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Prioritizing Invasive Species Management by Optimizing Production of Ecosystem Service Benefits

Final Report

Lisa A. Wainger, Dennis M. King, Richard N. Mack, Elizabeth W. Price, and Thomas Maslin

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Abstract

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Program Of Research On The Economics Of INVASIVE SPECIES MANAGEMENT

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Final Report to USDA ERS PREISM Program

ERS Project Manager: Craig Osteen, USDA, Economic Research Service
Resource Economics Division

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Cover Figure: Top image shows an Idaho landscape dominated by native vegetation and lower figure shows an Idaho landscape dominated by invasive annual grasses of cheatgrass (*Bromus tectorum*) and medusahead (*Taeniatherum caputmedusae ssp. asperum*). Pictured in photo from left to right are David Pyke (USGS), Mike Pellant (BLM, Coordinator Great Basin Restoration Initiative), Susan Gianettino (BLM - Idaho Deputy State Director for Resources).

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Executive Summary

Goals

Federal agencies that manage land in the Intermountain West, such as the U.S. Department of Agriculture's Forest Service and Bureau of Land Management, are concerned about finding a cost-effective way to preserve a range of social benefits generated by public lands, including those associated with food production, recreational opportunities, aesthetic enhancements, and existence values for species or natural systems. A potential threat to maintaining these ecosystem services is invasive species that can dramatically alter essential ecosystem function and put human health and welfare at risk. To manage risks, the managers of these lands devote funds to invasive species control. Since treatment funds are limited, managers must choose from many potential restoration sites and treatment intensities. Their goal in most situations is to choose combinations of treatment options that maximize social benefits based on available funding, labor, and equipment.

The goals of this project were to:

1. Demonstrate how economic principles could be incorporated into a decision support framework, usable by natural resource managers, to assess and compare the social benefits and cost-effectiveness of invasive species treatment alternatives
2. Evaluate effect on treatment decisions of explicitly accounting for spatial heterogeneity of costs, benefits, and risks

3. Evaluate whether promoting particular ecosystem services over others would change cost-effectiveness of site-treatment options
4. Evaluate whether existing data and information were adequate to create practical quantitative decision support tools
5. Assess the potential usefulness to managers of modern quantitative optimization techniques for choosing treatment options.

The aim was to demonstrate that decisions about where to invest in invasive species management could be based on concepts and applications of location and investment theory common to other types of management and investment decisions.

Major Findings

1. Decision-support tools that incorporate economic principles and reveal underlying assumptions and value judgments provide a basis for both expert and stakeholder involvement in decision-making and promote cost- and risk-conscious solutions
2. Available data and knowledge were sufficient to capture the decision process in a standardized framework that could be used to develop consistent decision criteria.
3. The optimization model results suggested that managers could increase benefits of treatment by selecting sites that maximized multiple service benefits simultaneously, rather than the apparent current practice of selecting sites that maximized only one service benefit.

Case Study

Researchers relied on a case study of one of the best studied invasive species in North America, cheatgrass (*Bromus tectorum*). After reviewing much of the available information about this species, models were developed to: measure relative benefit indicators of 4 ecosystem services by location; estimate treatment costs by location; and estimate restorability of ecosystem services by location. Site-specific measures of costs, benefits, and restorability were then combined within an optimization framework to generate optimal sets of treatment options that maximized a weighted sum of social benefit indicators subject to a fixed budget constraint. The ecosystem services selected for final evaluation were

1. Support of recreational antelope hunting;
2. Forage production for cattle;
3. Property protection from fire; and
4. Existence values associated with sage grouse.

The sensitivity of the optimal solution to different weights assigned to these four benefit measures was also tested.

The analytical framework and optimization model were developed by working closely with Federal land managers in the region to specify a framework that addressed local decision support needs. Local and regional technical expertise was elicited to build the underlying models of the decision framework using the best available research and data. However, it became apparent, that even for cheatgrass, one of the most-studied invasives in the United States, a great deal of quantitative information was lacking. As a result, the model relied more heavily than planned on the professional judgment of managers, scientists, and the research team. This situation should be expected for other invasives where the characteristics of the invasive species and its effects on ecosystem services are even less well understood.

This work built on a more general decision framework that had been previously developed by the authors. In this implementation, the availability of data and technical understanding of the cheatgrass problem were evaluated for their adequacy to support an optimization model aimed at selecting management options to maximize social benefits under budget and other constraints. Each management option consisted of a choice of treatment site and treatment intensity when applying a preventative treatment regime following fire.

A Spatially Explicit Approach to Evaluating Cost-Effectiveness

To capture the most important differences among treatment locations, both cost and risk-adjusted benefits were assessed and compared using as much location-specific information as possible. Site conditions and characteristics of the landscape surrounding treatment sites were incorporated to differentiate between the levels and qualities of ecosystem services protected or restored at sites, estimate likely treatment costs, and evaluate the likelihood that treatment would improve such benefits. Benefits were adjusted for treatment performance risk by reducing benefit indicators in proportion to the probability of successful outcomes of preventative treatment.

The approach, grounded in location theory, made use of available GIS data and analysis techniques to consider the effect of location on all aspects of cost-effectiveness, which formed the basis of site selection within the optimization model. For example, the research evaluated the effect of travel time on treatment costs, the effect of location on the number of potential recreational users, and how land cover configuration affected habitat quality. In addition, site and landscape factors that contributed to the expected *recoverability* and *restorability* of sites were used to generate risk-adjusted expected benefits of treatment. Such calculations would not have been possible without a spatially detailed approach.

As expected, many empirical obstacles were found in the development of statistically determined quantitative production functions that could link site characteristics to realized social benefits or site response to treatment with outcomes. Even the basic aspects of risk created by the invasive species were unquantifiable and of questionable certainty. Only cost accounting was based on statistical analysis of treatment spending. Best professional judgment was used to characterize restorability of sites (with treatment) and recoverability (without treatment), although interviews were supplemented by statistical analysis of an available database on restoration outcomes.

In lieu of quantitative relationships between invasive species and benefits, previous benefit evaluation framework and applied indicators of relative ecosystem service benefits were built upon to distinguish treatment sites and treatment intensities. First principles of ecology and economics, and best professional judgment, were used to link characteristics of the built and natural environment to ecosystem service quality. Economic literature and conceptual models were used, where applicable, to identify indicators of relative service value.

A parsimonious set of benefit indicators was selected to reflect aspects of relative value of ecosystem services produced at a site, including:

- Ecological site qualities
- Location attributes contributing to site quality
- Cost of access
- Presence of complementary goods
- Scarcity and substitutability of ecosystem service benefits
- Risk of service disruptions

An example for recreational hunting benefits (antelope hunting) was used to demonstrate how indicators were derived using conceptual models developed from models and understanding available in the economic literature.

Optimization Model

The results of the benefits, costs, and restorability modeling formed the basis of an optimization model developed using commercially available optimization software. The use and demonstration of such software was intended to facilitate increased use of such tools by resource management agencies. Although optimization models are often criticized for failing to capture important aspects of management decisions, a model was developed with an intermediate level of detail to screen many sites. Limitations due to data quality and the ability to demonstrate and quantify possible harm from invasive species were of greater concern than limitations of the optimization framework.

The optimization model was constructed from a social planner perspective, as it sought to maximize net social benefits through the choice of treatment sites and intensities. The budget constraint and weights applied to different ecosystem service benefits were adjustable to reflect alternative sets of potential management goals. Availability of spatial and non-spatial data was assessed for building such a system and it was apparent that a great deal of applicable data were available. However, the screening system would have been improved with the inclusion of additional regional data sets. In particular, a regional assessment of invasive species that included percent cover would have been valuable for assessing restorability and developing a regional treatment strategy.

Results

Despite the many concerns and caveats raised regarding the difficulty of developing the framework, the screening method, based on the best *regional* data and information available, was able to replicate choices made by managers who had detailed *site*-based information. **This result suggests that available data are sufficient to capture the decision-making process in a standardized framework that could be used to develop consistent decision-making criteria.** The value of a framework that replicates current decisions is that the process of site selection can be opened up to greater public and scientific input, rather than relying only on the negotiated solutions currently developed with limited public input. Since the framework could not be based completely on objective information, it seems particularly important that the approach developed here reveal the underlying assumptions, values, and tradeoffs associated with agency decisions; and enable researchers and stakeholders to understand how their input could be used.

Although able to replicate the choices made by the agency, the optimization model generated greater increases in benefit indicators by selecting different management options, suggesting room for improvement in agency decisions. When benefit categories (ecosystem services) were all given equal weight, the optimization model solution set had only one selected treatment site in common with the agency's choice set. When individual benefits were sequentially given full weight in the model and compared, to agency selections, all but one site was common to both sets. This indicated that agency managers picked sites to treat by choosing those that maximized change in benefits of a particular service (e.g., the best habitat site, the best fire prevention site) rather than choosing sites that jointly produced the highest net benefits from multiple services. Because the agency selected a different set of sites than the optimization model when all benefits were weighted equally (their stated goal), **the model suggests that the overall level of benefits might be improved by choosing sites based on their production of multiple benefits simultaneously rather than as superlative producers of individual services**.

The optimization results are somewhat unusual in that they did not suggest tradeoffs needed to be made among services, but rather that services were complementary.

Optimization of one service benefit tended to bring along all other benefits simultaneously, if sites were treated as joint producers of multiple services. High benefits co-occurred for some, but not all, site-treatment options, demonstrating the value of the optimization model in identifying sites where treatment resulted in multiple service enhancements. It should be noted, however, that the optimization model result was dependent on the assumption that native vegetation would be established during restoration, a simplification of real-world outcomes. If differential establishment of seeded native versus seeded non-native plants were modeled, it would have created conflicts among the services provided at some treatment sites. Data were not available to measure or model the details of treatment outcomes.

The optimization model constraint on the number of sites treated demonstrated the effect of resource limitations on the ability of managers to respond optimally to invasive species. The optimization model included the constraint of a maximum of 10 sites treated in order to match real-world constraints of managers with limited resources to allocate among sites. Model results were strongly affected by this constraint. When this constraint was removed, the optimization model solution included many small sites treated predominantly at low intensity, whereas the solution with the constraint of only 10 sites treated larger fires at low to high intensity. That smaller fires were selected by the optimization model when the number of sites was unconstrained, has important implications, as it indicates agencies might create more cost-effective treatment strategies by treating many small sites instead of a few large sites. Because this result is sensitive to many model assumptions, and because the optimization model does not necessarily find the globally optimal solution, further work is needed to confirm this result.

The framework appears to succeed in representing most of the factors that managers consider when setting restoration priorities. In this way, the framework could be used to focus public debate over how public lands are managed and enable agencies to accept input from the public and scientific community. Agencies struggling with such decisions may find these experiences helpful in forming their choice to use optimization modeling for decision-making. In addition, managers may recognize that the many data gaps identified might be filled through alternative recordkeeping methods and treatment protocols.

Overview of Cheatgrass Decision Support Model

Objective: Maximize the weighted sum of standardized benefit indicators for four ecosystem services:

- Recreational hunting (antelope)
- Forage production
- Existence values for Sage-grouse
- Property protection from fire

Constraints: Limits on spending and number of treated sites.

Control Variables: Allocate 5 different levels of treatment among 68 potential treatment sites.

Results: Optimal selection of treatment sites and treatment intensities that generate highest relative benefits and meet all constraints. The expected benefits, costs and risks of treatment differ from site to site based on specific sets of site characteristics and landscape context.

1. Introduction

Increasingly, agricultural lands are being managed to provide a range of ecosystem services in addition to the production of food and other commodities (e.g., USDA Sustainability Programs). In the case of publicly-owned lands, decisions about how to make tradeoffs among different types of benefits through alternate management strategies are particularly challenging due to the multi-stakeholder nature of the management problem. Federal agencies are under pressure from a wide array of stakeholders to maximize production of ecosystem services such as rare species habitat, hunting opportunities, and agricultural productivity. As a result, they must either implicitly or explicitly weigh the different types of social and private benefits to be achieved through various types of land management. The choice of land management actions can depend in critical ways on the weight applied to different benefits and on the ability of alternative management actions to achieve management goals. The efficiency of such decisions can depend on the tools used to target management actions (Babcock et al. 1997).

Invasive species are one of many forces changing landscapes and natural systems today and their influence on public lands in the U.S. spans at least the last 150 years (Mack 2003). Following a definition provided by Mack (2000), invasive species are defined as "species that establish a new range in which they proliferate, spread and persist to the detriment of the environment" and human welfare. Although non-native can refer simply to a species native to one part of the U.S. that has been carried to another part of the country where it has proliferated, these cases are less common. In the main, an invasive species in the U.S. refers to a species transported from outside the U.S. (most commonly from another continent) that has proliferated in its comparatively new non-native range in the U.S. The key here is the recognition that the transport or dispersal of almost all current invasive species to and within the U.S. has been driven and facilitated by human activity. Thus, there is an inextricable link between the character, number, areal extent and influence of invasive species and a diverse array of human activities (Ruiz and Carleton 2003).

Based on the growth in the diversity and extent of invasive species in the U.S. for the last 250 years (Mack 1991; 2003), there is every reason to anticipate this trend will not only continue but accelerate – a function of the increase in international commerce and human conversion of land to developed uses. This phenomenon is not unique to the U.S. – all parts of the planet and all ecosystems are susceptible to the establishment of invasive species, although to varying extents (Mack et al. 2000). Consequently, invasive species collectively should be legitimately viewed as one form of global change. Changes in the earth's climate can and probably already have altered the trajectories for some biological invasions because the distributions of organisms are directly affected by climate (Dukes and Mooney 1999). Thus, the invasions on public lands will likely be influenced in the next 50 years and onward by climate change. Paradoxically, invasive species can also affect the rate of change in the earth's atmosphere, as large regional changes in the form of plant cover on the planet change the rate of CO_2 sequestration (Mack et al. 2000 and references therein).

Based on observations over at least the last 150 years, we know that most recognized invasions have been irreversible. Even where the distribution of an invasion has contracted, it has rarely led to the re-emergence of native species, but more commonly, results in the invasive being supplanted by another invasive species (Mack 2000). These range contractions for invasive species have not been predicted and in some cases are difficult to explain, signaling that our predictive power could well diminish as climate change contributes to the multi-faceted aspects of uncertainty with respect to the range, intensity, and impacts of invasive species.

Many invasions by plants, insects and animals threaten agricultural productivity and other ecosystem services. While lost agricultural revenues are relatively easy to tally, the values associated with the loss of habitat, loss of resilience of agro-ecosystems to pests and disease, changes in water cycles, and other effects are harder to capture. Although damages from invasives can be ameliorated on working landscapes, usually only with recurring effort and cost, the losses generally cannot be checked, much less reversed, on the vast public lands in the western U.S. As a result, the diversity of ecosystem services provided by these vast holdings held in the public trust is being depleted in a seemingly irreversible manner.

Decisions about where to spend scarce management dollars on invasive species depend on a variety of factors but perhaps, most importantly, on the perspective and goals of the decision maker(s). Agency personnel respond to agency mandates, but may not have clear guidance for interpreting rules or setting priorities. By using a relatively formal decision-making framework that allows competing goals to be specified and weighted and that deals explicitly with tradeoffs and uncertainty, managers can better examine, explain, and justify the basis of their decisions.

1.1 Analytical Framework

With this project, we aimed to develop and test an analytical framework for prioritizing where to restore agro-ecosystems to maximize expected streams of ecosystem service benefits. In previous work, we established the basic framework of specifying a benefit accounting system that is based on 1) a wide range of potential social welfare impacts, 2) the probability of successfully restoring services that contribute to those social welfare impacts and 3) the costs of treatment. Here, we developed a case study, with the help of federal management partners, to test whether data and understanding were sufficient to rigorously evaluate and compare costs and risk-adjusted benefits of alternative management actions using this basic framework.

This case study implementation was designed to evaluate whether data and understanding were sufficient to demonstrate links between ecosystem service benefits and invasive spread and whether that information could be used to guide management choices on restoration to the most cost-effective options. Further, we wanted to demonstrate, using economic location theory concepts, how location, or landscape context, influences the relative value of ecosystem services and how the spatial heterogeneities of expected service values resulting from treatment might influence management decisions. In other words, we expected to demonstrate that decisions about where to invest in invasive species management could be informed by using concepts and applications of location theory and investment theory that are common to other types of management and investment decisions.

This report is organized around five main themes:

1. Background on the research approach and case study, including:
    a. Previous work on the cost-effectiveness framework
    b. History and ecology of cheatgrass invasion in the Intermountain West
    c. Management of cheatgrass in the Intermountain West
2. Rationale for and organization of revised analysis framework
3. Development of the benefit assessment module (idealized and practical)
4. Development of a treatment cost model
5. Development of a restorability model
6. Evaluation of options within an optimization model

1.2 Research Approach

Two commonly used tools for evaluating and comparing effectiveness of proposed activities are cost-benefit analysis and cost-effectiveness analysis. In cost-benefit analysis, all benefits are measured in terms of the monetary value of benefits to those affected. In cost-effectiveness analysis, both monetary and non-monetary measures may be used, usually to incorporate benefits that cannot be monetized, or to substitute non-dollar metrics for controversial monetary results. Cost-benefit analysis has the advantage that all benefits are measured in a common unit (dollars) that can be compared to costs to evaluate net social benefits of taking the action. Cost-effectiveness analysis is more commonly used to evaluate how to efficiently allocate funds that have already been dedicated and cannot be used to explicitly demonstrate net social benefits.

We initially chose a cost-effectiveness framework to assess and compare alternate management options because of its advantages in terms of being able to incorporate a wide range of management concerns and goals. We used non-monetary measures to *indicate* changes in relative benefits among management options and compared those changes to costs in order to evaluate change in benefit indicators per dollar spent. We accounted for *performance risk* by incorporating whether treatment outcomes matched management goals. Changes in benefit indicators were therefore weighed by the probability that treatment would succeed in restoring ecosystem services to generate risk-adjusted benefit indicators.

To provide more flexibility in evaluating which combination of cost-effective options generated the highest level of benefits, we input the costs and risk-adjusted benefit measures into an optimization model. Cost-effectiveness analysis can generate many similarly cost-effective options, so the optimization model allows us to combine options in many different combinations to arrive at a set that maximizes benefits for budget and logistical constraints. The optimization model still uses cost-effectiveness to judge options but allows us to allocate the budget most efficiently.

The indicators of benefits were constructed to represent changes in the supply and demand of ecosystem services that flowed from treating a site. Treatment involved preventing domination of the vegetation by a non-native invasive species through post-fire rehabilitation. The indicators were carefully selected to measure the quantity and quality of benefits that people would derive from site restoration. They did not represent willingness-to-pay or willingness-to-accept, but they did represent preferences and were intended to provide a practical means to capture relative social benefits.

We recognized that we could not represent all the details of the important ecological and economic linkages that could affect the selection of an optimal treatment plan and still have a workable framework. Therefore, we sought an intermediate level of detail that would further the goal of developing an economically-based decision support system that was simple enough to be used, but that did not ignore the most important aspects of ecological complexity.

The benefit assessment within both the cost-effectiveness framework and the optimization model were built upon a multi-criteria analysis of stakeholder objectives. The multi-criteria framework, in which different goals are represented and assigned weights, closely matched the decision-making techniques used by some of the regional resource management offices, even though managers did not arrive at their decisions through formal analysis. Our partners for this case study used a typical approach to decision-making whereby priorities for restoration were chosen through a negotiated process among a group of individuals, each of whom represented a particular interest such as hunting, ranching or an environmental concern. This approach matched our multi-criteria analysis framework which allowed users to assign weights to different benefit-generating ecosystem services that supported: hunting, ranching, property protection, and habitat (existence values).

The steps we used to develop the decision support framework for resource managers can be summarized as:

1. Characterize management problems and response options
2. Identify ecosystem services affected by problem and frame benefits analysis
3. Evaluate data available to inform quantification of benefits, costs & risks
4. Develop risk-adjusted cost-effectiveness framework for decision support
5. Test an integrated optimization approach for comparing options, including constraints and assessing uncertainty.

Our previous work developing the structure of the framework resulted in conceptual models for assessing benefits and costs and a detailed indicator system for measuring relative benefits of ecosystem services (Wainger and King 2002, Wainger et al. 2001). This project continued that work by fleshing out the specific restorability model that could be used to assess risk-adjusted benefits, developing more detailed methods to capture the *change* in social benefits due to a management action, and evaluating location-specific costs rather than relying on average costs of treatment per acre.

Because location matters when assessing supply of and demand for ecosystem services, our methods characterized many aspects of the spatial arrangement of resources to capture the effects of service quality, or quantity on value. The benefits assessment considered site-specific qualities within the site and within the surrounding landscape. We included aspects of location that typically drive supply or demand of services such as proximity of users, scarcity or substitutability of services, and risks of disruptions to service flows. Not all aspects identified as important could be included in the final framework due to constraints on data availability or lack of mechanistic understanding. However, a great deal of spatial and supporting data sets were available to evaluate costs, risks and benefits and we applied novel analyses using a Geographic Information System (GIS) to quantify site and landscape characteristics that reflected expected changes in streams of future benefits with and without treatment.

1.3 Why this approach?

In making prioritization decision, managers must typically compare many site-treatment intensity options and do not necessarily have the ability or the resources to monetize the many benefits derived from those treatments for each option. With this case study, we were testing whether it was possible to develop a system that managers could use to incorporate economic concepts into decision-making without introducing unsupportable cost or benefit estimates. The result was the development of a site screening tool that, by necessity, simplified economic and ecological complexity in order to create a manageable analysis tool.

Throughout this report we will refer to the characteristics measured by the indicators as “benefits.” Although the indicators do not and cannot represent an explicit demonstration of net social benefits, they describe characteristics that we know (or hypothesize) contribute to social benefits. The reader is hereby cautioned that we have not demonstrated benefits from treatment but rather these indicators provide a practical proxy for comparing elements of value among management options. We take this approach with the understanding that complete technical and social preference information is not available for conducting more traditional types of benefit assessments.

Despite the simplifications, our system more explicitly accounted for costs, risks and relative benefits than many types of economic decision-support methods, and as a result has the potential to be more revealing of assumptions and subjective judgments than less detailed models. Our management partners encouraged and cooperated with this approach out of their desire to enhance the quality of decisions and to create a more transparent method of decision-making. By revealing assumptions, it was hoped that stakeholders would perceive and appreciate the work of the agency to weigh benefits associated with competing interests with fairness. By revealing assumptions and developing conceptual models, this approach also provided a means to allow input by a broad range of scientists, managers and the public.

1.4 Case Study

For our case study we examined the rangeland invasive cheatgrass (*Bromus tectorum*) which is a management concern throughout the Intermountain West, a large area bounded by the Rocky Mountains on the east, and the Sierra Nevada and Cascades mountains on the west. This area includes eastern Washington, eastern Oregon, Idaho, most of Nevada, and parts of California, Montana and Utah. The boundaries are sometimes drawn by characterizing the combined area of the Columbia River Basin and Great Basin. Much of our focus was on a sub-region within Southern Idaho that corresponded to the management unit whose decisions we were modeling.

Cheatgrass is a non-native invasive plant with a long and well-documented history in this region (Mack 1981, Young and Evans 1978) and serves as a good test species because it is thought that this annual grass affects many services by displacing native vegetation, and increasing fire frequency and extent (Knapp 1996). The species is well-established in the Columbia River and Great Basins, but is not yet at the full extent of its range in North America. The species is exceptionally competitive with the native vegetation and apparently takes advantage of fires to spread and increase its dominance (Knapp 1996, Brooks et al. 2004) although some have questioned this relationship (Johnson et al. 2006).

1.4.1 History and Ecology of *Bromus tectorum* (aka cheatgrass and downy brome)

1.4.1.1 Native Range of Bromus tectorum

A synopsis of the biology (i.e., ecology and genetics) and history of the introduction of this small annual grass into North America can shed light on various aspects of its effect and future influence as well as those tools that could be used to control it. *Bromus tectorum* is a cleistogamous (i.e., almost totally self-pollinating) annual grass that occupies an enormous native range in Eurasia and the northern rim of Africa (Novak and Mack 2001). It is common, though rarely abundant, all across Western Europe, especially in countries surrounding the Mediterranean Sea (Pierson and Mack 1990). Its northward limits in Europe are currently in southern Sweden, into which it has apparently been expanding its range even in the last century. Eastward it extends into all countries in Eastern Europe, including the Ukraine (Tutin et al. 1980). Although common in European locales, it becomes more locally abundant on arid, treeless sites, encircling the Mediterranean (Upadhyaya et al. 1986). This last point is quite relevant to the Intermountain West, which is both arid and largely treeless, except along rivers and streams. The pattern of common occurrence but not abundance extends into the Middle East, where the grass is reported in a wide swath from Turkey to Rajahistan in Northern India. It is also reported to occur in Tibet, and other provinces in western China, although its status (native or introduced) is unknown (R.N. Mack, unpublished data). It is most abundant in its native range in a region that bears much climatic similarity to the Intermountain West – the arid steppes in Turkmenistan and Uzbeckistan.

1.4.1.2 Pre-adaptation to Environments in the Intermountain West

B. tectorum appears well suited to the environments in which it occurs. Much of its native range is arid, i.e., prolonged dry periods are routine and soil pH may exceed 8.4 (Daubenmire 1970). The timing of precipitation in such areas is usually unpredictable; and, as a result, for an annual species to persist there, it must display considerable phenotypic plasticity in response to environmental stochasticity. *B. tectorum* does indeed display enormous plasticity for a wide range of morphological and ecological traits: e.g. germination time, rate of growth, number of tillers, and fecundity. Flowering time – which is late spring, is one of the few phonological events for which the grass displays little site-to-site variation (Rice and Mack, 1991 a, b, c).

Such phenotypic plasticity for so many traits has direct bearing on the extent and persistence of this grass's invasion in the Intermountain West. Each rainfall event triggers a cohort of seedlings within as little as 72 hrs of the site receiving precipitation. Thus, a series of cohorts usually emerge from late summer into winter and onward as long as the soil surface is not snow-covered. Important here is that practically every cheatgrass plant alive on May 1 has produced at least one seed – the basis for the next generation (Mack and Pyke 1983, 1984). This remarkably long period of potential germination accounts for the difficulty in identifying a season in which control can be maximized. Even if control is concentrated in spring, some plants will survive and reproduce.

Other aspects of the biology of *B. tectorum*, in the context of the Intermountain West, explain its proliferation. In autumn, before most of the native species have germinated, cheatgrass seedlings rapidly extend their root system deep into the soil profile (Harris 1967). Establishment of a deep root system sets the stage for warmer conditions in late winter/early spring when these roots undergo further rapid growth and usurp the soil water and nutrients (especially nitrogen) before seedlings of the native species can become established. Although *B. tectorum* may offer little competition to the adults of the large species in these communities (*Artemisia tridentata, Festuca idahoensis, Poa secunda*), it eventually dominates the site by usurping soil resources from successive generations of these species' seedlings (Harris 1967). Over time, as recruitment among these species is effectively truncated, the adult plants, upon death, are not replaced in the community and cheatgrass dominates (Daubenmire 1970). Furthermore, cheatgrass seedlings can tolerate periodic (but not prolonged) drought, and also snow cover and a wide range of soil conditions with respect to texture, salinity, and nutrient levels (Mack and Pyke 1984). Cheatgrass is susceptible to many fungal parasites, which can devastate populations (Klemmendson and Smith 1964). Most serious among these parasites is *Ustilago bullata*, a smut, which infects the developing seed and forms a conspicuous mass of black spores in the aborted seed. Even where the vast majority of plants within a cheatgrass population are obviously infected, some plants appear immune or at least uninfected. As a result, populations are not completely destroyed by this or other parasites (Mack and Pyke 1984).

Despite its tolerance to an impressive array of environmental forces, mortality from drought, ill-timed freezes and voles can be extensive leading to significant interannual variability of biomass. Plants that germinate in late summer/autumn run the risk of desiccation in September-October if precipitation is not frequent. In winter, chief mortality agents are frost heaving of soil peds in which the seedlings have germinated (thereby breaking their tap root) and vole grazing. Although cheatgrass can tolerate prolonged snow cover, it is susceptible to conditions that produce bare ground (no snow fall) accompanied by temperatures well below 0^{o} that freeze the soil. Competition below ground for water and nutrients can greatly affect plant vigor, but does not usually cause plant death.

B. tectorum is highly susceptible to competition for light. This vulnerability is evident in its native and introduced ranges and is reflected by the fact that this grass dominates arid sites that usually have less than 100 % plant cover. As a plant community's canopy coverage approaches 100 %, there is increasing likelihood that insufficient light will reach *B. tectorum* for it to survive. For example, within the Intermountain West, *B. tectorum* reaches its highest abundance in the arid steppe once dominated by *A. tridentata* (Big sagebrush) or on other habitat types that once supported communities dominated by other native shrubs (*Purshia tridentata, Sarcobatus vermiculatus*), as well as in somewhat less arid sites co-dominated by *Agropyron spicatum* (*Pseudoroegneria spicata*) and *Festuca idahoensis*.

On the other hand, the ability of *B. tectorum* to compete is much lower in the meadow steppe that was once dominated by *Symphoricarpos albus* and *Rosa nutkana/woodsii* and a wide variety of broad-leaved perennial dicots (*Balsamorhiza sagittata, Geum triflorum, Geranmium viscosissimum*) (Daubenmire 1970). The grass does occur in the forest riverine communities within the Intermountain West, which are dominated by *Pinus ponderosa, Crataegus douglassi, Salix* spp. and *Populus trichocarpa* but in comparatively low numbers. It is rare to see abundant cheatgrass along streams and rivers in this region, even though these sites would seem to offer ideal growing conditions (abundant soil moisture and nutrients). Competition for light intensifies within the regional forests, where *B. tectorum* can occur in low numbers under the open canopy within Pinus ponderosa forests, but in forests upslope (beginning with those dominated by *Pseudotsuga menziesii*), the grass is very uncommon (Mack and Pierson 1990). Light limitation emerges as the chief environmental limitation on the invasiveness of this species in the region.

The grass's most common habitats in Europe and the Middle East are routinely disturbed, either by livestock or human activity. It is common to see *B. tectorum* within or growing beside cereal fields in southern Europe, alongside a road or path or in a pasture (R.N. Mack, pers. obser.). In contrast, many of the native angiosperms flower infrequently, are poor competitors, do not produce many viable seeds even in the years in which they do flower and are intolerant of livestock grazing and trampling (Mack and Thompson 1982). In summary, *B. tectorum* is superbly well pre-adpated (*sensu* Grant 1963) to the steppe environments into which it arrived. In some respects, its features and traits are superior to those species that evolved within this region.

Given this grass's common occurrence in its native range in association with agriculture and livestock, it is easy to envision the modes by which it could have arrived in the U.S.: as a seed contaminant in agricultural seed (especially the seeds of wheat, oats and barley), as a contaminant in any form of dry cargo or ballast (Mack 1981; Mack 2003). Its long awn (> 1cm) adheres readily to fur or clothing, so it could have arrived as a hitchhiker on clothing, although the number of individual seeds arriving in a single event via that mode seems to make this mode unlikely. At any rate, the first confirmed record of *B. tectorum* in the U.S. was ca. 1790 in Lancaster, PA (Bartlett et al. 2002). Muhlenberg, the doyen of American botany at that time, reports the grass in his Flora Lancasterensis (Bartlett et al. 2002). An examination of his collection at the Academy of Natural Sciences in Philadelphia by one of the authors (Mack) confirms that Muhlenberg had indeed collected *B. tectorum*. Determining whether any descendants from this early population has persisted is problematic because the collection record is silent for the next 70 years with the single exception of an enigmatic reference to a *Bromus sterilis* in Torrey's 1843 Flora of the State of New York, for which the description agrees with *B. tectorum*, not *B. sterilis*. Further searches (by Mack) at the Yale Peabody Museum, the Gray Herbarium (Harvard) and the National Museum of Natural History (Washington, DC) failed to undercover any pre-1859 specimens.

1.4.1.3 Spread of B. tectorum to East and West U.S. Coasts and Inland

The confirmed collection history of *B. tectorum* – which must always be viewed as first detection and not an accurate record of first arrival – is 1859 in West Chester, PA, about 80 km from Muhlenberg's 1790 collection site (Bartlett et al. 2002). It cannot be resolved as to whether this small, isolated population found in 1859 stems from the population in Lancaster or represents a separate introduction. We do know, however, that the collection frequency of *B. tectorum* quickens substantially after 1859. Mack has recently discovered a specimen also collected in West Chester in 1861. Its collection site is intriguing – a plant nursery that was actively engaged in receiving and distributing non-native ornamental species from Europe (Rofini 1986). The nursery's catalogues from 1860 onward do not list *B. tectorum* or any brome for sale (D. Rofini, West Chester Historical Society Collection, pers. comm.). Whether this collection was made from a population that had arrived as a contaminant in nursery stock or a population that had arrived independently of the nursery's overseas business cannot be determined. The strong circumstantial evidence however is that by 1870, at the latest, *B. tectorum* had been introduced repeatedly along the Eastern Seaboard from Philadelphia to Boston. Approximately 20 pre-1870 specimens have been examined from sites in this region (e.g. Providence, RI; Boston, MA; Philadelphia, PA; Camden, NJ; Long Island, NY; R.N. Mack, unpublished data) and provides strong evidence that even if *B. tectorum* had arrived pre-1859 along the Eastern Seaboard, it probably had remained in very small, isolated populations. The high level of collection intensity by local botanists and naturalists in that era was unprecedented and was certainly higher than the collection intensity today. If there had been any more widespread occurrence of this or any other non-native species in that region in the mid-19th century, it would have had a higher chance of detection than a similar event today.

By the 1870s, the collection intensity west of the Appalachian approached the level in Northeastern U.S. , yet no *B. tectorum* was reported in the Mid-West (Ohio) before 1886. This occurrence was joined, however, in the following decade by multiple collections that spanned the Mid-West (R.N. Mack, unpublished data). The collection history became trans-continental in the last decade of the 19th century. By 1900 cheatgrass had been found in Iowa, Colorado, Nebraska and Missouri. The occurrences in Colorado and Iowa are coupled with revealing and valuable observations of this grass associated with packing straw (Crandell 1893; Pammel et al. 1901). It remains equivocal whether the string of pre-1900 collection sites in the Mid-West are all attributable to the expansion of the railroad westward, but the pattern of early collection dates through Nebraska to Colorado, the route of first trans-continental railroad, suggests that the railroad was the chief vector and mode of transport.

Bromus tectorum arrived in the Intermountain West in the last decade of the 19th century. The earliest collection of the species was in 1889 in the Okanogan Valley in British Columbia, and not much later, in 1893, the species was found in the state of Washington (Ritzville). Clusters of other collection dates also occur in eastern Washington and Oregon in the years around 1900 (Mack 1981 and R.N. Mack, unpublished data). Comprehensive examination of all pre-1900 specimens of *B. tectorum* in this study failed to discover any earlier records than those reported in Mack (1981), although a full record has emerged with newly-discovered records in Washington and Idaho [R.N. Mack, unpublished data]).

These pre-1900 dates are significant for at least two reasons. Although they are contemporaneous with the earliest collection dates in the Great Plains (Iowa and Colorado), there are no collection records for the grass in the area, approximately 1000 miles wide, between eastern Colorado and eastern Washington during the last decade of the 19th century. Although it was becoming possible at that time for *B. tectorum* to have been carried by the newly finished railroads across the Northern Rockies into the Far West (Mack 1981; Meinig 1968), the lack of any detection of the grass in the intervening 1000 miles remains curious. Furthermore, the most prevalent genotype of this species, by far, in the Pacific Northwest today is termed Got-4c, which has not been detected in any other populations in the US, east of eastern Wyoming (Novak et al. 1993). The isolation of these early collection dates coupled with the prominence of a unique genotype among these populations strongly suggests that plants with the Got-4c genotype entered the Pacific Northwest from a route other than across the continent, i.e., they arrived on the West Coast and were transported inland.

Spread of *B. tectorum* in the Far West, including the Great Basin moved swiftly after 1900 (Mack 1981); the pre-adaptation of the grass to the arid environments of the region no doubt facilitated this invasion greatly. By 1900, the first records appeared in the Salt Lake City area, northeastern Oregon, and many more records appeared in eastern and central Washington. By 1915, the arid region of Washington had innumerable localities reporting the occurrence of cheatgrass to the extent that it was already being recognized as a harmful weed in wheat fields (Mack 1981). Many new farmers to the region, in their zeal to claim free farmland from the government, attempted to settle claims that had soils far too saline/alkali to support wheat. However, by breaking the thin cryptobiotic crust in the sites, experiencing crop failure and then abandoning the land altogether, they inadvertently set the stage for rapid occupation of these sites by *B. tectorum* (Mack 1981). Habitat types with this soil chemistry (and fate) are the *Sarcobatus verminculatus/Disticlis stricta* and *Grayia spinosa/Distichlis stricta* (Daubenmire 1970), which are primarily in south-central Washington, which is the most arid portion of the steppe in Washington; similar communities occur farther south in the region.

If *B. tectorum* first advanced into the Intermountain West from west coast points of disembarkation (Puget Sound and the mouth of the Columbia River), what role did California ports play as early points of introduction? Apparently, any role these early major ports played in the grass's spread occurred later and does not appear to have contributed the Got-4c genotype of this species that is so prevalent today. One of us (Mack) has now examined all the *B. tectorum* collections at university and other herbaria in California (e.g. UC Berkeley, Humboldt State, Fresno State, Rancho Santa Ana Botanical Garden, California Acad. Of Sciences) and found no pre-1900 collections. Nor do any pre-1900 journals or articles by plant collectors in that region mention *B. tectorum* Mack has now examined these journals (e.g., Zoe and Hilgardia) through the relevant years, and although "no detection" cannot be unequivocally equated with "no occurrence" (Mack 2000), the relatively strong collection intensity in California in the decades around 1900 suggests that the grass had not yet arrived.

In the late 19th and early 20th century San Francisco, and soon thereafter Los Angles, became home to numerous amateur, but nonetheless knowledgeable, plant collectors. Recognizing that they resided in a region with a poorly known flora of both native and introduced species, and further recognizing that new introduced species were continually arriving, a cadre of eager plant collectors combed the streets of San Francisco looking for new plant to report. These collectors routinely (and much more often than yearly) reported the new findings from their surveys of the burgeoning cities of California. In many cases, their collections survive and can be annotated. The earliest California specimen of B. tectorum is dated 1899 (R.N. Mack, unpublished data) – and was collected in northern California, far removed from the metropolitan ports. This suggests that this collection may represent the southward spread of B. tectorum from the North (the Got-4c genotype) rather than representing the early entry of the grass in California and its spread northward. Collections of the species do appear with increasing frequency in northern California and at Santa Barbara, a seaport, 1900-1908.

Entry of *B. tectorum* via San Francisco does likely explain another genotype that eventually reached the western fringes of the Intermountain West near Reno, NV. Mack has traced this genotype (Pgi-2b) to its native range in southern France and the Iberian Peninsula. It is not surprising hat *B. tectorum* would have immigrated from that part of the Mediterranean to California, given the long association of California with Spain. It is surprising that the first American botanists in California, who arrived in the latter half of the 19th century, did not report detecting *B. tectorum*. Nevertheless this genotype is common around Truckee, CA and Reno, NV; yet has not spread widely into the Great Basin; which is anomalous.

The invasion of *B. tectorum* in the Intermountain West followed a common trajectory of a lag phase (about 20-25 years) in which range expansion began from a few isolated locales within a restricted area, followed by a logistic growth phase (15-20 years) in which range expansion accelerated, such that most of the habitat in which *B. tectorum* could persist was occupied by ca. 1930. Given the grass's attributes discussed above (successively germinating cohorts, enormous phenotypic plasticity, ability to usurp soil resources, tolerance of grazing and trampling, tolerance of frequent fire), its spread was, in retrospect, not surprising. Collectively these traits allowed *B. tectorum* to transform the steppe communities in the Intermountain West within no more than 45 years (ca. 1890-1935) and within probably less than 25 years (ca. 1910-1935) (Mack 1981 and unpublished data).

1.4.1.4 Ecological Genetics of B. tectorum in the Intermountain West

Two genotypes, reflecting at least two independent introductions of *B. tectorum* to the region, brought about most of this transformation. Of these, the aforementioned Got-4c genotype was likely most responsible, followed by a genotype that is widespread in western Europe (MCG). Two other genotypes (the aforementioned Pgi-2b) and a trace occurrence of a fourth genotype (Mdh-1b/2b) are also present but in isolated areas. The role of the Got-4c genotype is remarkable because its native range in Europe is quite restricted. Based on approximately 150 populations that have now been examined, the Got-4c genotype is restricted to a small area in eastern Germany and the adjacent portion of the Czech Republic. The climate in this region, while perhaps the most arid in Germany, is hardly similar to climates in the Intermountain West. And yet, in common glasshouse experiments, this genotype consistently showed high vigor and fecundity when compared with other genotypes from Western Europe, including the widespread MCG genotype (Novak and Mack, 1993; 2005).

Ironically, the Got-4c genotype that invaded the Intermountain West is comparatively vigorous. In extensive common glasshouse experiments Kinter (2003) demonstrated that these populations were likely composed largely, if not totally, of plants with the Got-4c genotype and were superior in a wide array of growth features to plants with the MCG genotype. The putative Got-4c plants ranked first in phonologic development germination, tiller growth, tiller number (a strong correlate for eventual plant fitness), panicle number, panicle biomass, vegetative biomass, and total biomass. In other words, the putative Got-4c plants from the Intermountain West emerged sooner, grew faster and taller, produced more biomass and ultimately more flowering panicles than plants of any of the other sources examined in the side-by-side trials. Not only was the Intermountain West invaded by a grass that was pre-adpated to the region's environment, it was invaded by the most vigorous and fecund genotype of this widespread Eurasian grass that has yet been detected.

1.4.1.5 Cheatgrass and Its Role among the Steppe Communities

Bromus tectorum is primarily an invader of the steppe (i.e., those grasslands dominated by perennial grasses) in the Intermountain West. This is a reflection of the strong tolerances and vulnerabilities that the grass displays, including highly tolerance for periodic drought, low soil nutrient levels, saline/alkali soils, snow cover, repeated grazing, routine disturbances of the soil surface, and microbial parasites. The species, however, is vulnerable to prolonged bouts of frozen, bare soils and competition for light. These species characteristics provide signals that can be used to predict sites on which it has proliferated as well as sites from which it is largely precluded.

The steppe communities of Washington illustrate the environmental spectrum of sites in the Intermountain West across which *B. tectorum* shows substantial variation in abundance. In general, *B. tectorum* is most abundant on those sites that are most arid (<250 mm annual precipitation). As a result, the grass is a clear dominant in the arid steppe (e.g. *A. tridentata/A. spicatum* zone); almost as dominant in the steppe in eastern Washington displaying the co-dominance of *A. spicatum* and *F. idahoensis.* The grass is also common, but usually not abundant, within the meadow steppe zone (S. albus/F. idahoensis) which receives the most precipitation (> 350 mm) and forms the lower timberline ecotone with P. ponderosa communities at characteristic elevations (Daubenmire 1970). The lower prominence of *B. tectorum* in the meadow steppe habitats is also attributable to the intensive cereal agriculture in this zone, which involves routine herbicide treatment for cheatgrass. Further west (*A. tridentata/A. spicatum* and *A. spicatum and F. idahoensis* zones), farming is less prevalent and much more of the landscape is devoted to livestock rearing. In addition to supporting native communities that have low canopy cover (the chief limitation for cheatgrass), these economically marginal lands do not usually receive herbicide treatments. It is the large areas in the Intermountain West that support the arid steppe environment, now dominated by cheatgrass, that present the most difficult areas in which to attempt any program to control, much less eradicate, this species.

Cheatgrass is one of several plant invaders on agroecosystems of the Intermountain West and appears to be part of a trend of irreversible changes in plant type and structure that adversely impact a range of valued ecosystem services including those supporting agricultural productivity. The primary mechanisms by which ecosystem services are impacted are through direct habitat loss and indirectly through enhanced fire frequency. The specific links between ecosystem services and welfare changes are discussed below.

1.4.2 Burned Area Rehabilitation and Emergency Stabilization and Rehabilitation Programs

The specific management decision we modeled involved prioritizing the allocation of funds within Burned Area Emergency Stabilization and Rehabilitation programs in southern Idaho. In these and similar rehabilitation programs, burned areas are rehabilitated through seeding and other activities to prevent erosion and promote the establishment of desirable species. In areas where cheatgrass is present, treatment is undertaken to prevent the species from taking advantage of the disturbance in order to quickly establish and become dominant before native species have the opportunity to rebound. Cheatgrass infestations in the Intermountain West are typically managed jointly by Bureau of Land Management and USDA Forest Service, so similar techniques are used in both agencies. Managers at BLM were willing to share databases with us, so we focused on decisions made by BLM in the Twin Falls District of Southern Idaho (Figure 1). The types of decisions this management unit must make are typical of many other natural resource agencies who receive funds from this program or similar programs.

The post-fire period appears to be the most important time to treat since cheatgrass and other annual invasive grasses are well-adapted to fire and take advantage of this disturbance to increase their extent and dominance. A fairly low level of cheatgrass cover, can rebound post-fire to become the dominant species. Cheatgrass survives these recurring fires via its buried seeds, which have been produced before onset of the fire season. In contrast, many of the native herbaceous perennial species and native woody species, such as *Artemsia tridentata*, are eliminated from these sites by cheatgrass fires (West and Yorks 2002). Within a few decades, any site with prominent coverage of cheatgrass is converted to complete dominance by cheatgrass (Knapp 1996; Evans et al 2001).

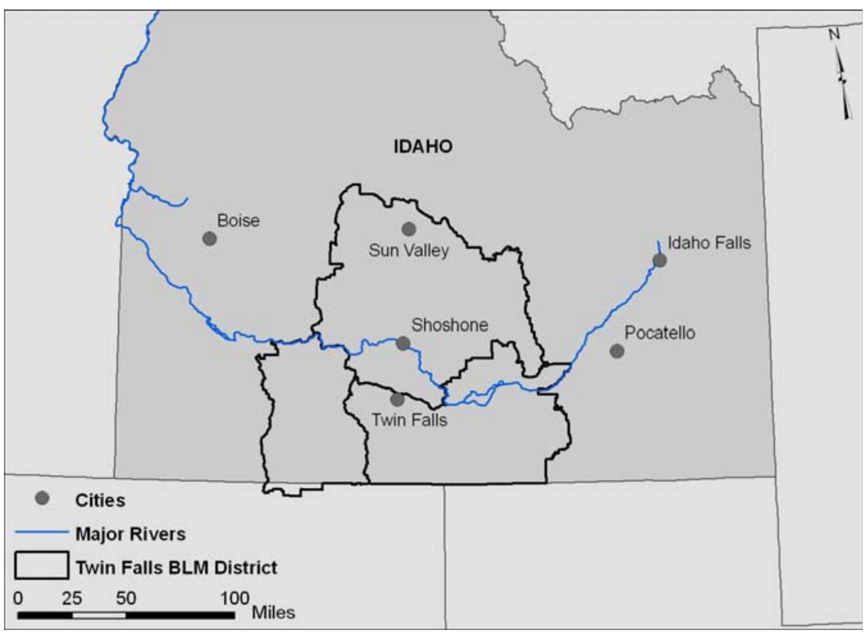

Figure 1. Site Map of BLM Twin Falls District in Southern Idaho
Boundaries of management unit used in case study.

1.4.3 Treatment Options

Ideally, a cost-effectiveness analysis should address the full range of management options to directly or indirectly generate benefits, typically through damage avoided. While the strategies that we will address here involve primarily direct options, some indirect strategies may also be appropriate to prevent harm. For example, if the primary concern with cheatgrass were enhanced risk of fire to property, an appropriate indirect management action might be to fund programs that created incentives for property owners to use fireproof materials in buildings.

The full range of direct options for addressing an invasive species problem include:

1) Ignore	No response.
2) Monitor	Routinely measure the intensity and extent of the weed threat.
3) Contain	Prevent a weed that has already been introduced to a region from spreading to new areas.
4) Control	Remove weeds and limit their growth in areas already infested.
5) Eradicate	Implement control until the weed, seeds and propagules are eliminated from infested areas.
6) Mitigate	Offset the effects of the weed infestation, for example, by creating, restoring, or enhancing alternative wildlife habitat.
7) Adapt	Allow businesses and households to respond as best they can to the spread of weeds; this may be a no-response option associated with an unavoidable weed threat (e.g., when human disturbance is likely to result in reinfestation, or when no acceptable treatment method is available).

For direct cheatgrass control following fire, treatment options available to managers include various seeding techniques combined with physical manipulations and/or herbicide treatments. On rare occasions, sites may be hand planted with seedlings. For purposes of our analysis, we defined five levels of treatment (including no treatment) that represent increasing intensities of effort (person-hours and spending) that could be directed at a given burned site. The management decisions being addressed, in other words, involved choosing which of the five level of treatment intensities to apply at various fire sites, subject to constraints on budget, person-hours, and other factors.

Cheatgrass Treatment Options

1. No treatment
2. Aerial seeding only (mix of native/non-native)
3. Aerial seeding + chaining
4. Aerial seeding + chaining + herbicide
5. Aerial seeding + chaining + drill-seeding

Aerial seeding involves the placement of seeds on burned areas from small fixed-wing aircraft or helicopters. *Chaining* is a technique used to increase soil-seed contact and enhance germination, but can only be used where sites are accessible to tractors. *Herbicide* treatments may be conducted with hand sprayers or aerial application. *Drill-seeding* involves use of special equipment that digs, furrows, drops, and then covers seeds. Drill-seeding enhances germination rates, but is typically applied over small areas, and can only be applied where slopes are gentle and where areas are accessible by tractor.

While treatment techniques may be mixed and matched in different combinations, managers typically apply a low, medium, or high intensity effort to sites based on expected benefits. The specific combinations of techniques listed above are typical, and are used in our analysis as proxies for intensities of treatment effort because doing so allowed us to use actual cost and effectiveness data and information in our analysis.

An important decision the manager must make is whether to place native or non-native seeds on a rehabilitation site. The *non-native species* used include perennial grasses such as crested wheatgrass which have much higher germination rates than native perennial grasses. They offer benefits such as high quality forage and some ability to withstand cheatgrass invasions, but do not offer the same habitat benefits for some native species and are not as well adapted to severe droughts as the native species. The *native species*, on the other hand, have low germination rates, are generally more expensive than non-native species, and do not offer much resistance to cheatgrass on disturbed sites. However, the range of native shrubs has been significantly reduced as a result of the invasion of non-native species. Managers have a goal to retain and restore this characteristic vegetation that supports many native birds that annual grasslands cannot.

A technique we did not consider in our analysis but which warrants further investigation is the use of "flash grazing" or "targeted grazing" where sheep or cattle are stocked at high rates for short periods in order to reduce aboveground portions of all vegetation. When applied at the proper time of year, this technique has the potential to reduce annual grass populations and favor native perennial grasses (Mosley 1996, Davison pers. comm.). The technique is highly controversial due to the potential for permanent harm to soil crusts and sensitive ecosystems, but appears to be a cost-effective method of cheatgrass control that may be appropriate for systems where soil crusts are not present, habitat is degraded, or other methods of treatment can not be applied, such as rocky areas. Evidence suggests the technique is successful at controlling cheatgrass in the year it is used and the subsequent year, but treatment must be repeated to achieve ongoing success (Davison pers. comm.). Long-term studies have not yet been conducted to evaluate potential for long-term control.

1.4.4 Framing the Benefits Assessment

As a widespread invasive that "cheated" wheat farmers out of their expected crop yields decades ago, the impact of cheatgrass on human welfare is relatively well understood compared to most invasive species. Yet, the magnitude of the harm is not well constrained for many ecosystem services. Cheatgrass is thought to create harm by displacing native plants and altering plant type and structure, which leads to associated changes in ecosystem functions such as soil nutrient storage and fuel conditions (Brooks et al. 2004). Functional changes result in habitat losses for certain species and changes in the system's fire dynamics. Fire is a natural component of much of the Intermountain ecosystem we investigated, although some evidence suggests that fire recurrence interval and extent has increased since the spread of cheatgrass and other annual grasses.

As an annual plant, cheatgrass dies immediately after maturation of the next generation's seeds. This maturation coincides all across the Intermountain region with the onset of summer drought. With little or no rain from mid-June until mid-September, cheatgrass adults rapidly die. These dead plants are highly combustible and the monocultures of cheatgrass that have arisen across much of the area create a large fuel load. High air temperature, low rain, and the potential for high winds in summer create ideal and recurring conditions by which these masses of fuel ignite and burn rapidly. In contrast to cheatgrass, the native species mature later and maintain green vegetation later into the summer, thereby shortening the period of dry vegetation. The later date of seed set means natives are more easily eliminated from sites during fires.

An important distinction between a landscape dominated by native perennial grasses and shrubs and annual invasive grasses such as cheatgrass and medusahead (*Taeniatherum caputmedusae ssp. asperum*) is the patchiness of the landscape. The native grasses, referred to as "bunch grasses" form clumpy patches on the landscape in a matrix of shrubs and bare ground. This patchiness is thought to reduce the extent of fires relative to landscapes dominated by invasives grasses, which tend to form a more continuous landscape of grasses (Pyke et al. 2003).

The evidence for increased fire frequency due to cheatgrass is somewhat circumstantial and anecdotal (Roberts 1990, Stewart and Hull 1949, Whisenant 1990). A recent formal study (Pyke et al. 2003) may be the first to quantitatively assess the change in fire frequency due to invasion by annual grass. Their evidence suggests at least a doubling of fire risk in the recent past where cheatgrass cover is dense and a four-fold increase in average fire size, and earlier workers suggested a risk of fire 10-500 times higher in cheatgrass-infested rangelands, requiring 5 times the resources to control (Hull 1965 cited in Roberts 1990).

The prime evidence for increased fire frequency seems to be that the nature of a cheatgrass infestations (dense production of fine-textured biomass that matures and dies earlier than native vegetation) creates a landscape that is more prone to burning, increases the fire season for almost 2 months, and increases the probability that native species will be harmed by fire (Knapp 1997, Young and Blank 1995). Regardless of the ability to document change in fire frequency, what seems clear is that cheatgrass is a fire-adapted species that is able to take advantage of fires to increase its dominance of the sagebrush landscape, thereby suppressing native vegetation growth (Young and Evans 1978).

The enhanced risk of fire due to cheatgrass presence and spread may never be defined because, unlike forest fires that leave scars on trees, grassland fires do not leave evidence that can be retrospectively evaluated (Young and Blank 1995). No method for evaluating historic fire frequency in grasslands that lack trees has been developed, to our knowledge. Simulation models have shown that fire extent is sensitive to fuel fragmentation and load (Duncan and Schmalzer 2004) suggesting that the more continuous cover of cheatgrass would be predicted to increase fire extent relative to the patchy cover of the native grass and shrublands.

A recent report has suggested that there is insufficient evidence to reject the null hypothesis that abundance of invasive plants is not changed by wildfire (Johnson et al. 2006). This report combined data from the peer-reviewed literature on several invasive grasses and forbs in several ecosystem types in an effort to establish a statistically significant effect of fire on non-native species abundance. Unfortunately, poor data quality limited their ability to draw firm conclusions. Many studies they found had small sample sizes, short duration, or used sites with substantially different landscape contexts, making it difficult to draw conclusions given the high natural variability of the systems. The authors document a disturbing tendency on the part of researchers to claim an effect on non-natives when evidence is mixed or variable. However, the evidence to reject the null hypothesis appears as weak as the evidence suggesting an effect, particularly for sagebrush areas. While the scientific method requires preserving the null hypothesis in the absence of evidence to reject it, decades of observational evidence must carry some weight when the quantitative evidence is poor.

Because we had no firm evidence to contradict the widely-held belief that cheatgrass cover is increased by fire and in turn leads to enhanced fire frequency, we accepted the common wisdom of our management partners for purposes of developing our analysis. If this assumption is later proven to be invalid, then the relative benefits of cheatgrass management and the spatial pattern of benefits will likely be misrepresented here. In particular, the benefits assessment for commercial forage production and property protection are highly sensitive to the assumption that cheatgrass reduces the duration of the fire-free interval. On the other hand, for benefits derived from changes in habitat, namely those derived from hunting and existence values, only modest changes in relative benefits might be expected if this assumption proves to be invalid. Treatments aimed at preventing cheatgrass dominance still improve on-site conditions for these services and reduce the chance of cheatgrass spreading to uninfested habitat.

1.4.5 Potential Welfare Changes with Cheatgrass Spread

To inform our selection of benefit indicators, we first evaluated potential welfare effects from cheatgrass spread. Our final analysis did not evaluate these welfare changes explicitly in a cost-benefit framework, but the framing of the benefits is important to correctly identify potential impacts on valued ecosystem services. A full assessment of monetary impacts to conduct a cost-benefit analysis would require prohibitive levels of resources from resource management agencies. However, managers may find this discussion of welfare effects (as opposed to purely ecosystem service changes) as useful for informing choice of ecosystem services to consider when describing and measuring program benefits.

A variety of changes in ecosystem services result from cheatgrass becoming a dominant species in agro-ecosystems of the Intermountain West, and many of these changes can be linked to impacts on human welfare. Effects on ecosystem services were identified through a literature search and interviews with government managers, agricultural extension agents, researchers in academic and government institutions, and private businesses. We evaluated these effects on ecosystem services to identify potential impacts on social welfare (Table 1) and eventually narrowed the focus of our work to four main types of services that appeared to be the primary pathways by which cheatgrass adversely affects people.

Some potential benefits of cheatgrass were also identified for recreators and commercial ranchers. For example, off-road vehicle (ORV) users take advantage of burned areas and therefore appear to derive benefits from increased frequency or extent of fires. However, ORV users also avoid cheatgrass-infested areas at certain times of the year in order to avoid starting fires (M. Pellant, pers. comm.). Therefore, benefits of using burned areas were at least partially offset by the inconvenience of avoiding cheatgrass infestations. Due to the relatively small size of the net benefits, the potential benefits of cheatgrass were not included in the optimization model.

The effect of cheatgrass on commercial forage production is mixed. Ranchers are able to use cheatgrass-infested areas for grazing in the spring and the forage quality at that time can be superior to native species, particularly if the cheatgrass is less than 50% of cover. The fact that cheatgrass dies off earlier than native species is not a significant problem in parts of the study area because ranchers must typically move cattle to higher elevations (above the zone of cheatgrass infestation) in summer to find water sources. Currently, cheatgrass is not widespread at elevations above 5000 feet (Pierson and Mack 1990) however, a new cheatgrass genotype may have emerged that seems to be able to dominate at high elevations (Brown and Rowe, in press). Cheatgrass productivity is thought to be less consistent than native vegetation since it is more susceptible to drought than natives (Roberts 1990).

Despite the lack of direct harm to forage quality at low percent cover, ranchers appear to have growing concerns about systems becoming dominated by cheatgrass (M. Pellant pers comm., various agricultural extension websites) because of the direct impacts of fire on their operations, the loss of late spring-summer grazing areas, and a perception that annual grasses such as cheatgrass have more variable production than perennial species. Fire appears to have the most direct adverse impact on ranchers leasing land. If a grazing allotment is burned, it is BLM policy to prevent ranchers from using the land for 2-3 years while the land recovers. The effect of BLM's policy toward lessees on individual producer profits depends on whether the rancher has alternative allotments to use, and how well he can adapt by changing stocking rates or buying/producing supplemental feed. Ranchers are generally opposed to this policy of being removed from the land, indicating that they see an adverse impact of fire, or, more specifically, of government's response to fire, on profits and/or their personal welfare. Other evidence that ranchers are increasingly worried about cheatgrass spread is the development of private associations of ranchers and others that are aimed at controlling cheatgrass (M. Pellant, pers. comm.).

Anecdotal evidence from managers and ranchers indicates that the variability of forage production can be greater when a system is dominated by annual grasses, such as cheatgrass, as opposed to perennial grasses. Such a claim is supported by ecological principles that predict higher variability of biomass production in an annual plant species relative to a similar perennial species, over the long term. Research also supports the concept that cheatgrass can have variable productivity year to year and plot to plot (Bradley and Mustard 2005, discussion in Johnson et al. 2006). In the end, we identified the primary effect of cheatgrass on commercial ranching profits as the lost profits of being removed from the leased grazing allotment for a 2-3 year period.

The effect of cheatgrass on hunting opportunities is similar to the ranching story. Many popular game species are able to use areas dominated by cheatgrass, implying little to no welfare effects from post-fire treatments.[1] However, two hunted species appeared to be directly at risk from cheatgrass: antelope (*Antilocapra Americana)* and sage-grouse (*Centrocercus urophasianus*). These species do not readily use cheatgrass-dominated areas and therefore require native vegetation to thrive. Neither species is listed as threatened or endangered. However, managers anticipate future problems for these species since their habitat areas are shrinking and populations of sage-grouse appear to be declining from historical levels. The conservation status of sage-grouse is considered "near threatened" (BirdLife International 2006). In addition, managers expressed some concern that deer overwintering habitat was also threatened by cheatgrass since deer move to low elevation in winter and cheatgrass that dominates at these lower elevations does not provide any food value for deer at that time of year. However, the deer population appears stable for the moment, so the degree of this threat is not clear.

A list of ecosystem services that we identified as being affected by cheatgrass is provided in Table 1 which includes a brief description of the mechanism by which cheatgrass infestations are thought to induce welfare changes. The specific ecosystem services that were the focus of attention in our benefits analysis are shown in Table 2.

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[1] Cheatgrass dominated areas are widespread and plentiful, so hunting opportunities are not lost by treating cheatgrass areas. Popular hunting species that use such areas, such as the chukar (*Alectoris chukar*), are generally abundant and, for the chukar, the conservation status is "least concern" (BirdLife International 2005).

Table 1. Potential Welfare Impacts of Cheatgrass

Ecosystem Service Category	Mechanism of Welfare Change	Potential Impact on Welfare (consumer or producer surplus)
Hunting (sage-grouse, antelope, and potentially mule deer)	Fires destroy habitat & reduce resilience of populations/herds.	Loss due to: increased costs of access, reduced bag rates, reduced quality of experience.
Hiking / Dog Walking	Generates time of year restrictions on natural area use since seeds are a nuisance to dogs and people. Animals get infections from embedded seeds which require operations to remove.	Loss due to: increased costs of access since users avoid infested/burned areas; reduced quality of experience.
All terrain vehicle use	Users avoid when fire-prone, but burned sites are preferred.	Potential gains or losses due to increased access of burned areas or loss of access in fire-prone areas.
Property protection	Increase in fires	Private or public losses associated with increased costs for fire prevention (e.g., building requirements), property replacement, and higher insurance rates.
Health Risks	Increase in fires cause undesirable air quality which can trigger health problems (Ferguson et al. 2003)	Losses due to increased incidence of disease and health care costs or insurance rates; lost worker productivity.
Aesthetics	Burned areas offer degraded views and wildlife are displaced.	Losses due to reduced quality of views and lost wildlife viewing opportunities associated with driving or recreation.
Soil Stabilization	Fires create erosion events that can block roads and degrade stream habitat (includes degradation of salmon spawning areas) (Klemmendson and Smith 1964; USACE 2002)	Losses associated with public costs to remove material. Effects on salmon difficult to characterize as losses because other stressors on populations dominate.
Existence values	Sagebrush (*Artemesia*) and sage-grouse (among other species) are displaced by cheatgrass and are becoming increasingly scarce. Widespread loss of sagebrush in this area appears irreversible without human intervention (Whisenant 1990) particularly in salt desert communities where natural fires were historically rare. Raptors are indirectly affected through loss of rodent population (Brooks et al. 2004, US DOI 1996)	Losses due to diminished population of characteristic valued species.
Producer losses by ranchers	Increased fires. Fires reduce productivity and/or increases costs. BLM policy is to shift ranchers off burned areas for 2-3 years. Without fire - increased variability of forage; costs associated with seeds becoming embedded in animals.	Lost producer profits if ranchers have no substitute land after fire. Potential reduction in profits from having to provide supplemental feed or reduce herd size.

Table 2. Ecosystem Services Used in Benefits Analysis

Benefit Variable	Service Description
b_1	Recreational Antelope Hunting
b_2	Forage Production for Commercial Ranching
b_3	Property Protection (public and private)
b_4	Habitat Support for Sage-Grouse (characteristic species)

1.4.5.1 Off-Site Benefits of Site Treatment

Treatment at a site not only improves on-site conditions for providing certain ecosystem services and related benefits, but also improves the expected future streams of services and benefits from neighboring sites by preventing their infestation or reducing the risk of fire. To capture the relative level of protection to neighboring sites of treating any particular site, we relied on measures of landscape connectedness. Our assumption was that a site would be more likely to confer advantages to other sites if it were well-connected (ecologically and biophysically) to areas showing no cheatgrass cover. The degree of connection was measured using landscape metrics described in Section 3.2.2.

2. Problem Formulation

Conceptually, the manager's decision problem is to choose specific sites to treat and specific intensities of treatment at those sites so as to maximize change in total benefits subject to a budget constraint. In practice, only a small proportion of sites can be treated with the available budget. When different services cannot be provided simultaneously at the same site, the manager must make tradeoffs among services and the services are considered *antagonistic*. On the other hand, if services are *complementary* and treatment of a site enhances all services simultaneously, the manager may be able to maximize aggregated benefits without having to explicitly trade-off services. In our formulation of the problem, we employ a weighting scheme that assigns relative importance to different services, providing a means to trade-off services when aggregating benefits. The service weights must be the same for all individual sites, but since different sites will have different capacities to enhance particular services, these weights can have proportionately different effects on the overall benefits from treating different sites.

Many potential treatment sites must be compared for cost-effectiveness. In Idaho alone, over a three-year period (2001-2003) about 4000 fires burned on all lands while only 50 or so were treated on BLM lands under the two programs we examined: Emergency Stabilization & Rehabilitation and Burned Area Rehabilitation (Figure 2). We used information for 46 of the treated fires over the three-year period to evaluate treatment spending patterns.

Within BLM, managers within a given district office (sub-state level) must decide each year which fires within their district will be treated. For the Twin Falls District that we were modeling, in 2002, 114 fires were recorded on BLM lands. In our cost-effectiveness analysis and optimization modeling, we compared in detail 68 of those fires that had been mapped within the GIS database of fire occurrences and that were larger than one acre.

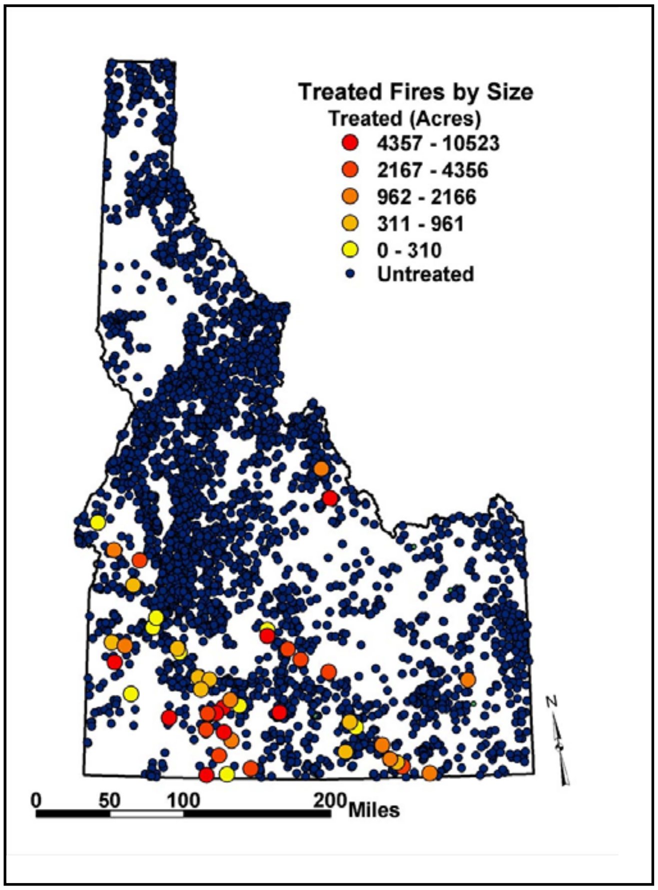

Figure 2. Treated and Untreated Fires 2001-2003
Fires shown in yellow, orange or red were treated under Emergency Stabilization & Rehabilitation and Burned Area Rehabilitation programs in Idaho in 2001-2003. Blue dots are fires of any size occurring during that same period, on any public lands, that were not treated in these programs.

2.1 Optimization Model Overview

A generalized version of the optimization model can be summarized as follows:

Let B_t equal the aggregated net benefits at time t of improvements in the set of ecosystem services $\boldsymbol{j}$ from treating a set of sites $\boldsymbol{i}$ with treatment $\boldsymbol{k}$[2] where b_{ijk} is a vector of services (b_{i1k} = recreational antelope hunting, b_{i2k} = forage for commercial ranching, b_{i3k} = property protection, and b_{i4k} = benefits associated with sage-grouse) and aggregate net benefits (B_t) are calculated as:

$$B_t = \sum_i \sum_j w_j \left(b_{ijk}^{with} - b_{ijk}^{without}\right) \qquad \textit{(Eqn 1)}$$

b_{ijk} = benefit, with or without treatment
w_j = weight applied to service j
i = location i, (sum over treated *subset* of burned sites)

[2] Note that treatment may result in improvements of some services at the expense of other services, so the direction of change in benefits with treatment is not necessarily positive.

j = service j (sum over service benefits : recreation, forage, property protection, existence values for sage-grouse)
k = treatment intensity applied
Benefits are evaluated at a single point in time, 2-3 years following restoration.

The budget constraint is set to the allocated program budget and cost of treating a site is calculated as:

$$TC_{\boldsymbol{ik}} = FC + JC_k * JT_i + TCA_{ik} * A_i \qquad \textit{(Eqn 2)}$$

FC = Fixed Costs
JC_k = Journey (Travel) Cost, personnel and equipment
JTi = Journey Time (hours)
TCA_{ik} = Treatment Cost per Acre
A_i = area treated (acres)

The factors under control of the manager are:
i = site treated
k = treatment applied to burned area i (expressed as treatment intensity level)
w = weights assigned to benefits $\boldsymbol{j}$
The treatment intensity, k, includes the use of multiple trips to sites and costs reflect both inputs per trip and number of trips.

2.2 Short vs. Long-Term Benefits Analysis

Benefit calculations can be conducted over a short or long time period. In the world of natural resource decision-makers, social benefits are generally maximized when the expected *future stream* of benefits is maximized. Someone making an investment in preserving or restoring a natural asset typically wants the investment to pay out over some period of time. For this analysis, we set a single evaluation point for benefits of 2-3 years following treatment, to increase the tractability of the system for use by managers. Managers can reasonably evaluate short term success of a restoration treatment 2-3 years following rehabilitation and, from that short-term success, can infer potential long-term success. We recognize that the use of a single short time-frame in the benefits equation, while a great advantage for simplifying calculations, may emphasize short term gains over long-term results.[3]

To counteract this apparent emphasis on the short-term, we incorporated two approaches in calculating net benefits to reflect long-term impacts of treatment. The main approach was to define benefits in terms of the change in the percent invasive cover and to limit benefits to less than 100% of maximum, even if 100% native cover would be expected at the post-treatment evaluation 2-3 years following treatment (see detailed discussion of benefits function below). Short-term gains in native plant cover are expected to translate into higher likelihood of long-term benefits (e.g., in terms of species survival and preservation of hunting options), but full potential benefits would not be

[3] By comparison, a multiple time period or dynamic analysis would allow us to project benefits in each of many future time periods, apply a discount rate to future benefits, and examine residual land values at the end of the time horizon to better model investment decisions.

realized at that point in time. Using this construct, the benefits function reflected the concept that some benefits would not be realized until native cover was maintained over some future time frame (e.g., 30 years hence).

A secondary approach to reflect potential future benefits was to include a term in the benefits calculation that projected the importance of a site for preventing spread of infestations to neighboring areas. This term was an indication of the persistence of benefits, produced by neighboring sites, that would be maintained by treating site *i*. In other words, treating the site of interest would be expected to slow or prevent infestation of neighboring sites, or reduce the risk of fire to neighboring sites, thereby allowing those sites to produce a greater flow of benefits over the long-term.

3. Benefits Assessment

Our method to assess benefits of treatment relied on indicators of benefits and did not directly quantify economic value of services in monetary terms. The endpoints we measured were evidence of changes in economically important ecological services, but had limitations in terms of being able to quantify probable use, substitution among sites, and user benefits. We recognized these limitations, but expected that this system would provide adequate information about benefits to enhance decision-making by resource managers and potentially increase economic efficiency of resource allocation decisions.

The main difference and advantage of an indicator system of benefits rather than a system of biophysical endpoints (such as number of sites, or acres of grazing area) as a management target, is that the system incorporates components of asset quality, scarcity and value. Although the analysis is not without its judgments about preferences, the value judgments are made explicit and we reveal assumptions about relative values that are inherent in resource allocation decisions but not always made explicit.

The approach to measuring benefits of treatment that we used resulted in benefit indicators that were site-specific and took into account spatial heterogeneity of land cover and land use both on and in the vicinity of burned areas. Quantifying relative benefits at a particular site involved several steps. The description of our benefit estimating approach below progresses from the general, what we were trying to measure, to the specific, the equations and relationships we used to calculate relative benefits for each case of site and treatment intensity. We also describe how benefits were risk-adjusted using expected outcomes and different types of benefits weighted to reflect management goals.

3.1 General Benefits Model

Let b_{ijk}, the benefits from service *j* of treating a given burned area *i* at treatment intensity *k*, be a function of the maximum possible benefits from the site in the absence of invasive species, $b_{ij}^{\max}$, the site restorability or treatment effectiveness, S_{ik}, and the off-site benefits derived from protecting neighboring sites from infestation or reducing the risk of fire in neighboring cells in any given year, C_i.

$$b_{ijk} = f\,(b_{ij}^{\max}, S_{ik}, C_i, A_i) \qquad \text{(Eqn 3)}$$

$b_{ij}^{\max}$ = maximum possible site benefits at site i for service j

S_{ik} = restorability outcome, measured as % native vegetation, for a given site and treatment level

C_i = contribution to maintaining off-site benefits, measured as the effect of the site on contiguity / connectedness of native vegetation

A_i = size of treated area

Benefits derived from treatment are thus defined as a function of innate site characteristics that affect expected change in benefits in three ways. First site and landscape characteristics are used to measure a maximum potential level of benefits, $b_{ij}^{\max}$ (in the absence of invasive species). Second, site and landscape condition determine the site's ability to recover post-fire (recoverability), the outcome of treatment (restorability) and the difference between the two, which is used to generate net benefits. Finally, the protection conveyed to existing sites from treatment is quantified in terms of a descriptive statistic of land cover spatial pattern, C_i, that is essentially a proxy for invasion or fire risk to neighboring lands. See further description below.

The site restorability, S_{ik}, was measured as the estimated percentage of native vegetation cover post-treatment. The estimation of S_{ik} is described in Section 5.

Unfortunately, little empirical information was available to judge the expected relationship between how ecosystem service benefits changed in response to different levels of invasive cover. For example, as discussed earlier, the relationship between invasive cover and enhanced fire risk has not been quantified, so the change in fire risk could not be quantified. Such a lack of quantitative relationships between an environmental change and change in ecosystem service production level or quality, is common, even when harms have been documented. Therefore, knowing the prevalence of this problem, we relied upon ecological and economic first principles to create a theoretical relationship between change in vegetation and benefits.

3.2 Change in Benefits with Treatment

If b_{ijk} represents the realized benefits at site i with either treatment or natural recovery, we compare b_{ijk} with and without treatment to estimate the change in benefits for a particular management option. For each site, we consider the maximum possible benefits in the theoretical absence of invasive species[4], the percentage of native species realized after a management option is applied, S_{ik}, and a fixed parameter, m, which is used to define a hyperbolic function of benefits that is specific to site, benefit and treatment.

[4] We were not able to infer the value of all benefit metrics in the absence of invasives, however, we made the assumption of no invasives when data allowed.

Let b_{ijk}, with or without treatment, be represented by:

$$b_{ijk} = \left(\frac{b_{ij}^{\max} S_{ijk}}{m_j + S_{ijk}} \right) \qquad \textit{(Eqn 4)}$$

We used this hyperbolic form for the benefits function (after Monod kinetics equation) because it had the desired form (continuously increasing function with decreasing slope, Figure 3) and because the parameter *m* had an intuitive interpretation. Parameter *m* represented the percent native vegetation (value of S_{ik}) at which half of maximum benefits were realized. Parameter *m* can be elicited from managers and researchers and adjusted for each benefit to alter the responsiveness of the benefits to changes in infestation level. In other words, the slope will vary as m varies. The parameter *m* was site-independent and benefit-specific, while $b_{ij}^{\max}$ was site- and benefit-specific.

The benefits equation had the property that at 100% native vegetation, full benefits were not achieved. It was useful to apply this equation for our purposes in order to reinforce the notion that we were evaluating success before full long-term benefits were realized (i.e., over a long time horizon). Therefore, we did not suggest that 100% native vegetation equated to full benefits early in the site recovery. Not until the plant community is mature (especially for shrubs), or until successive years pass without cheatgrass becoming the dominant species, would full benefits be realized. Therefore, as constructed here, the percentage of native vegetation, evaluated 2-3 years following treatment, represented a relatively early point in the restoration trajectory and was interpreted as a leading indicator of future long-term benefits.

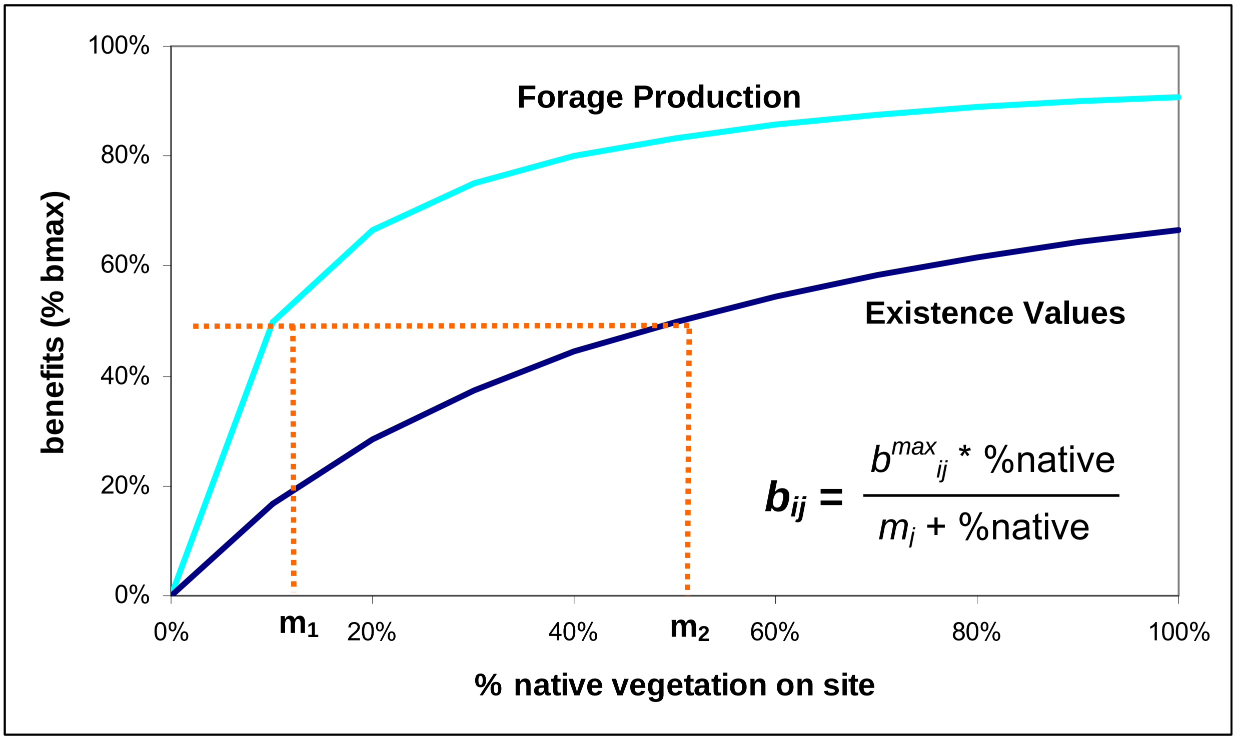

Figure 3. Benefits Equation Showing Responsiveness of Benefits to Parameter *m* and Percent Native Cover
The parameter *m* is varied for each ecosystem service to change the slope of the function that relates level of infestation to benefits. See Equation 4.

The functional forms used in benefit evaluation can be tailored to different services, depending on the information available to relate invasion to benefits. For this case study, the understanding of these relationships was tentative. Therefore, we did not diverge from this simple formulation for different services. Further work is needed to review empirical evidence and elicit best professional judgment on the nature and form of benefit response curves that more accurately reflect the causal relationships between treatment, biophysical effects, resulting changes in ecosystem services, and related benefits.

3.2.1 Scaling Change in Benefits to Effects on Neighbors and Fire Size

To derive the change in benefits, we calculated b_{ijk} with and without treatment and scaled the result by two factors:

$$\Delta b_{ijk} = \left(b_{ijk}^{with} - b_{ijk}^{without}\right)\left(1 + \Delta C_i\right)\left(A_i\right) \qquad \textit{(Eqn 5)}$$

where C_i, or the change in C_i, had a value between 0 and 1, and scaled the benefit function in proportion to the importance of the site for maintaining connections between native vegetation in the landscape surrounding the site. This factor is an indicator of the likely level of spillover benefits conferred on neighboring sites. The variable A_i scales the benefits with the size of the burned area.

The C_i term reflected the importance of a given burned site for maintaining connections between uninfested areas. The change in C_i was calculated as the difference between the pre-fire connectedness of native vegetation and post-fire connectedness, assuming the burned site became dominated by invasive plant cover and acted as a "hole" in the mosaic of native vegetation (Figure 4). The change in landscape connectivity reflected the importance of the site for maintaining connections between native vegetation and therefore the importance of the site for preventing further decline of the neighboring sites.

C_i was a proxy for the relative ability of site *i* to protect the stream of benefits derived from neighboring sites. The greater the loss of connection between native vegetation, the more site *i* acted as a "fire break" both in the literal sense and in the sense of potentially preventing degradation of services. The connectivity measures calculated using spatial data of native vegetation reflected both the proportion of the landscape in native vegetation and the spatial arrangement of that vegetation. Depending on the spatial arrangement of the native vegetation patches, cheatgrass seeds may have a greater ability to invade new areas, fire may have a greater tendency to spread, and organisms may have enhanced abilities to make use of the native vegetation.

An individual patch of native vegetation differs in its ability to maintain connections (and thereby support services at neighboring sites) based on the configuration of surrounding land. For example, if the patch is an island of native vegetation surrounded by invasive species, the change in landscape-scale connectivity with invasion will be lower than a site that serves as a bridge between two patches of native vegetation across an area dominated by invasives. Therefore, we assume that the greater the change is in connectivity with the loss of the burned site, the greater the likelihood that services at neighboring sites will be disrupted.

3.2.2 Scaling Benefits to Size of Treated Area

The change in benefits was multiplied by area of the burn (in acres) to account for size when comparing benefits. A linear response of benefits with fire size is probably more acceptable for some services than others (e.g., forage vs. habitat). This topic is ripe for further research, but given the lack of demonstrated functional relationships between size and level of benefits for these particular services, we chose the simplest assumption.

3.2.3 Spatial Pattern Metrics Used to Reflect Landscape Fragmentation

The level of landscape fragmentation before or after a fire (C_i) can be calculated using many of the spatial pattern metrics developed to represent aspects of land cover fragmentation (O'Neill et al. 1988, Gufstafson 1998). These descriptive spatial statistics are used to evaluate the heterogeneity of landscapes and characteristics of land cover pattern through analysis of digital land cover maps. Land cover data are most commonly represented as a lattice of values or equivalent grid cells, as displayed in many types of imagery or Geographic Information Systems (GIS) system raster files. Typically, contiguous patches of one or more habitat types are identified, delineated, and described through appropriate metrics, so that habitat qualities can be evaluated.

A variety of techniques are used to calculate pattern metrics; some of which reflect information theoretic concepts such as the probability of encountering a given cover type adjacent to a site *i* within a landscape (McGarigal et al. 2002). Metric values are affected by data resolution, typical patch size within a given landscape, and by the classification system and other methods used to create the landscape data (Wickham et al. 1997). Therefore, care must be taken to characterize the landscape in meaningful ways before applying such metrics. Of course, as is the case for all statistics, such metrics are only as good as the source data from which they are calculated.

These metrics have been applied within numerous ecological and economic analyses in which fragmentation is of interest. For example, they have been applied in economic hedonic models to assess the effect of neighborhood characteristics on home values (Geoghegan et al. 1997). Landscape ecologists typically apply such metrics to evaluate aspects of habitat quality (Fahrig 2003, Villard et al. 1999) or to assess effects of human actions on biophysical condition (O'Neill et al. 1997, Jones et al. 2001, Hunsaker and Levine 1995).

3.2.3.1 Calculating Change in C_i

We calculated fragmentation metrics for a standardized area or "window" surrounding each fire to evaluate the connectivity of the landscape. The size of the window was selected to be 10,000 hectares and was chosen to be smaller than the maximum patch size (contiguous area of native vegetation) and about twice as large as the largest fire size, in keeping with best practices for measuring effects in patchy data (Fortin and Dale 2005).

In practice, no single metric can fully capture the aspects of risk to neighboring sites that we aimed to measure. We explored using several metrics ranging from the simple percent of area in uninvaded cover, to more complex metrics of patch shape and landscape fragmentation such as average perimeter to area ratio, largest patch index (LPI), and landscape characteristics such as contagion. Contagion is one of a class of metrics that measures interspersion and juxtaposition of land cover types. All metrics were calculated using FRAGSTATS software (McGarigal et al. 2002).

Our final choice for C_i was based on finding a metric that responded predictably to fire size and that complemented other metrics used in benefit calculations. After evaluating metric results, we chose largest patch index (LPI) to scale neighborhood benefits. This metric had the advantage of an intuitive definition: patch index is the ratio of the area of largest patch to total landscape area, and thus is a measure of dominance of the native vegetation in the window surrounding each burned area. Also, this metric demonstrated a clear positive relationship with fire size (Figure 5) preventing this scaling variable from competing with fire size for effect on the landscape.

Although we avoided using a landscape metric that would compete with fire size, it could be said that we are skewing the importance of fire size by using two scaling measures that reflect size. However, the increase in LPI with increase in fire size is modest for all but the differences between the largest and smallest fires, so this was not a major concern. Alternatively, other metrics, such as contagion, can represent effects on the landscape connectivity but have less of a relationship to fire size. However, it is important to understand what the pattern metric is measuring to ensure the metric reflects the effect of interest. Many metrics are highly correlated and can substitute readily (Riitters et al. 1995) however, some can behave irregularly with environmental change, making interpretation difficult.

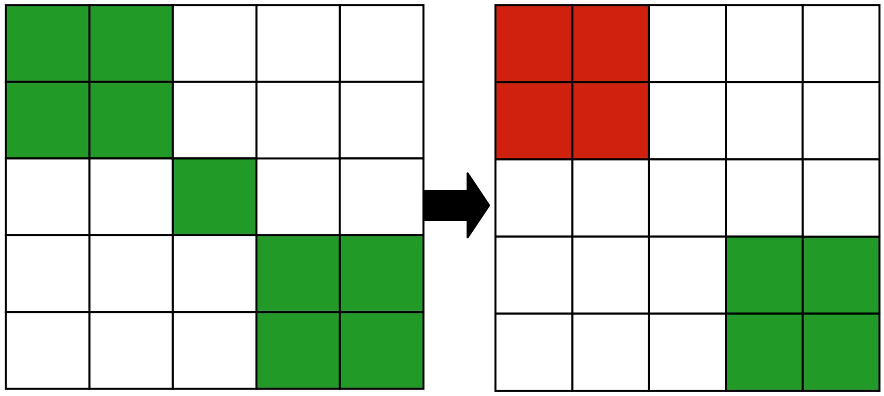

Figure 4a. Defining patches used in spatial pattern metrics
The two boxes represent two schematics of vegetation maps with the pixels outlined. Native vegetation is shown as colored pixels and invasive vegetation as white pixels. All pixels that share a side or a corner are considered part of the same patch which is indicated by a common color in the map representation of native vegetation. When a pixel or a patch (group of pixels) changes from native to invasive plant cover, it can disconnect two patches that were previously part of the same patch. A fragmentation of the single native patch in the left side box occurs when the center pixel transitions to non-native vegetation (shown as a change from green to white). The two native vegetation patches in the right side box are now represented by two different colors indicating the two patches are now unconnected.

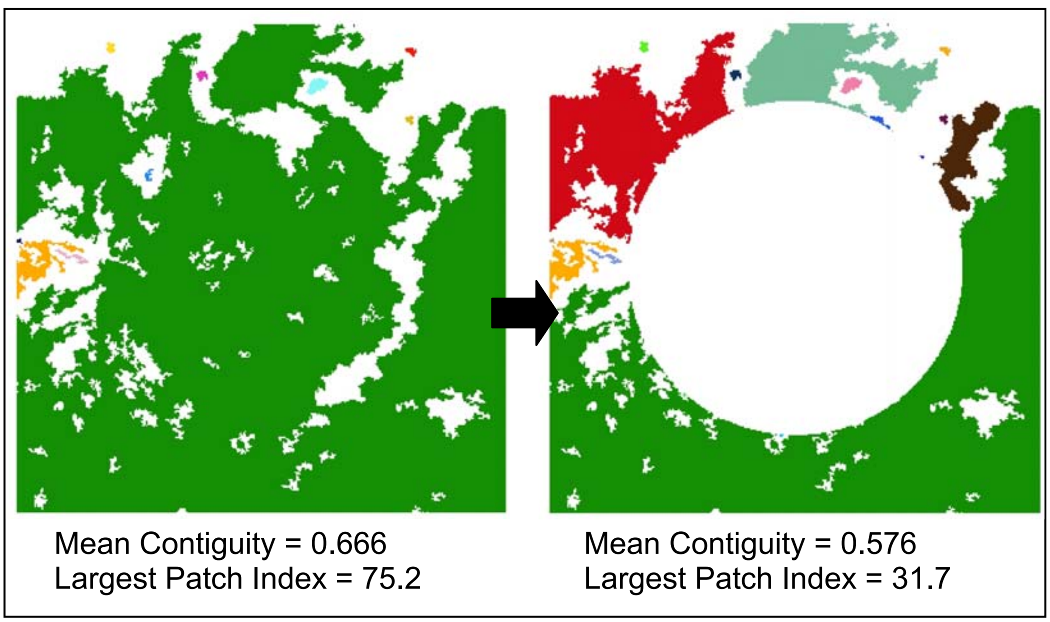

Figure 4b. Effect of Fire on Landscape Connectivity
As with the top figure, these two maps of vegetation show native vegetation as colored areas and white areas as non-native invasive vegetation. Like colors indicate pixels are part of a connected patch of native vegetations, such as the large green patch that covers most of the left-hand box. The effect of the burn (white circle) is to create new unconnected areas of native vegetation shown as a change in color from green between the left and right boxes. This change indicates that, in the right box, an organism traveling in the green patch would no longer be able to reach the red patch without traversing at least one cell dominated by the invasive species. Mean contiguity declines between the left and right boxes (with the fire) indicating the reduced occurrence of adjacent cells containing native vegetation. Largest patch index, the ratio of the largest patch area to total area, also declines showing lower dominance of native vegetation in the landscape.

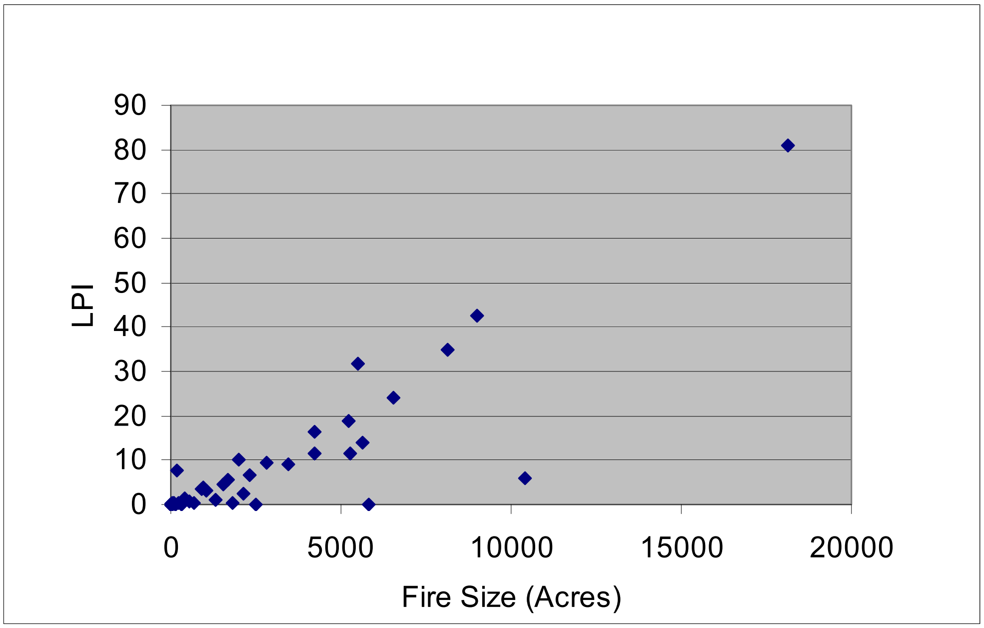

Figure 5. Change in Largest Patch Index (LPI) with Fire Size for 46 Burned Areas in Idaho and Nevada
The change in largest patch index represents the change in dominance of native vegetation after fire, assuming vegetation becomes dominated by cheatgrass. The change in LPI is correlated with fire size, one consideration in choosing this metric to represent fragmentation due to fire.

3.2.4 Explicitly Measuring Invasion Risk to Neighbors

Despite the inexact nature of C_i as an indicator of risk of loss of services from neighboring parcels, we considered it preferable to other available measures of invasion risk. It was beyond the scope of this project to develop our own model of invasion risk and thus we explored available models. While many models of invasion risk have been developed (Higgins et al. 1996), some for this specific region (M. Wisdom, pers. comm.), we did not find a model suitable for our purposes. In cases where managers have a more direct measure of risk of harm to neighboring areas, such a metric could be substituted.

Our experience in searching for an ecological risk model demonstrates a typical problem of applying existing ecological research to economic analyses. We found a promising invasion risk model that had been developed for the Columbia River Basin (discussed in Wisdom et al. 2003) in which invasion risk by location was modeled as a function of several spatial variables. However, the model had some limitations for our purposes. Our initial concern was that risk of invasion was modeled exclusively as a function of biophysical variables and did not include any factors related to human activities. The second, and more important limitation, was that the model was developed and tested for a particular set of land use conditions and could not reflect the dynamics of seasonal fires. This was not a failing on the part of the researchers, rather it reflected a common impediment to successful risk analysis. The ideal of dynamic models using up-to-date data is extraordinarily difficult to realize with current data collection and spatial modeling methods. Improvements can only be made if data are collected more consistently in time and space.

3.3 Measuring b_{max} - Maximum Potential Benefits of a Site

The core of the benefit evaluation system was a framework of indicators constructed based on the economic concepts underlying production functions and location theory. (For further explanation see Wainger et al. 2001, Boyd and Wainger 2002, Boyd and Wainger 2003). The indicators captured elements of the site and location that allowed a service to be produced (using a conceptual production function), and also reflected relative quality and location preferences of users. For example, a site may have the potential to serve as a recreational site, but if there is no access, the potential service cannot be realized. The indicators were used to measure the quality of the site for the intended use and evaluate whether complementary qualities were present to allow the service to be produced. Additional indicators were used to assess potential supply and demand of services produced at sites.

The framework used to evaluate b_{max} incorporated a hierarchical set of indicators covering five aspects of value:

- Site quality
- Landscape quality
- Site opportunity
- Scarcity / substitutability
- Risk of service disruption (service reliability)

Site and landscape conditions were the primary factors responsible for basic service quality, but additional aspects of location (site opportunity) were envisioned to consider the complementary nature of adjacent lands (e.g., presence of campsites for recreation) and to weight the potential value of the services by proximity to users.

Scarcity or substitutability of the service was another economic concept explored through the indicators. We assumed that, all else equal, greater scarcity and lack of substitutes increased the value of an ecosystem service. However, scarcity can only be effectively evaluated relative to demand, creating methodological challenges that we were not completely able to overcome due to constraints on data and analysis time. However, scarcity of services drove our selection of the services we evaluated.

The risk of service disruption was assessed through indicators that examined risk of land conversion. Risk of land conversion was assumed to decrease relative service value since most benefits were derived from having the land in natural vegetation. The exception was the case of property protection benefits, which increased with the risk of land conversion to housing or business uses.

Most valuation exercises incorporate the effect of income on willingness-to-pay for services. In this analysis, we ignored income effects because we were considering publicly owned land and taking a public land manager's perspective rather than a business or household perspective. Government decision-making does not typically embrace using income-related factors in resource allocation.

Indicator choices for each service were devised based on a conceptual model of factors driving the quality of and demand for service at a particular location. As a result, indicators selected to measure these qualities and the scale of analysis per service were usually distinct for each service. Indicators of service quality and accessibility, were developed to reflect preferences of stakeholders and were developed using existing economic theory and valuation studies. In addition, interviews with resource managers were used to develop other aspects of the production functions such as factors limiting service production and identifying the scarcest resources.

3.3.1 From Conceptual Model to Indicators: Recreational Hunting Example

Using the ecosystem service of recreational hunting support, we can illustrate in some detail how the indicators of site benefits were constructed by using literature sources to create conceptual models, and then how components of those models were used to create indicators of relative benefits. Our goal with both the conceptual model and the indicators chosen was to capture the potential change in recreational benefits to hunters with a change in site conditions. Within the environmental economics literature, changes in benefits are generally modeled as changes to consumer surplus associated with engaging in a particular activity before and after some environmental change (see Rosenberger and Loomis 2001 for review). Such models effectively generate a *price* that recreators are willing to pay to engage in an activity before and after an environmental change. Other models are used to evaluate the *quantity* of recreation demanded as a function of site characteristics. Often these two types of models are estimated within a single modeling framework. We explored both types of models to inform our conceptual model of site potential for providing recreational benefits.

Our first goal in model development was to identify the pathway through which the invasion might change consumer surplus of hunters. In southern Idaho, cheatgrass impacts recreational hunting for pronghorn antelope by potentially 1) reducing the number of locations available for hunting, 2) reducing the size of the herd and 3) reducing the quality of the hunting experience. These effects can potentially increase costs to hunters through increased travel or other costs. Hunters might have to drive farther to avoid cheatgrass-infested sites, buy special gear to avoid problems with seeds getting embedded in socks and shoes, or incur costs from other adaptations. Or the presence of invasives might lower the perceived value or willingness to pay for a site, if the quality of that site decreases due to reduced bag rates and reduced opportunities to see wildlife. Extensive survey data has shown that hunters often care more about seeing game and the beauty of an area than killing game (Duda et al. 1995).

Given these avenues for harm, models that accounted for effects on benefits from changes in travel costs, gear costs, bag rates, and aesthetics or ancillary wildlife viewing opportunities were of the most interest. We found a variety of relevant recreational hunting models or studies and supplemented those with additional outdoor recreational benefits models. Most importantly, the models had to provide us with a means to evaluate hunting participation rates or perceived quality based on site-specific characteristics that could be captured using available spatial data or that could be derived from spatial data through analysis.

Recreational demand in a region is a function both of the number of people participating and the intensity of their participation. Traditionally, recreational hunting participation rates have been well modeled as a function of demographic characteristics (e.g., income, age, gender, education level) and intensity of participation (total trips taken by participants) has been modeled as a function of costs, site quality, location characteristics and income or other characteristics (Phaneuf and Smith 2005). Models focused on assessing changes in site quality or changes in total economic benefits typically employ survey data showing how far people travel to participate, what substitute sites they considered, what they purchased, and the relative satisfaction derived from their visit (Bockstael et al. 1987, Bockstael et al. 1989, Loomis 1988). These multi-site travel cost models are used to compare how much people valued a visit with how much they spent to participate. This information is then used to examine the values people hold for an environmental change in site condition.

In addition to models developed from site-specific surveys, large survey databases are available to characterize recreation demand by region and activity. National or regional surveys of recreation participation can provide comprehensive information on participation rates, characteristics of participants, average trip distance and other information [National Survey on Recreation and the Environment (NSRE) 2000-2002, Cordell et al. 2005, National Survey of Fishing, Hunting and Wildlife-Associated Recreation, (USFWS 2001)]. However, site-specific surveys that contain more detailed information than national surveys are generally needed to develop site-specific benefit models. Although we obtained data on hunting area (large regions), we found the data inadequate for differentiating site quality and instead used costs of access and site quality to improve the spatial resolution of site demand characterization and to better understand the potential for sites to act as substitutes.

Visitation databases or hunting permit records can also provide useful information about popularity of sites. High visitation indicates sites are popular, but not whether it is because of high quality, low-cost or both. Without further information, this is insufficient to understand how users will make substitutions between sites.

Time cost of travel has been a component common to spatial allocation models of recreation trips and associated benefits of visitation, because of the information it conveys about level of demand and willingness to pay. It is understood that the more costly it is to access a recreation site, the lower the visitation, all else equal (e.g., McConnell and Strand 1981). However, if site quality is sufficient to offset higher travel costs, then sites with different costs may still produce the same level of consumer surplus. Since travel cost is also often the major site-specific expenditure (on a per trip basis) associated with outdoor activities such as hunting, travel time, by extension, it is a good measure of relative cost among sites. Travel time is the main location-specific variable responsible for travel costs since travel cost is typically modeled as proportional to time spent in travel.

3.3.1.1 Models of Trip Allocation

Even with a good site-specific measure of cost of access, the aggregate behavior of how recreators distribute themselves among multiple recreational sites is difficult to capture in a simple model, and many alternative modeling strategies have been proposed. Difficulties arise from several factors including the complex spatial relationships between options such as substitution and complementarity effects. Substitution between sites is not well understood, but researchers have documented that availability of substitute sites in proximity to one another can, in some cases, reduce the number of trips to a given site, or, in other cases, increase trips taken to both sites. The latter, known as agglomeration effects, results from sites acting as complements (Kim and Fesenmaier 1990).

Despite these complicating factors, researchers have attempted to model the spatial distribution, or allocation of recreational trips among sites, according to the variables that explain most of the response. One approach has been to use gravity models to demonstrate spatial relationships between population centers and alternative recreation sites (Freund and Wilson 1974, Sutherland 1982, McCollum et al. 1990, Isard 1972). These models are largely efforts to match observed patterns of aggregate trip distribution rather than attempts to model individual behavior. These models capture the effect of costs of access on site choice since they are based on the relative "pull" of sites given their distance from population centers of various sizes.

Early gravity models were criticized for not controlling for site quality and generally failing to capture economic behavior of participants. However later models have attempted to address some of these concerns. McCollum, Peterson et al. (1990) applied a reverse gravity model in which they examined the number of trips to site j, as the joint probability of trips from potential origin sites $\boldsymbol{i}$ using data from a large recreational survey. A second linked model was used to estimate total trips generated based on user costs, site characteristics and market area characteristics.

An allocation model, such as a typical gravity model, estimates how trips will be distributed among sites, given a fixed number of trips. However, total number of trips must also be evaluated by examining participation rates for given recreational activities. A major problem with all spatial allocation models is that they require observations of origin and destination of trips, which is typically collected through time-consuming site-based surveys. Such information is needed to parameterize the gravity model or similar pattern-matching models.

In our case study area, numerous alternative hunting sites are available since much of the publicly-owned land is open to hunting. Therefore, we sought a method that could be used to easily assess relative potential recreational demand at any particular site since scarcity did not seem to be an issue. We sought to develop a trip allocation model based on information known about average recreator behavior.

If aggregate recreation demand at a site is typically modeled as:

$$E(x) = f(p, \mathbf{Q}, y) \qquad \textit{(Eqn 6)}$$

where $E(x)$is expected number of trips to a recreation site and:

p	=	travel costs
$\boldsymbol{Q}$	=	site-specific characteristics
$\boldsymbol{y}$	=	demographics of source population (income, education)

Then, we applied this model and our indicator framework described earlier, to generate a conceptual model of relative site demand for recreation that uses available data:

$$E(x) = f(p, q, r, s, u) \qquad \textit{(Eqn 7)}$$

where:

p	=	indicator of site opportunity, population-weighted travel time (number of people living within 1.5 hours travel time from site)
q	=	indicator/index of site quality for hunting (quality for hunting and simultaneous activities such as bird viewing)
r	=	indicator of availability of complementary features for hunting (e.g., availability of campsites)
s	=	indicator of substitutability (presence of alternative sites of comparable quality within same travel distance)
u	=	risk of site becoming undesirable for hunting within the next 30 years (e.g., due to residential development)

In our case, p represents the number of people with access to the site given that the majority of recreational hunting trips in this region occur within a 1.5 hour drive (based on travel distances documented in English et al. 1993). Travel time is closely related to travel costs but does not consider the cost of operating a vehicle or the opportunity cost of a recreator's time. The population living within the 1.5-hour travel time (p) corresponded to the "site opportunity" class of indicator used in our indicator framework. Other variables represent site quality. We did not try to measure s since substitutes of equal quality appeared numerous, at least given our ability to distinguish site quality.

We did not include information on demographics of the source population, even though these data would be available and such variables are standard for statistical models of participation (Bowker et al. 2006). Instead, because we did not expect major differences in demographic makeup across these areas, we simplified our framework by assuming that relative participation rates were constant across the case study area. In other words, if one in ten people engage in hunting in south Idaho in a typical year, we assumed that rate held for all populations equally and, therefore, reducing p by 90% everywhere would not change our measure of the relative demand for hunting at a given site.

We present this model to show what drove our selection of indicators and to assist others who seek to find practical methods to measure benefits derived from natural systems. We note that despite our use of the expectation operator $E(x)$ in equation 7, we did not fit this model in the same way that researchers with survey data fit a probabilistic model. Rather we used the elements of this model to create indicators that reflected the relative number of expected number of trips and compared those values across sites. With our model, we were able to rate potential areas for their ability to be both high-quality and low-cost sites, which would be sites that would tend to generate the highest consumer surplus, although we did not attempt to measure this directly. Our measures for site quality, landscape quality, site opportunity, scarcity and risk of service disruptions were captured through separate indicators and combined with indicators of potential demand to compare relative desirability of sites for hunting. Specific metrics used to evaluate these indicators and details of how they were combined into a single metric are described in the next section.

3.4 Metrics Used to Measure b_{max} for all Services

Similar to the recreational hunting example, conceptual models were developed for all four ecosystem services we considered and indicators were selected to measure relative demand for a service at a site. Indicators were selected for the characteristics of service quality previously described: site quality, landscape quality, site opportunity, scarcity and risk. The indicators selected are shown in Table 3 along with a positive or negative sign indicating whether they were deemed to be positively or negatively correlated with benefits.

Table 3. Indicators Identified to Assess Relative Economic Benefits of Four Ecosystem Services

Benefit	Site Quality	Landscape Quality	Site Opportunity	Scarcity	Risk
Recreational Hunting (pronghorn antelope)	Sage-grouse habitat quality (high, medium, low) (+) (proxy for % cover of native vegetation, etc.)	Camping availability (developed or back-country campsites) (+)	Population within 1.5 hours (+)	Alternative sites of same quality within same travel distance (-)	Risk of urban development within 3-mile radius (-)
Forage Production	Animal Unit Months (AUMs)/hectare (+)	Distance to markets (-) Availability of water sources (+)			Risk of urban development within 3-mile radius (-)
Existence Values for characteristic species, sage-grouse	Sage-grouse habitat quality (high, medium, low) (+)	Local level of fragmentation (spatial pattern metric) (-)	Regional connection to habitat patches (spatial pattern metric) (+)	Probability of population falling below minimum viable population size (locally or regionally) (-)	Risk of urban development within 3-mile radius (-)
Property Protection	Population within 3 mile radius (+)	Within or adjacent to major urban area (+)			Risk of urban development within 3-mile radius (+)

Sign following metric indicates whether correlation with benefits is positive or negative. Grey text indicates indicator was not used due to lack of data or because of correlation with another variable. See text for further explanation.

3.4.1 Site Quality

For each service, an indicator of site quality was developed to reflect the quality of the service derived from on-site characteristics such as level of natural function.

3.4.1.1 Existence Values for Sage-Grouse

A composite measure of quality of habitat for Greater sage-grouse (*Centrocercus urophasianus*) had previously been developed by the Bureau of Land Management and was adapted for our purposes. BLM, using expert judgment, divided sage-grouse habitat into four classes that identified habitat importance and restoration potential. We eliminated the class that showed potential for habitat restoration because it had limited extent and the parameters for defining restoration potential were not available. We ordered the remaining classes on habitat quality into high (best in original data), medium (good in original data) and low (moderate in original data) quality (Figure 6). Areas outside these defined areas were not considered to be habitat. Burned areas were evaluated by examining the site quality class at the location of the fire.

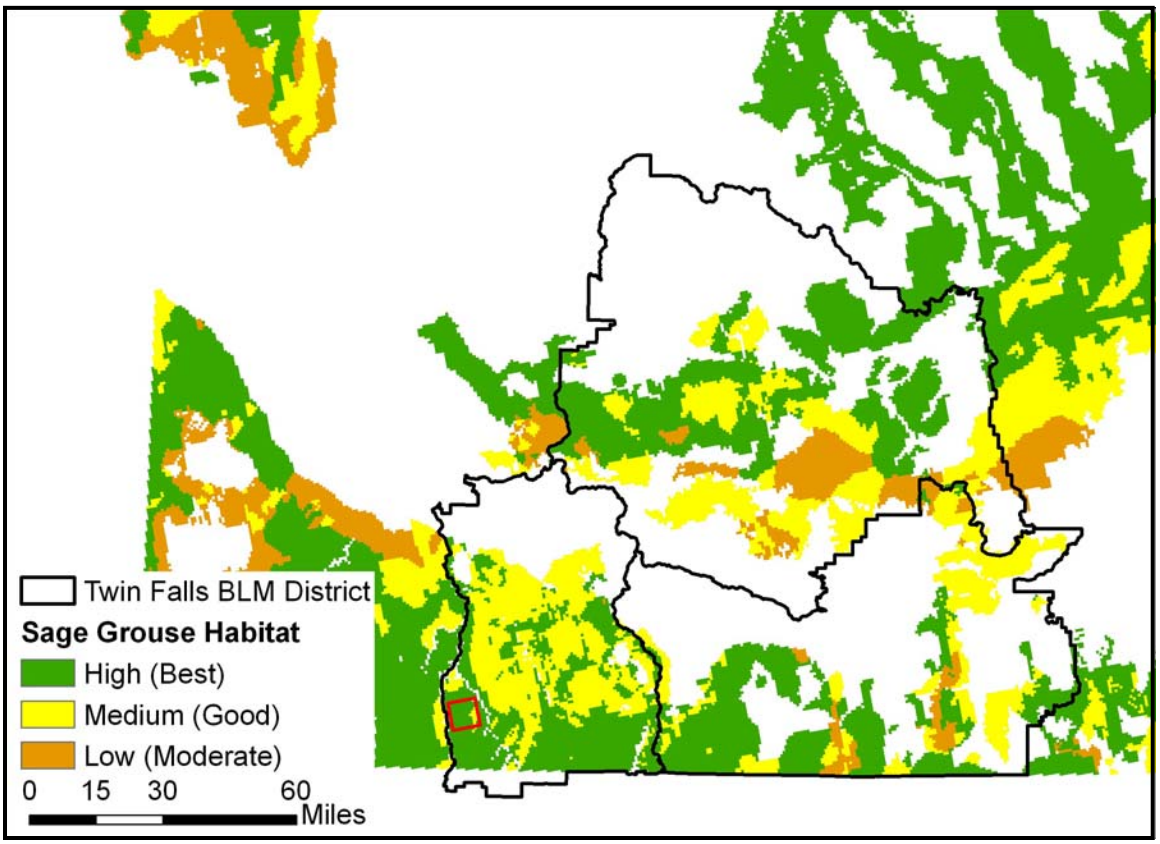

Figure 6. Habitat Quality for Sage-Grouse
Source data from ICBEMP 2004.

3.4.1.2 Hunting Quality for Pronghorn Antelope

Site quality for pronghorn antelope (*Antilocapra americana)* hunting was judged based on the suitability of the site for another indicator species, sage-grouse. Since both species are dependent on sagebrush, and because hunters prefer areas where they would have an opportunity to simultaneously engage in viewing native species (Duda et al. 1995), this was the best available marker of quality of hunting experience given that we did not have data on antelope habitat directly. Scores for this indicator were thus the same as scores for the existence values for sage-grouse.

3.4.1.3 Forage Production

Site quality for production of animal forage for ranching was characterized using animal unit months (AUMs) per hectare on grazing allotments (Figure 7). GIS data showing allotment boundaries and accompanying data on characteristics of active grazing allotments were obtained from the Idaho State Offices of the U.S. Bureau of Land Management (BLM). Other site quality factors were considered, such as availability or quality of water supply and suitability for feed crop production, however data could not be found to characterize these factors. This measure of potential yield could readily be turned into a value of production using price of commodities and other variables, however, information on types of animals being raised was not available by allotment within the database, although such data may have been available elsewhere. Also, if we assumed the most common type of cattle was raised at all sites, the values would not vary over the study region, and therefore would not change our measure of relative value among sites.

3.4.1.4 Property Protection

For property protection, the indicator of site quality was the number of people living within a 3-mile radius of the burned area. The assumption was that the greater the number of people within this area, the more structures at risk and the more people would potentially benefit by treatments to reduce fire risk. Data on property value or presence of vulnerable infrastructure could be used to enhance this metric, but data were not readily available for the entire region. Values for nearby residential population were normalized to a 0-100 scale with 100 being the highest number of people in the vicinity.

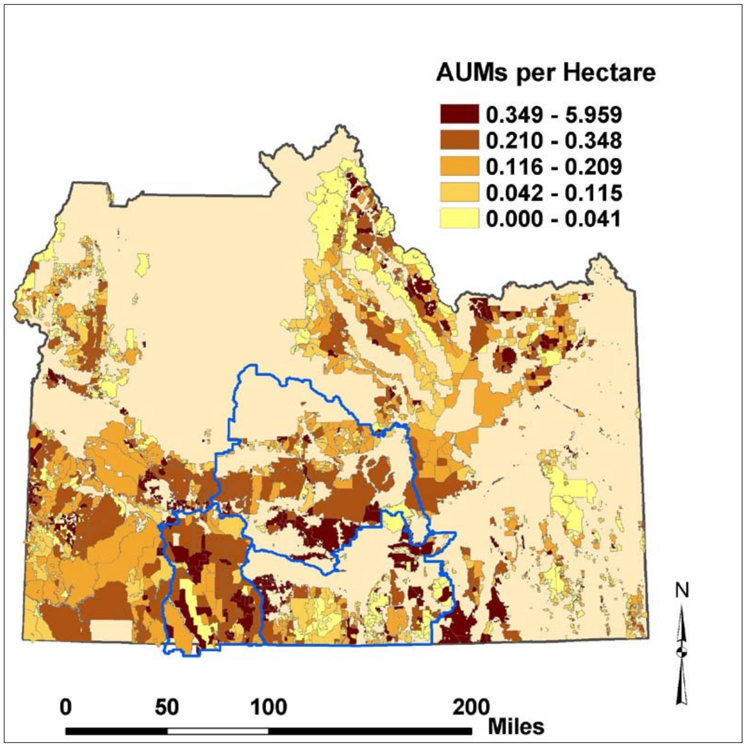

Figure 7. Map of Forage Production Quality Indicator: Animal Unit Months per Hectare
Source data courtesy of BLM.

3.4.2 Landscape Quality

Landscape quality captures elements of landscape configuration that complement site quality in order to enhance the overall service quality. These indicators describe conditions of the surrounding landscape that allow sites to produce quality services if on-site conditions are also good. For example, a site might have high quality native vegetation but be isolated from surrounding habitat by urban development. Such site context would prevent the site from realizing its potential for habitat for a sensitive species. Landscape quality indicators were developed for the services of recreational antelope hunting and existence values for sage-grouse. Landscape indicators were combined with the site indicators for these two categories to create an overall quality metric for each service. Scores for all services were adjusted so that the total number of indicators used did not affect total scores.

3.4.2.1 Recreational Antelope Hunting

The number of people within 1.5 hours (~75 miles) of the site was the landscape quality indicator developed for recreational antelope hunting. The distance that recreators travel was constrained by using the upper bound discussed in English et al. (1993, Table 11) that 75% of hunting trips in the Rocky Mountain region, which contains Idaho, took place within 75 miles of the originating site. This estimate was somewhat lower than some other travel distances that had been estimated by other researchers. For fishing, one study suggested sites within a 4.5 hour drive are reasonable substitutes (Whitehead and Haab 2000, for the southeast US). At 40 mph average trip speed, 4.5 hours translates to approximately 180 miles. Other studies used 100 miles from home as the relevant trip distance (Feather and Hellerstein 1997), although they acknowledged that different activities have different scales.

3.4.2.2 Existence Values for Characteristic Species

Sage-grouse are dependent upon contiguous tracts of sagebrush for habitat and recent studies suggest this species is particularly sensitive to human disturbance and avoids developed areas and roads (Naugle et al. 2006). Therefore, to develop a landscape quality indicator that reflected this component of their habitat requirement, we considered several landscape pattern metrics designed to examine habitat fragmentation. In contrast to the spatial pattern metric used to evaluate the effect of restoration on neighboring pixels (Section 3.4.2), this metric was developed as a specific landscape habitat quality measure affecting sage-grouse. These spatial habitat metrics are commonly used to examine whether patches of potential habitat form migration corridors or provide sufficient undisturbed area to provide high quality habitat, given the specific life cycle requirements of a species.

Because sage-grouse avoid sagebrush adjacent to disturbed areas, the total core area metric, calculated for the landscape scale, was chosen to reflect landscape habitat quality. Core area is a metric that refers to the area within a patch that is greater than a specified distance from the patch perimeter. The metric was calculated for a window around each fire site pre- and post-fire (assuming full conversion to cheatgrass) to examine the potential maximum effect of fire on total core area available to sage grouse.

This landscape quality indicator was calculated using several GIS processes, which were automated in Python and the core area metric was calculated using FRAGSTATS (McGarigal et al. 2002). Because the sage-grouse responds to fragmentation over short distances, we sought fine resolution data to evaluate this aspect of fragmentation. Relatively fine scale (30 meter cell size) vegetation data with detailed vegetation categories were obtained from the USGS GAP analysis for Idaho (Scott et al. 2002). All vegetation types containing sagebrush were considered as potential sage-grouse habitat. A 10,000m by 10,000m (10,000 ha) window surrounding each fire was evaluated for each site and a buffer of 30 meters (one cell) was used around patch edges to define non-core area. The metric is returned as a value between 0 and 100 that measures the percentage of the landscape in core area.

To develop a single metric of habitat quality for sage-grouse that incorporated site and landscape characteristics, we combined the habitat site quality developed by BLM with the landscape quality indicator of % of habitat core area surrounding each site. The core area indicator (scored between 0 and 100) was multiplied by a site quality factor based on the habitat quality (high, medium or low) developed by managers. The highest quality site value was assigned a scaling factor of 1, medium quality used a scaling factor of 0.75, and low quality a factor of 0.5. Therefore, the highest indicator values occurred where the landscape in the vicinity of the site showed the highest percentage of core area and where managers had assigned high potential habitat value based on multiple site and landscape characteristics.

It could be said that the site quality developed by managers would reasonably include fragmentation, given the sensitivity of the species to this aspect of habitat. The experts who drew the general boundaries of best, good, and moderate quality habitat sites undoubtedly considered this component of the landscape. However, the generalized boundaries they created were not intended for comparing sites at a fine scale. Therefore, somewhat counter-intuitively, our landscape scale indicator provides more detailed site-specific information to judge relative habitat quality than our site quality indicator.

Site-specific quality indicators can be among the most difficult to find in a GIS format when conducting regional screening. Typically, coarse scale data are used to suggest site quality when conducting regional screening. However, such data cannot capture site-specific conditions that can limit site quality such as fences or concentrated disturbances such as toxic waste sites or legacies from historic land use. Nonetheless, it is widely recognized that landscape context can serve as a strong indicator of site quality (Brooks et al. 2006), especially in the absence of specific limitations that are undetectable at coarse scales.

3.4.3 Site Opportunity

3.4.3.1 Recreational Antelope Hunting

As previously discussed, we evaluated the portion of the population living within a 1.5 hour drive of sites as the site opportunity indicator for hunting. We initially evaluated two techniques to estimate the population able to reach potential recreation destinations. In one technique a distance-decay function was estimated using a kernel density estimator. This function was used to probabilistically distribute population from multiple population centers assuming equal travel speed in all directions in the landscape. In the other technique, a travel-time model was developed using the road network and applicable road speeds. Both techniques are available within the GIS software package, and are readily available for managers to use. Each offers advantages in terms of evaluating potential patterns of travel, however the travel time model was eventually used in comparing benefits because of the greater realism in estimating travel times and thus costs. The major limitation to this technique is that it cannot easily distribute population from multiple population centers to a destination site. We describe the techniques for both methods in the interests of informing choices by managers.

Estimating Relative Accessibility Using Kernel Density Estimation

The purpose of kernel density estimation is to interpolate an underlying density curve given a set of observed data. In constructing this estimate, a curve is fit to each observed data point. The function of the fitted curve, which must integrate to 1, is called the kernel function. The density estimate is not overly sensitive to the kernel function and the most commonly used kernel function is the normal or Gaussian function due to its mathematical simplicity. The sum of these fitted curves produces the kernel density estimate and yields the following equation:

$$f(x) = \frac{1}{nh}\sum_{i=1}^{n} w\left(\frac{x - x_i}{h}\right) \qquad \textit{(Eqn 8)}$$

where x_i is the observed data point (for i = 1,2,…,n), *w(.)* is the kernel function, *h* is an arbitrary parameter known as the bandwidth, and the term (1/nh) is a normalizing constant to make certain the probability function integrates to 1.

The process of this estimation has been automated through a built-in function in ArcGIS. This function fits a smooth surface to each observed data point where the surface value is greatest at a peak over the data point and decreases with distance from the observed point (Figure 8a). The kernel function used to fit the surfaces to multiple points on the landscape is a quadratic function commonly used in three-dimensional density estimation. A radius is then specified to determine the area over which the density will be estimated. Since each data point has a unique surface and these surfaces overlap, the values of the surfaces within a certain raster cell must be summed, and the sum is the density estimation at that raster cell.

Using this kernel density estimator in ArcGIS, density of trip potential was predicted (as seen in Figure 8a). For each raster cell in this figure, the result is an estimation of the number of people who would likely travel to this area. For this estimation, the observed data used was population, and the radius of maximum trip length was 75 miles (English et al. 1993, Table 11). When combined with other indicators of service quality (Figure 8b), such trip density measures can be used to examine substitutable areas (Figure 8c) and areas offering the highest probability of providing continued future service (Figure 8d).

The kernel density estimator allows for a density distribution of trips, with more trips occurring close to population centers and fewer occurring far from population centers. However, the assumption of uniform travel costs in all directions is unrealistic except for limited circumstances.

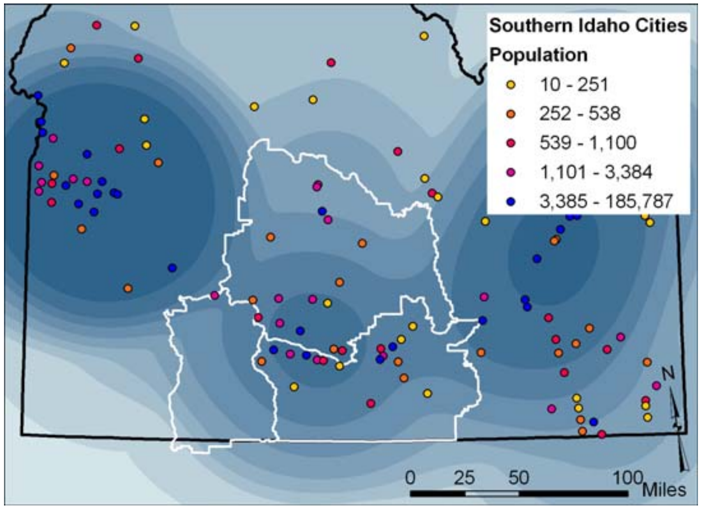

Figure 8a. Potential Trip Density
Map showing result of fitting a probability density (distance-decay) function to population centers.

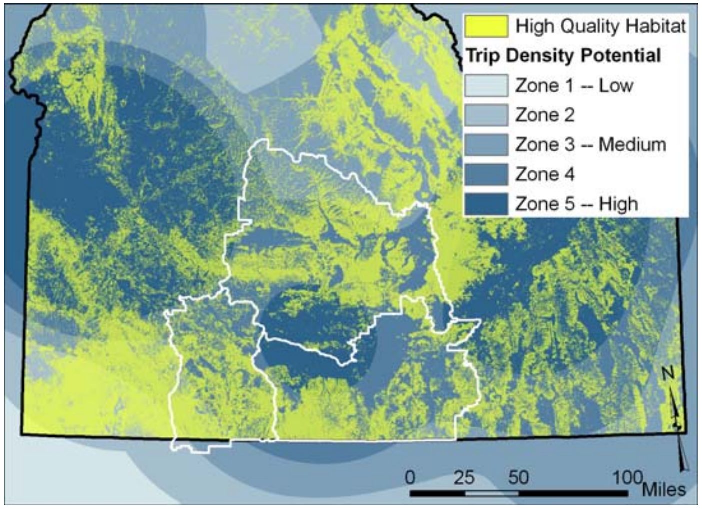

Figure 8b. Habitat Quality over Trip Density Zones
Map of sagebrush communities (proxy for habitat quality) over trip density zones from Figure 8a.

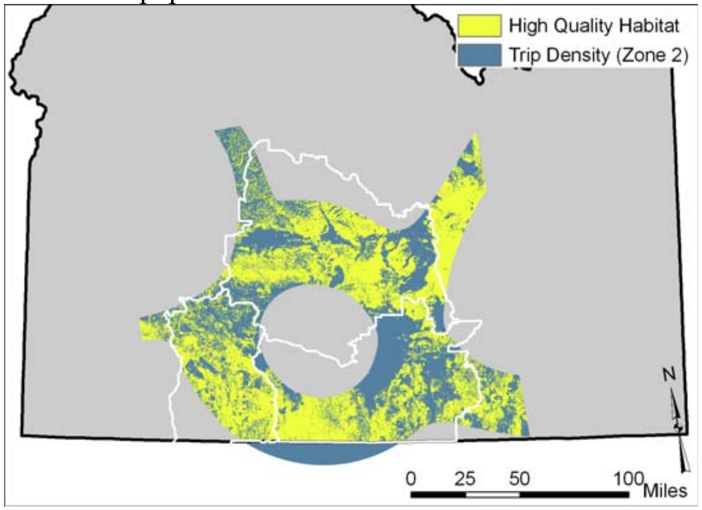

Figure 8c. Substitutable Sites
Zone of potential substitutes is developed using sites within the highest trip density zone that co-occur with areas of high quality habitat.

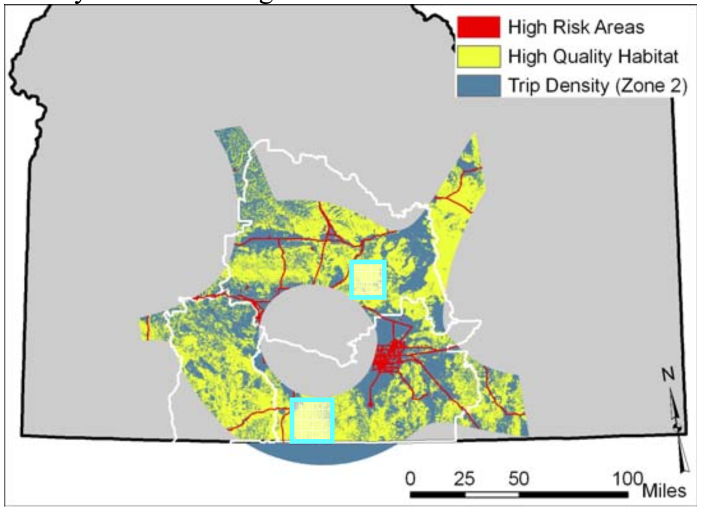

Figure 8d. Lowest-Risk - Highest Benefit Sites
Map shows roads (in red), a factor in risk of land conversion, overlain on Figure 8c. Blue boxes outline sites with highest (relative) probability of providing service streams in the future. Sites selected were those far from roads and within unfragmented sagebrush indicating highest quality habitat, assuming all else equal.

Travel Cost Model of Recreation Accessibility

The travel cost model uses the road network to more realistically evaluate site accessibility, although it does not provide the ability to distribute population according to a density distribution. The 75-mile travel radius used in the kernel density function was translated into 1.5 hours travel time and it was assumed that the number of recreational trips would be a constant proportion of available population. All of the fire sites were on public lands and were therefore accessible for hunting.

The number of people within 1.5 hours of the site was determined for each fire site through a series of GIS analyses (Figure 9). First, a "cost surface" map was created for the region using a combination of road and slope data (see Section 4.3.1.2). In this cost surface, each 1,000m by 1,000m raster cell was assigned a value that corresponded to the amount of time it would take to cross that cell. This time was determined by assigning speed limits by road class, or, in areas without roads, assigning a speed based on slope class (e.g., slope <10%, slope 11-30%, slope >31%). A raster file representing population was created using US Census data and assigning population to each grid cell based on the population density of the corresponding census block group. Using a series of GIS analyses (ESRI ArcGIS Desktop v. 8.3 software automated in a Python script) the number of people living within a 1.5 hour radius were summed for each fire. Values were scaled to a 0-100 scale, with 100 being the maximum people observed living within a 1.5 hour radius.

3.4.3.2 Existence Values for Characteristic Species

Conceptually, we identified the need for a variable of site opportunity for sage grouse habitat quality that represented the spatial connection between a population at a particular site and the broad-scale habitat mosaic within the Intermountain West. Such a metric would represent the importance of any given site relative to alternative habitat that the species might use. Such metrics are particularly important for migratory species that need stopover sites to be located within a day's flying distance of one another in order to maintain a migratory pathway. Similarly, a habitat mosaic may provide insurance that if a local population dies out, it remains connected to a source of new recruits. However, in practice, when we identified the particular habitat needs of the sage-grouse we did not feel such a metric could be supported with available data and understanding.

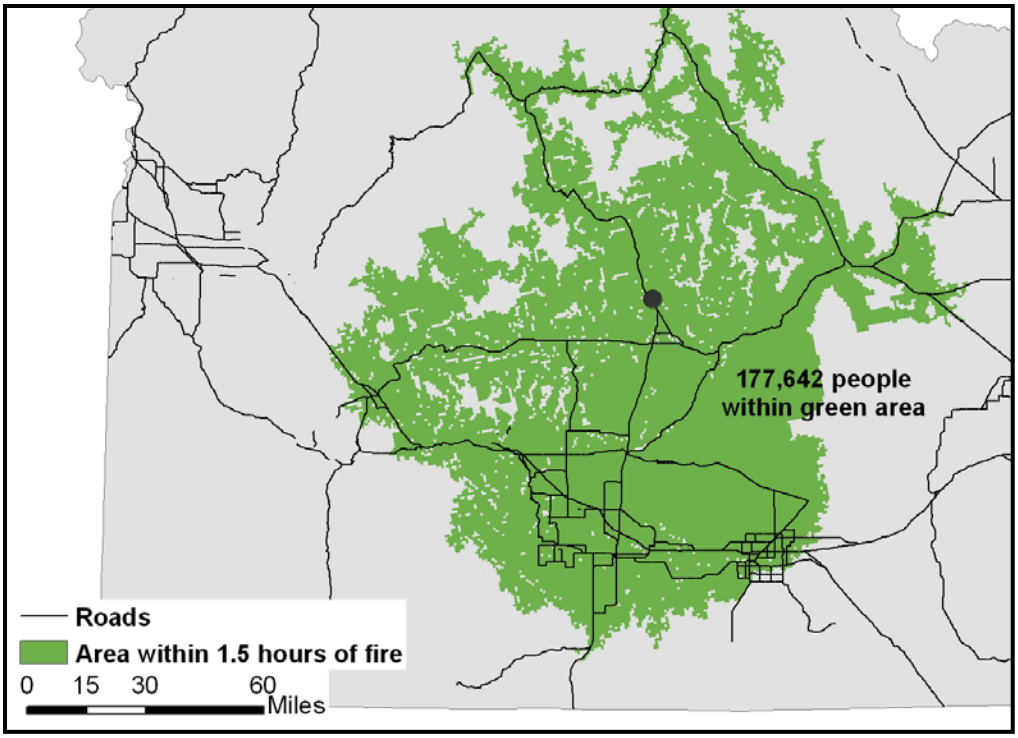

Figure 9. Area Accessible within 1.5 Hours Travel Time of Burned Area
Burned area is represented as a black dot.

3.4.4 Scarcity and Substitutability Indicators

Our main aim with the scarcity metrics was to demonstrate the availability of substitute sites to the user population of any particular service.

3.4.4.1 Recreational Antelope Hunting

We envisioned scarcity indicators for use services such as recreational hunting that could demonstrate the scarcity of sites for particular services and how people would be likely to make trade-offs among sites. For example, we expected to compare the number of equivalent recreational sites available to a given population and rank the scarcity of particular sites in terms of availability. Our decision to use the cost-distance model for assessing travel time for recreation services limited our ability to easily evaluate substitutability of sites among many populations. However, we demonstrated our intended methodology for assessing substitutability with the kernel density estimator (Figure 8c).

In addition to methodological challenges, such scarcity/substitutability indicators turned out to be difficult to capture for these services in this landscape context, primarily because of the abundance of substitutes for most services. The abundance of lands open to hunting in this region dramatically reduced the need for such indicators. However, antelope hunting opportunities appeared to be in decline with advancing cheatgrass. More spatially refined data on favorite hunting locations would be needed to better model how hunters make substitution decisions between sites and therefore, where local scarcity for services might be present.

3.4.4.2 Existence Values for Characteristic Species

Of interest for evaluating the value of sites for providing existence services is whether a species is maintaining a sustainable and resilient population. For the sage-grouse, the population does not appear to be endangered across its entire habitat since legal petitions to list the species as threatened or endangered were rejected in 1995 (USFWS 1995). At that time, the US Fish and Wildlife Service found that a large population remained, although it was in modest decline when evaluated over its entire range. In some areas, the population appeared to be stable or increasing.

For services that depend on maintaining a particular species or ecosystem, managers have several types of analyses they may consider to evaluate if the population is viable over the long term. "Population viability analysis" is an umbrella term for several techniques aimed at determining whether a site is sufficient to maintain a successful breeding population that is also resilient to disease or other perturbation (Coulsen et al. 2001). Such analyses will provide indicators of risk that the population cannot be maintained on a given site.

We explored several indicators for habitat scarcity of sage-grouse and hunted species (antelope and deer) including irreplaceability (Lawler et al. 2003, Ferrier et al. 2000), critical habitat areas (e.g., overwintering habitat), and scarcity of regional habitat. Managers had identified deer overwintering habitat as a resource that was potentially becoming scarce since these low elevation areas are most affected by cheatgrass. However, we did not include this component since our existence service was narrowly defined for sage-grouse. Other metrics were rejected due to lack of appropriate data.

3.4.5 Risk Indicator

The risk indicators were intended to reflect risk of service loss that was beyond the control of the rehabilitation project. Just as people decide not to fix an expensive component of a broken car when too many other things are likely to break soon, managers may decide not to rehabilitate a site that has a high likelihood of losing the ability to produce a service due to human activities (e.g., land conversion) or natural processes (e.g., ongoing plant succession). Managers may also consider the risk that a service will lose value in the future, such as waning popularity of hunting a particular species.

3.4.5.1 Residential Development Risk

For all ecosystem services, we used the same risk metric to capture risk of land conversion to residential use. In southern Idaho, land managers identified the greatest threat to habitat, other than invasive plant species and fire, as being the risk of conversion to residential development. We searched for existing research on residential land conversion risk for the area by calling government land use planning agencies and conducting web searches. We have previously been able to find assessments of future land development risk from land use or infrastructure planning agencies at the local or state level. However, in this case, we did not find any government research, nor did we find any academic models of conversion risk for this region. Digital zoning maps were also not readily available, although we were able to obtain maps of land ownership from the US Forest Service (ICBEMP 2004).

A variety of models are available to estimate risk of land conversion (see reviews by Irwin and Geoghegan 2001, Parker et al. 2003) although most require time-consuming data development and statistical analysis. The extraordinarily high percentage of government-owned land and the rural nature of the case study setting limited the applicability of most land use change models.

An in-depth analysis of conversion risk was beyond the scope of this project, so we developed an indicator to serve as a place-holder for our case study demonstration. We selected three metrics that are common to land use change models: travel distance to a main city in the district, proximity to major roads, and existing low density residential use. Using distance to a central business district is a common determinant of a component of risk of land conversion (e.g., Bockstael 1996, Landis 1995) and existing density and roads are typically used in pattern-matching type models of land conversion (Theobald 2005, Verburg 2006, Parker et al. 2003).

Such pattern-matching approaches use historical land conversions and presence of major predictors such as roads to demonstrate the spatial "contagion" common among subsequent land conversions. We excluded publicly-owned land from consideration thereby eliminating all categories of protected land from possibility of conversion to developed uses. Such an ad-hoc system of indicators can only provide a rough indication of conversion risk given the many factors that affect attractiveness for development. Managers who do not have access to carefully developed predictions of conversion risk may choose to ignore this factor or substitute professional judgment of realtors or land use planning professionals.

To generate the risk indicator for conversion of agricultural or natural land to residential or commercial uses, we developed three metrics for private lands within the Twin Falls BLM district. Government-managed lands were excluded from the analysis of development risk and not given any indicator scores. The first risk metric was travel time to the city of Twin Falls, which was used to reflect access to the principal business district. The areas closest to Twin Falls were given the highest risk classification. Distance to primary roads was the second metric developed. Primary roads included Interstates, US highways and state highways (with or without limited access). We used the log of distance to primary roads to represent the enhanced risk for parcels in close proximity to roads.

The third metric used was risk of development due to proximity to low-density housing. This indicator was measured by examining a map of land use developed to identify wildland urban interface (WUI) areas. The map contains 14 land cover classes describing combinations of level of residential development and vegetative cover, of which we designated three low density housing cover classes as the areas of interest for new development. A 3x3 cell neighborhood around each 30-meter cell was examined, using a moving window within the GIS, for presence of these low density housing land classes. Risk was assigned to a cell in a binary fashion with high risk given to sites that had presence of low density housing and no risk given to sites without low density housing in the window.

The three risk factors were combined by adding the three metrics, whose scores had all been assigned based on a 0-3 scale, with 3 being the highest risk. The final risk raster contained values ranging from 2 to 9, which were then used to adjust other benefit indicators, as described below. Because government lands were excluded, the majority of sites were not ranked for development risk and were effectively given no risk of development.

3.4.6 Combining Indicators into a Single Metric of Benefits

We were able to simplify the difficult issue of how to combine indicators for each service into a single benefit index by limiting the number of indicators used. Indicators were kept to minimum by carefully selecting only the most representative metric for each of the five indicator categories (Table 3). Based on our interpretation of the literature, interviews with managers and best professional judgment, indicators were developed that 1) represented the variable that would explain the majority of the variability among sites and 2) was represented in existing datasets.

Data and analysis complexity concerns also served to limit the number of indicator categories calculated and scarcity indicators were deemed largely unnecessary given the relatively high number of substitute sites for our selected services in the near-term. For the services of Forage Production and Property Protection, only a single quality indicator was developed creating the need to combine two indicators for site quality and risk. The services of Recreational Hunting and Existence Values were evaluated with three indicators (site quality, landscape quality/site opportunity and risk) that needed to be combined.

The technique used to combine indicators was to scale all continuous variables between 0 and 100% and multiply indicator values together. Essentially, we treated indicator scores as probabilities of achieving the highest possible service benefit and combined them as we would any probabilities. This crude method of developing the "production function" for service benefits was a necessary construct to fully demonstrate the analysis, although we do not deny that the objective information available to create such production functions is limited. We used our own best professional judgment based on literature review and interviews with managers, however our decisions were meant to serve as placeholders for a more open process that we envisioned would involve expert elicitation using groups of local experts.

Not all indicator variables were initially measured as continuous variables and therefore additional methods were applied to create comparable scores from the ordinal values used for sage-grouse habitat quality and urban development risk. For the sage-grouse habitat indicator, values between 0 and 100 % were assigned to each of the three levels of habitat quality (high, medium, low) for each of the two services being evaluated, Recreational Antelope Hunting and Existence Values for Sage-Grouse (Table 4). The values were developed based on our team's best professional judgment (developed from a literature review and manager interviews) of how a level of site quality would affect the benefits derived from that service.

When these site factor scores were multiplied by the indicator scores for Landscape Quality or Site Opportunity, they either acted to maintain high scores when the best conditions co-occurred or they became downward adjustments when lower quality conditions occurred on a site. The downward adjustments were more substantial for recreational hunting than for sage grouse habitat because hunters were considered to be more sensitive to degraded habitat because of their ability to move between sites and thereby easily find substitute high quality sites rather than accepting lower quality sites.

Table 4. Site Quality Factors Used when Combining Service Benefit Indicators

	Recreational Hunting	**Existence Values for Sage-Grouse**
Sage-Grouse Habitat Quality	**Continuous Site Quality Score**	**Continuous Site Quality Score**
High	100%	100%
Medium	75%	90%
Low	50%	80%
Not sage-grouse habitat	0%	0%

The Risk of Service Disruption indicator was interpreted as a probability of service persistence into the future. Higher risk of new land conversion was used to reduce potential service benefits for services related to hunting, forage production and existence values. For all services except property protection, benefit indicator scores were decreased by a maximum of 25% (risk indicator value = 75%) for the highest risk category of 9 and were not adjusted for the lowest risk scores of 2 (risk indicator value = 100%). Values in between 2 and 9 were scaled proportionally, so, for example, a risk score of 6 resulted in a benefit score reduction of about 15% and a risk score of 3 resulted in a benefit indicator reduction of about 4%. Risk-adjusted scores are shown in Table 5.

Table 5. Benefit Indicator Scores for Selected Sites

(normalized values of 0-100%)

Site ID	Site Quality				Landscape Quality	Site Opportunity	Risk (Urban Development)	Risk Adjusted Benefits			
	Existence Values	Recreational Hunting	Forage Production	Property Protection	Existence Values (Sage-grouse)	Recreational Hunting	Recreational Hunting	Existence Values (Sage-grouse)	Recreational Hunting	Forage Production	Property Protection*
52	100	100	18.3	1.0	56.1	29.7	100	56.1	29.7	18.3	1.0
53	90	75	23.5	0.5	37.4	82.4	100	33.7	61.8	23.5	0.5
55	0	0	0.00	2.3	3.8	35.8	92.8	0.0	0.0	0.0	2.3
56	100	100	22.9	1.0	64.6	33.8	100	64.6	33.8	22.9	1.0
58	90	75	0.0	0.1	38.3	0.0	100	34.4	0.0	0.0	0.1
59	90	75	34.9	0.2	62.6	37.9	100	56.3	28.4	34.9	0.2
60	80	50	35.9	0.5	68.9	46.6	100	55.1	23.3	35.9	0.6
61	90	75	13.9	0.1	46.9	32.3	100	42.2	24.2	13.9	0.1
63	80	50	36.5	1.5	4.9	30.8	100	4.0	15.4	36.5	1.5
66	90	75	69.2	0.1	40.2	0.0	100	36.2	0.0	69.2	0.1
67	90	75	0.1	0.1	57.5	25.8	100	51.8	19.4	0.1	0.1
69	100	100	35.9	0.4	62.0	38.1	100	62.0	38.1	35.9	0.4
81	100	100	24.2	0.1	60.3	23.7	100	60.3	23.7	24.2	0.1

* Property Protection values were highly skewed so that even though values range from 0-100 over all sites, scores shownare quite low.

3.5 Benefit Measurement Discussion

Our attempts to quantify the net benefits of treatment met with many roadblocks largely due to limitations on the quantitative evidence for cheatgrass impacts on ecosystem service benefits. However, much qualitative and descriptive information was available and we were able to apply data, information and best professional judgment to create a system to capture expected effects of treatment using indicators of ecosystem service benefits. The system we developed relied on applying basic ecological or economic functional relationships, yet, demonstrated the methods one might use to characterize site and location characteristics that contribute to location-specific service value. We showed that aspects of future site benefits could be captured within a static analysis and that appropriate information was available to characterize a wide array of ecosystem services. With the rest of this report, we develop complementary information on costs and probability of effective treatment so that this information on relative benefits can be used in making prioritization decisions using cost-effectiveness analysis or an optimization model that maximizes change in benefits for a budget constraint.

4. Evaluation of Treatment Costs

4.1 Facility Location – Application of Location Theory

For purposes of estimating costs of enhancing ecosystem services at sites, we characterized potential treatment sites as "factories" of ecosystem services and our investment decisions were then analogous to facility-siting problems. For facility-siting problems, locations are chosen that will minimize costs of production and delivery of goods and services (Brandeau and Chiu 1989). Locations may also be chosen to minimize losses from competing producers or to take advantage of agglomeration effects. In our analysis of the costs and outcomes of prevention strategies, we similarly sought to evaluate the costs of production (treatment) and benefits of final products (ecosystem services) at specific locations.

The issue of maximizing returns through facility location choice is a well-studied concept in economics. Alfred Weber (1909) is generally acknowledged as having formally introduced location theory by examining the problem of siting a warehouse to minimize travel distance between the warehouse and customers, although he was not the first to consider spatial pattern of economic activity. Weber's work was expanded upon by many, including groundbreaking work by Walter Isard, who examined facility location, optimal land use, routing problems, and other location-dependent questions (see Brandeau and Chiu 1989 for review and Isard et al. 1972). Advances in computing allowed sophisticated network analysis to be used to determine travel times and find efficient routing solutions (Jensen and Bard 2003). Some recent applications of facility location have included siting bio-energy facilities to minimize costs of inputs (Noon et al. 2001).

To understand the potential net benefits of enhancing ecosystem services by treating invasive species, the costs and benefits specific to that site must be examined. Our location-based approach to estimating costs departs from previous analyses of invasive treatment that assume costs are a constant per acre value (e.g., Denne 1998, Zavaleta 2000). Since site accessibility to labor and equipment affects cost of treating at a particular location, we evaluated the costs by considering treatment "delivery" times to location *i*. Economies of scale were important to capture because it is well-established that treating small areas is generally more costly per acre than treating large areas (Mills and Bratten 1988, Hesseln et al. 2004). However, it has been suggested that treating small outlier populations of invasive plants is the best strategy for long-term control of invasives plants (Moody and Mack 1988). Therefore, it was important to understand the relative costs and risk-adjusted benefits of small vs. large and remote vs. near sites.

We took an approach to estimating costs that considered site and location factors for two reasons. First, the effect of location on treatment cost increases when repeated trips to the site are needed, as is the case for most invasive species treatments. In the case of small treatment areas, travel costs can represent a large fraction of treatment costs (Mills and Bratten 1988). Second, we wanted to be able to compare the marginal costs and benefits of treating different size areas in order to find an efficient size of area to treat in each location. Locally heterogeneous conditions may dramatically affect the relationship between average cost and size of area treated. Therefore, a constant per acre cost may not capture important heterogeneity of costs with location or allow the relationship between size of area treated and net benefits to be explicitly examined.

The need for such site-specific factors in cost estimation can be weighed against the increased time required to make such estimates. In deciding on treatment options, managers must choose both which sites to treat and how intensively to treat each site. Keeping cost estimation systems simple is important because decision-makers typically need to screen many areas to evaluate the best use of resources. Yet, spatial analysis tools and available spatial datasets are making such spatially-detailed estimates increasingly quick and easy. Where good data exist, use of spatial analysis within Geographic Information Systems (GIS) allows a wide variety of location factors to be considered simultaneously, although some factors may still need to be assessed through site visits. By using such tools to create maps of costs and cost factors, managers can readily compare trade-offs between treatment locations.

4.2 Cost Model Framework

To evaluate costs by location, we created a spatial cost model and evaluated data available to parameterize the model. We developed the model with the general case in mind of cost estimation for any type of invasive species treatment decision and eventually parameterized a simplified model based on the specific treatments used for burned area rehabilitation programs in the case study area. The costs of treating a site were conceptualized as a function of fixed and variable costs with variable costs made up of travel time, search time and treatment time. These categories incorporated costs related to equipment, materials, personnel and transportation.

The cost of treating a single cell at location *i* on the landscape with a treatment intensity *k* (PC_{ik}) was characterized in terms of the following variables:

$$TC_{ik} = FC_k + JC_k{*}JT_i + SC_k{*}ST_i + CA_{ik}{*}A_i + DC_k{*}D \qquad (Eqn\ 9)$$

FC_k	=	Fixed Costs for treatment intensity k
JC_k	=	Journey (Travel) Cost, or the hourly cost of transporting all personnel and machinery for treatment k from the origin cell to treatment destination i
JT_i	=	Journey Time (hours) to site i is a function of road networks, streams and slopes
SC_k	=	Search Cost, hourly cost of searching for area to treat for treatment k, which is a function of treatment method
ST_i	=	Search Time (hours) is a function of site characteristics
CA_{ik}	=	Treatment Cost per area, cost of treating location i with treatment k
A_i	=	Area treated (acres)
DC_k	=	Daily Costs, added costs beyond the hourly rate incurred for each overnight stay (e.g., overtime pay, hotel and meal costs, etc.)
D	=	Number of overnight stays
i	=	Site location
k	=	Treatment phase

To simplify calculations of treatments that involving multiple trips, we measured costs of all trips to complete treatment regime *k.* Therefore, journey costs and treatment costs for *k* could reflect multiple trips.

We were able to simplify calculations of fixed costs because much of the treatment work was contracted out by the federal agencies that manage the land. Therefore, costs could reasonably be estimated by focusing on the short-term costs of treatment and assuming that management and facilities costs were fixed. However, we included agency personnel costs to plan, oversee and monitor treatment in this model. Overhead costs were ignored because they were generally a fixed percentage of total costs and therefore did not contribute to heterogeneity of costs among sites.

Because we were primarily interested in the heterogeneity of costs in space, we loaded many of the costs that did not vary greatly by location into the fixed costs. This included the initial scoping and planning, treatment design, and permitting. Certain costs increase for every day in the field and are not a smooth hourly function, for example such as lodging costs, equipment rental, and overtime pay. These costs are handled using the Daily Cost (DC_k) variable that incremented such costs by the day.

Based on previous work (Robichaud et al. 2000, Bohlen and King 1995) and interviews with restoration practitioners, we identified site and landscape factors that we hypothesized would affect heterogeneity of costs across a region. The vector ***s*** of site characteristics that affected search time (*ST*) and treatment cost (CA_{ik}) influenced each treatment type differently. Based on interviews, we determined that site factors that affected search time included the infestation size and density, whether the site had burned, and the distinctiveness between the invasive and native vegetation.

Interviews and a literature review also revealed that the factors most likely to affect treatment cost included soil type, presence of rocks, proximity to urban areas and croplands, protected areas, slopes and aspect (Hesseln et al. 2004). Such factors can change the type of equipment used, recommended levels of herbicides and levels of safety precautions needed to protect homes or crops, which would translate into more hours of labor and materials and alter equipment needs. Areas with sensitive environmental resources, such as those in protected areas or the wildland urban interface (WUI) would be expected to have higher treatment costs since contractors and agency supervisors would need to take extra care to ensure the safety of such resources. Specific expenses might include additional supervising personnel, insurance requirements, and extra time required for permitting.

Some site factors did not affect treatment cost when aerial methods were used. However, slope remained important in both ground-based and aerial applications since slope influences the pattern of application and aircraft altitude, thereby affecting costs. Other factors affecting treatment cost were the restoration goals and, in particular, whether native or non-native plants were being restored since native seeds tended to be more expensive than non-native seeds.

4.2.1 Treatment Regimes Considered

In evaluating costs, we considered various levels of treatment intensity for both preventative and control treatments. When controlling cheatgrass, the first step in treatment is typically to burn the area, therefore, our analysis of burned area rehabilitation costs (where treatment is applied to a burned area) was applicable to a broad range of treatment currently being conducting. Following a burn, herbicides may be applied before reseeding or reseeding may occur without interim treatment. Seeds are spread using aerial (fixed wing aircraft or helicopters) or ground-based equipment. Application of herbicides can also be spread aerially or using ground-based equipment. Aerial seeding is sometimes followed by chaining, where ship anchor chains are dragged across the ground surface to increase contact of the seeds with the soil. Mechanical seed planting equipment (drill seeders) may also be used and are generally considered more effective than aerial seeding. Fencing is used to protect newly seeded areas from animals.

Aspects common to all treatments we investigated were: 1) multiple trips to treatment sites were needed to establish extent of potential restoration area (burned or infested) and to conduct multiple phases of treatment, and 2) native or non-native seeds were chosen based on meeting site-specific restoration goals. Native species, while desirable for their hardiness under extreme weather conditions such as drought and their ability to support native fauna, are difficult to establish and less competitive with cheatgrass than non-native species. Non-native species used to restore sites often have forage value (e.g., crested wheatgrass - *Agropyron desertorum*) and a higher germination rate than native seeds. Most commonly, a mix of native and non-native species were used, with the proportion of natives varying by site conditions, seed availability and management preferences.

4.2.2 Estimating the Cost of Treatment Model

We applied statistical modeling, GIS analysis and other estimation methods to generate parameters for Equation 9. A statistical model was fit to a database on treatment spending for two BLM rehabilitation programs. GIS analyses were used to develop data on journey times to treatment sites.

4.2.2.1 Database on Restoration Spending

The database used to fit the statistical model of treatment costs contained restoration spending information submitted by BLM offices for the Emergency Stabilization and Rehabilitation (ESR) and Burned Area Rehabilitation (BAR) programs. Projects were implemented from 2001 though 2003 in western states. Included in the database were offices in Arizona, California, Colorado, Idaho, Nevada, Oregon, Utah, and Wyoming that had implemented projects to mitigate impacts from fires. Cheatgrass is being managed cooperatively in this region by USFS and BLM, but all cost data were provided by BLM for BLM lands. Collection of spending data was not standardized across BLM offices, therefore, we selected data for burned areas in Idaho and Nevada only, to limit variability due to different accounting methods.

The same burned area may receive ESR and BAR funds, so data from both programs were combined to more accurately reflect total restoration spending per site. Fire numbers were used to combine data from the two sources by location and total amount spent in both programs was combined. The combined database for 2001-2003 had a total of 65 unique observations that included the critical spending and site attributes, 35 for Idaho and 30 for Nevada. Database attributes for sites included location descriptors, size, and costs of treatment. Treatment spending was broken down into costs for aerial seeding, drill seeding, miles of fencing, and herbicide use. The types of seeds used (native vs. non-native) and whether native shrubs had been planted were described. All sites received aerial seeding over some portion of the burned area and a subset received more intensive practices such as herbicide and drill-seeding. The majority of sites showed spending on fencing (72%) and this spending was broken out into repair of permanent fence or installation of temporary fencing that was either electric or non-electric.

4.2.2.2 Data Development within the GIS

The spending database was joined to a spatial coverage of reported fires (BLM 2004) to allow rehabilitated areas to be geolocated. The GIS coverage of fire locations provided a single point corresponding to fire location. A unique fire ID was available in both databases to conduct an accurate linking.

Using the fire location points to represent the approximate center of the burned areas, we used supplemental spatial data and GIS analyses to calculate or associate additional attribute variables with each burned area. In cases where we were interested in conditions across the entire burned area, we estimated the location of fire extent. Because we did not find data on exact fire footprint on the landscape, we assumed fires were circular and centered on the point location. Fire size was used to calculate an appropriate circle radius. These circular areas were used to determine average conditions of variables across the burn, such as average slope, and used to determine presence or absence of certain conditions, such as whether the protected areas occurred within the burned area. The complete list of variables calculated in the GIS is shown in Table 6.

Table 6. Variables Developed Using GIS Data and Analysis to Test Effects on Treatment Costs

Variable	Calculation
Elevation	Average value within burned area (DEM from ICBMP 2004)
Slope	Majority of cells within 0-10%, 10-30%, or > 30% slope; calculated from elevation (DEM) data
Within Urban area or Wildland Urban Interface (WUI)	Co-occurrence with urban land use defined in WUI (Radeloff et al. 2005)
Rocky areas	Co-occurrence with "Barren" land category (Idaho Gap Analysis, Landscape Dynamics Lab, 1999)
Within Protected Area	IUCN protected status: IA, IB, II, and IV; (Conservation Biology Institute 2005, DellaSala et al. 2001)
Airport Travel Time	Travel Time/Cost Analysis (see text)
Field Office Travel Time	Travel Time/Cost Analysis (see text)

Travel Time Estimation Methods

To evaluate travel time and costs, we first used a method to determine least cost travel paths between locations. An algorithm available within the GIS software selects paths between origin and destination cells by minimizing the cumulative time cost of travel (ESRI 2005). Travel time between any two cells is calculated by

$$t_i = f(Distance, Impedence) \qquad \text{(Eqn 10)}$$

where travel time (t) is a function of cost-weighted distance represented as the impedence or friction of traveling between cells. The impedence represents relative travel speed and is measured through an impedence map that associates travel speed with each cell on the landscape. The impedance map was created using four variables that affected travel time. The road network was mapped and road segments were weighted by road class in proportion to speed limits by road class. Other variables were added to affect travel off-road. Large streams and rivers were used to create barriers to travel off the road network and slopes were used to slow travel in areas with no roads by dividing slopes into three classes and weighting travel speed within each class. These factors were weighted and combined such that the value of each cell in the impedence map equaled the time required per unit distance to pass through the cell (Figure 10).

The cumulative time to move from the nearest origin point to a destination point *i* (T_i) is the sum of the impedence of all the cells along the least-cost route:

$$T_i = \sum_i t_i \qquad \text{(Eqn 11)}$$

Total travel times between points were estimated for several sets of origin and destination points. To represent personnel travel times, field offices were used as the origin points. Destination cells consisted of all invaded or potentially invaded lands. To represent contractor driving time associated with certain treatments, we calculated driving times from airports to represent required travel time for contractor vehicles that supported aircraft. Invaded and invadable areas were identified using a map we created of cheatgrass cover (Appendix A) and by assigning invadability status to current land cover (ICBEMP 2004) using best professional judgment of R. Mack who has conducted field studies of cheatgrass in the region for over 30 years. Using these origin and destination points several versions of travel cost maps were developed (Figure 11).

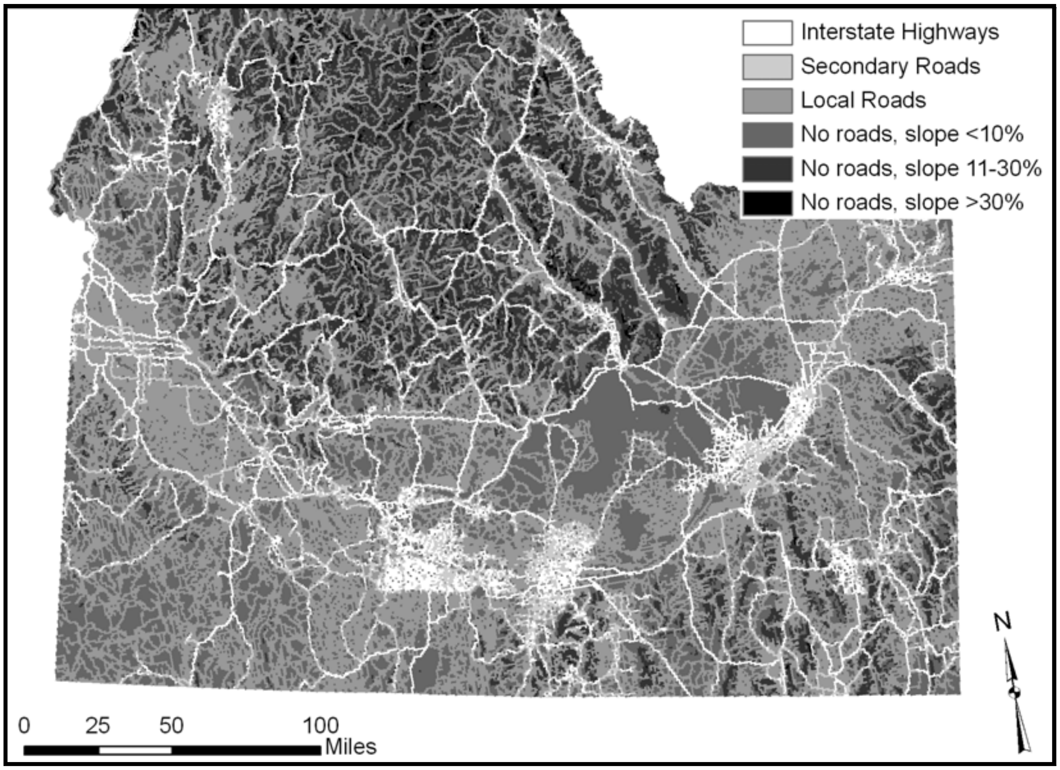

Figure 10. Impedence Map Representing Travel Time per Landscape Grid Cell
Map showing the relative time required to travel through an individual grid cell on the landscape. Darker shades represent slowest travel times; lighter shades faster travel times. Major roads, which had the fastest travel times, appear as white areas.

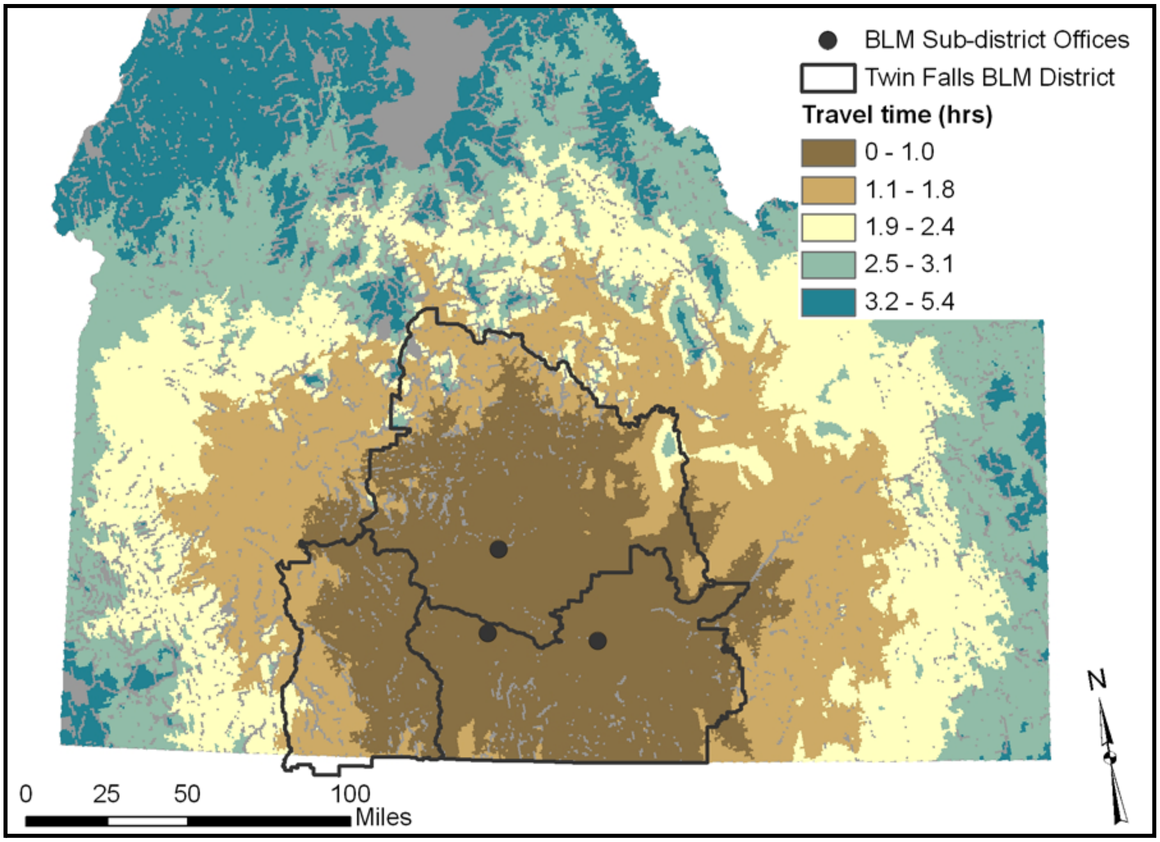

Figure 11. Cumulative Journey Time Map
Map showing the sum of travel times along the "least cost" path between BLM district offices and all potential treatment locations. A least cost path is determined through a routing algorithm using the impedence map, shown in Figure 10 and then total travel time is the cumulative sum of time time along that path.

4.2.3 Model Estimation

Our objective with the cost modeling was to identify variables that could explain heterogeneity of total treatment costs (TC_i) among treatment sites and to develop a model that could be used to predict costs for untreated sites. However, the statistical modeling that was fit to the treatment spending database could not encompass BLM personnel time and other costs because the majority of observations within the treatment spending database accounted for materials and contractor costs only (Tom Roberts, pers. comm.). As a result, our statistical model only estimated treatment cost per area (CA_{ik}) which would be able to represent expected contractor bid amounts by location and treatment. Planning costs and supervisory costs by agency personnel were estimated elsewhere (as described below).

As previously discussed, we hypothesized that treatment costs would vary due to factors affecting material, labor and time costs of treatment. Of the variables we identified as affecting costs, we were able to gather data or create methods for measuring slope, elevation, presence of rocks, location in protected areas, location in WUI, and remoteness (measured as driving time from airports).

We also expected higher intensity of treatment to increase treatment costs and we tested the effects on costs of all treatment options identified in the database including use of: drill seeding, fencing, herbicides, and native seeds. We also tested whether differences in costs could be attributed to the state from which reports originated. We hypothesized such costs might differ due to variations in accounting, treatment, site conditions or other factors.

Economies of scale were anticipated, since larger burned areas would be expected to be cheaper to treat on a per acre basis. Previous work has suggested that fire treatment costs are a nonlinear function of size of fire. For example, fire suppression costs for large fires have been estimated to increase with the square root of fire size (Mills and Bratten 1988). We tested several nonlinear relationships between average costs per acre of invasive species treatment and total size of area treated. The economies of scale would be expected to be derived from spreading fixed costs, such as helicopter rental and transport to site, across more treated area.

4.2.3.1 Model Limitations for Prediction

The database presented many challenges to testing our hypotheses. Foremost in terms of estimating the effect of treatment intensity, was that the data on the proportion of the fire treated with each treatment methods were missing in many cases. Also, information on whether chaining had been conducted was missing. Since treatments were applied to very different proportions of the site, and chaining could potentially increase costs substantially where used, these were serious omissions. Although, we were not able to weight each treatment by area treated, we still chose to examine how use of a practice (binary yes/no) affected total treatment cost per acre.

Other challenges included selection bias in the data set. Using only treated fires to estimate costs for all fires is problematic because treated areas are likely to have different characteristics from untreated areas. Most troublesome was that more difficult sites were likely to be left untreated, creating a tendency within our model to underestimate costs for some sites because of effects we did not estimate. Yet, sites with high ability to recover naturally were also untreated and treatment costs may be somewhat lower in these sites, perhaps creating a competing effect to lessen the degree of bias in the model. Most importantly, the model would be expected to be most accurate for sites with the highest probability of treatment under current practices.

Several factors caused us to reduce the number of observations used in the statistical model from 64 to 46. We were forced to omit observations that could not be associated with points mapped in the spatial database of fire occurrence because of the need to measure complementary site attributes within the GIS. Therefore, the number of observations was reduced to 48 through this process, resulting in 28 observation within Idaho and 20 within Nevada. In addition, only one observation of a treatment using hand-planting was available, so this observation was discarded and one extremely expensive treatment site (outlier) was removed from the dataset because of concerns that this one data point was unduly influencing model results. This site was a particularly small burned area. Therefore a total of 46 observations were used in final model estimation.

4.2.3.2 Treatment Cost Model Specification

Our initial modeling used ordinary least squares (OLS) linear regression, to estimate a model of the form:

$$y = \alpha + \sum_{j=1}^{J} \beta_j x_j + \varepsilon \qquad \textit{(Eqn 12)}$$

Where   was a constant,   ’s were the estimated coefficients on the independent variables (***x***) and   represented unexplained variability. The variable ***j*** indexes the independent variables used to predict costs. The vector of independent variables tested (***x***) is shown in Table 7.

Table 7. Variables Considered for Empirical Cost Model

Variable	**Mean** (continuous) (n=46)	**Sum** (binary) (n=46)
Treatment options		
Fencing use (=1 if used)		34
Drill seeding use (=1 if used)		16
Herbicide use (=1 if used)		22
Native seed use (=1 if used)		29
Site and location characteristics		
State (=1 if Idaho)		27
Within WUI (=1 if within)		2
Within Protected Area (=1 if within)		6
Slope class (= 0 if 0-10%, =1 if 10-30%, none > 30%)		4
Burned area within BLM lands (acres)	4605	
Elevation (m)	1547	
Airport Driving Time (hours)	1.16	
Dependent variable		
Treatment cost per acre ($)	$102 all cases ($64 ID, $154 NV)	

Variable names shown in gray were not tested in model due to the limited number of observations.

Once the final database had been constructed, it became clear that we did not have sufficient observations in some classes to test effects. We quickly eliminated WUI location, slope class, and protected area location due to the lack of sufficient representation of such conditions. All continuous independent variables (burned area, elevation and driving time) were log-transformed, as was the dependent variable of cost per acre.

Using model fit statistics from both ordinary least squares and maximum likelihood estimation, a parsimonious model was selected that included five independent variables shown in Equation 13. The variables that were dropped: fencing use, elevation, and airport driving time either had little predictive value or collinearity problems with other variables. The final OLS model had the form:

$$ln(cost\ per\ acre) = \ + {}_1ln(fire\ size) + {}_2(state) + {}_3(herbicide) + {}_4(drill\ seed) + {}_5(native\ seeds) \qquad (Eqn\ 13)$$

This model had an R^2 of 0.608, an adjusted R^2 of 0.559 and was significant at > 0.001 level. Coefficient estimates are shown in Table 8 and, if we assume a valid model, all coefficients were significant at $p > 0.02$, although as we discuss below, these significance tests are suspect due to spatial autocorrelation of the residuals.

Table 8. Coefficient Estimates from OLS Model

	Coefficients	Std error	Standardized Coefficients	t statistic
Constant	6.440	0.509		12.649
State dummy	-1.222	0.217	-0.743	-5.620
Herbicide dummy	0.494	0.189	0.304	2.606
Ln(Area)	-0.306	0.066	-0.485	-4.642
Drill seed dummy	0.548	0.182	0.322	3.015
Native dummy	0.562	0.226	0.335	2.488

The OLS model results suggested that costs in Idaho were lower than in Nevada, when treatments applied were held constant. As expected, we found economies of scale with treated area size in which cost per acre was a decreasing function of the log of area. As part of fitting the model, we explored other relationships between area treated and cost per acre and found the log function provided the best fit to the data. Also as expected, extra treatments, such as using herbicides and drill seeding, increased per acre costs. Native seed use tended to increase costs.

The effect of state (Idaho vs Nevada) was highly significant indicating that treatment methods and or accounting methods were dramatically different between states. We combined data from both states to improve statistical power of the cost model and to create a more geographically diverse set of treated fires. However, combining data between these two states clearly created a less homogenous dataset.

Unfortunately, none of our location characteristics could be definitively linked to costs of treatment. This result was unexpected and we suspect part of the lack of predictive power of these variables stems from problems with data. Improved databases are needed to fully test the potential relationships and alternative model forms explored. The selection bias inherent in our data set appeared to eliminate difficult sites that would have higher costs and thus prevented us from testing the effects of difficult conditions on cost. The lack of many treated sites in areas where costs would be expected to be high (e.g., in urban areas, protected areas, areas with high slopes) suggested, and interviews with managers confirmed, that difficult sites were generally avoided.

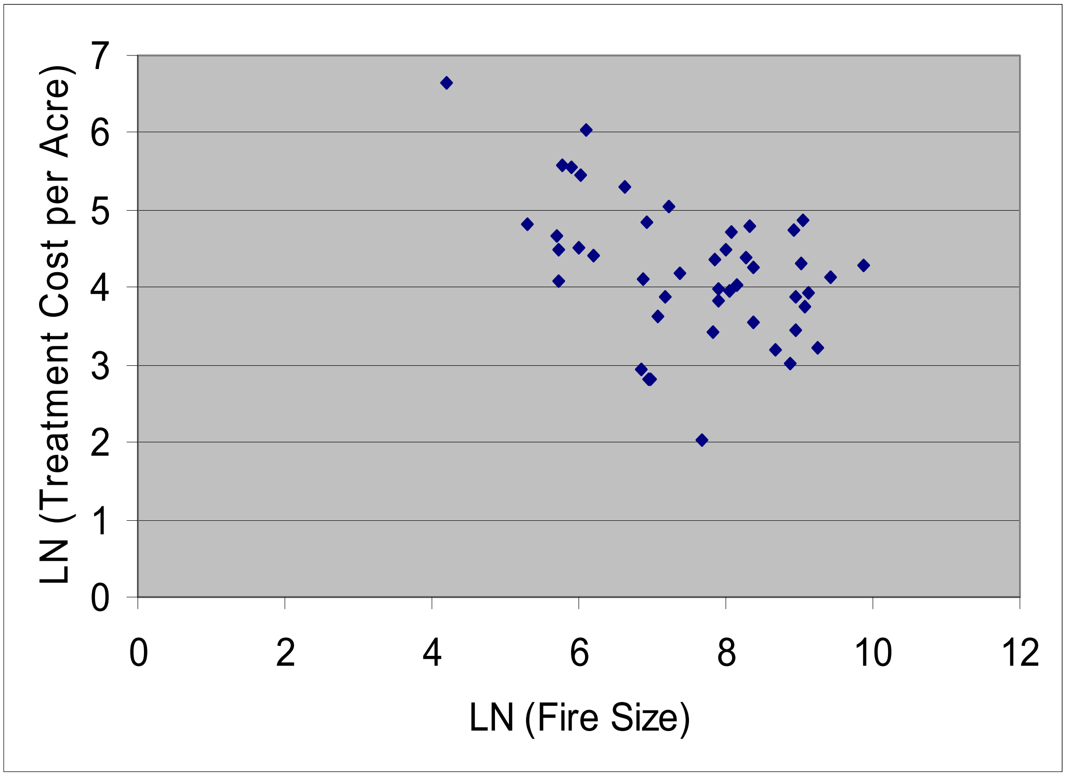

Figure 12. Natural Log of Cost per Acre vs. Natural Log of Fire Size
We tested several nonlinear relationships between fire size and treatment costs in our statistical model and the log of fire size provided the best fit for our dataset.

4.2.3.3 Test of Residuals for Spatial Autocorrelation

Spatial autocorrelation of data can arise when observations are close together in space. The minimum distance between burned areas was about 3.75 miles for our combined data set for Nevada and Idaho indicating a reasonable separation between data points. Nonetheless, we expected to find spatial autocorrelation of the data and a semi-variogram plot and a likelihood ratio test confirmed a moderate degree of spatial autocorrelation of the residuals from the OLS model. Because of this spatial autocorrelation, the parameter estimates for our model are unbiased but inefficient, and standard error estimates are biased thereby throwing significance tests into doubt.

A comparison of methods for dealing with spatial autocorrelation of errors demonstrates that significance tests may be strengthened or weakened by failing to correct for this bias (Bell and Bockstael 2000). While many statistical tests are available to handle such autocorrelation of the errors, decisions made in estimating alternative models can produce other sources of bias (See Bell and Bockstael 2000 for a comparison of methods and Anselin and Florax 1995 and Cressie 1991 for more complete discussion). Given that the selection bias of our data and other questionable aspects of the data weakened the strength of our OLS model, we did not expect the benefits of correcting for spatial autocorrelation of the errors to warrant the extra effort of fitting a different model. Rather, for this screening effort, we used the OLS model to estimate costs at untreated sites with the acknowledgement that this estimate could be improved. In the future, expected improvements in cost accounting databases will likely warrant such efforts to develop a model robust to spatial correlation errors.

4.2.3.4 Other Sources of Cost Data

We made the simplifying assumption that fixed costs were the same for all locations; therefore variation in costs between sites was determined by variable costs only. Because of poor data availability, we did not include a value for fixed costs in our total cost calculations. This omission has the effect that cost-effectiveness ratios will appear higher than if we had included fixed costs. However, this omission does not affect the budget constraint of the optimization or of the real world decisions of resource managers because the agency's treatment budget does not include fixed costs.

Search Costs

For the types of treatments applied here, where large burned patches were being treated, search costs (i.e., for invasive plants of interest) were ignored since individual plants or patches of infestation were assumed to be readily identifiable. However, search costs may be significant when only parts of patches are being treated.

Journey Costs and Daily Costs

We created coefficients for journey costs (JC_k) and daily costs (DC_k) by examining published reports, grey literature analyses of treatment costs, and previous databases we had created for restoration costs. In addition, we interviewed personnel within local field offices to gather additional information regarding practices and costs not directly associated with treatment such as personnel time and hourly rates. Overhead costs were ignored because they did not contribute to major heterogeneity of costs between sites. Fixed costs such as initial scoping and planning, treatment design, and permitting were included to a limited extent as an initial site visit with personnel and equipment costs. Information on fixed costs was not readily available from accounting personnel.

Comprehensive information about personnel costs was not available from BLM, but a representative from the Twin Falls district described typical makeup of crews that would visit sites for planning and contractor supervision. A typical initial site visit might be conducted by 6-7 people, typically all GS-11s. Once contracts are in place for treatment, 3-5 people will generally have responsibility for inspecting sites and supervising contractors depending on the number of contracts issued. Different treatments require different crew make-ups. From this information, we estimated average, low and high hourly costs to represent different types of crews visiting a site.

The journey costs by treatment (JC_k) used for subsequent modeling and shown in Table 9, incorporated the average hourly crew rate for initial inspection and one follow-up inspection trip. A per vehicle cost of \$0.315/ mile was applied, based on the IRS reimbursement rate for large vehicles such as vans or trucks. Daily costs were not used because sites were not sufficiently far from offices to indicate BLM personnel would be required to stay overnight.

4.3 Cost Model Discussion

The importance of cost-screening tools should not be underestimated since a manager typically decides which sites to treat based on little or no explicit cost information. Much of the treatment work is contracted to outside businesses and therefore, according to BLM personnel, the details of cost estimation are not well understood within the agency. Also, the cost of personnel time to administer contracts and monitor sites is not well-tracked. Clearly, if effort is going to be allocated based on cost-effectiveness of all direct and indirect costs, improved accounting would be needed.

We were able to generate tools to estimate costs of treatment at untreated sites by creating 1) estimation equations of treatment costs by treatment intensity and 2) maps of journey time that could be created within the GIS and combined with personnel and vehicle costs (Table 9) to generate journey costs for agency personnel and equipment. It was likely our cost model underestimated total costs because of the variables we were not able to include.

The statistical modeling exercise of treatment cost per acre did not find that site or location characteristics were useful for predicting costs by location. This result is in contrast to other work that found that site characteristics were important in predicting costs (Hesslen et al. 2004). We attributed our inability to link site characteristics to costs to problems with the data that likely prevented us from uncovering these relationships. However, through the journey cost analysis, we captured a major factor determining variability of cost by location.

Table 9. Journey Costs per Hour and per Trip

Treatment level (k)	# trips	# vehicles	Vehicle operation cost ($/hr)	Average cost crew ($/hr)	Low cost crew ($/hr)	High cost crew ($/hr)	Inspection crew cost ($/hr)	Total Journey Cost (all round trips)
No Treatment								
Aerial seeding only	1	1	$18.90	$160.59	$114.56	$206.61	$34.61	**$466**
Aerial seeding + chaining	2	1	$18.90	$160.59	$114.56	$206.61	$34.61	**$863**
Aerial seeding + chaining + herbicide	3	1	$18.90	$160.59	$114.56	$206.61	$34.61	**$1,260**
Aerial seeding + chaining + drill-seeding	3	1	$18.90	$160.59	$114.56	$206.61	$34.61	**$1,260**

5. Restorability

5.1 Introduction

The benefits of post-fire rehabilitation are inextricably linked to the successful outcome of such efforts. The perceived value of the investment, as with any risky investment, depends on the certainty of the outcome. For a typical risk-averse investor, risk reduces the perceived expected value of an investment, and the reduction in value is a function of 1) the degree of risk as measured by the variability of potential outcomes and 2) the degree of risk aversion of the investor. This means that, all else equal, risk-averse managers will prefer to do a restoration that is a “sure thing” rather than pursue an option that has a high probability of failure.

How might we take the effect of risk into account when weighing alternative investments? The basic formula investors use for comparing uncertain outcomes is to weight expected value by the probability of alternative outcomes using a formulation such as:

$$E(B) = [\,b_{ijk}^{succeed} * p] + [\,b_{ijk}^{fail} * (1 - p)] \qquad (Eqn\ 14)$$

where the expected value of benefits $E(B)$ is the sum of the expected benefits of each alternative outcome (b_{ijk} as measured if the project succeeds or if the project fails) multiplied by its probability of occurrence. If there are only two outcomes, success and failure, and p is the probability of success, then the probability of failure is $1 - p$. The benefits of a successful project are multiplied by the probability of success and the benefits of failure are multiplied by the converse probability, allowing the total expected values of outcomes to be estimated. This model provides the decision-maker with a logical means to compare benefits for projects that vary in risk levels.

Success of restoration depends on how success is defined. The broad management goals may be to reduce erosion potential, to limit the cover of invasive species and to restore vegetation appropriate to the expected uses of the site. Areas with the potential to provide quality habitat may have additional goals set forth to establish native species, whereas site with limited ability to provide habitat may still serve as productive areas for livestock grazing or other uses.

In the case of restoring a burned area, success was defined, through discussions with managers and scientists, as several sub-goals that would allow the overarching goals to be realized: 1) preventing regrowth of cheatgrass that was present before the fire, 2) preventing new cheatgrass recruits from establishing, and 3) ensuring long term survival of desirable species, which may be natives or non-native forage species (Figure 13). Many factors had the potential to affect the probability of success of each sub-goal. We identified three general categories of factors that determined restorability, or probability of success: site characteristics, location or landscape characteristics, and stochastic variables such as rainfall patterns. Each category had a main influence on one of the three sub-categories of success and a lesser effect on the other categories as shown in Figure 13.

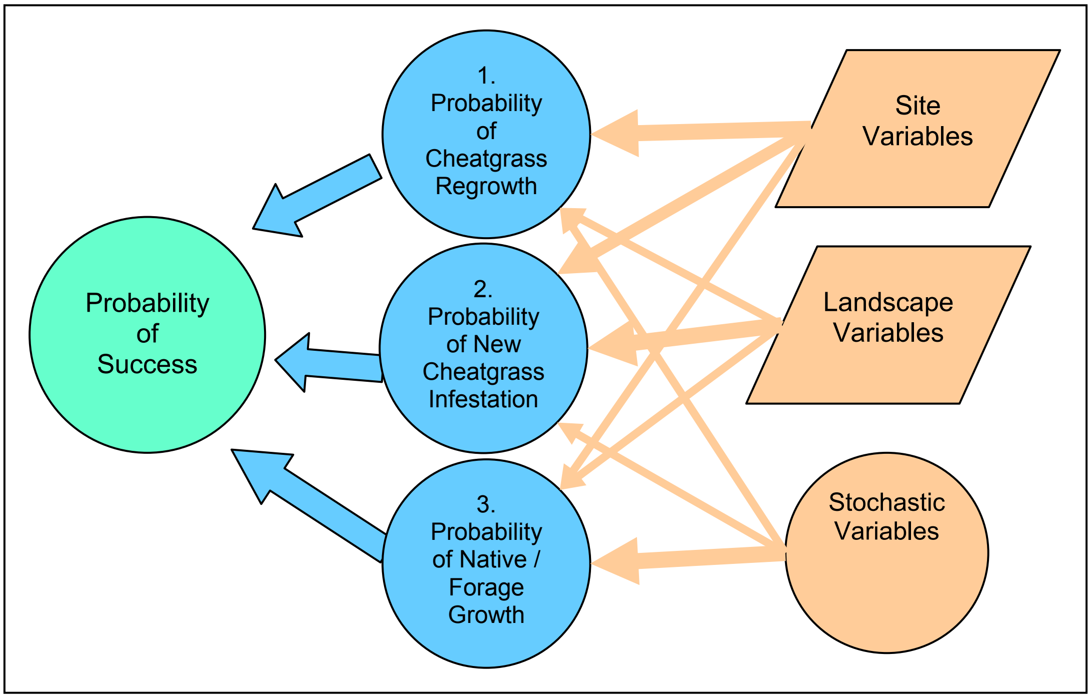

Figure 13. Influence Diagram of Restorability
The overall probability of success (circle at far left) was conceptualized as the joint probability of 1) preventing regrowth of cheatgrass present before the fire, 2) preventing new cheatgrass recruits from establishing, and 3) ensuring long term survival of desirable species (natives or non-native forage species). The probabilities (circles) in center column were functions of the observed factors (parallelograms and a circle in rightmost column). Site variables included characteristics such as % cover of cheatgrass pre-fire and elevation. Landscape variables included factors such as distance to nearest cheatgrass-infested area. Stochastic determinants included factors such as timing and quantity of rainfall, and availability of experienced pilots. Width of arrows shows relative importance of a given type of factor in determining the probability.

Restorability was not considered to be independent of treatment effort. Given unlimited resources, almost any site can be successfully restored unless we lack technical knowledge or source materials. Such a case of irreversibility appears to exist, for the moment, in parts of Southern Nevada. Fire now occurs regularly in a landscape that did not evolve in the presence of regular fire. It appears that following fire, an invasive non-native grass, red brome (Bromus rubens), is increasing its range and prevalence; cheatgrass may also be adapting and invading into the Mojave desert in southern Nevada. As a result, plants have limited capacity to rebound following fire and we currently lack the materials and expertise to propagate the plants that are being lost (T. Roberts pers. comm.). Nevertheless, in many systems invaded by non-native plants, we expect to be able to increase the probability of successfully restoring services by increasing the level of inputs, although a non-linear (e.g., exponential) relationship between inputs and success is typical (Figure 14).

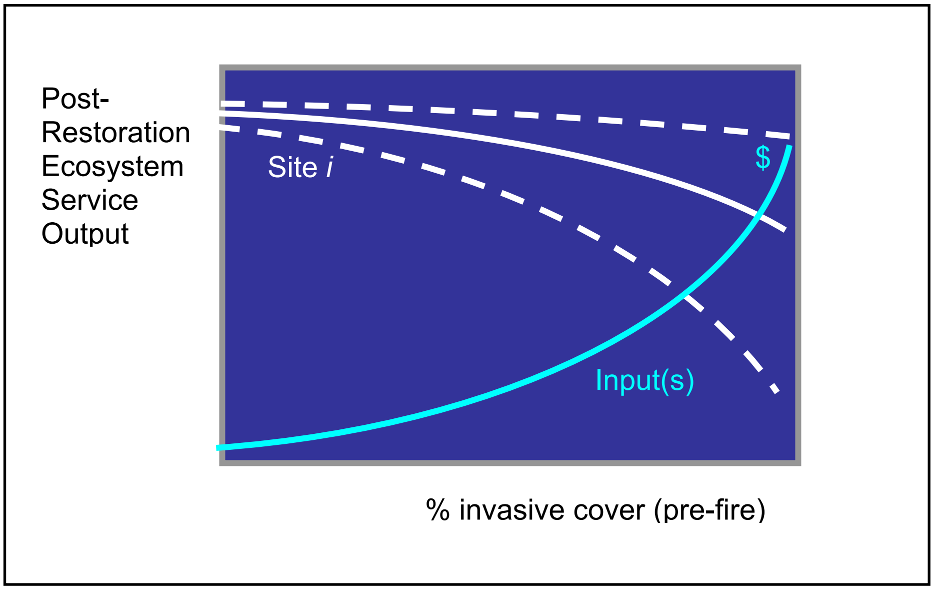

Figure 14. Reversibility of System Decline
Theoretical representation of ecosystem service output after restoration as a function of level of degradation (% invasive cover) before fire. The solid white line shows the ecosystem service outputs after restoration. The blue line shows the increasing inputs required to achieve the level of restoration shown. The dashed lines show the range of potential outcomes and increasing uncertainty of outcomes at higher levels of initial degradation.

5.2 Measuring Recoverability and Restorability

Returning to Equation 3, we developed S_{ik} (restorability) as a function of natural recoverability and treatment effectiveness.

$$S_{ik}= r_i + \quad _k \qquad \textit{(Eqn 15)}$$

r_i = site recoverability
$\quad _k$ = treatment effectiveness
i = site
k = treatment applied

Thus, we envisioned restorability as a function of site characteristics that tended to increase or decrease the probability of success for a given treatment type. Note that in the absence of treatment, recoverability = restorability (S=r).

We found, as others have (Robichaud et al. 2000, Eiswerth and Shonkwiler 2006), that little empirical evidence exists to quantify probabilities of successful restoration of cheatgrass invaded sites. Even less information was available on the ability of systems to recover naturally, let alone provide information on variables that would affect variability of response. Therefore, we used a combination of techniques to identify conditions that might be predictive of restoration outcomes. A literature review and interviews with managers and researchers provided initial information which we used to develop an interview protocol to guide further discussions with managers. Within interviews, we encouraged managers, ecologists and restoration practitioners to weight the relative importance of different factors (Appendix B). We also informed our restorability analysis through a statistical exploration of the qualitative restoration outcomes included in the treatment spending database used for the cost modeling.

Our goal in developing data for recoverability was to characterize biophysical attributes that could be mapped to allow managers to weight risk of benefits within our spatial cost-effectiveness screening tool. Our ideal data set for testing such biophysical factors, would have included multiple sites with varying site conditions and located within different environmental contexts, for each level of treatment applied. In other words, we would like inputs to have been held constant as we compared the probability of success across environmental gradients. However, the concepts of recoverability and treatment effectiveness were confounded in the database we examined and in the expert judgment we elicited from managers because treatment was applied as a prescription, tailored to site conditions. Managers, quite reasonably, adjusted treatment inputs in an attempt to hold probability of success more or less constant. This limited our ability to statistically analyze the data, and therefore we relied more on interviews to try to tease apart the effects of treatment vs. site conditions.

5.3 Literature Review of Restorability Data

We reviewed the literature for information on factors that affected the ability to control cheatgrass and to characterize the effectiveness of restoration practices and invasive species control techniques. Most of the information was available only as grey literature and not within peer-reviewed publications.

5.3.1 Restoration Techniques Effectiveness

The most systematic review of restoration techniques we found was an analysis by Robichaud et al. (2000) who examined the effectiveness of post-fire rehabilitation, in western Forest Service lands, for preventing erosion and establishing vegetation. They reviewed 321 restoration projects put in place over 26 years and evaluated relative effectiveness of specific techniques in addition to the effects of site and environmental factors contributing to success or failure of projects. Much of the evidence was drawn from post-fire monitoring reports, although interviews were also used to supplement reported data.

The Robichaud et al. work identified a variety of factors that might limit or enhance effectiveness ofpost-fire seeding including the ability to find proper inputs such as: contractors and aircraft operators, weed-free straws, and skilled pilots. Their work suggested that timing of seed application was important because it could affect probability of experiencing favorable weather such as rainfall and avoiding negative events such as high winds after seed placement. Other factors that tended to enhance success included successful protection from grazing (a combination of low existing grazing pressure and reducing pressure through effective fencing), avoiding slopes, and avoiding soils that were shallow, rocky, uneven or fine-textured.

The reported effectiveness associated with seeding tended to be higher when qualitative techniques were used to judge results and lower when quantitative techniques were used. Around half of the interviewees and monitoring reports that used *quantitative* information stated that aerial seeding had good or excellent effectiveness. On the other hand, 79% of monitoring reports using *qualitative* judgments considered seeding successful in the first year following fire (Robichaud et al. 2000, p. 45).

The same report showed that reported effectiveness also varied by geographic region. Reports from the region that including Idaho showed a higher proportion of responses stating that seeding was effective compared to other regions of the country. However, it should be noted that many of the restoration sites evaluated were in locations with higher precipitation levels than the case study sites examined in southern Idaho.

The difference in aerial seeding success rates perceived by those using qualitative vs. quantitative information, speaks to the need for more rigorous studies to determine effectiveness. In Robichaud et al.'s review of treatment effectiveness, they found the literature "limited." In terms of evaluating natural recoverability of sites, they also found that, "good methods for assessing native seed bank viability are lacking."

A study of restoration success that focused exclusively on rangeland in Nevada (Eiswerth and Shonkwiler 2006) used an empirical analysis to evaluate seeding success of natives and non-natives as a function of site conditions and seeding techniques. They found that 1) timing of seed placement 2) seeding rate and 3) whether grazing had resumed between fire and seeding were most predictive of success. Seeding was most successful in the fall and early winter. They also found the response to seeding rate was quadratic, indicating success peaked at an intermediate seeding density. Grazing between the fire event and seeding appeared to enhance cover of cheatgrass. From the limited database at their disposal, they were able to develop initial empirical relationships between site conditions and treatment effectiveness, but further work is needed to test these effects across regions.

A few other studies have tried to determine what makes arid grasslands susceptible to cheatgrass invasion. For example, Gelbard and Belnap (2003) found that sites within 50 meters of a paved road had a higher proportion of cheatgrass cover than sites adjacent to unpaved four-wheel drive tracks. The effect of elevation has been studied and there seems to be some consensus that some elevation threshold generally exists above which cheatgrass does not persist and that higher elevation is also associated with higher probability of successful recovery and restoration (Pyke et al. 2003, DiTomaso 2000). However, consensus on the exact elevation of the threshold is elusive and may vary by region.

5.3.2 Control Effectiveness

We evaluated literature that examined effectiveness of various mechanical, chemical, and biological control methods. Much of the literature involved observations on limited numbers of plots, often without control plots, and therefore the results were of limited use. However, we found reasonable documentation and quantitative studies on the effectiveness of herbicides to control cheatgrass and other invasives, most likely because it is a primary method of weed control in rangelands (DiTomaso 2000). Although most studies did not consider the effect of location on effectiveness, one study identified precipitation regime as an important restorability marker (Quigley et al. 1996).

For cheatgrass control, two studies showed that the herbicide OUST® dramatically reduced cheatgrass cover on treated sites. Studies showed that timing of application affected success with 95% control achieved on fall-treated and 60% control achieved on spring-treated. The treatment was more effective on a weed that typically co-occurs with cheatgrass, medusahead (*Taeniatherum caputmedusae ssp. asperum*), resulting in a 93% reduction in biomass on fall-treated and 82% on spring-treated plots (Shaw and Monsen 1999). Pellant et al. (1999) reported that OUST® was more effective than disking, burning or no treatment on research plots in Nevada. The herbicide treatment resulted in a 26% "frequency of occurrence" of cheatgrass compared to 72% occurrence on the control plots and 46% or 42% occurrence on the disked and burned plots.

The herbicide OUST® is no longer used by BLM to control cheatgrass, but they have sought permission to use another herbicide, Imazapic (Plateau®) (BLM 2005), and in the interim are using glyphosate (Roundup). We did not find any published studies on the effectiveness of these herbicides for controlling cheatgrass but we were told by researchers that both herbicides have the potential to be highly effective, with Plateau® offering residual (long-term) control (J. Volmer pers. comm., M. Pellant pers. comm.).

5.4 Using Elicited Best Professional Judgment to Score Restorability

Although we identified many factors that had the potential to affect restoration probability, the restoration practitioners and researchers we interviewed generally focused on only a few of the variables as being most appropriate for generating rules of thumb. The response of practitioners to early interviews steered us away from pursuing the use of a more structured survey instrument. Several factors contributed to a lack of enthusiasm for investing our research resources in a survey. First, empirical data were largely lacking and restoration practitioners appeared to have insufficient experience or information to judge potential success objectively, particularly given the high variability of annual grass growth. Second, only a subset of researchers were comfortable generalizing about conditions, especially since they viewed groups of conditions as being the most important determinants. Many could not isolate individual factors for evaluation, indicating that a complex survey design was needed (e.g., a contingent choice model), which would, in turn, necessitate having many respondents. Third, restoration experiences tended to vary with the particular ecosystem/ecoregion under study. Therefore, the number of researchers or practitioners with expertise in any particular system was limited and likely to preclude successful implementation of a complex survey.

We used the interview protocol we developed (Appendix B) to generate discussion and elicit responses from a variety of personnel engaged in restoration of rangelands. The group included biologists, ecologists, economists, program managers, and rangeland specialists and represented agencies including the USDA Forest Service, Bureau of Land Management, US Fish and Wildlife Service, and US Geological Survey and academic institutions.

The broad range of interviews or group discussions we conducted, among approximately 30 restoration practitioners, yielded the results in Table 10. To refine those results, we examined which variables were highly correlated spatially and found substantial overlap. The correlations allowed us to identify some variables as potentially redundant, however, we were not able to elicit sufficient information to allow us to combine remaining factors into a cohesive model of restorability.

Using these results and interviewee comments, we adapted our interview protocol to focus on the most important factors practitioners used when selecting appropriate sites for treatment. We interviewed restoration practitioners within the Twin Falls BLM district and gave significant weight to their judgments since they had the most experience conducting restoration in these ecosystems. As previously mentioned, restoration practitioners tended to adjust treatment to expected site outcomes and therefore used restorability "rules of thumb" to set treatment levels. For example, managers would use a higher proportion of native seeds on sites they deemed to have a higher probability of success and were more likely to use herbicides on sites with a low probability of success. In addition to identifying which factors affected success of treatment, we asked managers to identify factors and thresholds of such factors that would reflect the likelihood of a site to recover on its own (recoverability).

Table 10. Restorability Factors Rated by Restoration Practitioners (based on interviews)

	High Restorability	Moderate Restorability	Low Restorability
Precipitation zone (thresholds vary by locale)			
< 8 in (20 mm)			X
8-14 in		X	
> 14 in (36 mm)	X		
Cheatgrass cover 5-20%		X	
Cheatgrass cover >20% existing on site			X
Elevation			
>6000 ft.	X		
5000-6000 ft		X	
3000-5000 ft			X
Human Use			
urban use			X
intense recreational use			X
Soil Type			
droughty soils (sandy or granitic)*			X
high clay content soils*	X		
Ownership fragmentation (private landowner)			X
Habitat type			
Mountain big sage		X	
Wyoming big sage			X
Salt desert shrub			X

* Opinions differed on the effect of soil texture; high clay content (the opposite of droughty soils) was seen as an indicator of low restorability by some restoration specialists.

5.5 Statistical Evaluation of Restorability Data

The same database used to evaluate costs of treatment included some rudimentary information on restoration outcomes. We used that data on treatment outcomes to fit an empirical model that we hoped would allow us to predict probability of successful restoration. However, the database had significant limitations. The developers of the treatment database cautioned us that the data were not highly robust since they were gathered from numerous reporting sources and observations were missing for many variables. This warning from the database developers, and our own evaluation of the database, led us to limit our interpretation of the statistical findings for making predictions. However, we cautiously present the results of our statistical modeling to show how future data collection could be used to inform restorability and recoverability models.

The outcomes of restoration were described qualitatively by managers and we coded that information to allow us to conduct statistical modeling. Project results were judged by BLM managers using qualitative terms of "excellent", "very good," "poor," etc., which we assigned to a 0-5 scale with 0 being failure and 5 being "excellent" results. By assigning a 0-5 scale to the qualitative descriptions, we were able to put results in rank order and conduct statistical modeling to test whether the independent variables we had identified explained variability of successful outcomes.

Using ordinal logit (maximum likelihood) regression analysis, we explored the relationship between success ranking (0-5) and selected independent variables (Table 11). We tested the explanatory power of the site and landscape variables, treatment variables, and total spending to predict success rate. Treatment variables were included so that contributions of site conditions to restorability could be tested while controlling for treatment effort. Since we did not have data for untreated sites, we were not able to independently estimate natural recoverability and restorability, however the data offered some insights on these distinctions.

As with the cost modeling, we developed a database of spatial characteristics that had been identified as potentially important for predicting restoration success. We queried the GIS data to generate five site/landscape parameters identified by practitioners and researchers: elevation, slope, percent cover of cheatgrass prior to burn, isolation from human activity and proximity to roads. Precipitation regime and the particular sagebrush community, which were also identified as important restorability markers, were omitted due to high correlation with elevation.

Several spatial variables were measured indirectly. Our map of cheatgrass cover used a binary presence/absence variable and did not include the proportion of cover in cheatgrass. Therefore, to estimate proportion of site cover in cheatgrass, we evaluated the proportion of the surrounding landscape (fixed window) in cheatgrass and used that value as a proxy for percent cover on site. Isolation from human activity was measured as driving distance from the closest airport. A site was designated as near a road if it fell within a 500-m buffer of a primary or secondary road.

We controlled for size of treated area and treatment effort by including fire size and spending per acre. We evaluated all aspects of treatment choice for their influence on restoration success including techniques for spreading seeds, herbicide and fencing use, and whether native seeds and shrubs were used. As mentioned in our cost model fitting, explicit use of chaining was not available within the database as provided. The use of native seeds for grasses and/or shrubs was of particular interest because of the anticipated lower success rate in germinating such seeds. Treatment cost per acre and fire size were transformed by taking the natural log because of expected non-linear responses. We explored interaction terms between variables where we expected to see such effects.

Table 11. Variables Used in Restorability Statistical Model

	Variable	Mean (continuous) (n=45)	Sum (binary) (n=45)
Treatment options			
	Fencing use (=1 if used)		31
	Drill seeding use (=1 if used)		15
	Herbicide use (=1 if used)		21
	Native seed use (=1 if used)		28
	Shrub use (subset of native seed use) (=1 if used)		26
	Spending per acre ($)	89	
Site and location characteristics			
	State (=1 if Idaho)		26
	Slope class > 30% (1 if true)		No observations
	Within 500m of primary or secondary road (1 if true)		1
	Proportion neighborhood in cheatgrass	0.29	
	Burned area within BLM lands (acres)	4684	
	Elevation (m)	1544	
	Airport Driving Time (hours) (remoteness)	1.17	
Dependent variable			# observations
	Restoration Success (based on manager rankings)		
	0 = Failure		8
	1 = Poor		13
	2 = Mixed and Poor		6
	3 = Mixed and Fair		11
	4 = Mixed and Successful		1
	5 = Excellent (removed)		2

Variables in grey were not used in fitting model. Variables with known collinearity issues were tested separately (specific cases were 1) native dummy or shrub dummy; 2) spending per acre or treatment dummies).

5.5.1 Statistical Results

Developing a robust model with good fit statistics was challenging because of the many data issues. To explore the small data set, we fit a model with the fewest number of variables to understand the strongest correlations and to avoid overfitting the model. The best minimal model is shown in Table 12. The model was significant at $p > 0.01$, however, model fit statistics were barely adequate. Observations were insufficient to test the five levels of success we included in the model. Aggregating success rankingsinto three categories did not improve the model fit.

Despite the weakness of the model, the signs on the coefficients of the variables shown in Tables 12 and 13 were generally robust to model specification. Coefficients on native seed use, ln(fire size) and proportion of the landscape in cheatgrass were significant in many versions of the model. As expected, native seed use tended to reduce restoration success, as did larger fire size and greater proportion of the landscape in cheatgrass. Use of fencing tended to increase success and, as expected, being farther from an airport tended to increase success. We expected more remote sites to have higher probabilities of success, all else equal, due to the generally lower level of disturbance and lower cheatgrass cover of sites remote from human activities. All of these results were quite tentative and further modeling is required to confirm these relationships.

Table 12. Parameter Estimates for Ordinal Logit Model of Restoration Success

		Estimate	Std. Error	Wald	Sig.
Threshold	[SUCC_RATE = 0]	-5.412	2.125	6.488	.011
	[SUCC_RATE = 1]	-3.586	2.068	3.008	.083
	[SUCC_RATE = 2]	-2.749	2.044	1.808	.179
	[SUCC_RATE = 3]	-.486	1.972	.061	.805
	[SUCC_RATE = 4]	.462	1.998	.053	.817
Independent Variables	Proportion cheatgrass	-2.208	.980	5.076	.024
	LN(fire size)	-.563	.256	4.824	.028
	Airport driving time	.755	.551	1.882	.170
	Native seed use = false	2.217	.752	8.687	.003
	Fencing use = false	-.495	.800	.383	.536

Variables dropped from the model included elevation which was not significant and the sign on the coefficient was sensitive to model specification. We hypothesized a relationship between cheatgrass and elevation but the effect of this interaction term was minor in the model. Similarly, drill seed use and herbicide use, which were used on a small proportion of sites, did not appear necessary to the model. However, when we tested these variables, the coefficients on herbicide use and drill seed use were consistently negative (Table 12 shows positive coefficients for *not* using these practices), indicating their use was associated with lower success rates. If these results can be believed, they suggest that greater intensity of treatment was associated with lower levels of success. This result was not completely surprising given that managers were likely to expend more effort on more challenging sites.

Table 13. Summary of Restorability Model Results

Reduced Success (negative coefficients)	**Enhanced Success (positive coefficients)**
% cover cheatgrass Increasing fire size Native seed use Herbicide use Drill seeding use	Fencing use Driving distance from airport (remoteness) Elevation

The dummy variable used to test for differences between Idaho and Nevada was significant in some models, but the sign on the coefficient varied depending on which variables were included in the model. Therefore, even though Idaho had a greater number of failures (success = 0 or 1) and Nevada had more successes (success rating 4 or 5), their practices were sufficiently different that no clear pattern of greater success or failure emerged when different site and treatment factors were included in the model. In other words, depending on which treatment factors were being held constant, each state could be seen to have had relatively more success or failure. This difference likely stems from greater use of native seed in Idaho which tended to lower the overall success rate unless treatment factors were considered (Figure 15).

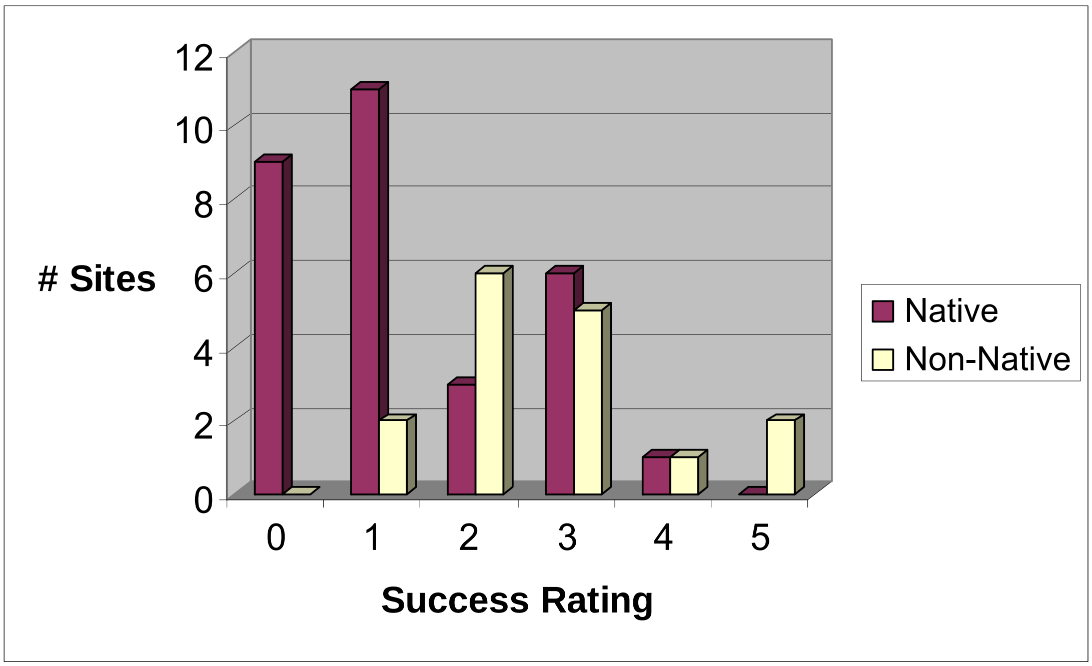

Figure 15. Seed Type (Native vs. Non-Native) and Project Success (2001-2003)
Use of any proportion of native seeds was associated with lower success rates compared to sites using entirely non-native seeds.

We did not consider the statistical modeling sufficiently robust to use the results to predict restorability. We also note that this model suffers from the same selection bias present in the treatment cost model. The selection bias resulted from managers being more likely to treat sites they thought had high probability of success. Therefore predictions made with the restorability model for a randomly selected site might overestimate the probability of success.

5.5.2 Operationalizing Restorability Results

Due to the limitations of the restoration database and the effect of those data limitations on statistical modeling, we did not apply the model results in our restorability estimates used in the cost-effectiveness and optimization modeling. Instead, in the interest of providing usable estimates of restorability, we focused on the apparent consensus among practitioners and researchers that the percent of cheatgrass pre-fire was one of the most important predictors of restoration success. For this variable, we were able to elicit probabilities of success from practitioners for multiple levels of the variable in order to hypothesize generalized response functions for 1) natural recovery and 2) restorability with treatment intensity. The example curves shown in Figure 16 represent expected responses of burned areas to recovery or treatment using the pre-fire cheatgrass cover as the only predictive variable. This approach of basing restorability on a single variable may tend to bias the overall level of success and including more variables could show that probabilities are higher or lower for different combinations of site conditions.

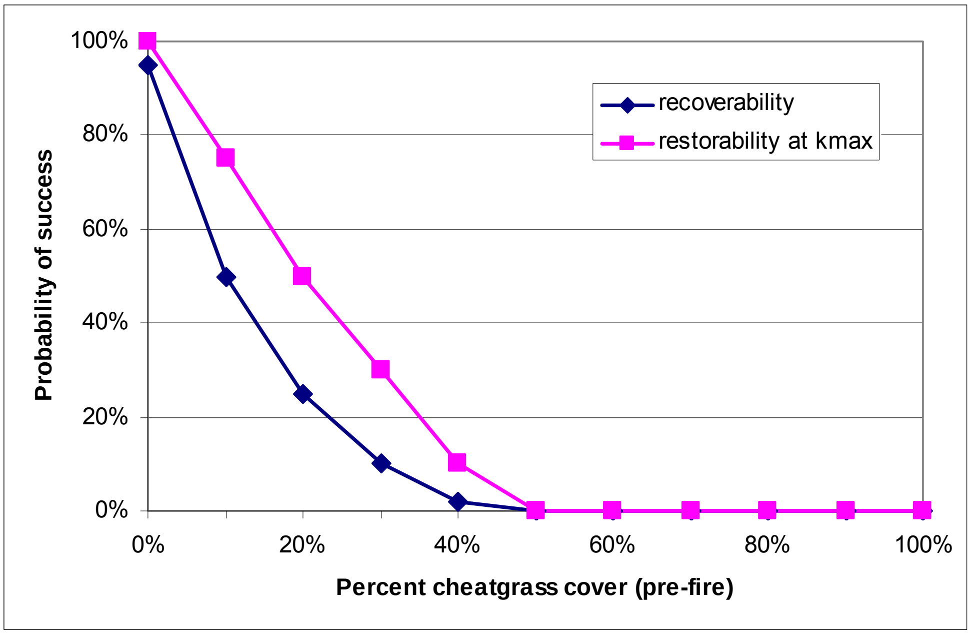

Figure 16. Hypothesized Relationships between % Cheatgrass Cover Pre-Fire and Natural Recovery or Restorability with Maximum Treatment Intensity

These relationships were developed through interviews with managers, a literature review and statistical exploration of a treatment success database. Restoration success was defined as establishing complete or near-complete cover of *desirable* plant species 2-3 years following the highest level of treatment (k_{max}) and restorability was the expected value of proportion of native cover. Desirable plants may include native or forage species. Note that this graph shows the contribution of only one factor of restorability; other factors were identified that might increase or decrease expected restorability at sites.

The curves created to judge response of sites to treatment were informed by our interviews with restoration practitioners, our literature review, and our statistical exploration of the treatment database, but were not a statistical fit of any data source. Therefore, these relationships should be used with caution, since they represent *hypothesized* relationships rather than those explicitly demonstrated through field studies. The weakest element of these response curves is that we had extremely limited information available to distinguish recoverability from treatment effectiveness and had to rely completely on opinions of practitioners. For treatment effectiveness, we used both available literature studies and best professional judgment of practitioners.

As shown in Table 14, the restorability relationships showed slightly increasing or a leveling off of probability of success as treatment intensity increased. These relationships would seem to contradict the results of the statistical analysis that suggested that increased intensity of treatment (use of drill seeding and herbicides) was associated with *lower* levels of success. However, we were able to rectify these seemingly conflicting findings by determining that managers will generally increase treatment intensity on sites they believe have lower recoverability. This reasonable approach to determining treatment has the unfortunate side effect of confounding treatment effectiveness and recoverability. The relative importance of site factors contributing to recovery vs. treatment effects will only be clarified by objective observation of treated and untreated sites under a variety of site and landscape conditions. The relationships we used here are based on the best, albeit weak, evidence available at this time.[5]

Table 14. Probability of Treatment Success by Treatment Intensity Given Percent Cheatgrass Cover Pre-Fire

Cheatgrass cover	Treatment Intensity				
	No Treatment	Aerial seeding only	Aerial seeding plus chaining	Aerial seeding, chaining & herbicide	Aerial seeding, chaining & drill-seeding
0%	95%	100%	100%	100%	100%
10%	50%	65%	75%	75%	75%
20%	25%	35%	40%	45%	50%
30%	10%	21%	24%	27%	30%
40%	2%	7%	8%	9%	10%
50%	0%	0%	0%	0%	0%
60%	0%	0%	0%	0%	0%
70%	0%	0%	0%	0%	0%
80%	0%	0%	0%	0%	0%
90%	0%	0%	0%	0%	0%
100%	0%	0%	0%	0%	0%

The restorability curves represented the proportion of treatments likely to result in "success." A specific definition of success was not available from our management partners, although it was clear that the goal was to prevent all cheatgrass growth where possible and achieve as low a percent cover of cheatgrass where complete prevention was not possible. Given this conceptual model of success, we used the restorability values we derived to generate the expected post-treatment cover of natives using this formulation:

$$\textit{Native cover (\%)} = 1 * S_{ik} \qquad \textit{(Eqn 16)}$$

where 1 represents the successful outcome (100% native cover) and S_{ik} is the probability of success. Therefore, the expected returns are expressed as percent native cover.

[5] Efforts are underway at BLM to improve reporting of treatments applied and success rates (Tom Roberts, pers. comm.)

Because of the high uncertainty associated with the restorability of sites, we eventually framed restorability as a stochastic variable when implemented in the optimization software. Using a Monte Carlo function, we defined the restorability as a value drawn randomly from a histogram probability function. The histogram probability was defined using a normal probability function and the estimated restorability at the mean. Making the substitution of the stochastic equation into Equation 16 yields:

$$\textit{Native cover (\%)} = 1 * \left[E(S_{ik}) = \sum_{z=1}^{6} p_z s_z \right] \qquad \textit{(Eqn 17)}$$

where $E(S_{ik})$ represents the expected value of the discrete random variable *s*. The probability of a value occurring within any range of restorability values (i.e., within bin *z*) is represented by p_z a value between 0 and 1, where the sum of the p_z's equals 1. Using Equation 17 to define restorability effectively generates a value for native cover for a site that represents an average site response assuming many implementations of the same treatment. This result was used to weight benefits of treatment as calculated in Equation 1.

5.6 Restorability Model Discussion

It was clearly a concern that the main variable used for assessing restorability was a variable for which we lacked spatial data. However, within a field office conducting such treatments, managers are able to develop this information for individual sites after fires. Ecologists are readily able to examine the site after a fire and determine pre-fire vegetation. Our estimate of regional cheatgrass cover was therefore only a place-holder value for proportion of neighborhood in cheatgrass, a variable that can be readily supplied through site visits. However, the lack of regional datasets on invasive cover is an impediment to conducting screening analyses or comprehensive regional planning for invasive management. In some areas (Peterson 2003), cheatgrass density has been mapped through remote sensing, demonstrating the possibility that such data could be developed comprehensively for other arid regions.

Perhaps the most important thing we learned from developing the restorability models was that treatments were only able to improve outcomes over a small range of site degradation. Sites with extremely low percentages of cheatgrass appeared to have the potential to recover on their own while sites with greater than 20% cheatgrass cover pre-fire had less than a 50:50 chance of recovery, even with the most intensive treatment. Pursuing treatment in most cases appeared highly unlikely to generate successful outcomes in most cases.

6. Optimization Modeling to Select Treatment Sites and Treatment Intensities

The models and information developed to measure benefits, costs and restorability of treatment options were implemented within an optimization framework to explore results and identify sets of options (site and treatment combinations) that could maximize various benefit measures subject to a budget constraint.

Steps for Conducting Simulation Optimization

Step 1	**Enter all factors** (inputs, outputs, weights, constraints, etc.),
Step 2	**Define all processes** (relationships among factors)
Step 3	**Identify which factors are uncertain** and their expected range and probability distributions.
Step 4	**Identify the target cell** to be maximized • Weighted sum of our four benefit indicators.
Step 5	**Identify adjustable cells** (control variables): • Weights on each benefit indicator (defined by the user) • Treatment intensity at each area (solved by the program)
Step 6	**Identify constraints** • "Hard" – budget • "Soft" – policy choices that limit the optimization
Step 7	**Assign a weight to each benefit** (user preferences)
Step 8	**Run the program and assess results**
Step 9	**Perform sensitivity tests** on assumptions, weights, etc.
Step 10	**Interpret results** and develop recommendations

6.1 Optimization Model Background

Optimization is a basic form of decision-making and, most generally, involves allocating scarce resources to meet competing objectives. Optimization may be as simple as choosing the shortest route to get from Point A to Point B in order to meet the objective of minimizing time and cost. However, optimization questions can become complex quickly as the number and range of resources and options for achieving objectives and, in some cases, the weights given to competing objectives increase.

In the application that will be described in the following section, for example, the optimization problem involves allocating limited dollars, labor, material, and equipment to different locations and sizes of burned areas in order to maximize the outcomes of cheatgrass preventative treatment. The objective being maximized in this case is a set of sometimes competing environmental and economic goals (e.g., improved wildlife habitat vs. improved livestock grazing). The optimization model was run using a variety of weights assigned to individual goals in order to test the sensitivity of results to preferences regarding potentially competing environmental and economic goals.

Initially, the optimization model specified to address this problem involved choosing one of five treatment intensities (including no treatment) to apply at each of 114 potential treatment sites, a problem with many trillion potential "optimal" solutions (actual number = 4.81 X 10^{79}). After an initial screening of options to exclude those that were clearly inferior in terms of minimizing costs or maximizing any specific environmental or economic objectives (based on very small fire size), the number of potential sites was reduced to 68. This screening reduced the number of potential "optimal" solutions, but the number remained in the trillions. Clearly this represents too many options to be compared in terms of expected costs and environmental and economic benefits without resorting to some kind of quantitative optimization method.

6.1.1 Quantitative Optimization Methods

Quantitative optimization methods are used routinely in commercial, industrial, and military applications to minimize costs or maximize performance in situations where objectives and options are too complex and/or too numerous to be assessed and compared in any other way (Optimization Online 2006). Such models are being used with increasing frequency to help prioritize and manage environmental conservation and restoration initiatives (Aravossis et al. 2006).

There are many different types of optimization methods (Troutman 2006, Papalambros and Wilde 2000, Nocedal and Wright 2006). They are usually categorized on the basis of whether they are: linear or non-linear, static or dynamic, constrained or unconstrained, and so on. The choice of the best optimization method depends on both the nature of the optimization problem being addressed and the availability of data to model the problem.

6.1.2 Optimization Techniques

The selection of the optimization method defines how the problem is characterized within a quantitative optimization program. Each optimization program, however, employs a particular optimization technique or algorithm for finding an optimal solution. These techniques can be very different and can yield very different optimal solutions. They fall into categories with names that reflect their general approach for finding the best option among many. These categories include hill climbing, ant colony, simplex, stochastic tunneling and particle swarm optimization techniques (Fox and Sengupta, 1969). Some techniques are based on what are called evolutionary algorithms, including the one used here, which is referred to as a "genetic algorithm" (Goldberg 1989, Rawlins 1991).

6.1.3 Genetic Algorithms

Genetic algorithms derive their name from the fact that they search for an optimal solution using the same kind of "survival of the fittest" simulations that were developed to trace and forecast the evolutionary development of biological populations. Using this approach, populations (options) are pitted against one another to determine which is the fittest (most optimal) with respect to certain desirable characteristics (objectives). Seemingly successful populations are then recombined, as in a mating process, and experience mutations and random variations that allow them to "evolve" into better solutions if possible. Genetic algorithms have advantages over other optimization techniques when considering large complex problems because they are relatively simple to program and can be more efficient at finding global optima. However, there is no guarantee that such algorithms will identify the optimal solution.

6.1.3.1 Applying Genetic Algorithms

Genetic algorithms can be applied to an optimization problem if it has two specific characteristics:

1. it must be possible to express the solution as a "string" of solution values, where a string is equivalent to a string of chromosomes in conventional genetics
2. it must be possible to calculate a value for each string in order to compare them with one another (where value is the contribution to an objective and is equivalent to the contribution to "fitness" in conventional evolutionary biology)

With the cheatgrass optimal restoration problem, we determined that these two conditions were met. The solution could be described as a "string" that consisted of the level of treatment effort applied to each of the treatment sites; and the value of each potential string could be measured in terms of the cumulative expected contribution of the string to a weighted sum of four benefit measures (our measure of fitness). The strings are equivalent to vectors of treatment choice.

Illustration

The cheatgrass optimization problem can be addressed using a genetic algorithm because it involves finding a string (more typically called a vector outside of this application) that consists of a level of treatment at each potential treatment site that maximizes the cumulative value of the weighted set of benefit indicator changes for each site. Table 15 illustrates the strings associated with a situation where there are only three potential treatment sites and only three potential treatment intensities.

Table 15. Illustration of Strings and Related Value Measures for Cheatgrass Optimization
(3 Treatment Sites, 3 Levels of Treatment, 3 Benefit Measures)

Combinations (Strings) of Potential Treatment Options

Site	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27
X	1	1	1	1	1	1	1	1	1	2	2	2	2	2	2	2	2	2	3	3	3	3	3	3	3	3	3
Y	1	1	1	2	2	2	3	3	3	1	1	1	2	2	2	3	3	3	1	1	1	2	2	2	3	3	3
Z	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3

The total benefit index for each string is the weighted sum of the individual benefit indicators associated with various outcomes (e.g., wildlife habitat improvement, livestock grazing improvements, or recreational benefits). Individual benefit indicators associated with each string will depend on site and landscape factors that determine the restorability of specific types of benefits at each site, the weights assigned to those benefits, and the intensity of treatment. (see Equation 1).

In this illustration, the total number of strings under consideration is 27. This is a far more manageable number than the trillions of strings under consideration in our actual cheatgrass case study. However, note that even in this limited illustration, examining the sensitivity of optimization results to changes in weights assigned to the three different measures of benefits that make up the cumulative "value measure" for each string could result in the need for thousands of program runs.

The next section deals with how the choice of an optimization program should also depend on the level of uncertainty about the values of critical parameters that link different treatment intensities at sites with different site conditions and landscape contexts to the various benefit measures. If the level of uncertainty regarding one or more parameters is significant, it may be beneficial to have their values selected from a probability distribution and generate a distribution of potential benefit estimates for each string rather than simply using average or typical parameter values to generate a single benefit measure for each string. If this approach is used, the number of program runs required to converge on an optimal solution, even in the limited illustration described above, could be in the hundreds of millions or billions. The number of iterations and the criteria for stopping the program from recombining and mutating solution values is determined by the user.

6.1.4 Dealing with Uncertainty

Besides being influenced by the significant number of potential treatment locations and intensity of treatment options, the selection of an optimization method to address the cheatgrass problem needed to take account of significant uncertainty about many of the parameter values and relationships that determine the links between levels of treatment at particular sites and various indicators of expected benefits. Typical optimization methods, whichever searching technique they employ, generate "optimal" solutions based on average or typical parameter values provided by the user. When there is significant uncertainty about the value of these parameters, relying on average or typical values can lead to sub-optimal solutions. In these situations, it may be far better to rely on the results of optimization programs that deal explicitly with uncertainty.

Traditional optimization programs, such as Evolver or Excel's built in Solver, require users to provide specific values for the parameters used to characterize the optimization problem (Frontline Systems 2006, Hallogram Publishing 2006). Some of them allow limited "brute force" approaches to testing sensitivity of results by running the model, adjusting some parameters, running the model again, and observing changes in results. This approach does not address the full range of uncertainty that exists in many situations, can be enormously time consuming, and makes it difficult for users to incorporate what they know about the range of uncertainty about each parameter into the search for an optimal solution. Fortunately, several new-generation optimization programs have been developed that combine simulation with optimization in ways that allow uncertainty to be incorporated directly into the selection of an optimal solution (Hegazy 2003, Pichitlamken and Nelson 2002).

6.1.5 Selection of "RISKOptimizer" Software

The particular optimization software that was selected for use in our research was RISKOptimizer (Palisades Corp. 2006), a program that combines genetic algorithm methods for optimization with Monte Carlo simulation to characterize uncertainty. This software is relatively easy to use since it is installed as an extension to Microsoft Excel spreadsheet software.

For characterizing uncertainty, the user has the ability to use one of many probability distributions to express uncertainty about parameter values, and to allow stochastic selection of parameters rather than being restricted to using deterministic values. For example, instead of assigning a value of 20 to a cell, the user can specify a triangular distribution where the expected value of the parameter ranges from 10 to 30, with a mean value of 20. As the genetic algorithm develops and examines "strings" that may be solutions, it selects values of this uncertain parameter based on this probability distribution along with values of all other uncertain parameters based on the uncertainty specified for them. In effect, the program initially selects the most "fit" strings of solutions based on the most probable combinations of parameters, and then uses the information provided about the uncertainty of each parameter to randomly change solution values until the likely values of the solution based on all combinations of parameter values begins to emerge.

6.2 Optimization Model Implementation

The optimization model was configured in the RISKOptimizer software using the framework discussed throughout this report and summarized in the inset box. The objective function was set to maximize the total change in benefit indicators by adjusting the levels of the treatment intensity for each burned area. [6] Five treatment intensities (including no treatment) were possible and two constraints were used.

[6] The default values for options controlling the behavior of the genetic algorithm were used: mutation rate, crossover rate, population size, random number seed, and sampling. We set the simulation stopping conditions to run 100 iterations before moving on to the next simulation.

Restoration Optimization Model

Maximize the *change in* benefits of multiple ecosystem services subject to a given budget constraint

Objective $B_t = \sum_i \sum_j w_j \left(b_{ij_{with}} - b_{ij_{without}}\right)$

i = Location i

j = Benefit j ($b_{recreational\ hunting}$, $b_{forage\ production}$ $b_{property\ protection}$, $b_{existence\ values}$)

t = 2-3 years following restoration

Control Variables

k = treatment level (0-4), the level of preventative/ restorative treatment provided to burned area i

w = weights assigned to benefits $\boldsymbol{j}$

Budget Constraint

Total Costs ≤ $1,000,000

$TC_{ik} = \sum_k FC + JC_k{*}JT_i + TCA_{ik}{*}A_i$

FC	= Fixed Costs
JC_k	= Journey (Travel) Cost, personnel and equipment
JT_i	= Journey Time (hours)
TCA_{ik}	= Treatment Cost per Acre
A_i	= area treated (acres)
k	= treatment intensity

Other Constraints

Hard Constraint: Number of Treated Fires ≤10 sites

To set up the decision, as one that paralleled that of the management agency, we used data for only one year, 2002. In that year, 114 fires were recorded on BLM lands within the Twin Falls District. We selected only the fires that burned one or more acres (excluding small fires) to ease computations and reflect likely management behavior. Then, we analyzed the necessary GIS data related to calculating benefits, costs and restorability and calculated risk-adjusted benefits and costs for the 68 burned areas that could be located in the GIS. Of these fires, four had been treated by BLM Twin Falls District in 2002. Other data indicated that a total of 8 fires had been treated by the district. However, some of these fires could not be matched to our spatial (GIS) database of fires and one fire was shown as having less than 1 acre area in the database we used for screening and was therefore screened out, although the treatment database showed higher acreage burned at this site.

The model was subject to two hard constraints: 1) the overall budget was limited to $1 million (based roughly on budgets of the two rehabilitation programs described in Section 4.3.1.1), and the number of burned areas that could be treated could not exceed 10. The constraint on treated sites was set up to reflect the limited equipment and time available to respond to fires. An average of about 10 burned areas were treated under the BAR and ESR programs, within the BLM Twin Falls district per year, based on our 3-year dataset. We tested the effect of the limit on fire sites treated by comparing results of optimization runs with and without the constraint, discussed below.

The benefits framework and optimization software were set up so that users could easily change the weightings applied to different types of benefits. Such weights were meant to be based on the relative importance given to different agency goals and to reflect necessary trade-offs. We initially applied an equal weighting to all services, in keeping with stated agency practices, and tested the sensitivity of results to this weighting by assigning all weight to each service, in turn.

6.3 Optimization Model Results and Discussion

6.3.1 Optimization Simulation Results Were Consistent Among Runs

We found almost complete consistency of results among optimization runs, leading us to conclude that the software performed well in defining the optimal choice set. Models were typically run for 20 hours (2-6 million simulations), by which time, model output showed that total benefits were improving by less than 1% between simulations. Although the choice of which sites to treat did not vary, we found treatment intensity levels varied somewhat across runs. Therefore, small differences in treatment intensity choices between model runs should not be viewed as significant differences.

We tested the sensitivity of the outcome to initial conditions and found that, given sufficient run time (~8 hours), model results were not sensitive to initial conditions or RISKOptimizer run parameters. The option to use Monte Carlo techniques to allow uncertainty of model parameters to be evaluated did not change the choice sets.

Model results showed consistent selection of certain treatments. The optimization typically resulted in the lowest intensity of treatment (aerial seeding only) being applied to the majority of sites selected for treatment. A modest number of sites were assigned the second level of treatment (seeding + chaining) and only a few small fires were assigned the most intensive drill-seeding treatment. The treatment involving herbicide, the third highest intensity, was never assigned.

6.3.2 Constraint on Number of Sites Treated Changed Average Size of Sites Selected

The constraint of treating 10 or fewer burned areas had a major effect on which sites were selected for treatment. When unlimited fires could be treated, the software chose to treat many fires, with a small average size (Figure 17, Table 16). With the 10-site constraint, the average burned area size increased, but total costs and benefits remained about the same (Figure 18, Table 16). This was not a major surprise since the main factor controlling site selection within the optimization program was the cost-effectiveness ratio, and both small and large burned areas can have similar cost-effectiveness. However, we had expected the economies of scale built into the cost model to make the larger fire sites more cost-effective and therefore be favored by the selection algorithm.

Optimization results generated with and without the 10-site constraint (weighting all benefits equally) showed no difference in average costs and average cost-effectiveness for optimization runs that treated 33 vs. 10 fires. Therefore, we concluded that the economies of scale were relatively minor and that other factors dominated the selection of burned areas to treat. This was confirmed when we plotted cost-effectiveness vs. size (Figure 19). While the plot shows that economies of scale are evident across the full range of fire sizes (small to large), the relationship over the range of small to mid-size fires is weak.

This constraint on maximum fires treated may have important implications for a regional strategy of cheatgrass control. Without the limitation on number of fires treated, the model selected many small fires to treat, probably reflecting a higher probability of success and higher potential benefits. However, benefits were not substantially greater when many small fires were treated, suggesting that more work is needed to evaluate trade-offs involved in treating many small fires. However, other research has shown that treating small infestations has a greater ability to control invasive spread (Moody and Mack 1988, Bangsund et al. 1996). Such work has shown greater effectiveness or cost-effectiveness of treating small sites to contain an infestation. In work that may have relevance to a regional treatment strategy, Bangsund et al. (1996) found an inverse relationship between leafy spurge infestation size and treatment payoff to ranchers when resilience of rangelands was considered.

Table 16. Comparison of Solution Sets without and with Limit on Number of Sites Treated

	No Limit on Fires Treated		Treatment limited to 10 fires	
Treatment Intensity	Number Fires per treatment level	Average Fire Size (acres)	Number Fires per treatment level	Average Fire Size (acres)
No Treatment	33	1,696	58	1,071
Aerial seeding only	19	3,138	6	7,194
Aerial seeding and chaining	3	17	3	3,179
Aerial seeding, chaining & herbicide	4	2	0	NA
Aerial seeding, chaining & drill-seeding	9	13	1	964

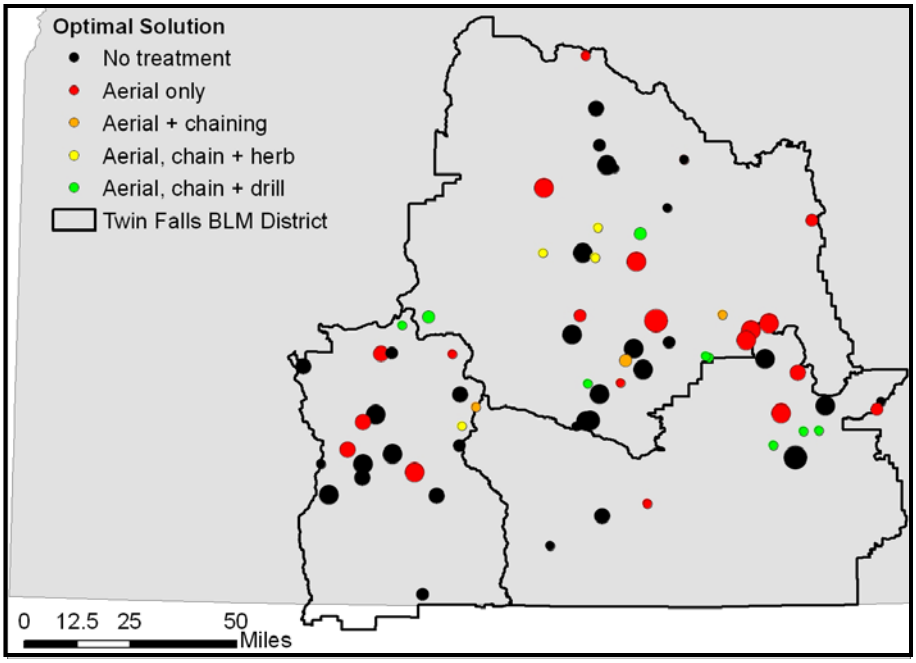

Figure 17. Optimization Model Solution with No Limit on Treated Fires, Equal Weight on All Benefit Categories)

Burned areas selected for rehabilitation by the optimization software are shown as colored dots. Black dots represented unselected sites. Dot radius represents relative fire size.

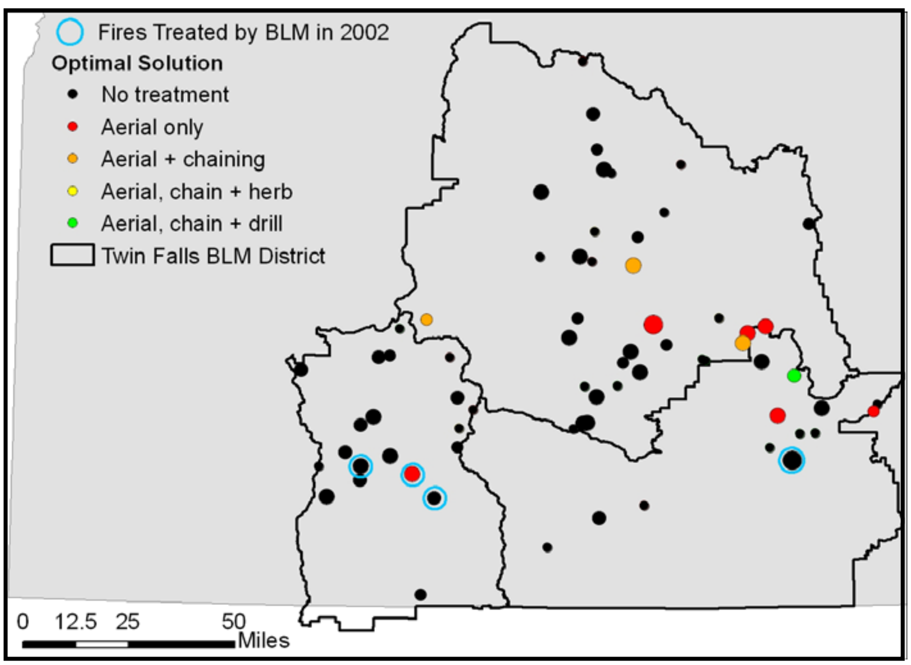

Figure 18. Agency Choices vs. Optimization Results with 10-Site Limit, Equal Weight on All Benefits)

Burned areas selected for rehabilitation by the optimization software are shown as colored dots. Black dots represent unselected sites. Burned areas treated by BLM in 2002 are circled in blue. (Only burned areas greater than 1 acre are shown. Not all sites treated by BLM are shown due to data limitations).

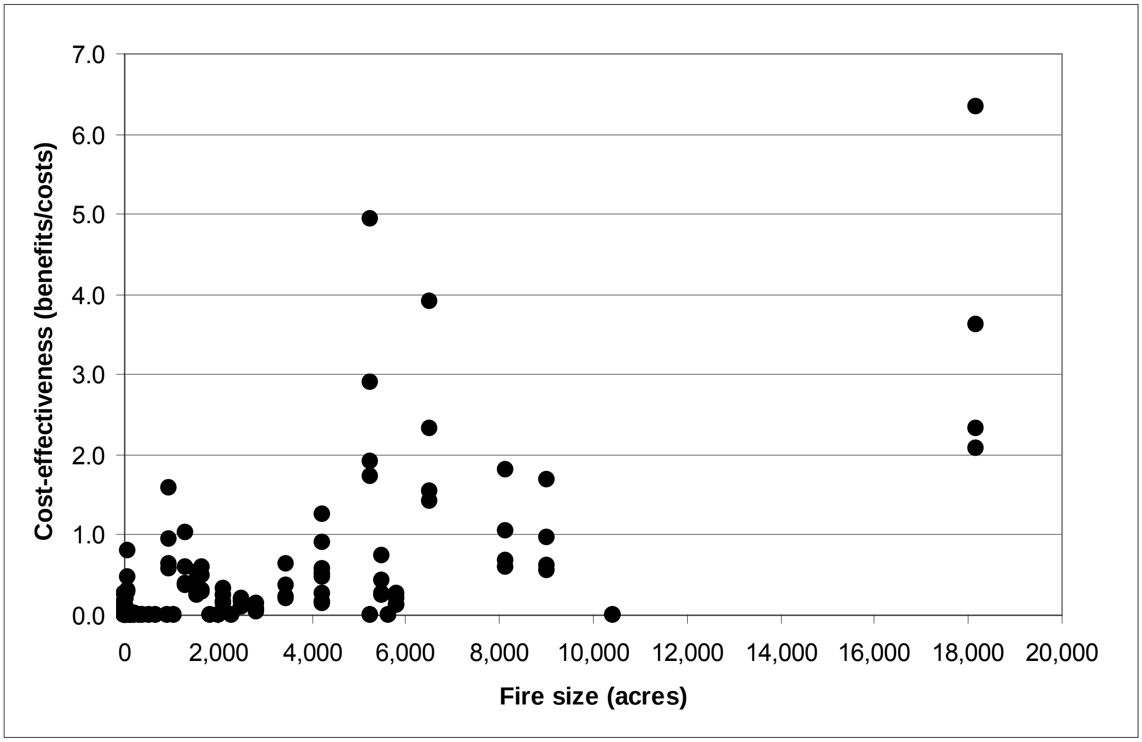

Figure 19. Cost-Effectiveness vs. Fire Size
Some economies of scale are evident since this graph shows a positive relationship between cost-effectiveness ratio and fire size. However, note that the few large fires drive the relationship and the economies of scale over the range of small to mid-size fires is weak.

6.3.3 Benefit Category Weightings Affected Site Selection and Overall Change in Benefit Indicators

The sensitivity of results (choices of sites and treatment intensities) to benefit category weightings was tested by conducting model simulations in which all weight was assigned sequentially to each of the four service ecosystem services. The results showed that the total change in benefit indicators was somewhat lower when forage production or property protection benefits were being maximized (Table 17). The specific site and treatment options selected did vary with different weighting schemes, as discussed in the next section.

6.3.4 Benefit Category Weightings Did Not Affect Proportion of Benefits by Service Category

Despite different weightings on benefit categories, the *proportion* of total benefits (as measured by change in indicator scores) achieved for any given benefit category did not differ substantially when weights were manipulated (Figure 20, Table 18). The relative proportion of total benefits realized in each of the four benefit categories remained roughly the same, regardless of which service benefit was being maximized. This result indicated that levels of benefits were highly correlated among the best site-treatment options. The bulk of the benefits, in all cases, were generated from existence values for sage-grouse and from recreational hunting benefits. The smallest proportion of benefits was derived from property protection.

6.3.4.1 Model Structure Strongly Influenced Ability to Produce Multiple Ecosystem Services at Sites

Before reviewing results of changes in benefit indicators in depth, it is important to clarify that aspects of the optimization model structure had the effect of minimizing the potential conflicts between some services. In particular, potential conflicts between forage production and services dependent on native habitat creation were minimized as a result of modeling choices. Conflict was minimized because of a simplifying assumption made regarding post-treatment vegetation cover. The change in benefits with treatment was based on the reduction in invasive species cover without specifying whether native grasses and shrubs were restored or whether non-native grasses were restored. In practice, conflict existed in this choice because native shrubs may be lower quality forage compared to non-native grasses that were commonly used in rehabilitation. Yet, sage-grouse existence benefits and antelope hunting benefits depended on the presence of shrubs.

Although failing to specify vegetation outcomes would seem to be a critical omission, it was difficult to avoid given available data and information. A mix of native and non-native species was typically used by this management agency, with the proportion of native seeds determined by initial site conditions. Sites with higher probabilities of success were given higher proportions of native seeds, up to 100%. However, native seeds had a much lower probability of successfully germinating. As a result, the ability to predict details of vegetation outcomes of restoration and understand the relative effects on benefits was limited.

With this model, we were able to distinguish the relative benefits derived from sites that differed in their likelihoods of successful treatment, as defined by preventing domination by cheatgrass. However, we were not able to model benefit indicator response to different mixes of native vs. non-native plants. To improve the model, information would be needed on the probability of different proportions of native vs. seeded non-native species establishing in response to different treatments. In addition information on how each service would respond to those different proportions of vegetation would also be needed, something we were not able to establish quantitatively. Therefore, our model structure glosses over some conflicts related to the specific vegetation established. In particular, the use of crested wheatgrass has been suggested to generate some ecological harms that we did not try to capture (Keeley 2004). Future efforts should endeavor to close this information gap since it is vital to understanding how services may compete when restoring sites.

Table 17. Optimization Solution Sets Summary for Different Ecosystem Service Weightings

Optimization Program Maximizing:	Total Change in Risk-Adjusted Benefits	Total Cost	Treatments Selected									
			No Treatment		Aerial seeding only		Aerial seeding and chaining		Aerial seeding, chaining, & herbicide		Aerial seeding, chaining & drill-seeding	
			# fires	Average fire size	# fires	Average fire size	# fires	Average fire size	# fires	Average fire size	# fires	Average fire size
All service categories equal	2,834,436	$999,969	58	1,071	6	7,193	3	3,179	0	NA	1	964
Recreational Hunting only	2,830,018	$996,835	58	1,072	6	7,194	2	4,733	0	NA	2	490
Forage Production only	2,670,331	$998,772	58	1,116	6	5,828	4	4,015	0	NA	0	NA
Property Protection only	2,554,258	$998,087	58	947	8	6,718	2	3,550	0	NA	0	NA
Existence Values only	2,885,233	$998,376	58	970	7	7,647	3	1,990	0	NA	0	NA

Table 18. Distribution of Benefits among Ecosystem Services when Service Weights Varied

Benefit Maximized	**Total Change in Benefits from Optimization Program Run**			
	Recreational Antelope Hunting	**Forage Production**	**Property Protection**	**Existence Values for Sage-Grouse**
All service categories equal	536,311	723,971	11,546	1,562,608
Recreational Hunting only	534,328	722,401	11,520	1,561,768
Forage Production only	463,879	777,673	9,180	1,419,599
Property Protection only	477,481	642,929	13,137	1,420,711
Existence Values only	522,091	724,743	11,596	1,626,803

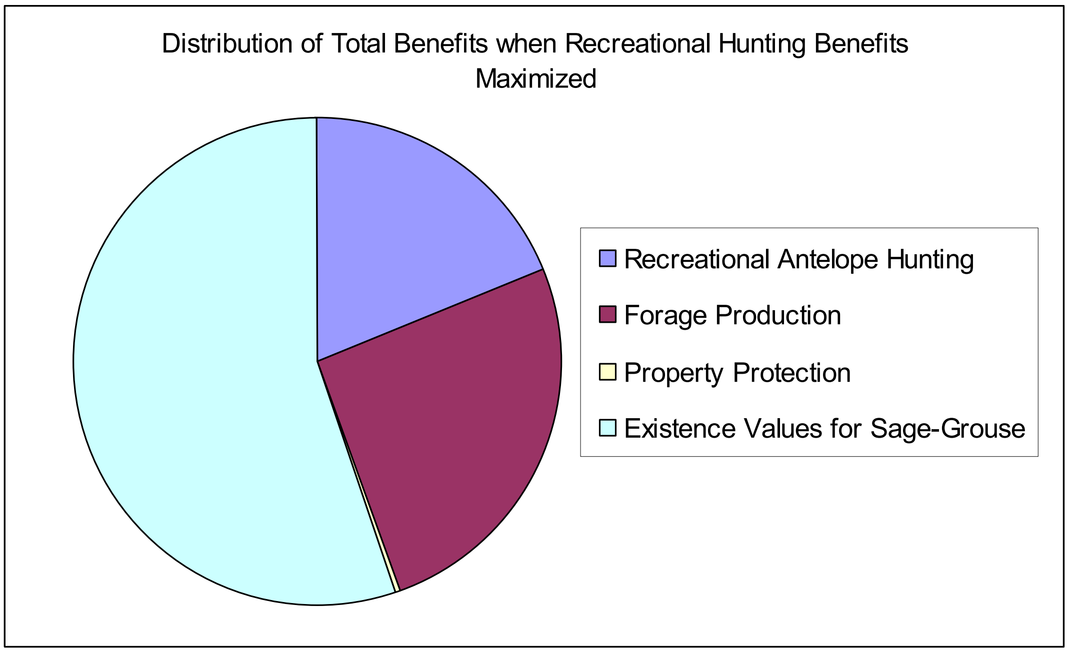

Figure 20. Typical Distribution of Benefits among Service Categories in Optimization Model Results
Figure shows resulting allocation of benefits among ecosystem service categories when all weight was given to Recreational Hunting in optimization model. This chart is representative of all model runs since the proportions of benefits by category varied only slightly for different benefit indicator weighting schemes.

6.3.4.2 Model Structure Influenced Proportions of Benefits Derived from Different Ecosystem Services

Several factors drove the relative proportion of the increases in ecosystem services. A primary factor was the scaling used to adjust service categories. Our scaling method assigned the maximum value of 100 to the highest score on the landscape and normalized all other values to that maximum value. This scaling method was sensitive to the shape of the distribution of benefit indicator values. The more consistent or even the raw scores were over the range of potential values, the greater the number of sites with high scores. Conversely, the more skewed the distribution towards high values, the fewer the sites that scored well. In the case of property protection, one fire was near an urban area, while the majority of sites were in remote locations. The urban site had the largest potential user population of fire protection services with 15,260 people within a 3-mile radius, whereas the mean score for all other sites was 252 people within the 3-mile radius. Therefore, when the maximum score of 100 was assigned to that urban location, which was an extreme outlier in the distribution, other sites were given proportionately low scores.

This result demonstrates how seemingly simple scaling rules can have a major influence on model results and may bias scores in unintended ways. In this case, we were not troubled by the model assigning a low score for remote sites for the benefit of property protection, since it accurately reflected the relative importance of preventing fire in an urban area vs. a remote area. However, managers should ensure that scoring rules match their priorities. Managers may want to score sites consistently across regions rather than just scaling values to the data at hand. This sensitivity to data distribution can be minimized by adopting alternative scaling rules or data transformations. However, all scaling choices reflect subjective choices that should be matched to management goals.

In addition to scaling, another factor that altered the relative responsiveness of benefits to treatment was the m parameter used in the equation relating benefits to change in native cover (Equation 4). Habitat was given the smallest m value, meaning it increased more slowly than other benefits as native cover increased. As a result, it did not level off as quickly as other benefits, making habitat more sensitive to changes in the ranges we were evaluating (80-95% native cover, see Figure 3). Conversely, the slope of forage production was small over the 80-90% range of native cover since the steep part of this response occurred at much lower levels of native vegetation. Unfortunately, this demonstrated that our results were quite sensitive to a parameter for which we had limited information.

Compromise Programming Framework for Scaling Benefit Indicators

Many alternative systems for scaling indicators exist. A class of scoring rules used in *Compromise Programming* (Zeleny 1973) creates scaling rules that can be used to quality-adjust the change in benefit indicators, such that changes in an indicator are evaluated in terms of the perceived benefits they produce. For example, changes of the same magnitude but at different points in the range of possible scores can be treated differently when they result in different levels of improvement. A net change of say 20 points, when it occurs close to an ideal score (e.g., moving from 80 to 100) might be given a larger weight than the same 20-point change when it occurs in a degraded site (e.g., a movement from 20 to 40). For our indicator scoring system, we did not explicitly weight the change in benefits in such a manner, rather, we used supplemental variables (landscape context C_i and restorability) to reflect the quality or importance of a change in a benefit index.

In a compromise programming framework, indicators are judged using ideal and anti-ideal sites. A change in benefit index score would be evaluated in terms of the distance a site moves towards an ideal site, measured in indicator state space, divided by the potential distance between the ideal and anti-ideal sites within the landscape. In other words, the deviation from the ideal is calculated as a percent of the total deviation observed in the landscape.

The advantages of such a system are that scores can be made consistent across sets of sites being evaluated and made consistent with management goals. In addition, by choosing an ideal site within the landscape being managed, goals are realistic with respect to limitations created by the landscape. A major disadvantage is that the ideal site may not be easy to identify or may not represent diverse viewpoints.

6.3.4.3 Level of Service Varied Spatially

Under different service weightings, the spatial patterns of treated sites differed noticeably between simulation runs (Figures 21 a-d) indicating that services were spatially heterogenous across the region. The sites chosen when habitat benefits were maximized included some more remote sites in the NW part of the region (Figure 21a). Whereas, when property protection benefits were maximized, the burned area in the SE that had been treated by the management agency was selected. However, some sites were common to all sets, despite different weightings on service benefit categories.

6.3.5 Agency Optimized Individual Services per Site While Optimization Model Optimized Multiple Services per Site

The comparison of optimization model results and agency choices demonstrated that more agency-treated sites were chosen (as part of multiple optimal sets) when full weight was assigned to a single service, than when all services were weighted equally. These results suggested that managers were choosing sites that produced the highest benefits for individual services and not evaluating the sites as producers of multiple services. Whether benefits could have been produced simultaneously at sites deserves further review since it was unclear whether current treatment practices result in services being generated simultaneously on sites. However, if benefits are not in conflict at some sites, our results suggest managers may be missing an opportunity to maximize benefits of spending by failing to choose sites that produce multiple services simultaneously.

The sole agency-treated site that was not selected by the optimization model either represented a problem with our data or model, or an opportunity to improve decisions. We suspected that some sites chosen by the agency might not fully reflect all costs or risks associated with sites, and this result bears that out. The treated site that was never selected for treatment in the optimization software scored high in terms of available benefits after treatment, but our data suggested that the site was in a location where it had a high likelihood to recover on its own. Therefore, the change in benefits with treatment, for this isolated site, would be negligible. However, our data on cheatgrass coverage and recoverability is admittedly weak. Therefore, further information would be needed to evaluate whether this decision to treat was inefficient, as the model results suggested, or whether it reflected knowledge not captured in the model.

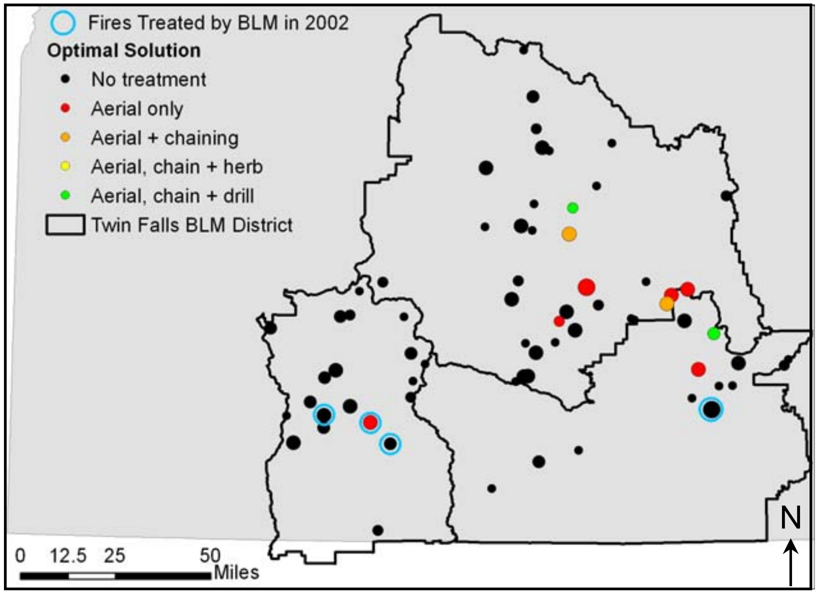

Figure 21a. Hunting benefits only.

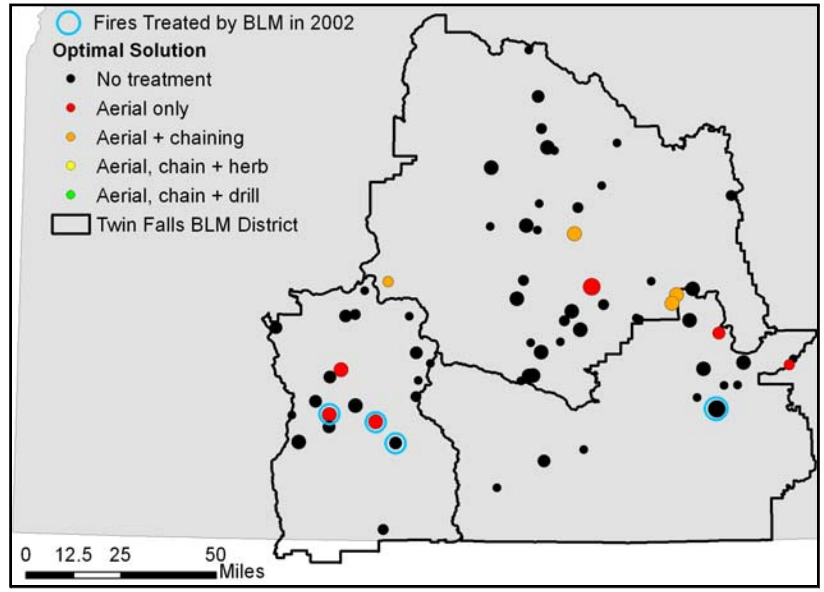

Figure 21b. Forage production benefits only.

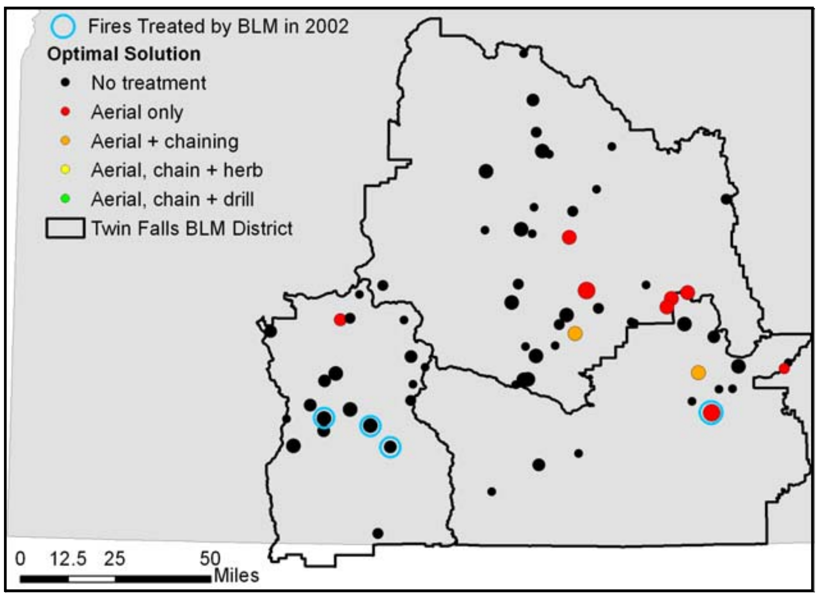

Figure 21c. Property protection benefits only.

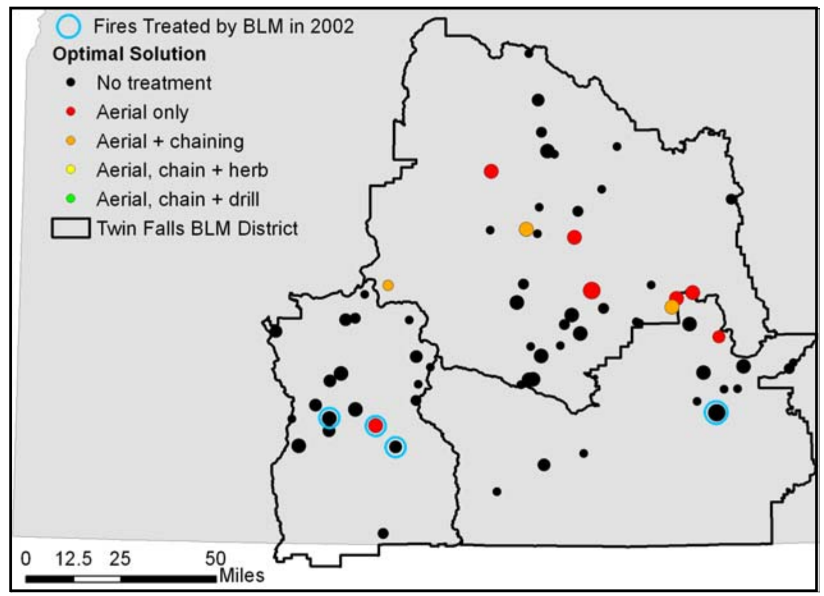

Figure 21d. Sage-grouse habitat benefits only.

Figure 21 a-d. Maps of Solution Sets from Optimization Model and Agency Choices in 2002
Colored dots represent model output; dot size represents fire size. Each map shows results when the specified benefit category was given full weight. Four of the agency selected sites are shown circled in blue. Note that all but one agency-treated site (the southernmost blue circle) was chosen in one or more model solution sets. Not all agency-treated sites could be evaluated due to data limitations.

6.3.6 Optimization Results vs. Agency Selections: Model Generated Higher Benefits by Selecting Options that Produced Multiple Services

A major question we addressed was, "Did the optimization model produce results different from those of the management agency?" We expected similar results for three reasons: 1) we constructed much of the benefits assessment framework to reflect manager conceptions of important environmental services produced on the landscape. 2) we used management opinions about restorability to inform our risk-weighting of benefits and 3) we used constraints that reflected those of the management agency.

On the other hand, we had reason to expect our results would differ for several reasons. First, we were not certain that managers, in their decisions, had captured the indirect costs of treating sites, such as those incurred in travel to sites and supervising contractors. We were able to include part of, but not all of, those costs in our model. Second, we used a maximum of 10 treated sites, which was slightly higher than the five mapped fires observed to have been treated in the year we compared.[7] Finally, we were not certain that the GIS data were adequately reflecting the detailed site-based information available to managers. For these reasons, we expected our results might deviate from management choices.

In a plot of benefits vs. costs, we identified that the choice sets of the optimization model and the management agency were not dramatically different in their cost-effectiveness (Figure 22). For both the optimization model and the agency, the selected sites generally fell close to or on the cost-efficient frontier. However, one point chosen by the management agency shows apparently low benefits for modest costs and is far from the efficient frontier. For the optimization model, points chosen that were close to but off of the efficient frontier appeared to reflect trade-offs made by the software to maximize benefits without exceeding the budget although it is also possible the software did not identify the most optimal set.

[7] We ran the optimization with a constraint of a maximum of 5 sites and found that the 5 sites selected were consistently captured within the set of 10 sites selected under the 10-site constraint. This result indicated that even if the agency were required to choose fewer sites, those sites should have been included in the set of 10 sites.

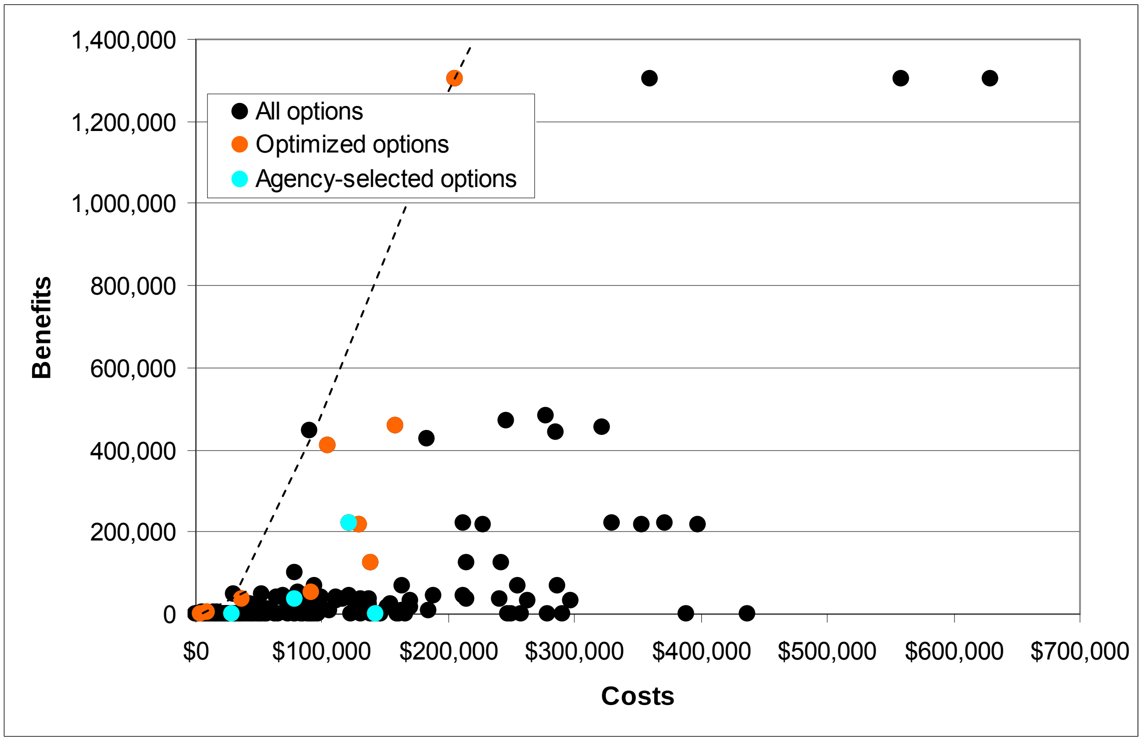

Figure 22. Risk-Weighted Benefits vs. Costs of Model and Agency Selected Options
The most cost-efficient sites are those along the cost-efficient frontier (dashed line). Sites in the bottom left are low cost–low benefits while sites in the upper left are low cost–high benefit.

When we compared the change in benefit indicators for the four agency-selected sites for which we had data and the top five sites chosen by the model, we found that the total benefits were substantially higher for the optimization model set than for the agency set. Such results should be viewed with caution due to the potential for inaccuracies of our model or the inability of the landscape data and screening-level analysis to capture the site-based information available to the agency. However, these results, and the results discussed in the previous section, suggest that the agency may be able to generate higher levels of ecosystem services through their restoration choices by selecting sites that can provide multiple services well, compared to selecting sites that provide a particular service well.

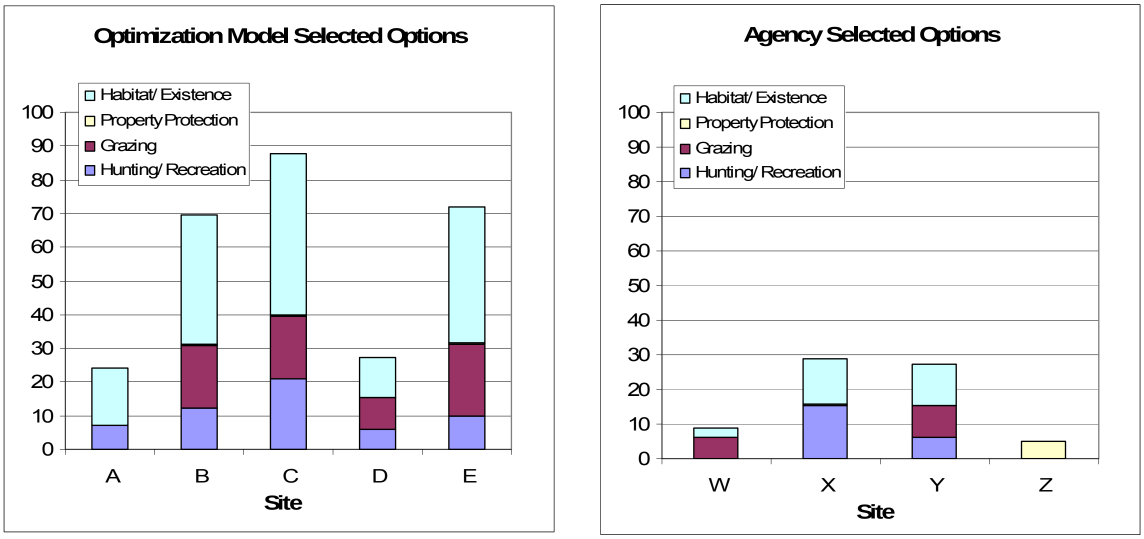

Figure 23. Comparison of Per Acre Benefits for Options Selected by Optimization Model and Agency
Bar graphs show the change in the bundles of ecosystem services produced by sites as a result of the selected treatment. The change in benefit indicators for each site is shown as the per-acre change in indicators to remove the effect of fire size. Note that all but one of the sites chosen with the optimization model generated substantial benefits across all categories of service while benefits of agency selections are more likely to be concentrated in a single service. Site D in the left side graph and site Y in the right-side graph are the same site. At least one burned area treated by the agency was not able to be represented by available data, so this graph does not show all fires treated by the agency.

6.3.7 Caveats on Scope of Analysis

Through this analysis that focused on site and neighborhood-based cost-effectiveness, we did not examine some of the regional trade-offs inherent in site selection. Our model constraints led the model to choose large rather than small fires, but this binding constraint needs to be examined further to determine how it fits into a strategy of regional control of cheatgrass. We did not explicitly examine how different types of preventative treatments in unburned areas might be used in a strategy to manipulate fire regimes (e.g., by creating regional fire breaks) and thereby create similar types of benefits to direct rehabilitation of burned areas. Also, we did not consider the potential harms of treatment, which are just beginning to be understood. These risks include the potential for treatment to facilitate invasion by even more harmful species than cheatgrass (Keeley 2006). In addition, we did not examine trade-offs having to do with the use of alternative methods of risk reduction such as providing incentives for homeowners to build in different locations or with different materials. Such issues are important to identifying optimal solutions for how to allocate funds among programs but are typically beyond the scope of analyses conducted by management agencies.

6.3.8 How Does Scale of Analysis Affect Perceived Benefits?

Because benefits of different services accrue to different sets of users, benefit weights attach relative importance to different scales of analysis. Through our analysis methods for devising indicators to measure relative benefits, we effectively assigned a representative scale of user benefits. We were interested to see whether total benefits differed when the scale of services being weighted changed. Specifically, forage production benefits were evaluated in terms of site qualities only, and thus relative benefits were measured at the plot scale. These forage-production benefits would be realized primarily by the lease-holder. Next in increasing scale, was the property protection service which benefited a relatively small area of nearby residential landowners (3-mile radius). At a broader scale, hunting benefits applied to a potential user group within a 1.5-hour drive (~75 miles).

For existence values, we did not have an indicator that explicitly identified the human population receiving benefits. Rather, we developed the index to reflect qualities of the landscape that were important to sage-grouse, intending that factors supported long-term sage-grouse survival would reflect the relative benefits to those who value sage-grouse. Sage-grouse are not just affected by site conditions, but also the landscape context in which sites exist. Therefore, the landscape indicators were used to assess aspects of habitat quality over a 10,000 hectare area surrounding each burned area. However, implicit in our indicator selection was the concept that the location of the existence service did not affect the values people hold for that service, as long as habitat quality was held constant. Since the service was defined as a non-use service, we did not need to weight the service by accessibility by people.

Although an existence (non-use) service should imply that all sage-grouse populations are substitutable, we did not think the beneficiaries within the management area would agree. Conceptually, we envisioned the user group as being the affected population within the management district, since maintaining this characteristic species *locally* was a clear goal of the local management agency. Other populations of sage-grouse throughout the Intermountain West may be substitutes for beneficiaries outside the region. Therefore, the main beneficiaries of this service were the local residents, at least until the population of sage-grouse becomes sufficiently rare that all habitat is seen as critical to its survival.

By revisiting the optimization results generated when different service weights were applied, we can see that scale of analysis did have an effect on which sites were chosen for treatment and the relative level of benefits, as measured by our system of benefit indicators. Sites that maximized forage production benefits (site scale) were different from sites that maximized property protection or hunting benefits. However, sets had some overlap regardless of which service benefit was being maximized, suggesting scale of analysis would affect treatment priorities to some degree. As would be expected, the benefits were seen to increase as more services (increasing scales) were included in the analysis.

7. Conclusions

Our goals with this project were to 1) demonstrate how economic principles could be incorporated into an indicator-based framework used for assessing and comparing the social benefits of invasive species treatment alternatives; 2) evaluate whether and how optimal treatment decisions would differ as benefits of different ecosystem services were given different weight; 3) evaluate effects on treatment decisions of explicitly accounting for spatial heterogeneity of costs, benefits, and risks; 4) evaluate whether existing data and knowledge were adequate to create practical quantitative decision tools; and 5) assess the potential usefulness to managers of modern quantitative optimization techniques for choosing optimal sets of treatment options. In developing these tools, the overarching goal was to keep methods accessible to managers and to fit economic tools to current decision support tools, such as GIS databases and analysis, in order to encourage the use of economic concepts in decisions made by non-economists. The incorporation of economic principles, even imperfectly, has the potential to generate recommended treatment regimes that compare favorably with or perhaps improve upon ad hoc decisions.

We used a case study of one of the best studied invasive species in North America, cheatgrass (*Bromus tectorum*). After reviewing much of the available information about this species, we developed models to: 1) measure indicators of relative social benefits for four ecosystem services by location, 2) estimate treatment costs by location, and 3) estimate restorability of ecosystem services by location. Then we combined our site-specific measures of costs, benefits, and restorability within an optimization framework to generate optimal sets of treated areas and treatment intensities that maximized a weighted sum of social benefit subject to a fixed budget constraint.

We worked closely with management partners from federal agencies to capture their knowledge about the problem and to incorporate their treatment goals into our analysis framework. We further informed the optimization framework with the best available research and data, although we found that even for cheatgrass, one of the most-studied invasives in the United States, a great deal of information was lacking. As a result, the model relied more heavily on ecological and economic first principles and best professional judgment of managers, scientists and our research team than was initially planned. This suggests that until databases improve, developers of decision-support models for less-studied invasive species should probably focus as much attention on methods of eliciting parameter and indicator values from experts as finding them in the literature or estimating them from data.

7.1 Management Implications

Despite the many concerns and caveats we raised on the difficulty of developing an objective evaluation of benefits, costs and risks, the screening method we developed, based on the best information available, was able to largely replicate choices made by managers who had detailed site-based information. This result suggests that current data and knowledge are sufficient to capture the decision process in a standardized framework that can be opened up to public and scientific input. Since we were not able to base the framework on completely objective information, it seems particularly important that the approach we developed reveals the underlying assumptions, values and tradeoffs associated with agency decisions and enables researchers and stakeholders to understand how their input can and is being used.

A major distinction between the results of the optimization model and the agency choices was that when we gave benefit categories equal weight, in keeping with stated agency practices, the selected set had only one treatment site in common with the agency-selected set of sites. However, when we gave full weight to individual benefits sequentially, and compared all optimization model selections to agency selections, all but one site was common to both sets. This indicated that managers picked sites to treat by choosing those that maximized a particular service benefit (e.g., the best habitat site, the best property protection site) rather than choosing sites that jointly produced high levels of multiple services. Because the agency selected a different set of sites than the optimization model, when all benefits were weighted equally, our model suggests there is an opportunity cost of choosing sites as producers of individual services rather than as producers of joint services.

Some questions remain regarding the ability of services to be simultaneously provided at all sites. Several simplifying assumptions we made in our analysis framework to accommodate data constraints, may have minimized the potential conflict between forage production and sage-grouse habitat or antelope hunting services. The agency choices could be more accurately reflecting conflicts that our framework was not able to reveal. However, even if only some sites can be managed to produce multiple services, our results suggest it may be possible to generate a higher level of benefits by considering joint production of ecosystem services where they can be achieved.

7.1.1 Summary of Lessons Learned:

4. The ecosystem service benefits generated from treating cheatgrass might be improved by viewing sites as simultaneous producers of multiple services rather than selecting sites that maximize only one service benefit.
5. Analysis details are important and relevant to managers, so a screening analysis must incorporate at least an intermediate level of detail to be acceptable to managers, but also must make simplifying assumptions to remain tractable.
6. Benefit response functions to invasive species are not well documented; but screening-level economic analysis based on first principles of ecology and economics can duplicate analyses based on site-specific information
7. Data are sufficient to perform screening level analysis, but results are sensitive to the functional form of benefits, which is not well-constrained
8. Weights on benefits are only critical when benefits compete; when they do, weights can significantly change the optimal allocation of treatment effort and treatment intensities
9. Decision-support tools that follow sound economic principles and reveal underlying assumptions and value judgments provide a basis for both expert and stakeholder involvement in decision-making and promote cost- and risk-conscious solutions

7.1.2 Use of Cost-effectiveness Analysis and Optimization Models

The practical use of optimization models to select high priority preservation or restoration areas is a hotly debated topic because such models typically have difficulty capturing the real-life constraints of manager's decision problem (Drechsler 2005). A significant problem, in the case of static models, is that the dynamics of making selections are ignored. In other words, once an area has been selected for restoration, the goals for and potential benefits of making other selections change. However, a static model cannot incorporate these dynamics. More complex dynamic models have the potential to overcome this problem, but they also present significant data and analytical challenges and frequently some level of detail must be forfeited when developing a dynamic model.

The benefits of using a cost-effectiveness and optimization framework are that even when the models rely in part on best professional judgment, they expose assumptions, reveal the influence of different assumptions, and allow competing assumptions to be compared on the basis of various benefit measures, cost-effectiveness, or some combination. The drawbacks are that it is difficult to support the generalizations needed to allow many sites to be screened.

In our attempts to build a decision support framework with an intermediate level of complexity, we quickly found that we could not find enough hard evidence to support our intended simplifications. It was our conclusion that the goal of creating a screening tool to assist managers and potentially increase the benefits that can be derived from restoration programs is severely hindered by the lack of relationships to characterize system response to change. Even with all the simplifying assumptions we made, the decision support tool we eventually developed turned into a fairly complicated evaluation technique based on many weakly supported assumptions. This reflects both the messy nature of natural resource decisions and the poor quality of information available to characterize how people derive benefits from natural assets.

Clearly, much work remains to specify functional relationships that represent change in ecosystem service levels as a function of invasive cover that are also tailored to specific services. We used a stylized relationship to suggest a likely relationship, however, the functions do not represent quantitative relationships developed from observed relationships. We may never be able to achieve such an ideal for these relationships given the complex nature of ecosystems. In which case, we should continue to test the sensitivity of our management choices to our assumptions of such relationships.

Our framework appears to have succeeding in representing most of the factors that managers consider when setting restoration priorities. In this way, our framework can be used to focus public debate over how public lands are managed and can make it easier to incorporate new input on the scientific and economic basis of such decisions. Agencies that are struggling with such decisions may find the experiences described here, with respect to cheatgrass in our study area, to be helpful for informing their choices.

The research described in this report demonstrated that it is possible to develop useful decision support tools to help assess and compare invasive species treatment options. However, it also demonstrated the difficulties inherent in making prioritization decisions in this area, and the difficulties of capturing the multi-faceted aspects of social value and their ecosystem linkages in a single quantitative decision-support framework. To develop models that are more rigorous and generate more credible results, future efforts of this kind will need to be well-supported so that managers and decision support tool developers have the time to review existing scientific information together, elicit best professional judgment, and create decision-support models that match the level of knowledge and amount of data that are available. The plans of BLM and USGS to improve invasive species monitoring databases will greatly increase the tractability of creating these types of decision-support frameworks. However, we still have a long way to go before we will have the capacity to capture, test, and properly interpret the accumulated knowledge about particular invasive species problems, and to then incorporate them into a reliable and generally applicable decision support tool.

7.2 Recommendations for Further Study

7.2.1 Ecological Information Required to Improve Risk Analysis

Improve analysis of enhanced risk of fire due to cheatgrass/ annual grass cover (see work by D. Pyke and others)

Continue research to establish efficacy of treatments by evaluating sites in a range of environmental conditions (see work by N. Shaw and others)

Evaluate whether treatments provide opportunities for new invasives to establish

Untreated control sites should be monitored long-term to evaluate natural recoverability and establish dynamics of multiple invasive species

Develop spatial time-series data of cheatgrass cover to allow better risk analysis and improve regional strategic planning

7.2.2 Economic Research Needs

Although we demonstrated that a static analyses can incorporate concepts of benefit persistence, further research is needed to develop methods that adjust benefit measures to changing regional losses of ecosystem services

Dynamic models are required to directly assess the value of increasing fire-free interval as a result of cheatgrass treatment

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