted variable bias and a correctly specified model.

The possibility of multicollinearity was investigated using a correlation matrix Table 3.3 and variance inflation factor (VIF) Table 3.4.

Table 3.3

	lntot_exp	lntot_acre	lnnfdrs_wgt_fl	lntot_20	lnfil	d1995	d1996	d1997	d1999	d2000	d2001	d2002	d2003	d2004	d2005	d2006	d2007
lntot_exp	1.000																
lntot_acre	0.547	1.000															
lnnfdrs_wgt_fl	0.180	0.061	1.000														
lntot_20	0.197	-0.186	-0.014	1.000													
lnfil	0.357	0.132	0.070	0.177	1.000												
d1995	-0.086	-0.065	-0.008	0.065	0.037	1.000											
d1996	-0.154	-0.142	0.091	0.036	0.060	-0.051	1.000										
d1997	-0.101	-0.084	-0.085	0.003	-0.016	-0.018	-0.044	1.000									
d1999	-0.087	-0.067	-0.050	0.007	0.016	-0.032	-0.080	-0.028	1.000								
d2000	-0.068	0.090	0.040	-0.089	0.026	-0.065	-0.161	-0.056	-0.101	1.000							
d2001	0.000	-0.134	0.023	0.024	0.073	-0.036	-0.089	-0.031	-0.056	-0.112	1.000						
d2002	0.116	-0.040	0.014	0.000	0.066	-0.048	-0.120	-0.042	-0.075	-0.151	-0.083	1.000					
d2003	0.068	0.036	0.031	-0.001	-0.076	-0.039	-0.097	-0.034	-0.061	-0.122	-0.067	-0.090	1.000				
d2004	0.027	-0.081	-0.077	0.059	0.013	-0.032	-0.080	-0.028	-0.050	-0.101	-0.056	-0.075	-0.061	1.000			
d2005	-0.018	0.012	-0.166	0.065	0.032	-0.034	-0.085	-0.029	-0.053	-0.107	-0.059	-0.079	-0.064	-0.053	1.000		
d2006	0.110	0.103	0.029	-0.025	-0.103	-0.058	-0.145	-0.050	-0.091	-0.183	-0.101	-0.136	-0.110	-0.091	-0.096	1.000	
d2007	0.177	0.215	-0.014	0.047	-0.038	-0.056	-0.139	-0.048	-0.087	-0.175	-0.096	-0.130	-0.105	-0.087	-0.092	-0.157	1.000

Table 3.4

Variable	VIF	1/VIF
d2000	4.97	0.201262
d2006	4.47	0.223617
d2007	4.40	0.227404
d1996	3.84	0.260090
d2002	3.57	0.279798
d2003	2.81	0.355329
d2001	2.56	0.391046
d2005	2.52	0.397348
d2004	2.33	0.429905
d1999	2.30	0.434854
d1995	1.58	0.632834
d1997	1.43	0.697751
lntot_acre	1.25	0.801404
lntot_20	1.14	0.877880
lnfil	1.13	0.888337
lnnfdrs_wgtfl	1.07	0.936254
Mean VIF	**2.59**	

The strongest correlation is between *lntot_exp* and *lntot_acre* (0.547) - not surprising since expense and acreage burned are related in terms of scale. The next strongest correlation is between *lntot_exp* and *lnfil* (0.357). The strength of this relationship is understandable since *lnfil* picks up both weather and vegetation effects on fire intensity. No other variables have a correlation stronger than 20 percent. Table 3.4 reflects the VIF's, none over five, but approaching five in years 2000, 2006, and 2007. A VIF of five is generally regarded as reflecting multicollinearity. Since correlations are low, dummy variables are intended to absorb annual effects, and multicollinearity still allows for unbiased estimation, no adjustment will be made for potential multicollinearity.

Endogeneity is not indicated in the results. The correlation between the dependent variable *lntot_exp* and *lntot_acre* is 0.5472. An instrumental variable (IV) was constructed to test for consistency of the structural equation using an instrument representing prior year total acres burned by forest. For forests that had no wildfire information for the prior year, the average acres burned from 1995 – 2007 for that

particular forest were used. Unless completely cheatgrass dominated, acres burned in the

previous year would not provide vegetation available to burn in the current year. Thus,

this IV represents the available acres to burn in a given year.[10] After estimating both the

instrumental variable regression and the OLS, a Hausman test (Hausman 1978) was

performed Table 3.5. The results fail to reject the null hypothesis based on a $Chi^2 = 1.38$

(p>1.00). Thus the original structural equation is the consistent estimator and the original

structural equation can be used without an instrumental variable.

Table 3.5

	(b) iv	(B) ols	(b-B) Difference	sqrt(diag(V_b-V_B)) S.E.
lntot_acre	1.059236	.6064564	.4527795	.51409
lntot_20	.0563798	.0427558	.013624	.0174928
lnfil	.6687555	.9784422	-.3096868	.3557216
d1995	-.1409943	-.0736018	-.0673925	.352097
d1996	.290361	.4200426	-.1296816	.2495793
d1997	.4146926	.4018289	.0128637	.3692208
d1999	.4032331	.6142772	-.2110441	.3391894
d2000	.0409634	.5622059	-.5212425	.6518834
d2001	1.476537	1.450517	.0260198	.2575656
d2002	1.530196	1.823212	-.2930163	.4094359
d2003	1.005087	1.598551	-.5934641	.6980667
d2004	1.477661	1.677595	-.199934	.3401165
d2005	.3429516	.8882758	-.5453242	.6510983
d2006	.855999	1.515061	-.659062	.7829913
d2007	.5300212	1.444211	-.9141901	1.063486

```
                       b = consistent under Ho and Ha; obtained from ivreg
         B = inconsistent under Ha, efficient under Ho; obtained from regress

   Test:  Ho:  difference in coefficients not systematic

             chi2(15) = (b-B)'[(V_b-V_B)^(-1)](b-B)
                      =          1.38
              Prob>chi2 =      1.0000
```

Finally, the underlying distribution of the model was examined using Cameron &

Trivedi's information matrix test (Cameron 1990) and the Jarque-Bera normality test

(Jarque 1980). The Jarque-Bera test rejects the hypothesis of normal distribution for both

skewness (p>0.000) and kurtosis (p>0.000). The Cameron & Trivedi's test fails to reject

---

[10] Many assumptions are made including, all prior and current year acres burn completely, the acres burned in a current period represent the total acres available to burn, and no vegetation burned in a previous year can regenerate to provide fuel for the next year. These assumptions are overly simplified, but not intended for estimation, but rather to test for endogeneity.

skewness (p>0.0802) yet rejects kurtosis (p>0.1817). The skewness is not surprising as large wildfires rarely are "typical" in size, location, or intensity and are individually rare events (Holmes 2008). Studies have indicated that other data related to large wildfires tend to follow a non-normal distribution (Holmes 2008). These results along with mixed tests for heteroskedasticity suggest OLS may not be the best functional form. Instead, a generalized least squares fixed effects (FE) model conditioned on USFS forest management unit is estimated. Within the Great Basin region there are a total of 18 USFS management units, 15 of which are represented in this dataset. Management units were chosen as the fixed effect because terrain, vegetation, and weather patterns define the unique characteristics of a region and have been shown to influence wildfire suppression cost (Donovan 2005). Additionally, each forest has unique management resources and requirements that are factored in to wildfire fighting decisions (GAO 2008).

Table 3.6 summarizes the results of the FE model. The F test for the FE model rejects the null hypothesis that the USFS management unit fixed effects are all equal to 0 (p>0.0359). The proportion of variance explained by the effects (rho) is 0.0889. A random effects model was also attempted. The Breusch-Pagan LM test (Breusch 1979) failed to reject the null hypothesis (p>0.1285) that the $\mu_i=0$, thus random effects are not present and fixed effects is the preferred model. Even though the FE model explains very little about the variance due to management differences, it's preferable to OLS because of the data issues of non-normal distribution and heteroskedasticity.

Table 3.6

						Marginal Effects	

Fixed-effects (within) regression   Number of obs   397
Group variable (i): fs_unit   Number of groups   15

R-sq: Obs per group:
within   0.5230   min   5
between   0.4968   avg   26.50
overall   0.5293   max   66
corr(u_i, Xb)   0.0138   F(16,366)   25.09
    Prob > F   0

lntot_exp	Coef.	Std. Err.	t	P>t	Δ in y	Δ in β_i
lntot_acre	0.5969	0.0479	12.46	0.00	0.597%	1%
lnnfdrs_wgt_fl	0.0838	0.0260	3.23	0.00	0.084%	1%
lntot_20	0.0421	0.0075	5.65	0.00	0.042%	1%
lnfil	1.0518	0.1551	6.78	0.00	1.052%	1%
d1995	0.0693	0.6722	0.10	0.92	31.453%	=1
d1996	0.6768	0.4668	1.45	0.15	64.914%	=1
d1997	0.5852	0.7328	0.80	0.43	50.497%	=1
d1999	0.7180	0.5320	1.35	0.18	65.471%	=1
d2000	0.6828	0.4416	1.55	0.12	66.056%	=1
d2001	1.7607	0.5108	3.45	0.00	187.797%	=1
d2002	1.9269	0.4793	4.02	0.00	225.254%	=1
d2003	1.6746	0.4958	3.38	0.00	173.622%	=1
d2004	1.7991	0.5320	3.38	0.00	193.005%	=1
d2005	1.1707	0.5372	2.18	0.03	102.676%	=1
d2006	1.7304	0.4563	3.79	0.00	187.073%	=1
d2007	1.5254	0.4673	3.26	0.00	151.619%	=1
_cons	4.7336	0.5212	9.08	0.00		

sigma_u   0.4557   F(14, 366)
sigma_e   1.4590   F test that all u_i=0:   1.81
rho   0.0889 (fraction of variance due to u_i)   Prob > F: 0.0359

3.4.1 Results

Total large wildfire suppression expenses for the 397 events included in this data set are $583,291,539 with an average cost of $1,469,248 per fire. Total acres burned were 2,719,735 with an average event size of 6,851 acres. Summary statistics for all variables are presented in Table 3.2.

All variables are individually significant at least the 5 % level except year proxies for 1995–1997, 1999, and 2000 which are not significant at any level. The F test that all β_i's are jointly = 0 is rejected (p>0.000), thus all variables in the model are jointly significant.   The overall $\overline{R^2}$ = 0.5293 suggesting that a majority of the variation in *lntot_exp* is explained by the given model.

The variable *lntot_acre* representing total fire size has a positive coefficient as expected and is highly significant at 1 percent (p>0.000).   The interpretation of the coefficient β_1 = 0.5969 is that a 1 percent increase in total acres burned will increase cost per acre by 0.597 percent. The log-log relationship also reflects the elasticity of acre size on cost per acre which is 0.597. This inelastic relationship means that changes in acre size have a relatively small change on the cost per acre. This is likely due to the economies of scale that occur at larger levels of production (Schuster 1997) and because the majority of total financial cost is generated by the number and type of firefighting assets which are allocated based on fire severity, resources threatened, and availability rather than fire size.

Two variables were selected to represent the contribution of environmental attributes to wildfire cost.   Both are highly significant at 1 percent, β_2 = 0.0838, *lnnfdrs_wgt_fl,* (p>0.000) and β_4 = 1.0518, *lnfil, (*p>0.000) and both have expected positive signs. Unlike *lntot_acre*, both are ordinal variables and β_i must interact with the ordinal value to reveal the marginal effects for each variable category. These marginal values are reported in the Table 3.7 and Table 3.8 below. The difference in marginal

effects across vegetation types and FIL's is quite significant. The impact of the differences in marginal effects for each fuel will be more fully examined in Ch.4.

Table 3.7

NFDRS	Example of Type	Frequency	Weight	Marginal Effect	95% Confidence Interval	
A	Annual grass and forbs	24	0.40	-0.0768%	0.013	0.034
L	Perennial grass	2	0.75	-0.0241%	0.025	0.063
C	Open timber / grass	55	8.78	0.1821%	0.287	0.736
T	Sagebrush / grass	78	11.25	0.2028%	0.368	0.942
R	Hardwoods (summer)	3	14.13	0.2219%	0.462	1.184
B	Mature Chapperal	18	472.50	0.5160%	15.455	39.583
H	Closed, short needle conifer (normal dead)	80	663.75	0.5445%	21.711	55.605
F	Intermediate brush	43	816.75	0.5619%	26.715	68.423
K	Light slash	4	1,636.50	0.6201%	53.529	137.097
G	Closed, short needle conifer (heavy dead)	88	12,523.50	0.7907%	409.636	1049.151
I	Heavy Slash	2	26,264.00	0.8527%	859.079	2200.255

Table 3.8

ln(FIL)	Flame Length at Front of Fire in Feet	Frequency	Marginal Effects
0.00000	0-2	34	0.0000%
0.69315	2-4	59	0.7291%
1.09861	4-6	90	1.1555%
1.38629	6-8	53	1.4581%
1.60944	8-12	47	1.6928%
1.79176	12+	114	1.8846%

The relative level of marginal effects is higher for *lnfil* values than *lnnfdrs_wgt_fl*. This is not surprising as *lnfil* is based on vegetation type as well as weather factors. Lastly, because both variables have a log-log relationship with the dependent variable, elasticity is easily interpreted. Elasticity of vegetation on cost per acre is relatively inelastic ranging from -.07% to .85% but increasing as quantity of fuel available to burn (represented by increased "weight" value) increases. Additionally, the grass *lnnfdrs_wgt_fl* categories reflect negative cost effects. It is not uncommon for few if any fire fighting assets to be deployed for swift, low intensity grass fires. Elasticity of *lnfil* on total cost also varies across levels of fire intensity from 0.00 percent to 1.88 percent. Here the lower two levels are inelastic while the top four levels are elastic. Again, lower

intensity fires have relatively little effect on cost while higher intensity fires require significantly more fire fighting assets. As the fire increases in intensity, effort increases at an increasing rate so that the highest level of fire intensity has the strongest effect on cost.

A single variable represents the value of threatened resources, *lntot_20*. It is highly significant at 1 percent (p>0.000) with an expected positive sign. Like the previously discussed variables, this variable reflects a log-log transformation so that for every 1 percent increase in total housing value within 20 miles from the point of ignition, total expense per acre will increase by 0.042 percent. This may seem like a small marginal effect on cost, but is not insignificant when actual housing values are considered. The mean value of *lntot_20* for the full dataset is $2.2 billion, with a minimum value (not including zero valued observations) of $315,000 and a maximum value of $47 billion. The log-log transformation also represents the elasticity of housing values on total cost. The change in housing values within 20 miles of the ignition point is inelastic suggesting that a change in the value of threatened resources will have a very small effect on total wildfire expense. Basically, while suppression tactical decisions consider threatened resources when choosing cost increasing inputs, they have relatively little effect on total expense compared to environmental attributes. This may seem counter-intuitive when considering the increasing impact of the WUI wildfire suppression cost (Mercer 2005). However, this variable does not imply WUI characteristics thus a specific elasticity is not expected.

The remainder of the model reflects dummy variables where a value of 1 is reported if a fire occurred in the given calendar year. As discussed previously, these variables are intended to proxy several effects, particularly asset prices and asset constraints for a given fire. Variables *d1995–d1997, d1999, and d2000* are not significant at any conventional level yet do exhibit the expected positive sign. An F test fails to reject (p>0.5886) that years 1995-2000 are jointly equal to zero. Of the remaining year variables *d2005* is significant at 5 percent (p>0.03) while the remaining $\beta_{(dyear)}$'s are all significant at 1 percent. The significance of the post 2000 dummy variables is not surprising as 2000 was a watershed year in the wildfire community (FWFMP 2001; GAO 2007). A record \$2.0 billion was spent on wildfire efforts to suppress approximately 8.4 million acres of public lands (GAO 2002). This resulted in a paradigm shift in wildfire management and strategy including creation of the National Fire Plan (NFP) (FWFMP 2001). There is a noticeable change in the dummy variable significance levels between 2000 and 2001. This may reflect the policy and procedure changes resulting from the 2000 wildfire season. An additional dummy variable was constructed where a value of one was taken for observations with fire events in years 2000 and earlier. This variable was significant at 5%. Two additional variables created in an attempt at parsing out the differences post-2000. Interacting total acres with the pre-2001 dummy variable proved insignificant while interacting the same dummy variable with the length of fire in days proved significant at 1 percent.

The Halvorsen-Palmquist semi logarithmic transformation (Halvorsen 1980) was employed to correctly interpret the dummy variables given the logged dependent

variable. Considering the impact of significant dummy variables only, the cost of annual fire occurrence ranges from a minimum increase of 103 percent (if a fire occurred in 2005) to a maximum increase of 225 percent (if a fire occurred in 2002). Effects for each year are detailed in Table 3.6. The signs are positive as expected. It should be noted that the relative scale of the year dummies to the other variables in the model is not necessarily meaningful since they simply serve as a proxy to absorb model "noise".

3.5 Chapter Summary

A fixed effects generalized least square model was used to estimate total wildfire suppression expense in the Eastern and Western Great Basin during 1995-2007. The model included total acres burned, a vegetation variable weighted to reflect available fuel load, fire intensity level, and value of private residences within twenty miles of ignition. Dummy variables for years 1995-1997 and 1999-2007 were included to proxy annual effects associated with firefighting assets and weather effects. In general, most variables in the model are highly significant and the model as a whole is well fitted.

This model can now be used to estimate total suppression cost using different values and parameters. In Ch.4 total suppression cost will be estimated using the marginal effects of *lnnfdrs_wgt_fl*. A cost savings analysis will compare the benefits of vegetation treatment with the cost of no treatment in terms of changes in suppression cost. The variable *lnnfdrs_wgt_fl* will be interacted with the appropriate logged weight value. *lntot_acre, lntot_20,* and *lnfil* will be interacted with the logged mean value given a vegetation type. Dummy variables for years 1995–1997, 1999, and 2000 will not be included because of their insignificance. Coefficients on dummy variables for 2001-2007

will be averaged and set equal to one for all vegetation types to ensure equal proxy effects are applied equally across vegetation types.

Thus the following equation represents the base calculation for the cost savings estimation: (3.8)

$$\hat{TC}_v = \beta_0 + \beta_1 lntot\_acre * \overline{x_v} + \beta_2 lnnfdrs\_wgt\_fl * x_v + \beta_3 lntot\_20 * \overline{x_v} + \beta_4 lnfil * \overline{x_v} +$$
$$\overline{\Sigma}(\beta_{10} 2001 + \beta_{11} 2002 + \beta_{12} 2003 + \beta_{13} 2004 + \beta_{14} 2005 + \beta_{15} 2006 + \beta_{16} 2007) + \mu_v$$

Chapter 4.ESTIMATING COST SAVINGS OF WILDFIRE SUPPRESSION

Like other aspects of public land issues, decisions made regarding vegetation management are impacted not only by the different types of public land users, but also how the different types of users impact each other. In the Great Basin ranching, development, and recreation are the most common user types and they each exact net costs on the land. Additionally, fire events and life cycles of plant and animal species generate their own set of costs. Add in impacts to air and water quality related to use, fire, and species change and it becomes evident that choosing the optimal course of vegetation treatment is not only complex, but crucial to meeting the demands of many. While ideal, estimating the costs and benefits of vegetation impacts to all affected parties is beyond the scope of this study. The complexity of this issue likely explains why a system has not yet been developed to address such a pressing question, but estimating the marginal effects of vegetation type on wildfire suppression cost is a useful and necessary first step. The cyclic relationship between wildfire and invasive species is well documented to be the most pressing land sustainability issue in the Great Basin (GAO 2007; Pellant 2004; Knapp 1996). Since no estimations of this kind exist, restricting the analysis to wildfire cost alone still provides a significant contribution to policy devoted to maximizing the social value of land.

In this chapter I will perform a wildfire suppression cost savings analysis using parameter estimates from chapter three. The benefits of treating vegetation to move it to a more desirable state will be compared with the costs of foregoing treatment and allowing vegetation to decline to the least desirable state. Five of the eleven vegetation

types included in the regression analysis from chapter three will be considered in this analysis: annual grass (A), intermediate brush (F), mature chaparral (B), open timber (C), and sagebrush (T). These specific vegetation types were chosen to better focus the analysis on vegetation states most closely associated with the flatter, lower elevation portions of the Great Basin where invasive species are most deeply entrenched.

Annual grass (A) is dominated by cheatgrass and medusahead. Most importantly, this type is characteristic of a Great Basin vegetation state that has transitioned over a threshold and cannot be repaired in a reasonable time frame without substantial energy inputs. Type A is the worst case scenario. Intermediate brush (F) represents closed stands of piñon pine-juniper habitats. Larger vegetation is mature and overgrown ground cover is scarce due to crowding out by mature upper story vegetation. This type of vegetation state is at a high risk of invasive species domination after a wildfire event (Chambers 2007). Mature chaparral (B) is quite similar to type F, but also contains decadent sagebrush. This state is not necessarily closed as there is more plant variation, but much of the ground fuels are woody. Risk of wildfire is similar, but variation of plant types provides more protection against cheatgrass invasion post fire event (Chambers 2007). Open Timber (C) represents a healthy pinion-juniper habitat at higher elevation levels than any of the other vegetation types considered in the cost savings analysis. Not only are the dominant natives healthier than types B or F, but a healthy ground cover of native grasses and forbs are present. Type C is at a lower risk for wildfire events because of the lack of invasives. In turn, it also is far less likely to be infiltrated by invasives (Chambers 2007). Sagebrush (T) represents a healthy sagebrush state. Native grasses

are present but the overall area is more than one-third shrub cover. This state has little or no invasives present. Risk for wildfire or cheatgrass invasion is similar to type C. Types C and T are the healthiest vegetation states represented in this cost savings analysis.

4.1 Model Application: An Overview

Parameter estimates obtained for representative factors that affect wildfire suppression costs from chapter three are used to calculate cost savings due to vegetation management. To predict average fire suppression costs under the five vegetation types, coefficient estimates for wildfire size (A_{0j}), environmental attributes (α), threatened resources (R_j), and temporal trends (dYEAR) were interacted with representative values appropriate for each of the five vegetation types. The sum of the interacted values and coefficient estimates represents the predicted total cost of wildfire for a given vegetation type. The predicted total cost for each vegetation type is normalized by average total wildfire acres by type to avoid distortion of the final results for vegetation types with disproportionately small or large burned acreage. The predicted total cost is then adjusted for the probability of fire in any given year based on the fire regime for a given type of vegetation. This annualized, expected total cost of wildfire suppression per acre is used as the basis for calculating the expected, long-run cost of wildfire suppression under two distinct scenarios. The first scenario "treats" vegetation to move the vegetation to the next healthier vegetation state. The second scenario assumes that no treatment is given to vegetation resulting in conversion to less healthy vegetation states.

4.1.1 Base Calculation

In constructing predicted values of total wildfire suppression cost, the estimated coefficients for *lntot_acre, lntot_20,* and *lnfil* from the econometric model in chapter three are interacted with the means of the respective variables calculated for each of the five vegetation types. Means for vegetation types Table 4.1 are used to capture wildfire attributes unique to a given vegetation type.

Table 4.1

Vegetation Type		Obs	tot_exp	tot_acre	tot_20	fil
Annual grass / forbs	A	24	1,575,427.00	10,245.53	5,000,000,000.00	3.12500
Intermediate brush	F	43	445,139.50	1,132.02	8,160,000,000.00	3.95349
Mature Chaparral	B	18	829,865.20	1,776.78	8,300,000,000.00	4.72222
Open timber / grass	C	55	1,268,652.00	4,149.86	1,230,000,000.00	3.89091
Sagebrush / grass	T	78	550,382.30	3,373.36	1,170,000,000.00	3.80769

For example, of the five vegetation types, mature chaparral (B) has the highest mean value *lnfil* while annual grass (A) has the lowest.   This is due to the quantity of fuel available to burn based on their biology. Quantity of fuel is directly related to cost. Since the goal of the cost analysis is to capture the marginal vegetation differences, it would be counter productive to use an overall mean and lose this effect. The natural log of the mean of *tot_acre, tot_20*, and *fil* is the interacted value because using the individual mean values of *lntot_acre, lntot_20*, or *lnfil* results in a downward bias of the true mean value. The coefficient for *lnnfdrs_wgt_fl* interacts with the natural log of the weighted moisture hours value for a given vegetation type.

The base suppression cost is constructed using effects base on post-2000 regimes[11]. The 2000 wildfire season was the worst in fifty years (Reese 2001). By September, 6.5 million acres of public and private land had burned and nine of the eleven

[11] Describes the patterns of fire occurrence, size, and severity in a given area or ecosystem.

USFS regions had simultaneous wildfire events (Reese 2001). Wanting to avoid similar seasons in the future, wildfire managers, land managers, and elected officials reconsidered existing wildfire policy, and developed new policies and programs resulting in a paradigm shift for wildfire and land management. The 1995 federal Wildland Fire Management Policy (WFMP) was updated in 2001 - most notable, new policy elements addressing ecosystem sustainability, rehabilitation and restoration. These elements acknowledged ecosystem health and vitality as a core component of economic and social well-being. Along with several other new policy elements, the 2001 WFMP was integral in administrative, procedural, and planning developments including the National Fire Plan (NFP), Healthy Forests Initiative (HFI), and Fire Program Analysis (FPA). All programs focus on integrated forest and wildfire plan development and oversight to ensure that objectives are reached. Results of the regression model seem to capture the paradigm shift. Years 2000 and prior are insignificant while 2001-2007 are highly significant, likely capturing improved management and increased oversight effects on suppression cost. Coefficients on dummy variables for 2001-2007 are averaged and this average value is interacted with a value of one for all vegetation types. The interacted coefficient values are summed by vegetation type then exponentiated to arrive at a value representing the predicted total suppression cost $\hat{TC}_v$ for a wildfire given a vegetation type with average total burn acres, average fire intensity level, average value of threatened resources, and defined weighted moisture hours for that vegetation type plus the averaged annual effect for 2001-2007.

Incorporating annual probability of fire based on vegetation type completes the base value construction. The unique biological properties of each vegetation type influence how often that type of vegetation will burn – also called a fire regime. "Regime year" in Table 4.2 describes for each vegetation type the average number of years between fires. For example, under annual grass (A) a fire is expected to occur every two years, while under sagebrush (T) a fire is expected to occur every 70 years. For simplicity, this fire-regime information is translated into annual probability of fire. Thus, a fire is assumed to occur with a probability of 0.50 (=1/2) under A and 0.01 (≈1/70) under T each year. The probabilities reflected in Table 4.2 are multiplied with $T\hat{C}_v$ to obtain expected predicted total cost of fire suppression for each vegetation type.

The fire probabilities being defined are crucially dependent on the scale of fire size for each vegetation type. These probabilities are derived based on number of years between fires, which is a concept applied to unburned areas. Wildfire probability is applied annually in this cost savings analysis. At the same time, an average total wildfire size is assumed for each year (this is built into the cost basis for each vegetation type). Yet practically speaking, acres burned in a preceding year limit potential fire size resulting in zero wildfire probability for the burned group of acres in the following year. In order to remove the scale dependency of the fire probabilities resulting from simultaneous assumption of wildfire scale, it is necessary to normalize it to a unit cost, in this case per-acre cost. The annualized, expected total cost of wildfire suppression per acre is given by:

$$T\hat{C}b_v = (T\hat{C}_v \ / \ \overline{tot\_acre_v}) * regime\_prob_v. \tag{4.1}$$

4.1.2 Fire Suppression Cost without Treatment

Choosing not to treat a rangeland and allow it to convert it to the next less healthy vegetation types is one course of action. The predicted fire suppression cost of this choice is calculated. In doing so, each vegetation type is assumed to convert to the next less healthy type until it reaches the least healthy type given the 200 year discounting period. The transition is assumed to happen over the intervals shown in Table 10 as "conversion time." For example, mature chaparral (B) after staying in the state for 50 years, will convert into and remain as intermediate brush (F) for five years, then finally into annual grasses (A) in perpetuity. Under each "initial" vegetation type, the expected annual fire suppression cost without treatment is discounted at 4 percent discount rate and summed over 200 years. A two hundred year timeframe was chosen to capture at least one transition for each vegetation type. The discount rate was chosen because it is the rate commonly used for governmental project analysis. Because annual grasses cannot reach a less healthy state, the "status quo" value $T\hat{C}b_v$ is discounted at 4 percent and summed over 200 years. This value $T\hat{C}n_v$ represents the cost of fire suppression per acre over 200 years when vegetation is not treated but allowed to convert to less healthy types. It is formally given by:

$$T\hat{C}n_v = \sum_{1}^{200} T\hat{C}b_{v-1} + \sum_{200-L}^{200} T\hat{C}b_{v-1} \qquad (4.2)$$

where l = the life cycle of the base vegetation type.

4.1.3 Fire Suppression Cost with Treatment

Choosing to treat a rangeland in order to restore it to the next healthier type is the other course of action. The predicted cost of fire suppression when such an action is taken is calculated. Assumed for the purposes of this thesis is that a single treatment, if successful, results in immediate conversion to the next healthier type and will be managed to maintain the "new" vegetation type for 200 years. However, treatments may or may not be successful. An additional probability multiplier is applied to account for the probability of treatment effectiveness given each vegetation type (Table 4.2).

Table 4.2

Type	Name	Prob	Regime Year	Prob of trtmnt success	conversion time (yrs)
A	Annual grass / forbs	0.50	2	10%	NA
F	Intermediate brush	0.20	5	50%	5
B	Mature Chaparral	0.10	10	100%	50
C	Open timber / grass	0.05	20	100%	80
T	Sagebrush / grass	0.01	70	NA	150

To calculate the expected fire suppression cost under treatment, the probability of treatment success is multiplied by fire suppression cost under the next healthier vegetation type or $T\hat{C}b_{v+1}$ and added to the probability of failure multiplied by the status quo cost or $T\hat{C}b_v$.

Under each "initial" vegetation type, the expected annual fire suppression cost under treatment is then discounted at 4% discount rate and summed over 200 years. For sagebrush (T), which is the healthiest vegetation type in this application and therefore cannot be improved by treatment, the discounted sum of the status quo cost $T\hat{C}b_v$ is

used. This value $T\hat{C}t_v$ represents the fire suppression cost per acre over 200 years when

the vegetation is treated to maintain at the next healthier vegetation type and is formally

given by:

$$T\hat{C}t_v = \sum_{1}^{200} (T\hat{C}b_{v+1} * trtmnt\_success\_prob_v + T\hat{C}b_v * trtmnt\_failure\_prob_v) \quad (4.3)$$

4.2 Results

For each initial vegetation type, subtracting from the fire suppression cost with

treatment $T\hat{C}t_v$ the cost without treatment $T\hat{C}n_v$, the cost savings due to treatment is

revealed. The results are presented in Table 4.3 showing striking differences in cost

savings across initial vegetation types.

Table 4.3

		Total Expense per Acre ($)						Total Expense ($)
Vegetation Type		$T\hat{C}_v$ per acre	$T\hat{C}b_v$ per acre w/ regime prob.	$T\hat{C}b_v$ per acre w/ regime & trtmnt prob.	$T\hat{C}n_v$ 200 Yr NPV Adjusted No Treatment	$T\hat{C}t_v$ 200 Yr NPV Adjusted w/ Treatment	- 200 Yr NPV Cost Savings	200 yr NPV Cost Savings Total Avg Ac by Type
Cheatgrass dominated	A	113.69	56.85	63.00	1,477.41	1,637.31	(159.90)	(1,638,236)
Closed P-J conversion prone	F	591.85	118.37	87.62	1,762.26	2,277.16	(514.90)	(582,874)
Mature brush 25% decadent	B	568.65	56.87	12.65	2,983.58	328.79	2,654.78	4,716,961
Healthy P-J w/ natives	C	253.02	12.65	3.75	1,038.93	97.45	941.48	3,907,001
Healthy Sagebrush w/ native grass	T	262.47	3.75	NA	98.01	97.45	0.55	1,869
Total					**7,360**	**4,438**	**2,922**	**6,404,721**

Analysis of the results is most meaningful after first observing the effects fire

regime and probability of treatment success have on the per acre basis values. For

example, type F *tot_exp* value is four percent larger than type B ($591.85 and $568.65

respectively) prior to applying probability of fire. After accounting for fire cycle, *tot_exp*

for type F is 108% larger than type B *tot_exp*. The scale of the expense discounted

over time is highly dependent upon the vegetation fire cycle. Fires cycles decrease when

moving to a healthier vegetation type with treatment application. Over an equivalent amount of acres, *tot_exp* values then decrease and cost savings are realized. Probability of treatment success further adjusts the basis. Types A and F experience less than 100% treatment rates so their adjusted *tot_exp* is greater than the *tot_exp* of the next healthiest type. In fact type A, which contains large quantities of invasives, is the only vegetation type that reports a *tot_exp* that is greater with treatment than without. This is both because the treatment success probability is so low (10 percent) and because the next healthiest type is so much more expensive. This result is as expected since grass fires are relatively inexpensive per acre compared to other vegetation types because the available fuels are so fast burning.

The final results report mature chaparral (B) exhibiting the largest cost savings due to treatment at $2,655 per acre while intermediate brush (F) exhibits the greatest savings lost of $515 per acre due to treatment. Mature chaparral (B) also reflects the largest cost savings at approximately $4.7 million when comparing average fire size for each type. Under the same comparison, annual grasses (A) reports the greatest savings lost at approximately $1.6 million. Cheatgrass fires, while low in intensity, often burn over extremely large quantities of acres. The effect of average acres burned by vegetation type is also striking when considering average fire size effect on cost for types B and C. Mature Chaparral (B) fires are quite costly, yet since they have a 100 percent probability of treatment success and the next healthiest type is 66 percent less expensive per acre, they report the largest cost savings both per acre and for an average fire of its type. However, it is also significant to note that while Open Timber (C) fires realize 186

percent less cost savings than type B on a per acre basis, when considering average fire size for each given vegetation type, type C realizes only 20 percent less cost savings than type B.

It should be noted that the concept of "savings lost" reported for types A and F is theoretical and intended for comparison of differences in *tot_exp*. However, the "unrealistic" negative values follow state and transition ecological theory suggesting treatment of vegetation types that have crossed a threshold are not reversible to a more favorable state in a reasonable time frame without substantial amounts of energy (Stringham 2003). In practice, neither type A nor type F would be treated and any acres not already at the unhealthiest state would eventually succeed.

Revealing wildfire cost savings differences based on marginal effects of vegetation driven by type characteristics is a useful decision making tool for land managers. These values can be used as a basis to evaluate which vegetation types should receive treatment. Similar analysis based on cost of different treatment types with applied success probabilities and discounting over the same 200 year time frame will result in the maximum justifiable cost of treatment when netted against suppression cost. Ch.5 will discuss more completely study conclusions, contributions of cost savings analysis, and suggestions for future research.

Chapter 5. SUMMARY AND CONCLUSIONS

5.1 Review

Virtually all current academic and federal government literature on the topic of wildfire in the Great Basin suggests that the relationship between large wildland fires and invasive species takes the form of an amplifying feedback loop (Chambers 2007; Zouhar 2003). Large fires damage existing native plants creating a niche for invasive species. Invasive species germinate early and rob natives of the scarce water and nutrients available in the arid desert. Invasives are also less fire resistant than natives and as they infiltrate vegetation stands with large quantities of dead and diseased fuels from a century of aggressive wildfire suppression, the probability of a severe fires increase while fire cycles decrease. State and transition ecology models suggest that stochastic events like wildfire and invasive species can transition an unhealthy vegetation state across a threshold that is irreversible (Stringham 2003; Friedel 1991). Once vegetation crosses a threshold opportunity costs are exacted in terms of forgone benefits associated with healthier vegetation states. These costs are borne by society as public lands offer social values such as ranching, recreation, and development. Sub-optimal vegetation undermines land productivity and thus social value.

A cost savings analysis was estimated to provide a useful framework for predicting the marginal effects of vegetation on wildfire suppression costs for large wildfires in the Great Basin. A regression model of total suppression cost for 1995-2007 was estimated using data obtained from RMRS. The parameters of this model were then used to calculate a base per acre total cost by vegetation type. The basis was adjusted to

include ecological parameters to incorporate by vegetation type wildfire regime, treatment success, and succession life cycle. This adjusted basis was then used to calculate a NPV assuming treatment moves vegetation to the next healthier state as well as a NPV assuming no treatment and succession to the least healthy state over the 200 year discounting period. The NPV's were then netted to reveal total cost savings for each vegetation type. The NPV for mature chaparral revealed the largest cost savings both per acre and by average fire size by type for a mature chaparral (B) fire. The NPV's for both annual grasses (A) and intermediate brush (F) reported negative NPV suggesting that no wildfire cost savings are gained when these vegetation types are treated to improve vegetation health.

5.2 Conclusions

The cost savings analysis conducted in this study reflects significant differences in marginal effects on wildfire suppression cost which can be attributed to vegetation type. The results of this estimation align with state and transition theory. Vegetation types that have been completely or heavily infiltrated with invasive species reflect negative suppression costs suggesting that treatment actions would be ineffective in returning them to a healthier state. Vegetation with intermediate levels of invasive infiltration and/or high fuel loading characteristics (e.g. closed stands, significant quantities of ground fuel) reflects the highest suppression cost savings. Native vegetation without significant levels of natives or ground fuels report positive, but low suppression cost savings for the discount period. It is meaningful to consider suppression cost savings scale differences between per acre values and average fire size given vegetation.

For example, Type B reflects suppression cost savings 180 percent greater than the savings for Type C on a per acre basis but this comparative suppression cost savings relationship drops to 20 percent when comparing savings gained from a wildfire of average size for the given vegetation.

Marginal analysis is useful when allocating limited resources to maximize benefits from a socially valued public asset like land. Often instincts lead managers to choose actions with the lowest current cost or manage for a single dominant user group (Finnoff 2007) rather than considering the marginal impacts related to categorical or temporal differences. This approach can fail when making land management decisions since marginal differences between vegetation types result in very different impacts to land productivity. Additionally, quantity or quality changes in land characteristics or rangeland user groups over time must be incorporated into decision making to capture the true cost of the management decision.

5.3 Limitations

While the process developed in this study to estimate the marginal effects of vegetation on wildfire suppression cost provides a meaningful decision making framework there are a few relevant limitations worth discussing. First, there is a significant amount of uncertainty in the model. Stochasticity related to meteorological effects and biological processes contributes to uncertainty in the parameter estimates of the empirical model. Likewise, non-market value losses due to quality changes in air, water, and soil are un-observable and also contribute to uncertainty in the empirical model. The conceptual, theoretical, and empirical models also assume away impacts

other land users (e.g. ranching, recreation) may have on wildfire suppression cost as a function of vegetation management. Lastly, in the cost savings analysis probabilities of wildfire occurrence and treatment success along with the vegetation lifecycle were held constant for each vegetation type across the entire 200 year discount period. These values would likely shift in response to marginal health changes within a regime and also in response to general ecological changes.

The results of the cost savings analysis suggested that several vegetation types had reached an ecological state of irreversibility. It has been shown that when making decisions under irreversibility and uncertainty that decisions will be biased towards losses (Arrow 1974). For the land manager the uncertainty and irreversibility has likely contributed to the loss bias of under-funded vegetation management actions.

5.4 Recommendations

Given the limitations discussed in the previous section along with the NPV estimation process developed for this study a sensitivity analysis is recommended. Especially since the uncertainty impacts the model through several factors, a sensitivity analysis can lend insight to the most heavily weighted factors.

The second recommendation is estimation of expected net benefits of wildfire suppression due to marginal effects of vegetation which is observed in changes in ecological goods and services. These changes might include decrease wildfire frequency and severity or increased vigor of fire resistant natives. Estimation of total wildfire cost minimization as a function of vegetation will result in different management

combinations for a given target acre size when minimizing total cost versus maximizing public land value.

Addition of confidence intervals to the NPV's is also recommended. However, a non-normal distribution and NPV calculation complicate the process. Confidence intervals essentially estimate the reliability of a parameter assuming it was drawn is normally distributed. As discussed in Sec.3.4 this is not the case for the study data sample. Secondly, calculation of confidence intervals for the NPV's is not possible because the standard error associated with the vegetation parameter becomes obscured with the addition of environmental parameters, not to mention, each present value iteration – all of which act as a scalar to the original vegetation parameter. Perhaps a standard error for the NPV's could be estimated by interaction of each scalar with a covariance or by bootstrapping, but both are beyond the scope of this study. Confidence intervals were estimated for each vegetation type by fuel weight in Table 3.7.

Finally, the existing structure could be amended to see if and how vegetation marginal effects on wildfire suppression costs are impacted by a variety of factors. Delay in treatment could be incorporated by allowing the vegetation type to decline for several periods before applying treatment. Treatment types could be directly included in the NPV calculation using a multiplier representing cost or success probability. Treatment methods per vegetation type could be similarly discounted and the resulting NPV could then be compared the wildfire cost saving NPV. WUI factors could be added to the regression model. Population growth in the WUI over time could also be incorporated.

Fire fighting assets could also be incorporated into the regression model to examine vegetation impact on management choices. The estimation structure is very adaptable.

5.5 Contributions

The NPV of wildfire suppression cost given marginal effect of vegetation can be a useful tool for land managers and researchers. While the model certainly has limitations, a similar estimation has not previously been published despite acknowledgement that such a model can contribute to research about the impacts of the wildfire-cheatgrass relationship (Epanchin-Niell 2009; Abt 2008; Gebert 2008; 2005 QFFR; 2002 HFI). Researchers are likely to find the current NPV formulation valuable in representing vegetation impacts. The current estimation process can also be easily adapted to change the values of the environmental parameters, include additional parameters, or expand the analysis to include all vegetation types included in the regression analysis. The constructed vegetation variable *lnnfdrs_wgt_fl* may be of some interest to researchers seeking to incorporate vegetation marginal effects into their study.

As calculated, the NPV's provide land managers with a useful decision tool for making vegetation treatment decisions based on the cost minimization objectives of wildfire suppression management. Land managers can also use the NPV's to compare per acre wildfire suppression costs savings by vegetation to per vegetation treatment costs. Over the past decade escalating suppression costs, driven in part by the cheatgrass-wildfire cycle, have exceeded federal wildfire funding and required fund reallocation from other federal land management accounts including fuel reduction and land rehabilitation accounts (QFR 2005). For the years 2005-2009 the reallocation funds

comprised 27% of total federal suppression expenses annually, approximately $500 million per year (QFR 2009). Reallocation of vegetation management funds to cover wildfire suppression cost may exacerbate the wildfire-cheatgrass cycle. Comparison of wildfire cost savings NPV with a similarly discounted vegetation treatment NPV can provide justification for securing proactive vegetation management.

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