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Methods

Can the concept of ecosystem services be practically applied to improve natural resource management decisions?

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A R T I C L E I N F O

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A B S T R A C T

Applying ecosystem service valuation principles to natural resources management has the potential to encourage the efficient use of resources, but can decision support systems built on these principles be made both practical and robust? The limitations to building such systems are the practical limits on managers' time to develop or learn tools and the state of the science to support decision-making components. We address this question by applying a cost-effectiveness analysis framework and optimization model to support the targeting of restoration funds to control an invasive grass (*Bromus tectorum*) in agro-ecosystems. The optimization aims to maximize benefits derived from a suite of ecosystem services that may be enhanced through site restoration. The model combines a spatially-varying cost function with ecosystem service benefit functions that are risk-adjusted to capture the probability of successful restoration. We demonstrate that our approach generates roughly three times the level of ecosystem service benefits (as measured through indicators) compared to the current management strategy of selecting restoration sites that are superlative producers of one ecosystem service. The results showed that spatial (GIS) data and ecosystem understanding were sufficient to formally capture the managers' informal decisions and that cost-effectiveness of restoration could be improved by considering the ability of sites to jointly produce multiple ecosystem services and adjusting expected benefits by the probability of success.

1. Introduction

1.1. Use of Ecosystem Services in Decision-making

The ecosystem services framework is promoted as an approach capable of integrating ecological and economic outcomes in a manner useful for making tradeoffs in natural resource management (Barbier, 2007; Carpenter et al., 2006, NRC, 2005; Polasky, 2008; Tallis et al., 2009; Wainger and Boyd, 2009). Ecosystem services are variously defined, but here we define them as the benefits derived from nature for which people can express preferences that allow tradeoffs to be evaluated (see Wainger et al., 2001 for further discussion). This definition primarily distinguishes valued end uses (e.g., preventing flood damage) from the ecosystem processes from which they are derived (e.g., hydrologic regulation). The ultimate goal of using ecosystem services is to balance competing interests when deciding how best to manage and allocate natural resources. Yet, in applying an ecosystem services approach, particularly to local natural resource management, the devil, truly, is in the details.

The management decisions potentially aided by the ability to tally changes in ecosystem services include a range of land and resource management choices, including prioritizing investments in restoration, land set-asides, judging equivalency of market credits, or conservation easements. To date, the question of how to choose among land protection options has received the most attention (see Egoh et al., 2007 for review), but the approach is equally appropriate for targeting incentives for best management practices, controlling invasive species, or other management choices that can co-occur with developed uses of the land or water. Yet, despite development of many such tools, examples of application by public land managers are rare (Newburn et al., 2005).

1.2. Invasive Species Management Decisions

The ecosystem services approach has particular relevance to managers of public rangelands who are increasingly urged to manage these systems as "agro-ecosystems," i.e., create integrated systems that produce agricultural commodities, while simultaneously providing ecosystem services for other services, including those derived from wildlife habitat, hydrology, and fire management (Fischer et al., 2008; Maresch et al., 2008; Swinton et al., 2006). Rangeland managers make a variety of decisions that affect the production of ecosystem services on public lands, but a growing concern is the management of

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invasive plant species. Rangelands, grasslands and pasture, which comprise 26% of US land area (USDA ERS, 2009), are increasingly under threat from non-native invasive plants that harm the ability of lands to support agricultural production and habitat goals alike (DiTomaso, 2000; D'Antonio and Vitousek, 1992; Mack, 1981). According to the Bureau of Land Management (BLM) and the US Forest Service (2009), "Each year, the BLM inventories over 10 million acres [of rangeland] for invasive and noxious weeds; treats weeds on over 300,000 acres; and monitors and evaluates nearly 500,000 acres of weed treatments." This spending represents a major public expenditure and managers of such funds can benefit from a streamlined approach to prioritising response options.

Because invasive species control is expensive and because ecosystem service benefits can conflict as invasive species are controlled, a decision framework, either formal or informal, can help to assess tradeoffs and use allocated funds efficiently to promote public welfare. While cost-effectiveness analysis is used widely in allocation decisions, restoration choices are often *ad-hoc* and data-limited (e.g., Palmer et al., 2007). Where ecosystem service decision-support systems have been created, they typically capture a great deal of the understanding of environmental features and processes, but may miss information on costs and risks that can inform tradeoffs between different types of services (e.g., ICBEMP, 2000; Chan et al., 2006). Costs are typically estimated in ways that ignore important differences between sites (e.g., Denne, 1988), and performance risks are often ignored completely. Few systems capture the probability that the restoration will succeed in restoring ecosystem services, yet, as we will show, this short-coming can profoundly affect expected benefits for invasive species management. This inattention to costs and risks of different actions can prevent agencies from getting the most "bang for the buck" when allocating funds to land purchase or restoration.

Cost-effectiveness can form the basis of a formal decision framework, however, many difficulties plague the application of such a framework, including the time required to bring current research and cost-accounting to bear on the decision process and the lack of ecological and economic information. Shortfalls in the ecological research include a lack of quantification of the damage inflicted by invasive species, beyond a few isolated ecosystem services. Shortfalls on the economic side include the lack of quantification of public values and tolerance for irreversible loss of a sufficiently broad range of ecosystem services to inform tradeoffs. Yet, managers make decisions with incomplete information because waiting for complete information is deemed risky, given the ability of invasive species to spread and irreversibly alter ecosystems (e.g., Brooks et al., 2004).

A decision support system cannot overcome all these challenges, but it does present an opportunity to apply the best available data and test effects of uncertainty in order to promote efficient use of limited resources. A key challenge for such systems, given data gaps, is capturing the benefits of enhancing or restoring ecosystem services (see Barton et al., 2009 for discussion). Either monetary (e.g., Holmes et al., 2004; Loomis et al., 2000) or non-monetary metrics (e.g., Boyd and Wainger, 2002; van Wilgen et al., 2008) can be used in an optimization framework aimed at achieving cost-effectiveness, as long as the non-monetary metrics can be meaningfully aggregated across the bundle of services being maximized. However, only monetary metrics can be used to demonstrate net social benefits of an invasive species control strategy, as illustrated in some dynamic optimization models (e.g., Cook et al., 2007; Higgins et al., 1997; Leung et al., 2002; Settle and Shogren, 2002).

1.3. Monetary vs. Relative Measures of Ecosystem Services

Models that monetize benefits of invasive control may be ideal for comparing management strategies, but they are not necessarily useful for prioritizing restoration on a site-by-site basis. Much of the historic valuation literature has failed to meaningfully capture ecosystem functional quality, suggesting that site-specific monetary values (most likely derived through benefit transfer) may not accurately reflect the ecological quality or level of service provided by each site (Spash and Vatn, 2006; Ready and Navrud, 2005). If the monetary value does not capture the ecological productivity of the site when it is required for generating a service, then that value fails to distinguish highly functional parcels (e.g., ones that provide scarce habitat, flood control, etc.) from low functioning parcels (e.g., ones that provide little habitat value and flood control), and can ultimately result in the environment being degraded if such values are used in offsets or trading.

For many decisions, particularly those to avoid, minimize or mitigate harm, the most important information is the *relative* value of services, which can be well-represented with non-monetary metrics. However, to be useful in decision-making, relative benefit indicators must nonetheless capture details relevant in the decision context and focus on changes in human welfare, not just ecological changes (Heal, 2000; Wainger et al., 2001). The metrics for assessing benefits of an ecosystem service should encapsulate the characteristics that would inform a purchase, if ecosystem goods were bought and sold. Namely, metrics should capture how much users need or want that service, how much it costs to access the service, how easily the service may be substituted or replaced, and how reliable that service is over the long term.

1.4. Applying Multi-objective Optimization

Multi-objective optimization (multi-criteria decision analysis and multi-attribute utility analysis) is an increasingly common approach to applying non-monetary metrics (or a mix of monetary and non-monetary metrics) in resource use tradeoffs (see Kiker et al., 2005 for review). The basic optimization approach maximizes the production of a weighted set of objectives, subject to one or more constraints, such as a budget. For example, the objective may be to maximize the change in the benefits derived from a bundle of ecosystem services produced with a given management action. Benefits associated with different services are weighted to reflect their ability to meet management goals, and the weighted sum of benefits is maximized in the optimization algorithm. The ability to use a variety of metrics typically allows a broader set of ecosystem services to be considered and overcomes the problem of having to apply controversial monetary metrics to non-use services that may be better captured through ecological indicators. Furthermore, the multi-objective optimization model is appealing for guiding management decisions because of its ability to mimic real-world goals and constraints and integrate relatively complex information and tradeoffs.

Multi-objective decision methods have been used for decades (Keeney and Raiffa, 1976; Clemen, 1996) but, more recently, have been expanded to include spatially-detailed approaches that apply GIS tools and modelling environments to compare site-specific management actions for their ability to induce systems to produce multiple ecosystem services. Decision support tools, some of which are optimization models, link management actions to outcomes and prioritize the actions that maximize ecosystem services. Many models are produced (e.g., National Oceanographic and Atmospheric Association's Habitat Priority Planner, NatureServe's Vista) but not necessarily published in peer-reviewed literature, making it difficult to summarize the state of the art.

Among spatial decision support models, few apply cost-effectiveness analysis (CEA) or return on investment (ROI) approaches to capture potential efficiency of spending on ecosystem services. Models that have applied CEA or ROI deal with a range of terrestrial and marine ecosystems (Klein et al., 2008; Leslie et al., 2003; Marshall and Homans, 2006; Murdoch et al., 2007; Naidoo and Ricketts, 2006) and the coastal interface (e.g., Stoms et al., 2005). Most of these models are aimed at promoting land conservation through agricultural land retirement (e.g., Marshall and Homans, 2006) or through land purchases (e.g., Murdoch et al., 2007; Underwood et al., 2008).

In contrast, our model is a departure from existing optimization models that incorporate cost-effectiveness because it maximizes a bundle of ecosystem services on working lands. That is not to say that optimization models for working lands are scarce, only that they have traditionally aimed to maximize agricultural production rather than a bundle of ecosystem services (e.g., Eiswerth et al., 2005; Karp and Pope, 1984). Models that examine how to maximize benefits from private and public farm lands are needed in order to promote efforts to generate ecosystem services where complete land retirement is either not possible or not desirable (e.g., Green et al., 2005).

We demonstrate here a structured, practical decision support approach to invasive plant management on western rangelands. The optimization model aims to assist a decision-maker in choosing which sites to treat and at what intensity to maximize the net change in a bundle of ecosystem services, subject to budget and other constraints. By choosing a multi-attribute utility approach, we are able to use a variety of metrics, monetary or non-monetary, to assess benefits of restoration.

Our approach expands upon traditional cost-effectiveness frameworks in three primary ways. First, we use a spatially-variable cost of treatment function rather than assume costs per acre are the same, regardless of location. Second, we develop functions for each ecosystem service to estimate change in benefits with treatment that are adjusted to consider the probability of successful restoration and evidence of diminishing marginal returns. As a result, a probability of success <1% (or 100% chance of successful restoration) results in lower expected benefits compared to unadjusted benefits. Third, we consider not only the on-site benefits of restoration, but also the benefits conferred to neighbouring patches by reducing invasive plant cover in the treated site. These location details enhance the model's ability to represent site distinctions of restoration performance.

2. Case Study

The non-native invasive grass, cheatgrass (*Bromus tectorum*), entered the U.S. more than a century ago probably as a wheat seed contaminant that “cheated” wheat farmers out of their expected crop yields (Mack, 1981). The impact of this widespread invasive on human welfare is relatively well understood compared to most invasive species (Brooks et al., 2004; Whisenant, 1990). Cheatgrass (*Bromus tectorum*) is a species of prime concern because it is thought to increase fire frequency and intensity compared to native plants (Pyke et al., 2003; Hull, 1965 as cited in Roberts, 1990), thereby affecting human welfare through a variety of pathways. The enhanced fire risk affects ranchers, property owners, recreational users, and those concerned with rare and characteristic regional species.

For our case study, we partnered with managers at the Bureau of Land Management in the Twin Falls management district in south central Idaho (Fig. 1), who decide how to control cheatgrass through rehabilitation and restoration of burned areas. Fire provides the disturbance that allows cheatgrass to establish and become the dominant vegetation (Brooks et al., 2004; Pyke et al., 2003; Whisenant, 1990; Hull, 1965). Therefore, managers decide how to allocate scarce fire rehabilitation resources to prevent the spread of and dominance by cheatgrass and meet other goals. Hundreds of fires occur annually, and managers must chose how many burned areas to restore, which species (native or non-native) to seed, and how intensively to conduct the restoration/rehabilitation (i.e., with or without use of heavy equipment to enhance seed-soil contact).

3. Methods

The optimization decision support model seeks to address the question, “If we have X dollars to spend, how do we select among many options, of site choice and treatment intensities, to generate the highest level of ecosystem service benefits?” The details of the analysis are developed through four main steps:

Fig. 1. Site map of BLM Twin Falls District in Southern Idaho. Boundaries of management units used in case study are shown in black.

1. Characterize specific goals of the land or resource manager and create a set of treatment response options.
2. Evaluate the no-action scenario to establish baseline benefits and losses (i.e., which ecosystem services are most damaged by the invasive).
3. Develop cost-effectiveness components: benefit measures, restoration/treatment costs, and restorability metrics (success probabilities) for each alternative.
4. Apply a simulation optimization model to maximize risk-adjusted benefits for a given level of spending and evaluate effects of uncertainty.

3.1. Characterize Management Goals and Response Options

The overarching management goal was to prevent the degradation of multiple ecosystem services through the spread of cheatgrass. We formalized this goal as the desire to maximize the net increase in a bundle of ecosystem service benefits through site restoration. The ability to meet the goal is constrained by a total budget and a limit on the number of sites that could be treated with available personnel and equipment.

Response options consisted of choosing which burned sites to treat and at what intensity. Managers in our case study typically apply a low, medium, or high intensity effort to sites based on expected benefits. For analysis, we simplified the options into 5 typical treatments from low to high effort: No treatment; Aerial seeding only (mix of native/non-native); Aerial seeding + chaining (dragging a chain across seeded areas to increase seed-soil contact); Aerial seeding + chaining + herbicide; and Aerial seeding + chaining + drill-seeding.

3.2. Characterize the No-action Baseline

The no-action baseline is to withhold site treatment. We estimate the effect of no action by first characterizing burned sites in terms of whether the site is within potential cheatgrass range based on a binary map of habitat suitability that we developed based on the current understanding of habitat preferences and the professional judgment of one of us (RNM) who has studied cheatgrass extensively. The preferred habitat of the species is well-established and cheatgrass is not widespread at elevations above 5000 ft (Pierson and Mack, 1990). However, the ability of the species to adapt leaves open the possibility of range expansion (Kao et al., 2008).

We apply this map and other information to evaluate the likelihood that a site would remain in native vegetation or become dominated by cheatgrass without treatment. This likelihood is defined as the *recoverability* probability. Recoverability was estimated using methods similar to those used for *restorability* (ability of treatment to prevent cheatgrass dominance) and methods are described together in a Section 3.3.2.

3.3. Cost-effectiveness Analysis

The cost-effectiveness analysis relies on three components: 1) Benefits of restoration, 2) likelihood of successful restoration, and 3) costs of restoration.

3.3.1. Benefits

Benefits of cheatgrass control depend on how users value the ecosystem services from a site and how those values change with increasing cheatgrass density. Since it is impossible to represent all ecosystem services in the optimization model, we select the most significant services by identifying those that create the biggest changes in user welfare. We first establish the causal pathways through which the invasive increases or decreases the welfare of ecosystem service users and then evaluate potential magnitude of those changes. Impacts from cheatgrass are caused by the direct displacement of native species and by the indirect effect of enhanced fire frequency or intensity. Because of the increased incidence and intensity of fire with cheatgrass, the primary benefits of cheatgrass control are damage costs avoided.

We narrowed the list of potential impacts to four major categories: 1. Recreational antelope hunting, 2. Property protection from fire, 3. Existence values for sage-grouse, and 4. Forage production to support ranching. These impacts are representative of the major concerns of the management agency and have the potential for the greatest change with cheatgrass invasion. All effects on welfare were negative, although the effect on ranching may be positive in the short-run.

The mixed effect of cheatgrass on forage production explains, in part, why cheatgrass management has not been pursued more aggressively. Although cheatgrass is reputed to offer higher spring forage quality than native species, ranchers have long held concerns about sites becoming dominated by cheatgrass because of the direct impacts of fire on their operations and the loss of late spring–summer grazing areas (Pellant, 2005, various agricultural extension websites, e.g., www.msuextension.org). In addition, cheatgrass productivity is thought to be less consistent inter-annually than native vegetation (Roberts, 1990) thereby increasing the variability of returns. We assumed the long-term impacts of increasing cheatgrass cover on forage production would be a net harm to ranching.

To quantify the magnitude of harm to each of the ecosystem services, we sought to quantify the enhanced fire risk and enhanced damage associated with that risk due to cheatgrass. Although cheatgrass has been extensively studied, data were inadequate at that time to develop a quantitative (probabilistic) damage function. We substituted conceptual models representing best professional judgment (elicited through interviews) of researchers and managers for characterizing and quantifying changes in ecosystem service benefits with increasing cheatgrass dominance. Although not ideal, this substitution represents the only feasible option in the absence of quantitative, published evidence.

3.3.1.1. General Benefits Model. The benefits model considered both site conditions and the landscape context of the burned site to estimate potential changes in benefits with restoration. Let b_{ijk}, the risk-adjusted benefits of treating a given burned area i, in terms of ecosystem service j, 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 restoration outcome, S_{ik}, and the *off-site* benefits derived from protecting neighbouring sites from infestation or reducing the risk of fire in neighbouring cells in any given year, C_i.

$$b_{ijk} = f(b_{ij}^{\max}, S_{ik}, C_i, A_i) \tag{1}$$

$b_{ij}^{\max}$	Maximum possible site benefits at site i for service j
S_{ik}	Restoration outcome, measured as % native vegetation, for a given site and treatment level; in the absence of treatment S_{ik} = recoverability or the natural recovery outcome
C_i	Contribution to maintaining off-site benefits, measured as the effect of the site on contiguity of native vegetation (see Supplemental material)
A_i	Size of Treated Area
i	Burned Area/Rehabilitation Site
j	Service
k	Treatment intensity

Beneficial outcomes due to treatment were thus defined as a function of innate site characteristics that affected expected change in benefits in three ways. First, site and landscape characteristics were used to measure a maximum potential level of ecosystem services, $b_{ij}^{\max}$ (in the absence of invasive species). Second, other site and landscape conditions determined the site's probability of restoration (i.e., achieving $b_{ij}^{\max}$) with a given treatment or no treatment. Third, the protection conveyed to neighbouring sites from treating site i, was quantified in terms of a land cover spatial pattern, C_i, that was a proxy for invasion or fire risk to neighbouring lands and used to scale benefits.

3.3.1.1.1. Maximum Potential Site Benefits (b_{ij}^{max}). Potential benefits of a site are represented through conceptual models of how user preferences vary with site characteristics. Preferences were identified through a literature review and interviews and then represented in conceptual models based on a benefits framework previously developed (Wainger et al., 2001). Five types of indicators were used in the conceptual models to represent the maximum potential service benefits at a site: site quality, site opportunity, complementary inputs, scarcity and reliability.

All of the factors in the conceptual model are meant to represent aspects that a buyer might consider when deciding how much to pay for a service. *Site quality* encompasses the productivity of the site for providing the service, and is usually derived from biophysical characteristics such as plant cover, soil type, etc. The *site opportunity* metrics identify the location factors that make the site more efficient at providing the service and can be either biophysical or sociodemographic variables (e.g., proximity to users or position within movement pathways of migratory species or nutrient-laden runoff). *Complementary inputs* are typically built infrastructure that allow a potential service to be realized (e.g., proximity of roads that provide access or campsites that enhance desirability). *Scarcity* can be measured by examining the availability of substitutes and ideally incorporating a level of supply relative to demand. For example, data on the ratio of hunting permits granted per application received provides a measure of hunting scarcity. *Reliability* captures whether the site is expected to provide a stream of services into the future and not be compromised by incompatible adjacent land use, degraded by processes beyond the control of restoration (e.g., land conversion to developed uses), or lose its appeal because of changing user preferences.

The indicators selected for each service are shown in Table 1 along with a positive or negative sign, indicating whether they were deemed to be positively or negatively correlated with benefits. Indicators were kept to minimum by selecting variables that met three criteria: 1) most representative of the indicator category, 2) explained the most variability among sites, and 3) were available in existing datasets. Scarcity indicators were not developed for services delivered to private land holders since availability of substitutes could not be determined (rangeland leases) or were homogenous across all sites (fire prevention). The uniformly high number of substitute sites for hunting led us to omit calculation of this metric.

Note that for the Existence Value for sage-grouse (non-use) category, the metrics have a different meaning from those for use services of recreation, forage production and property protection. All of the factors for non-use services are related to ecological quality and whether the site can sustain a population, which promotes the species existence and therefore the *existence service* that people value. Relative demand indicators, in this case, consider the species of interest (i.e., sage-grouse) to be the "user" of the site, not people, and therefore, capture how close a site is to ideal habitat. For example, indicators aim to represent whether a patch is sufficiently connected to other habitat patches to attract the species and provide viable long-term support. The reliability indicator for this non-use service is the same as for the other services (development risk of site), but it would ideally be supplemented by a population viability analysis for the particular patch of interest to recognize that the existence service would be more valuable if a parcel contributed to population survival and not just survival of a few individuals.

3.3.1.2. Combining Indicators into a Single Metric of Benefits (bmax). We simplified the difficult issue of how to combine indicators for each service into a single benefit index by limiting the number of indicators. To combine the selected indicators, we scaled all continuous variables between 0 and 100% and multiplied indicator values together. Essentially, we evaluated how each indicator contributed to the probability of achieving the highest possible service benefit and combined them as we would any probabilities. This simple method was a necessary construct to fully demonstrate the analysis. We used best professional judgment based on literature review and interviews with managers; our decisions were meant to serve as placeholders for a more structured elicitation process.

3.3.1.2.1. Change in Benefits with Cheatgrass Dominance. The site benefits achieved for any given level of cheatgrass infestation was represented with a continuously increasing function with decreasing slope to represent diminishing marginal returns as percent native cover increases (Supplemental Material, Fig. A1). This concave form for the benefits function (after Monod kinetics equation) is useful because the main shape parameter (m) has an intuitive interpretation. It represents the % cheatgrass cover at which 50% of the benefits are achieved. Thus, this parameter can be elicited through best professional judgment to inform the analysis.

3.3.2. Measuring Recoverability and Restorability

Discussions with managers indicated that restoration success could be defined as the establishment of substantial plant cover of native or non-invasive species, which served as a leading indicator of forage, habitat, and fire protection benefits. 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 is available on the ability of systems to recover naturally. Consequently, we used a combination of techniques to identify conditions that might be predictive of restoration outcomes including a literature review, statistical analysis of the available treatment database, and interviews with restoration practitioners.

We defined the percent of native plant cover to be S_{ik} (restoration outcome, from Eq. (1)) as a function of natural recoverability and treatment effectiveness.

$$S_{ik} = r_i + \tau_k \tag{2}$$

r_i	Site recoverability
τ_k	Treatment effectiveness
i	Site
k	Treatment applied

Restoration outcome was modelled as a function of site characteristics that tended to increase or decrease the probability of success for a given treatment type above what would occur without human intervention.

The recoverability and restorability factors were driven primarily by elicited judgment of restoration practitioners because a satisfactory statistical model could not be fit with available data. Variables that were significant in multiple versions of the statistical model and had consistent directions of correlation (Table 2) were used to structure interviews. An important insight from the data analysis was that higher intensity treatments were associated with lower success rates, most likely because more intensive methods were used on sites at higher risk of failure. The results characterizing the difference between ability of a site to recover without human intervention (r_i) and the improvement due to restoration as a function of cheatgrass cover (S_{ik}) are summarized in Fig. 2. Further information on methods can be found in Wainger et al. (2007).

Table 1
Indicators identified to assess relative economic benefits of four ecosystem services.

Benefit	Site quality	Site opportunity	Complementary inputs	Scarcity	Reliability
Recreational hunting (pronghorn antelope)	Sage-grouse habitat quality (high, medium, low; a function of native plant cover and other metrics) (+)	Population within 1.5h (+)	Camping availability (developed or back-country campsites) (+)	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) (−)		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 (+)	Value of property at risk (+)	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.

Table 2
Summary of restorability statistical model results.

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

3.3.3. Spatially Heterogeneous Cost Model

A cost model was developed to identify variables that explained heterogeneity of total treatment costs among restoration sites and to predict costs for all site-treatment combinations in the optimization model. The total cost (TC_{ik}) of treating site i with treatment k was formulated as:

$$TC_{ik} = FC + (JC_k{}^*JT_i) + (TCA_{ik}{}^*A_i) \tag{3}$$

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)
i	Site treated
k	Treatment applied to burned area i (expressed as treatment intensity level)

The treatment k, includes the use of multiple trips to sites and costs reflect both inputs per trip and number of trips. *TCA* was estimated through a statistical model and fixed costs and journey costs were gathered from government sources. Travel time was estimated through a GIS analysis described in Wainger et al. (2007).

To determine Treatment Cost per acre (TCA_{ik}), a statistical model was fit to a database of treatment spending on BLM rehabilitation program. Treatment was conducted as part of the Emergency Stabilization & Rehabilitation and Burned Area Rehabilitation programs in Idaho and Nevada in 2001–2003. Sixty-five unique observations were available describing location, and quantifying size of area treated 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 was noted. In addition, we tested hypotheses that treatment costs would vary due to site factors expected to affect material, labour and time costs of treatment including travel time, size of burned area, presence of sensitive species, and proximity to developed areas. The supplemental site and landscape characteristics were developed through a GIS analysis.

The OLS model had an adjusted R^2 of 0.559 and was significant at >0.001 level, suggesting the model was reasonably robust. However, due to spatial autocorrelation in the data, the significance levels cannot be considered valid, although the coefficients can be considered unbiased, creating a reasonable model for predicting costs. The final OLS model had the form:

$$\begin{aligned}\ln(\text{cost per acre}) = {} & 6.440 - 0.306[\ln(\text{fire size})] - 1.222(\text{state}) \\ & + 0.494(\text{herbicide}) + 0.548(\text{drill seed}) \\ & + 0.562(\text{native seeds})\end{aligned} \tag{4}$$

Economies of scale were evident because costs per acre declined with increasing natural log of fire size. Our result is similar to previous work suggesting that fire treatment costs increase with the square root of fire size (Mills and Bratten, 1988). The variable for state reflects that lower costs were associated with Idaho relative to Nevada, potentially due to different accounting methods or distinct environmental conditions. Further information about the modelling and choice of statistical techniques can be found in Wainger et al., 2007.

3.4. Integrated Optimization Model

The measures of costs and risk-adjusted benefits developed in the previous sections were used to examine the cost-effectiveness of different treatment options for each site. However, cost-effectiveness ratios leave the decision-maker with uncertainties about how best to combine a package of site-treatment options to generate the highest possible benefits for a given level of spending and logistical constraints. Options that generate low benefits at low cost can have the same cost-effectiveness as options with high benefits and high costs, creating the need to compare many potential combinations of site-treatment options to find the combination of sites that maximizes benefits for a given budget. Further, the number of sites able to be treated with available equipment and personnel provides another constraint. We developed an optimization model using off-the-shelf software (RISKOptimizer v. 1.0, Palisades Corporation, 2006) to compare the many potential combinations of site and treatment options that would maximize ecosystem service benefits while meeting other management constraints.

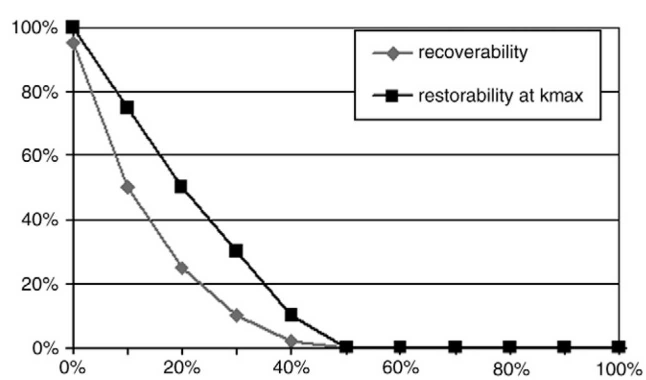

Fig. 2. Derived relationships between % cheatgrass cover pre-fire and natural recovery or restorability with maximum treatment intensity. The *y* axis shows the probable outcome of restoration (of maximum intensity treatment) measured as the % of native or non-native (non-invasive) plant cover on a site 2–3 years after treatment. The *x*-axis shows the variable identified by restoration practitioners as the most predictive of successful restoration — cheatgrass cover prior to the fire that induced restoration. These relationships were developed through interviews with managers, a literature review and statistical exploration of a treatment success database. Other variables were identified as being important, most notably elevation, but because a robust statistical model could not be developed to combine multiple variables, we used only the variable identified as most predictive of restoration success. Note that this figure represents only one curve used in the model to predict restorability; curves were developed for every level of restoration intensity. See Wainger et al., 2007 for further information.

3.4.1. Optimization Model Form

Within the optimization model, we maximized total benefits measured as the weighted sum of the net benefits created through treatment of a set of sites at particular intensities of treatment. The weights, which can be used to prioritize some services over others, were set equal for all four services to reflect stated agency goals. Thus the optimization routine chooses the sites and the treatment intensities that produce the greatest changes in the summed benefit indices.

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

Let B_t equal the total net benefits at time t of net improvements in the set of ecosystem services $\boldsymbol{j}$ from treating a set of sites $\boldsymbol{i}$ with treatments $\boldsymbol{k}$, where $\boldsymbol{b_{ijk}}$ is a vector of service benefits and total net benefits (B_t) are calculated as:

$$B_t = \sum_i \sum_j w_j \left(b_{ijk}^{\text{with}} - b_{ijk}^{\text{without}}\right) \tag{5}$$

b_{ijk}	Benefit, with or without treatment
w_j	Weight applied to service j
i	Location i, (sum over treated *subset* of burned sites)
j	Service j (sum over service benefits: recreation, forage, property protection, existence values for sage-grouse)
k	Treatment intensity applied

The model was subject to two hard constraints: the overall budget was limited to $1 million (based roughly on budgets of the two rehabilitation programs we were evaluating) and the number of burned areas that could be treated could not exceed 10. We tested the effect of the limit on fire sites treated by comparing results of optimization runs with and without the constraint, as discussed below.

4. Discussion of Results

A major question we addressed with the optimization model was, “Did the 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 agency conceptions of important environmental services produced, 2) we used agency 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 several reasons to expect our results might differ. First, we were uncertain that managers' decisions had captured the indirect costs of treating sites, such as those incurred in travel to sites. Second, we were uncertain that the GIS data adequately reflected the detailed site-based information available to managers. Lastly, we made many simplifying assumptions.

In the end, we found that we could replicate the management choices of sites to treat under certain conditions, but also found that the optimization model could identify a preferred management option (choice set) that produced greater benefits. When we compared the change in benefits for the set of agency-selected sites with sites chosen by the model, the total benefits were substantially higher for the model solution set than for the agency choice set (Fig. 3). In total, across the four sites compared,[2] the sites chosen for treatment by the optimization model produced over three times the service benefits of the agency selected sites. On a per-service basis, sites chosen by the optimization model delivered 2–5 times the level of any given service compared to sites selected by the agency except for property protection, which was substantially higher in the best performing agency chosen site.

[2] Only four sites are included in the comparison of agency and model results because our data showed only five sites were treated in the year we used for comparison. One of the sites could not be mapped, so we omitted the top-performing site from the optimization model set to improve the objectivity of the comparison.

4.1. Considering the Service Bundle is More Efficient

The initial optimization model was run with each of the four ecosystem services being given equal weight in order to conform to the stated agency goals. The model was also run to maximize one service at a time in order to test the sensitivity of solutions to service weights and to examine the spatial variability of services. When full weight was assigned to a single service, all but one of the sites chosen by the management agency were eventually represented in the solution sets — an unexpected result (Fig. 4). In other words, the model was better able to represent the agency choices when sites were chosen to maximize only one service sequentially rather than all services simultaneously. It is also apparent that agency-selected sites were more likely to be dominated by one or two services compared to the three or four services represented in sites selected by the optimization model (Fig. 3). These results suggest that resource managers choose sites that produce the highest benefits for individual services and are not necessarily evaluating sites as producers of multiple services.

4.2. Benefit Estimates were Sensitive to Assumptions About Service Complementarity

A key finding of our analysis is that managers can potentially improve outcomes of restoration programs by selecting sites in which multiple service benefits can be maximized simultaneously, rather than selecting sites for treatment that maximize only a single service benefit. However, our results are sensitive to the question of whether benefits can be produced simultaneously at sites using current restoration methods that encourage both native and non-native (but non-invasive) vegetation. Evidence was mixed as to whether current treatment practices result in all services being compatibly generated on sites or whether services, such as forage production and habitat for characteristic species, were partially or wholly in conflict. Our assumption that services were compatible deserves further review and strongly influences the conclusion that agency choices were less efficient than optimization model outputs.

4.3. Risk-weighting Crucial to Estimating Benefits Accurately

One site selected by the management agency was not selected in any runs of the optimization model. This site demonstrates how our economically-based framework provides an opportunity to improve decisions (assuming our data and assumptions are robust). This site scored high for available benefits after treatment, but our data suggested the site had a high likelihood of recovering without human intervention. Therefore, the *change* in benefits with treatment, for this isolated site, was estimated to be negligible. Managers may indeed choose sites for treatment for which there is a high probability of restoration, but without objective information on a site's ability to recover without intervention, restoration resources can be wasted on sites where intervention does not generate net benefits.

4.4. Low Intensity Treatments were Consistently Chosen

Although simulation optimization models do not always identify the optimal solution, we found model solutions were insensitive to certain modelling choices, thereby giving us confidence in the outputs. For one, our model results showed a consistent selection of low-intensity treatments, regardless of the model's initial conditions and software run parameters. The optimization model selected the lowest intensity of treatment (aerial seeding only) for the majority of burned areas selected for treatment (Table 3). This result was likely driven by the relatively low improvement in success rates for higher intensity treatments and lower cost-effectiveness ratios. The

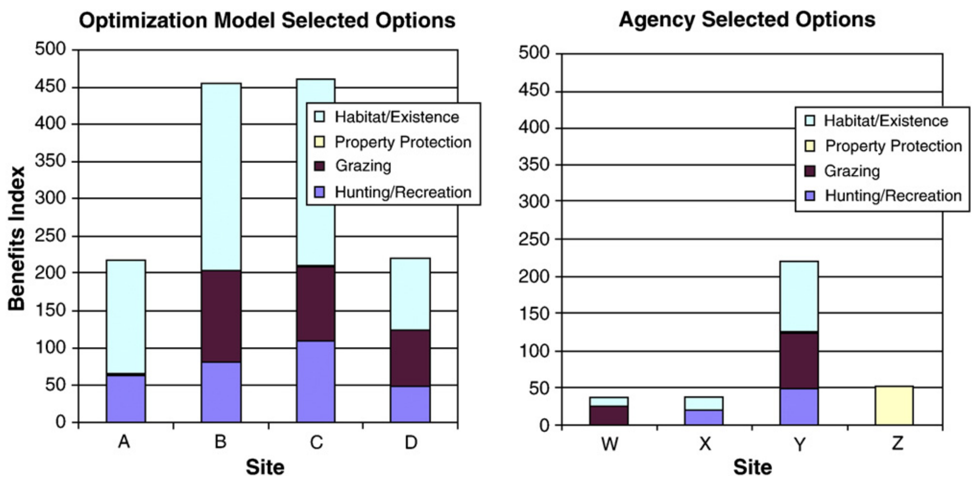

Fig. 3. Comparison of per acre benefits for options selected by optimization model and agency. Bar graphs show the change in the bundles of ecosystem services produced by sites as a result of the selected treatment. The model-selected sites produced over 3 times the service benefits of the agency-selected sites. 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 were more likely to be concentrated in one or two services. (Property protection is represented in most sites selected by the optimization model, but at levels that do not display well in graph.) Results suggest that the agency may be able to generate higher levels of ecosystem services by selecting sites that can provide high levels of multiple services, compared to selecting sites that provide a high level of a particular service. Notes: In the year shown, the agency treated only 5 burned areas through this program and one site treated by the agency was not able to be analyzed with available data. Therefore, we used the top 5 model-selected sites and excluded the top performing site to improve the fairness of the comparison between the model results and the agency selections. Site D in the left side graph and site Y in the right-side graph are the same site.

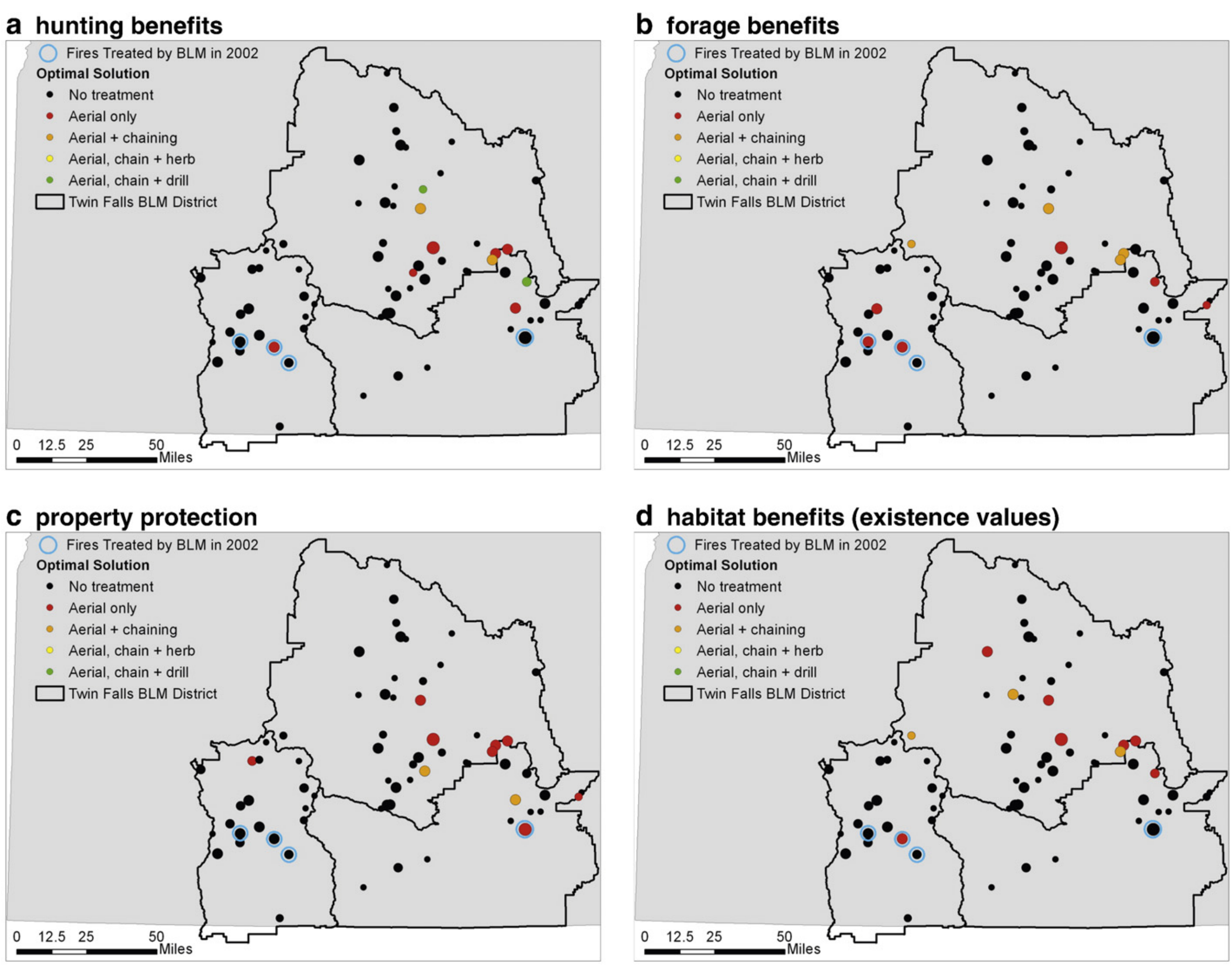

Fig. 4. a–d. Maps of solution sets from optimization model and agency choices in 2002 when full weight was given to the specified benefit category. Colored dots represent 10 model treatment selections; dot size represents site size. Four of the agency selections are shown circled in blue. Note that all but one agency-treated site (the southernmost circle) was chosen in one or more of these model solution sets in which only one service was maximized. When all services were maximized simultaneously, only one agency-treated site was chosen — the same one chosen in the set for hunting (a) and habitat benefits (d). Not all agency-treated sites could be evaluated due to data limitations.

Table 3
Optimization model results showing average fire size and cost-effectiveness ratios by treatment type.

Treatment intensity	# Sites	Average fire size (acres)	Total acres treated	Average benefit index/cost ratio
No treatment	58	1071	NA	NA
Aerial seeding only	6	7194	43,162	2.50
Aerial seeding and chaining	3	3179	8536	1.42
Aerial seeding, chaining & herbicide	0	NA	0	
Aerial seeding, chaining & drill-seeding	1	964	964	0.59
Totals for treated sites	10	789	53,662	1.98

consistency of results among model runs suggests that the model did identify the optimal set.

5. Conclusions

So, in answer to our question, “Can the concept of ecosystem services be practically applied to improving natural resource management decisions?”, the answer is a qualified yes. Although much of the data needed to rigorously address this question were absent, we demonstrated that with sufficient regional data sets, we could create decision tools that identified improvements over current decision-making. The model's ability to replicate the well-informed agency choices, under certain conditions, was an important test of the robustness of the model. Most significantly, this result implies that available data and knowledge are sufficient to capture the decision process in a standardized framework that can be used to develop decision criteria consistent with agency goals.

Even if the model had made the same choices as the agency, the approach would have value for a public decision-making process. A framework that replicates current decisions can be opened up to greater public and scientific review. Since we were not able to base the framework on completely objective information, it seems particularly important that the approach we developed revealed the underlying assumptions, values and tradeoffs associated with agency decisions. A structured decision approach can enable researchers and stakeholders to understand how their input can affect choices.

Our approach presented enhancements to ecosystem service decision support tools by incorporating important spatial variability and key economic concepts. In particular, adjusting benefits by the probability of restoration success is likely to be important in most restoration decisions. Our analysis of recoverability and restorability suggested that treatments were only able to improve outcomes over a small range of site degradation conditions. Sites with low percentages of cheatgrass cover prior to fires may be able to recover without human intervention, while sites with greater than 20% cheatgrass cover pre-fire had less than a 50:50 chance of recovery (defined as >50% desirable plant cover), even with the most intensive treatment. All restoration involves some gamble on the outcome, but the risk of failure may be acute when restoration is aimed at preventing the spread of invasive species, where success rates can vary widely.

The spatial detail we incorporated in our decision model appears to have enhanced the ability of the model to represent trade-offs encountered by resource managers. For example, the spatially-variable cost of treatment function allowed the extra cost from travelling to remote sites to be compared to the advantages of treating remote sites when such treatment limits the spread of invasives to uninfested neighbouring sites. The spatial detail also improved the ability of the model to mimic managers' decisions based on site-specific details.

Our work suggests that benefit indicators developed from conceptual models that incorporate economic concepts can improve decision-making even if benefits cannot be monetized. Specifically, our use of a concave benefit response function to incorporate diminishing marginal returns and the use of a multi-criteria weighting scheme to consider the joint production of multiple services appears to have enhanced the ability to maximize returns from restoration. The improved performance from model-generated choices did not require that we monetize benefits only that we generate logical proxies of benefits. Our findings support conclusions reached elsewhere (e.g., Weinberg and Claassen, 2006) that benefit indicators promote cost-effectiveness of restoration programs.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.ecolecon.2009.12.011.

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