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Fire and the joint production of ecosystem services: A spatial-dynamic optimization approach☆

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ABSTRACT

We develop a spatial-dynamic optimization model of landscape scale, jointly produced ecosystem services on a forest at-risk of catastrophic wildfire. We show optimal fuel management strategies vary over time in response to biomass growth dynamics, how biomass contributes to ecosystem services, the implied tradeoffs and complementarities between ecosystem service values, and the spatial dynamics of fire spread. Fuel reduction strategies increase expected ecosystem service values in the future by lowering the probability ecosystem services will be lost to catastrophic fire. However, fuel management also differentially impacts current ecosystem service values through the reduction in biomass. Ecosystem service values that are immediately enhanced by fuel reductions such as forage for grazing serve as an indirect benefit of fuel management. In contrast, ecosystem service values that are immediately harmed by fuel management such as recreational values serve as indirect costs of fuel management. Spatial spillovers, in the form of high wind, tend to lower tradeoffs between ecosystem services, and increase complementarities.

1. Introduction

Despite a lack of markets for natural capital and ecosystem services, society often makes choices that imply values for the goods and services natural capital provides (Turner and Daily, 2008). These implied values are especially apparent on public lands, where managers balance preferences of a wide range of stakeholders, potential impacts, and legal obligations (Binder et al., 2017 and Sills et al., 2017). Understanding tradeoffs that exist in ecosystem service provision is important because, like many goods and services, ecosystem services are jointly produced. A change caused either by nature or management to the natural capital that produces ecosystem services can have broad landscape-scale impacts on other ecosystem services (Bagstad et al., 2013).

In the American West, those management actions are often directed toward reducing the risk of wildfire to natural capital. Within natural return intervals and intensity levels, fire is important to maintain ecosystem health and many ecosystem services (Thom and Seidl, 2016). Values for recreation, pollinator services, and many types of wildlife habitat, for example, tend to be higher in thinner, patchy landscapes created by natural fire behavior (Hanula et al., 2016). However, a legacy of fire suppression has led to over-accumulation of fuels in many western forests, resulting in fires that burn hotter and larger than they otherwise would (Hessburg et al., 2005). In 2017, just over 10 million acres burned throughout the U.S., making it at the time, the second largest amount of acreage burned in U.S. history, just below the 10.1 million acres burned in 2015.[1] These so-called ‘mega-fires’ (fires that burn > 100,000 acres) have the potential to directly destroy homes, wildlife habitat, and commercial timber. Burned areas often see increases in chemicals, metals, and sediment in waterways (Neary et al., 2005; Smith et al., 2011) and declines in recreation access (Duffield et al., 2013).

Prompted by growing interest in forests for recreation, environmental uses, and the threats such uses face, forest researchers have expanded both the scope and tools of forest management by incorporating a variety of human values into traditional forest management models, though still with a heavy focus on timber. Early work by Hartman (1976) showed values like recreation and flood control can affect when or even if a stand should be harvested for timber. Johnson

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[1] https://www.nifc.gov/fireInfo/fireInfo_stats_totalFires.html, accessed November 13, 2018

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et al. (1980) generalized the Hartman model to account for multiple non-timber and amenity values. Their work led to the development of the FORPLAN model, used extensively during the 1980s for planning by the US Forest Service and highlighted many of the tradeoffs involved in forest management. With the expansion of computational power, researchers have turned more often to simulated production possibilities frontiers (PPFs), marginal cost analysis, and multi-objective optimization techniques to highlight tradeoffs between human values on a landscape. Arthaud and Rose (1996) were among the first to formulate a PPF for ecosystem service tradeoffs with a case study on aspen harvest and ruffed grouse habitat. They used computational iterations of a stand management program and a habitat suitability index. Calkin et al. (2002) solved for a PPF showing tradeoffs between timber harvests and northern flying squirrel habitat by maximizing timber harvests subject to various targets for habitat conservation. They showed that, even for their two-resource tradeoff functions, computational complexity remained high, and so relied on simulated annealing to optimize over both spatial and temporal ecological processes. Nalle et al. (2004) build off methods developed in Calkin et al. (2002) by expanding the complexity of the ecological model, particularly to highlight tradeoffs between species' habitat and to expand the geographic scope of the analysis. They argue that the information gains from more computationally sophisticated models are worth the added effort compared to non-spatial or static models. Similar arguments are made by Bekele et al. (2013), who use a genetic algorithm to map out a PPF for six ecosystem services in a watershed (corn, soybeans, hay, flood control, water quality, and carbon retention). By extending the list of ecosystem services considered, Bekele et al. also highlighted the role of complementary between some ecosystem services.

Newer models, combined with advances in GIS technologies, highlight the spatial interdependence between ecosystem services across a landscape. Spatial relationships are often modeled with ecological conditions that interact between locations, such that benefits from the entire landscape are larger than the sum of benefits from one location (Amacher, 2004; Bowes and Krutilla, 1985; Koskela and Ollikainen, 2001; Swallow et al., 1997; Swallow and Wear, 1993), and because reducing forest vegetation is often a management objective, timber harvests can be complementary to many other ecosystem services (Sims et al., 2014). Non-timber amenity values change with the stock of the forest, so optimal management over time may include parcels that are maintained in low-stocked levels of timber while other areas go virtually untouched as old growth forests (Bowes and Krutilla, 1985). In such cases, optimal forest management looks less like multiple use from a single stand and more like a zoning approach to management. In this regard, location of economic activity on a landscape is as important as the level of economic activity that takes place (Polasky et al., 2008).

Insights from ecosystem service models are increasingly applied to optimal management of wildfire risk (Minas et al., 2012). Lee et al. (2015), for example, use vegetation levels as a proxy for all ecosystem services, with ecosystem service values recovering post-fire based on the age of the mature vegetation. Rocca et al. (2014) look at the effect of fire regimes and fuel management practices on ecosystem services across forest types. They find fuel treatments are, in general, more effective at preserving ecosystem services in the lower and upper montane forests due to these forest types' relative productivity and potential for vegetation overgrowth compared to the other forest types. Vogler et al. (2015) combine a PPF framework with advanced computational optimization, GIS modeling, and simulations for vegetation growth and fire risk to examine tradeoffs between timber harvests, reductions in fire risk to the wildland urban interface, and reduction in risk of damages from insects (e.g., bark beetle). They highlight that, because smaller units of forest tend to be dominated by a limited set of ecosystem services, there are not only tradeoffs in how one manages a single forest stand, but also where on a large landscape to prioritize restoration.

The inter-temporal tradeoffs implied by managing fire risk and ecosystem services coupled with the spatial interdependence between ecosystem services across a landscape suggest the need for a spatial-dynamic optimization approach (Sanchirico and Wilen, 1999; Smith et al., 2009). However, combining space and time in optimal control problems creates significant degrees of analytical and computational complexity. To facilitate a solution, forest management researchers restricted the dimension of the problem by limiting the range of human behavior, number of ecosystem services, number of spatial patches, number of time periods, or the interdependency of spatial patches. For example, many studies utilize integer programming which implies forest management is constrained to a dichotomous choice such as to manage or not manage a given area (e.g., Hof et al., 1994). Others studies avoid such restrictions on management but facilitate a solution by limiting the number of time periods (Albers, 1996; Konoshima et al., 2010; Konoshima et al., 2008). Aadland et al. (2015) allow for a continuous gradient of management actions and a large number of time periods but restrict their attention to timber and non-timber ecosystem services to facilitate a solution.

We combine the literature on production of ecosystem services and spatial-dynamic optimization to develop a model of landscape scale ecosystem services that are jointly produced on a forest but subject to risk of catastrophic fire. We use nonlinear programming to show how fire is likely to affect values in the distant future and at distant points across the landscape and what optimal fuels management may look like when treatments affect this risk. We explore spatial and temporal spillovers to fuel treatments and illustrate when and where these spillovers matter. The work presented here contributes both with modeling techniques used to address spatial-dynamic optimization of ecosystem services and with conceptual methods used to think about how we should manage ecosystem services over space and time. We apply our methods to a case study of Montana watersheds, roughly based on forests around the capital city of Helena. The area includes grasslands, dense forests, and human uses ranging from intense grazing and timber production to wilderness. The area is also deemed high risk of wildfire, with significant departure from historic fire regimes (Schmidt et al., 2002).

2. Methods

2.1. Model

Consider a landscape with three watersheds where the objective is to manage vegetation levels over time to maximize discounted ecosystem service values. Each watershed i provides $s = 1, 2, \ldots, S$ ecosystem services $ES_{i,\,s}$. Ecosystem services are a function of the standing forest biomass (i.e., natural capital) in a watershed $v_i(t)$ which evolves according to

$$\frac{dv_{\mathrm{i}}}{dt} = g_i(v_i) - m_i v_i \tag{1}$$

where $g_i(v_i)$ is a growth function and $m_i(t)$ is the proportion of biomass removed for fuel management in period t. Ecosystem services may be either increasing in the level of biomass (e.g., recreation) or decreasing in the level of biomass (e.g., water supply). We focus on fires large enough to, at least in the short-run, completely eliminate all ecosystem services in the watershed. For example, recreational areas are often closed during and, often, for a period after large fires. Similarly, it may take a while for some wildlife species to return to a burned area while other species thrive shortly after a fire.

A fire in a watershed may originate within the watershed or spread from a neighboring watershed. Thus, the probability of a large fire in a watershed is a function of the fuel load in the watershed and the fuel load in neighboring watersheds

$$\varphi_i(t) = d_{i1}v_1(t) + d_{i2}v_2(t) + d_{i3}v_3(t) \tag{2}$$

where $d_{ii} = 1$ and $0 < d_{ij} < 1$ is the probability forest biomass in watershed j contributes to a catastrophic fire in watershed i. We assume the probability is larger for more proximate watersheds, and if watershed j is downwind of watershed i, $d_{ij} > d_{ji}$. We term the probability weighted biomass in Eq. (2) the *effective fuel load*. The timing of this large fire is unknown and characterized by a hazard function (Heines et al., 2018; Reed, 1984). Specifically, a fire occurs when an unknown effective fuel load threshold $\overline{\varphi}_i$ is crossed. The uncertainty in $\overline{\varphi}_i$ is controlled with the distribution function $F(\varphi_i) = \Pr(\overline{\varphi}_i \le \varphi_i)$ and density function $dF/d\varphi_i$ over support $[0, \infty]$.

To account for the effect of management on fire risk, a conditional probability density function is defined over changes in effective fuel load

$$h(\varphi_i) = \lim_{\Delta\varphi_i \to 0}\left[\frac{\Pr(\textit{fire in } (\varphi_i, \varphi_i + \Delta\varphi_i) \mid \textit{no fire up to } \varphi_i)}{\Delta\varphi_i}\right] \tag{3}$$

with $dh/d\varphi_i > 0$ reflecting a greater probability of a fire the larger the effective fuel load. The conditional pdf or hazard function represents the instantaneous probability rate of reaching a watershed's critical effective fuel load at φ_i given no large fires have previously occurred. Eq. (3) implies the probability a level of effective fuel load φ_i has caused a large fire in watershed i is

$$F(\varphi_i) = \Pr(\overline{\varphi}_i \le \varphi_i) = 1 - \exp\left(-\int_0^{\varphi_i} h(z)dz\right) \tag{4}$$

It follows that $F(0) = 0$. The specification of the distribution function in (4) critically depends on whether the hazard is unimodal or monotone. A flexible parametric form that accommodates both is the Weibull model with

$$h(\varphi_i) = \lambda\varphi_i^{\beta-1} \tag{5}$$

where $\lambda > 0$ and β is a shape parameter with a value greater than one. The parameters in the hazard function may vary across watersheds due to differences in physical characteristics of the watershed (e.g., slope, aspect, land cover), local climate (e.g., precipitation, temperature), or the presence of human infrastructure such as roads. Applying this specification to (4) gives

$$F(\varphi_i) = 1 - \exp\left[-\frac{\lambda\varphi_i^{\beta}}{\beta}\right] \tag{6}$$

Fire hazard can be reduced by investing in fuel management m_i. Fuel management lowers the fire hazard in watershed i and in neighboring watersheds ($d_{ji} > 0$). Thus the benefits of fuel management in watershed i spillover to neighboring watersheds. We assume non-decreasing costs of fuel management, which is true at least on the margin for a given management approach. For example clear cutting a large area may be cheaper than selective harvest of the same area, but for either method, removing vegetation near the road will be cheaper than removing more distant vegetation. This implies a cost of fuel management $C(m_1,m_2,m_3)$ with $\partial C/\partial m_i > 0$ and $\partial^2 C/\partial m_i^2 \ge 0$.

The forest manager's objective is to choose a time-varying fuel management schedule in each watershed $(m_1(t),m_2(t),m_3(t))$ to maximize the expected net present value of forest ecosystem services:

$$\begin{aligned} J(m_1, m_2, m_3) = \int_0^{\infty} e^{-rt}\Bigg(&[1 - F(\varphi_1)]\sum_{s=1}^{S} p_{1,s}ES_{1,s}(v_1) \\ &+ [1 - F(\varphi_2)]\sum_{s=1}^{S} p_{2,s}ES_{2,s}(v_2) \\ &+ [1 - F(\varphi_3)]\sum_{s=1}^{S} p_{3,s}ES_{3,s}(v_3) - C\Big(m_1, m_2, m_3\Big)\Bigg)dt \end{aligned} \tag{7}$$

subject to:

$$\frac{dv_i}{dt} = [1 - F(\varphi_i)][g_i(v_i) - m_i v_i]$$

$$m_i \ge 0$$

where $F(\varphi_i) = 1 - \exp\left[-\frac{\lambda(d_{i1}v_1 + d_{i2}v_2 + d_{i3}v_3)^{\beta}}{\beta}\right]$ and $p_{i,\ s}$ is the value of ecosystem service s in watershed i, with initial conditions $v_i(0) = v_{i,\ 0}$.[2]

The spatial and temporal dynamics of the problem make analytical solutions challenging. To gain intuition, we initially consider the aspatial problem in which the probability of fire is only a function of the biomass in that watershed: $F(\varphi_i) = F(v_i) = 1 - \exp\left[-\frac{\lambda(v_i)^{\beta}}{\beta}\right]$. In this setting, identifying optimal fuel management can be viewed as three independent optimization problems. The current value Hamiltonian for fuel management in watershed i is

$$[1 - F(v_i)]\left[\sum_{s=1}^{S} \mathrm{p_{i,s}ES_{i,s}(v_i)} - \mathrm{c(m_i)} + \pi_i\,[g_i(v_i) - m_i v_i]\right] \tag{8}$$

where π_i is the current-value shadow value for forests biomass. Forest biomass can be a valuable commodity if it contributes to ecosystem services: $\pi_i > 0$. It may also detract from ecosystem service values and increase the risk of fire: $\pi_i < 0$. The necessary conditions are:

$$m_i\,[1 - F(v_i)][-c'(m_i) - \pi_i v_i] = 0 \tag{8a}$$

$$\begin{aligned} &- F_i'(v_i)\left[\sum_{s=1}^{S} \mathrm{p_{i,s}ES_{i,s}(v_i)} - \mathrm{c(m_i)} + \pi_i\,[g_i(v_i) - m_i v_i]\right] \\ &\quad + [1 - F(v_i)]\left[\sum_{s=1}^{S} \mathrm{p_{i,s}ES_{i,s}'(v_i)} + \pi_i\,[g_i'(v_i) - m_i]\right] = -[\dot{\pi}_i - \rho\pi_i] \end{aligned} \tag{8b}$$

From the Kuhn-Tucker condition in Eq. (8a), we see that a positive amount of fuel management is only optimal when the shadow value of forest biomass is negative: $-c'(m_i)/v_i = \pi_i$. If the shadow value of forest biomass is positive, $-c'(m_i) < \pi_i v_i$ for all levels of biomass, it is optimal to refrain from fuel management.

Solving (8a) for π_i and substituting into (8b)

$$\begin{aligned} &- F_i'(v_i)\left[\sum_{s=1}^{S} \mathrm{p_{i,s}ES_{i,s}(v_i)} - \mathrm{c(m_i)} - \frac{c'(m_i)}{v_i}[g_i(v_i) - m_i v_i]\right] \\ &\quad + [1 - F(\varphi_i)]\left[\sum_{s=1}^{S} \mathrm{p_{i,s}ES_{i,s}'(v_i)} - \frac{c'(m_i)}{v_i}[g_i'(v_i) - m_i]\right] \\ &= \frac{c''(m_i)\dot{m}_i v_i - c'(m_i)\dot{v}_i - c'(m_i)v_i\rho}{v_i^2} \end{aligned} \tag{9}$$

Eq. (9) illustrates how the optimal amount of fuel reduction in a watershed equates the manager's marginal benefit and marginal cost of fuel reduction. The left-hand side is the marginal benefit of fuel reduction. The first term on the left is the effect of fuel management on the probability of a catastrophic fire. The value of the risk reduction is equal to the value of the ecosystem services the forest provides (the term in brackets). The second term on the left is the (probabilistic) effect of fuel management on current and future ecosystem service values. This would be a benefit if ecosystem services decline in biomass (e.g., water yield). It may also represent a cost if the watershed provides exclusively ecosystem services that increase in forest biomass. Thus, fuel management increases the expected duration of the flow of ecosystem services from the forest but it will not always increases the flow itself. The right side is the marginal cost of fuel reduction. Eq. (9) and

[2] We chose not to constrain the management decision with a budget constraint. One could match a budget to the optimal treatment curves since marginal benefits of treatments are equal across watersheds regardless of the aggregate level of treatment due to optimality. If provision of ecosystem services are extremely lumpy or strong complementarities exist, a small budget might imply treatments only in a single watershed.

the state equation for forest biomass can be solved to yield the optimal time path of fuel reduction $m_i^*(t)$.

Identifying optimal fuel reduction strategies becomes more complex when fire can originate in neighboring watersheds: $d_{ij} > 0$. Here, the marginal benefit of fuel reduction in watershed i depends on the fuel load and fuel reduction strategies in neighboring watersheds. Specifically, the left side of Eq. (9) will include additional terms that capture the spillover effects of local fuel reduction (i.e., the reduction in catastrophic fire risk in watershed j due to fuel reduction in watershed i). Optimal fuel reduction strategies in each watershed must be found simultaneously to ensure the marginal benefit of fuel reduction is equalized across watersheds. Otherwise the expected net present value of forest ecosystem services could be increased by reallocating fuel reduction effort to the watershed with the highest marginal benefit.

2.2. Illustrative example

To investigate the spatial dynamics of the model, we introduce an illustrative example based on forested watersheds in Montana. We calibrate the model to roughly represent the National Forests around the capital city, Helena (Helena – Lewis and Clark National Forests). We emphasize that our model and data are highly stylized and are not intended to imply specific management or policy suggestions. We selected six ecosystem services ($S = 6$) for the focus of this analysis: forests' role in removing sediment and pollutants in drinking water, provision of water for municipal and agricultural uses, access to outdoor recreational opportunities, provision of forage for grazing, provision of timber, and wildlife habitat to support hunting.

Because the spatial allocation of ecosystem services is important for our analysis, we focus on the hypothetical forested landscape in Fig. 1. The forested landscape is composed of three watersheds delineated by black lines in Fig. 1. Fuel management takes place on public lands in the headwaters of each watershed delineated by red lines. For exposition, we assume the forested landscape has 3000 mile^2 of public land evenly divided between the three watersheds, which is roughly the size of the Helena – Lewis and Clark National Forests. The northernmost watershed (watershed 1) contains heavily used public land that caters to a variety of recreational pursuits, grazing, and harvesting high-valued timber. The river flowing from this public land provides drinking water to a city with about 81,000 residents. The southwest watershed (watershed 2) contains a wilderness area with fewer recreational visits than the northern watershed and limited grazing. The river flowing from this wilderness area provides drinking water to a small town with about 68,000 residents. The public land in the southeast watershed (watershed 3) is used primarily for hunting, grazing, and low-value timber harvesting. The river flowing from this public land provides irrigation water for agriculture in the valley.

To isolate the effect of ecosystem service provision, we assume the vegetation characteristics are identical in each watershed. Fig. 2 shows how forest growth, percent forest cover, and fire hazard vary with the amount of forest vegetation in each watershed. Vegetation in the three watersheds ranges from dense coniferous forests to grasslands. Each watershed contains a mixture of spruce, fir, larch, lodgepole pine, and ponderosa pine depending on elevation. While admittedly a simplification, we use a single logistic growth function for vegetation throughout the study area: $g_i(v_i) = rv_i\left(1 - \frac{v_i}{K_i}\right)$ where $r = 0.05$ is the intrinsic growth rate of the mixed forest, and $K_i = \theta A_i$ is the carrying capacity of the forest. Vegetation is measured in million short tons, and $\theta = 0.019$ is calibrated to yield a carrying capacity of 19 million short tons which is the average biomass of overstocked forests in Montana. The current forest has 500,000 short tons of biomass in each watershed: $v_{i0} = 0.5$.

The percent forest cover in each watershed π_i is assumed to be increasing in the level of forest vegetation:

$$\pi_i = \frac{z_i v_i}{1 + z_i v_i} \tag{10}$$

where z_i captures the density of the forest in watershed i. While admittedly a simplification, we assume forest density is constant throughout the watershed, or that characteristics of the watershed can be described by an average density. For exposition, we assume $z_i = 0.158$ which implies that all watersheds are 75% forested when vegetation growth is at carrying capacity.

Fire hazard in each watershed is determined by two parameters that determine the hazard rate. We assume instantaneous fire hazard is increasing in the amount of forest vegetation in the watershed: $\beta = 2$. We assume $\lambda = 0.05$, which implies a 99% probability a large catastrophic fire has occurred by the time the forest reaches carrying capacity. Fig. 3 presents the two scenarios we use to illustrate the spatial dynamics in our model. The first scenario is a “no wind” scenario in which fuel load in a watershed is entirely determined by the biomass in that watershed: $d_{ij} = 0$ and $d_{ii} = 1$. With no wind-induced spillovers, the probability a fire has occurred with a given level of biomass is identical in each watershed. This scenario would represent the aspatial model investigated in section 2.1. The second scenario is a “high wind” scenario in which fuel load in a watershed is determined by the biomass in that watershed and watersheds that lie to the west. The wind-induced spillovers make the probability fire has occurred highest in watershed 3 and lowest in watershed 2.

The cost of fuel management is the sum of fuel management costs in each watershed: $C(m_1, m_2, m_3) = C_1(m_1) + C_2(m_2) + C_3(m_3)$. For exposition, the cost of fuel management in each watershed is a quadratic function of the amount of fuel reduction: $C_1(m_1) = c_{i1}m_i + c_{i2}m_i^2$ where c_{i1} and c_{i2} are non-negative cost parameters. To isolate the effect of ecosystem service provision, we assume the cost parameters are identical in each watershed: $c_{i1} = 50{,}000{,}000$ and $c_{i2} = 12{,}500{,}000$. In the following sections we describe the provision of ecosystem service values in each watershed. As illustrated in the previous theoretical section, many of these ecosystem service values accrue to the manager in the future while the cost of fuel management are incurred immediately. The forest manager uses a 4% discount rate to compare values that accrue at different points in time.

2.2.1. Removal of sediment and pollutants in drinking water (water quality)

A key role of forested watersheds is the provision of municipal drinking water and habitat for sensitive species. Vegetation filters water and holds sediment in place; benefits to water quality, therefore, generally increase with vegetation. We measure these benefits following Warziniack et al. (2017b), who look at the benefits of forestlands to municipal water providers in forested ecoregions. They find that a 1% increase in forest cover is associated with a 3% decrease in turbidity, and a 1% decrease in turbidity is associated with a 0.19% decrease in cost of water treatment. Following their work, we use the following equation to measure the impact of vegetation on water quality, and in turn the avoided costs of water treatment per million gallons treated:

$$AvoidCost_i = \left(\frac{\pi_i}{\pi_{i0}} - 1\right)\frac{C_{i0}}{3 * 0.19} \tag{11}$$

where C_{i0} is the initial water treatment cost at the initial forest cover π_{i0}. For exposition, we set the initial water treatment cost in all watersheds equal to \$105.32 per million gallons treated which is the average water treatment cost for utilities in Warziniack et al. (2017b).

Annual avoided water treatment costs are found by multiplying the avoided costs of water treatment by the number of gallons treated annually. We assume the city in watershed 1 treats about 8 million gallons per day and the smaller town in watershed 2 treats about 6.8 million gallons per day. Thus, the value of water quality ecosystem services (avoided costs of water treatment) for watershed 1 is $p_{1,1}ES_{1,1} = AvoidCost_1 * 8{,}000{,}000 * 365$ and in watershed 2, $p_{2,1}ES_{2,1} = AvoidCost_2 * 6{,}800{,}000 * 365$. Since the water in watershed 3 is

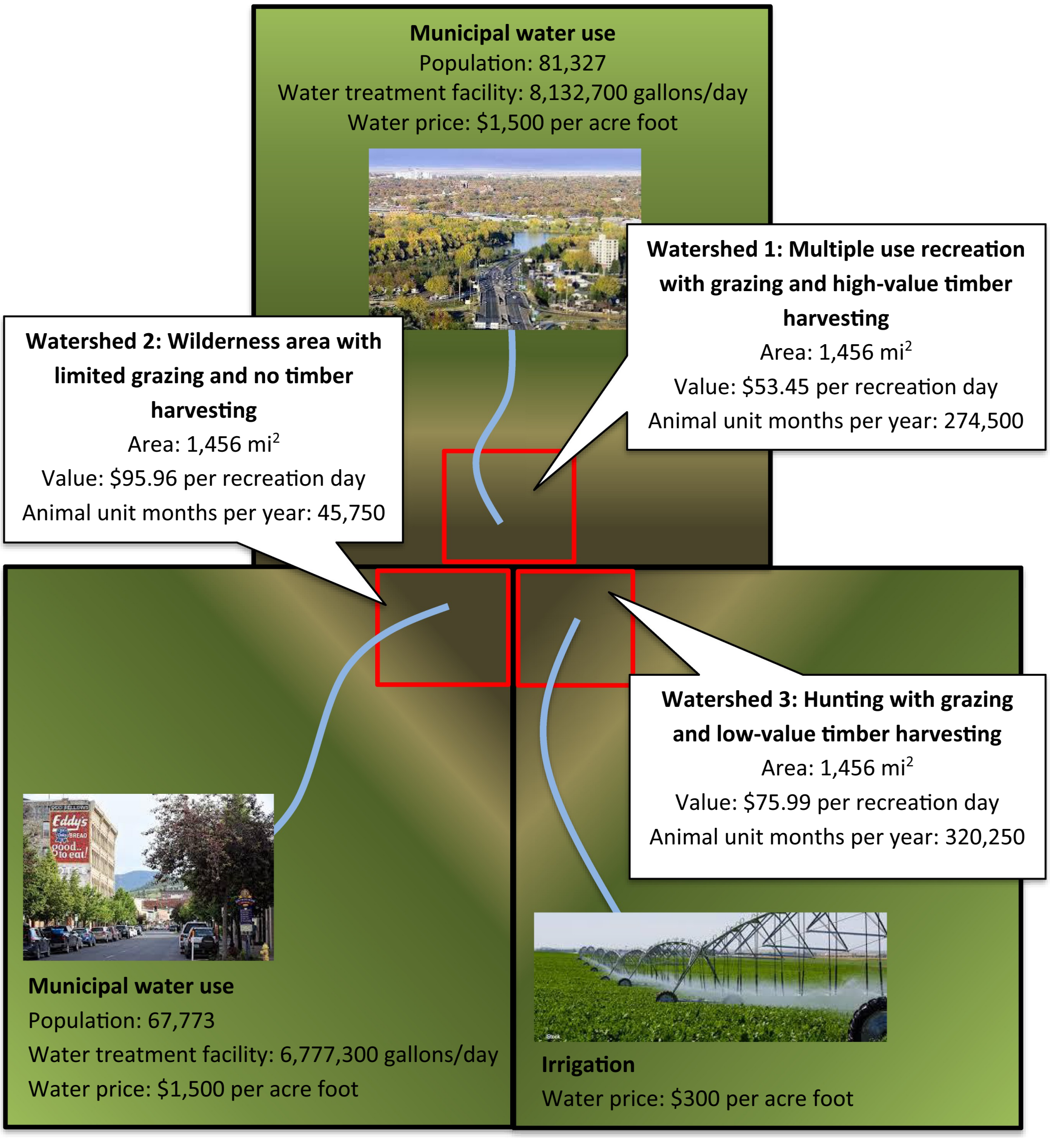

Fig. 1. Hypothetical forest landscape.

being used for irrigation, we assume there are no treatment costs associated with watershed 3.

As shown in Fig. 4, an increasingly forested watershed avoids more water treatment costs. Avoided treatment costs range from $0 when the forest cover is at its initial condition to $5 million per year when the forest in watershed 1 is at its carrying capacity. Increasing forest cover avoids water treatment costs but at a decreasing rate to reflect the reality that some treatment costs will have to be incurred regardless of the amount of forest cover in the watershed. Also, note that avoided costs are measured relative to the current state of the forest. Thus, costs are avoided when forest cover increases from its current state (i.e., positive avoided costs) but additional costs are incurred when forest cover decreases from its current state (i.e., negative avoided costs).

2.2.2. *Provision of water for municipal and agricultural uses (water yield)*

Paired catchment studies consistently show that water yield increases following removal of vegetation. For catchments in the Rocky Mountains, increases in water yield have been seen for vegetation removal > 15% of the catchment (Stednick, 1996), with 20–40 mm increases in yield for every 10% of vegetation removed in conifer forests (Bosch and Hewlett, 1982; Sahin and Hall, 1996). Without continued vegetation removal, however, water yields soon return to pre-harvest levels.[3] We account for these temporal changes in evapotranspiration rates using Zhang curves (Zhang et al., 2001) for each period with the following functional form:

$$\frac{ET}{P} = f\frac{1 + w\frac{Ez}{P}}{1 + w\frac{Ez}{P} + \left(\frac{Ez}{P}\right)^{-1}} + (1-f)\frac{1 + w\frac{Ez}{P}}{1 + w\frac{Ez}{P} + \left(\frac{Ez}{P}\right)^{-1}} \tag{12}$$

where w is the plant-available water coefficient specific to the type of vegetation, f is the percent of area in forests, and (1-f) is the percent of area in grasses. Zhang et al. estimate $E_z = 1410$ and $w = 2.0$ for trees, and $E_z = 1100$ and $w = 0.5$ for grasses. Average annual precipitation in Helena is about 49 in. according to usclimatedata.com, so Eq. (12) becomes

$$ET_i = 0.996369\ P + 0.003038\ P\ \pi_i = 48.82209 + 0.14884\ \pi_i \tag{13}$$

and water yield is given by.

$$Yield_i = P - ET_i = 49 - (48.82209 + 0.14884\ \pi_i) \tag{14}$$

[3] Forests also offer a service by retaining water and releasing it more slowly. Clearing too much vegetation could lead to more frequent flooding. Here we do not consider the possibility of flooding in this work.

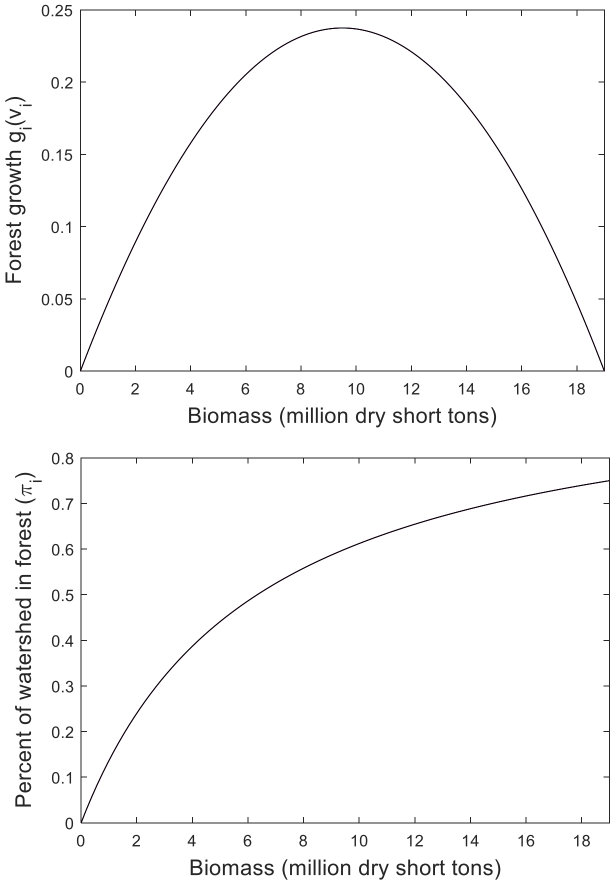

Fig. 2. Forest growth and cover in hypothetical watersheds.

Water values are based on a recent draft of the Montana State Water Plan (Tubbs, 2014). The largest consumptive use of water in Montana is irrigated agriculture, which accounts for 96% of all water diversions and 67% of all consumptive use (accounting for return flows). In the Yellowstone River basin, irrigation accounts for 83% of consumptive use. Although population is increasing in the study area, water demand for urban uses has not increased significantly; even in the most populated regions, households' share of consumptive use is below 4%. The marginal value of water in agriculture is routinely an order of magnitude lower than the marginal value of water for municipal uses, particularly in areas of recent population growth. Prices for municipal uses are $290 to $3145 per acre-foot, whereas prices for leased agricultural water diverted for in-stream conservation are $42 to $3614 per acre-foot. In general, prices increase for more senior water rights and when few other options for obtaining water exist in the area. Current rates paid by agricultural users of water from Bureau of Reclamation and Montana Department of Natural Resources and Conservation facilities are $2.32 to $7.50 per acre-foot per year, or a capitalized value of $76 to $244 per acre-foot. Accounting for delivery and operating costs, the capitalized costs of agricultural water range from $189 to $615 per acre-foot.

Given this information about water usage in Montana, the value of water yield for watersheds 1 and 2 are $p_{1,\ 2}ES_{1,\ 2} = 1500 * Yield_1$ and $p_{2,\ 2}ES_{2,\ 2} = 1500 * Yield_2$. The value of water yield in watershed 3 (where water is used for irrigation) is $p_{3,\ 2}ES_{3,\ 2} = 300 * Yield_3$. The value of water yield services in each watershed is shown in Fig. 4.

With small amounts of forest in the watersheds, the value of annual water yield services range from nearly $15 million in those watersheds that use the water for municipal purposes (watersheds 1 and 2) to under $3 million in watershed 3 where the water is used for irrigation purposes. When forests in each watershed are at or near carrying capacity, these services fall to a little over $5 million per year in watersheds 1 and 2 and less than $2 million per year in watershed 3. Note that the differences in the value of water yield services across watersheds is due only to differences in the value of the water that flows from these watersheds and not in the quantity of water.

2.2.3. Access to outdoor recreational opportunities

Provided vegetation is not too dense to restrict views or use, most studies find recreational visits increase with the age of the forest (Englin et al., 2001; Starbuck et al., 2006). Englin et al. (2001) show that recreational visits increase shortly after a fire when the forest is open, but decrease over time as brush thickens. Eventually, as the forest canopy closes, recreational visits increase with forest maturity. We focus on the latter stage of forest development and assume recreational benefits increase with forest vegetation, though at a decreasing rate:

$$Rec_i = a_i v_i^{0.3} \tag{15}$$

where Rec_i is the number of recreation visits in watershed *i*. We use the National Visitor Use Monitoring (NVUM) survey to determine a_i in watersheds 1 and 2. The number of recreation visitors can vary dramatically across forests. For example, according to the NVUM survey, the Custer National Forest had about 2 million visitors in 2003. This corresponds to 1077 visitors per square mile. We assume that if the forest in watershed 1 were at carrying capacity, it would attract an equal number of visitors after accounting for the size of the watershed: $a_1 = \frac{1{,}077 * A_1}{K_1^{0.3}} = 445{,}370$. Because a_1 is a function of watershed area, it also relates vegetation levels to vegetation density, which is more directly observed by recreational visitors. Using the USGS Benefits Transfer Toolkit (https://my.usgs.gov/benefit-transfer/activityCalc/index), we calculate the average value for general recreation to be $53.45 per person per day (2014 dollars). Assuming each visitor is on a day trip, the value of recreation in watershed 1 is $p_{1,3}ES_{1,3} = 53.45 * a_1 v_1^{0.3}$.

We assume fewer people visit the wilderness area in watershed 2. According to the National Visitor Use Monitoring survey for the Helena-Lewis and Clark National Forest, about 530,000 people visited the forests in 2003, or about 182 visitors per square mile. We assume that if the forest in watershed 2 were at carrying capacity, it would attract an equal number of visitors after accounting for the size of the watershed: $a_2 = \frac{182 * A_2}{K_2^{0.3}} = 75{,}079$. We assume the primary recreational activity in the wilderness area is hiking with an average value of $95.66 per person per day according to the USGS Benefits Transfer Toolkit. The value of recreation in watershed 2 is $p_{2,3}ES_{2,3} = 53.45 * a_2 v_2^{0.3}$. We assume hunters visit watershed 3 but general recreation visitors do not, $a_3 = 0$.

As shown in Fig. 4, the value of recreation services declines as the forest biomass declines. Relatively small changes in visits occur provided forest biomass stays above 1 million short tons. Once vegetation levels fall below 1 million short tons, recreational visits in each watershed drop off drastically. The differences in recreation values in watershed 1 and 2 are due to differences in the number of visitors and the value of the recreational activities. Watershed 2 attracts hikers who place a higher value on recreational experiences than general recreationists but attracts far fewer visits than watershed 1.

2.2.4. Wildlife habitat to support hunting

According to National Visitor Use Monitoring data, about a third of all visitors report hunting as their primary activity on the National Forests. Hunters prefer watersheds with a mix of forest areas that can be used for wildlife refuge and open areas that aid in hunting. To capture hunter preferences for a mix of forested and open areas, hunting ecosystem services are largest at intermediate levels of percent forest cover:

$$Hunting_i = g\pi_i - g\pi_i^2 \tag{16}$$

We parameterize g such that watershed 3 receives approximately

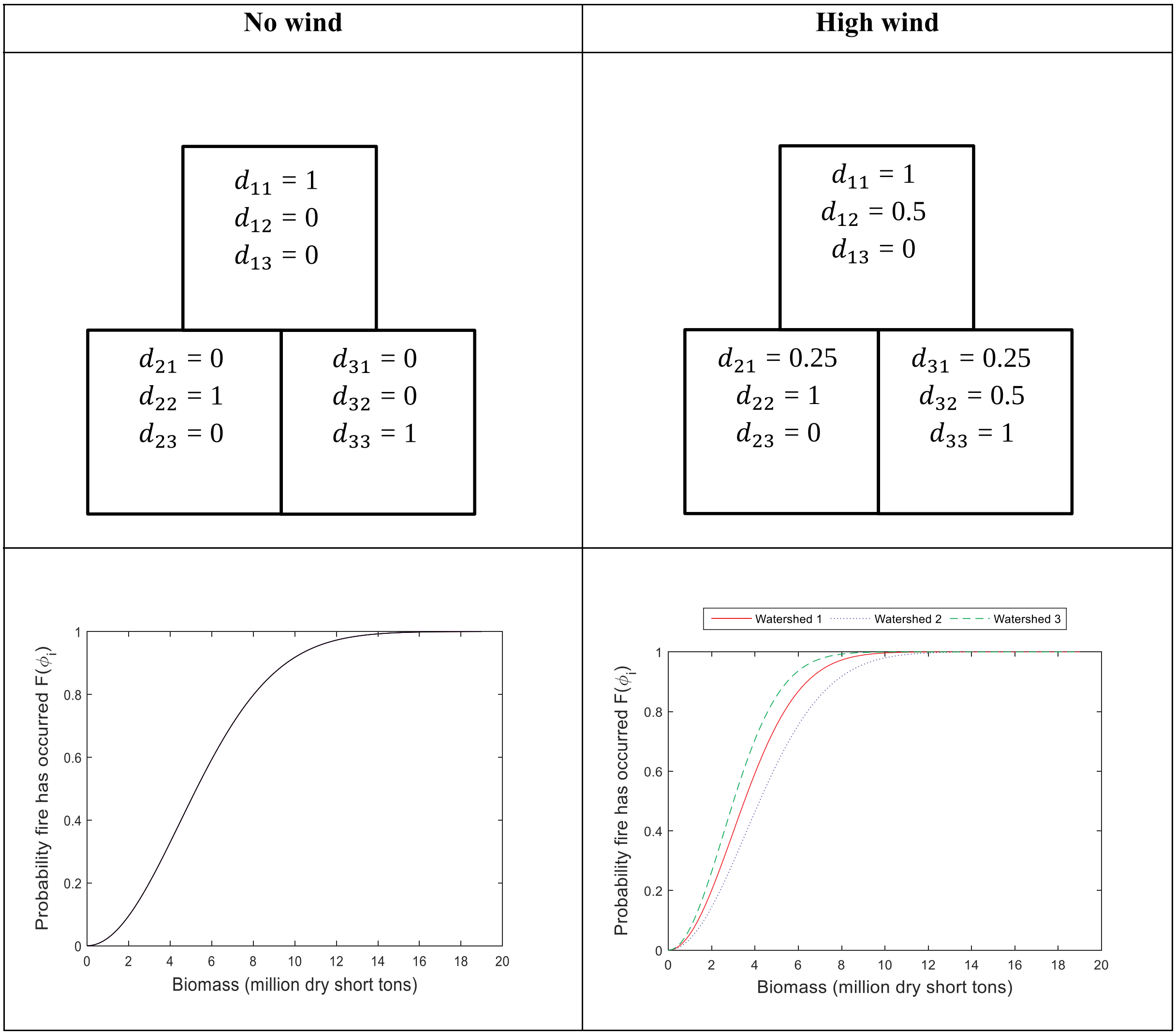

Fig. 3. Fire hazard and wind scenarios which determine spatial spillovers associated with fuel management.

400,000 annual hunting visits: $g = 1{,}551{,}300$.

According to the USGS Benefits Transfer Toolkit, a day of big game hunting in the area is valued at \$87.07 per person; small game hunting is valued at \$64.90 per person. We take the average of these two values to get a single value for hunting in the area equal to \$75.99 per person per day. The value of hunting ecosystem services in watershed 3 is $p_{3,4}ES_{3,4} = 75.99 * Hunting_3$. We assume no hunting takes place in the other two watersheds. Hunting visits, and thus hunting values, peak when percent forest cover is 50% or about 6 million short tons of biomass (see Fig. 4).

2.2.5. Provision of forage for grazing

Grazing livestock is a long-standing use of public lands. Since feed is the single greatest expense for most ranchers, grazing is the cornerstone of agricultural economies in many states in the western U.S. Provision of forage for grazing is a critical ecosystem service in support of this use. Popular forage species such as grasses and legumes grow in open areas or low-density forest. Thus, grazing ecosystem services, measured in total animal unit months per watershed, decline as forest cover increases:

$$Grazing_i = AUM_i \frac{(1 - \pi_i)A_i}{12.566} \tag{17}$$

where $(1 - \pi_i)A_i$ is the non-forested area in watershed i suitable for grazing and 12.566 is the average size (in square miles) of grazing allotments in the Helena-Lewis and Clark National Forest. The animal unit month per grazing allotment per year, AUM_i, represents the quality of grazing in watershed i. A lower value for AUM_i might reflect slower growing forage species or restrictions on the number of livestock that can be grazed on an allotment. For example, grazing opportunities in wilderness areas may be limited due to the lack of roads to access grazing allotments and restrictions on grazing activities following an official wilderness designation. As shown in Fig. 4, we assume relatively good grazing opportunities in watersheds 1 and 3 ($AUM_1 = 274{,}500$; $AUM_3 = 320{,}250$) but limited grazing opportunities in watershed 2 due to the presence of the wilderness area ($AUM_2 = 45{,}750$). Grazing fees are \$1.69 per AUM.

2.2.6. Provision of timber

Timber production is another long-standing use of many public forestlands. According to the 2016 Forest Inventory and Analysis (FIA), nearly 125,000 thousand board feet (MBF) was cut from national forests in US Forest Service Region encompassing the northern Rocky Mountain Region; though, only a little over 1% of this volume was harvested from the Helena-Lewis and Clark National Forest. To capture a reasonable relationship between the provision of timber and forest biomass, we calculated cut value per watershed and biomass per watershed for each national forest in the western U.S. (Forest Service Regions 1, 2, and 4). There is a clear positive relationship between these

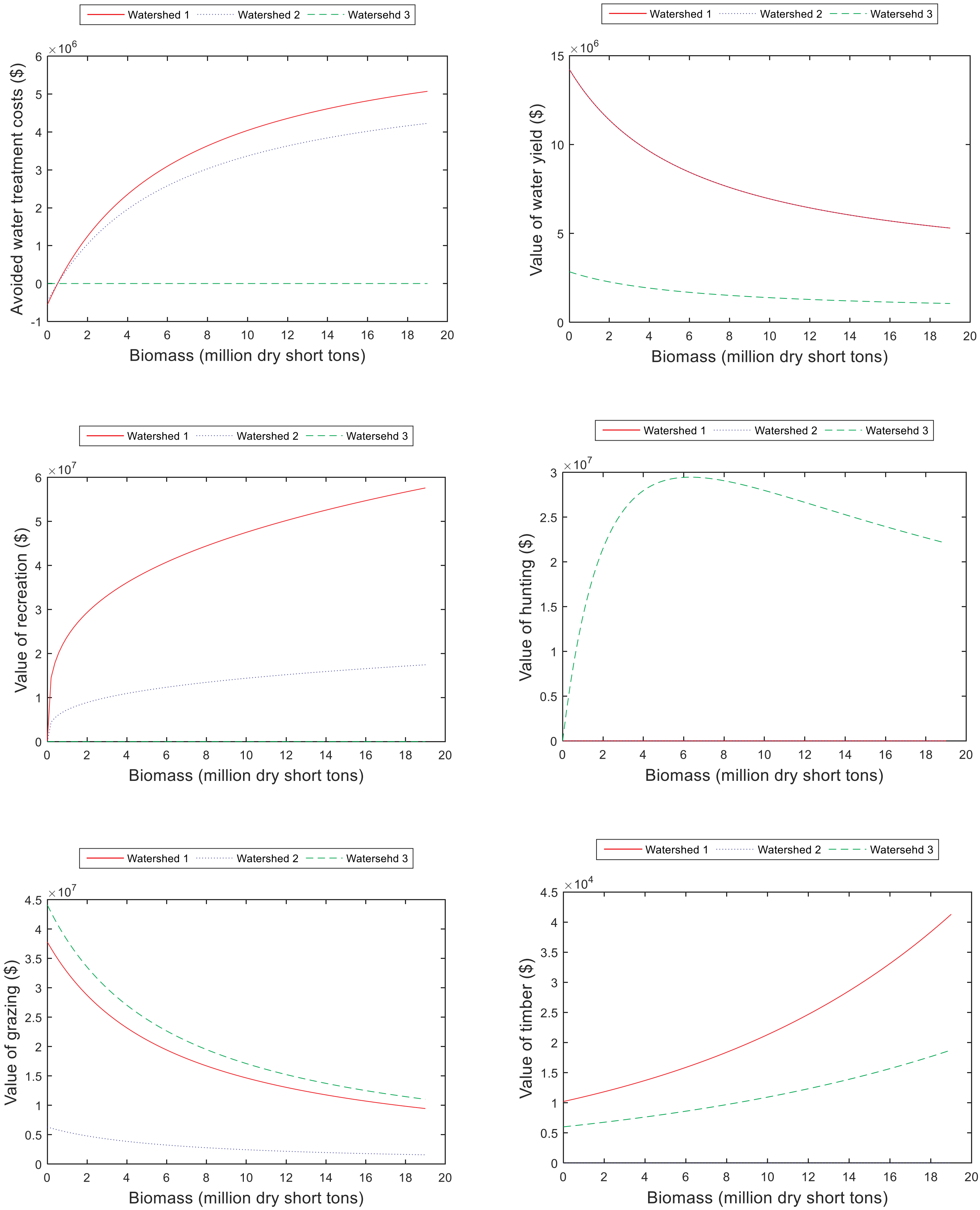

Fig. 4. Impact of forest biomass on the value of ecosystem services in each watershed.

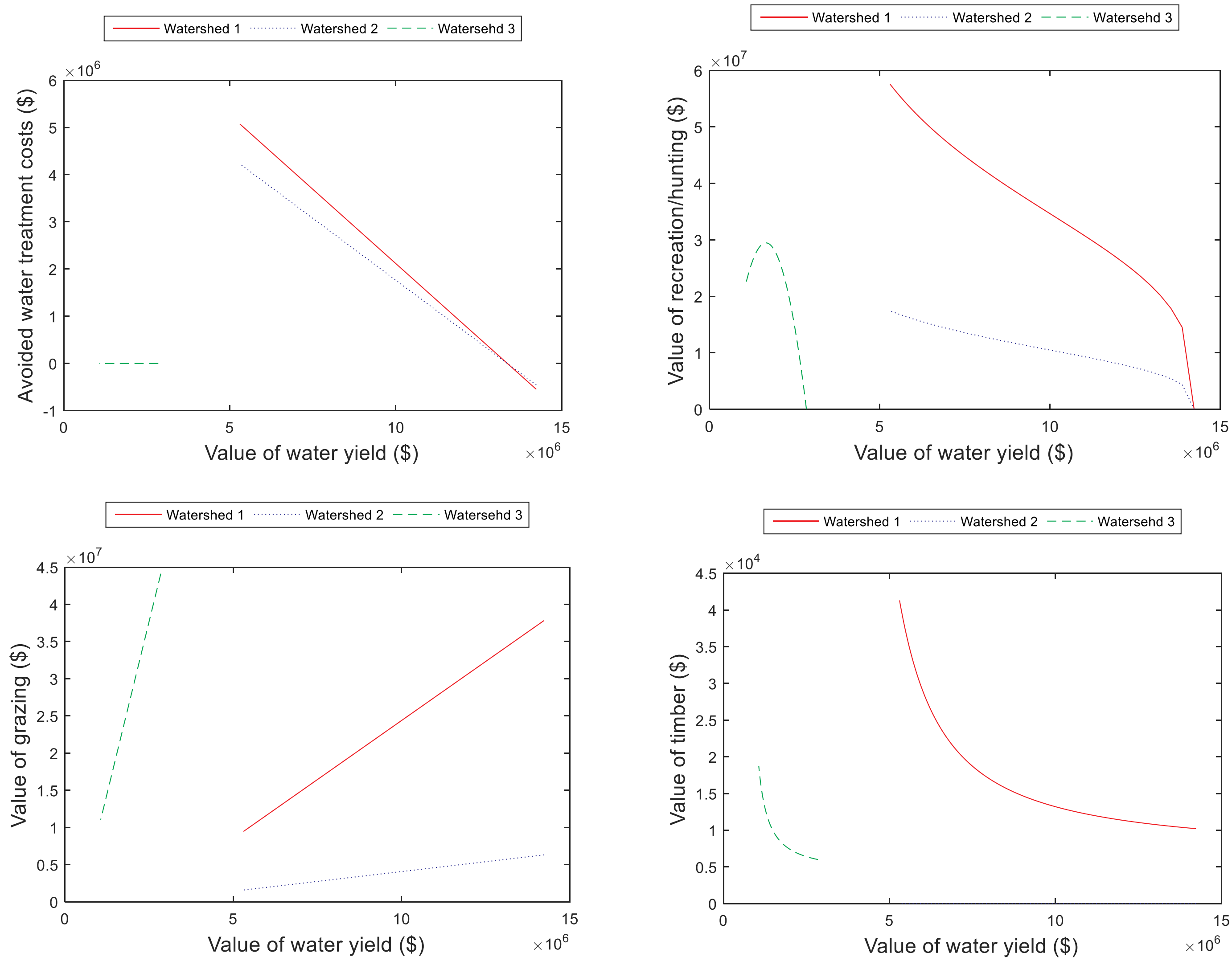

Fig. 5. Tradeoffs associated with water yield services. The top left panel compares water yield and water quality; the top right panel compares water yield and mixed use recreation in watersheds 1 and 2 and hunting in watershed 3; the bottom left panel compares water yield and grazing; the bottom right panel compares water yield and timber harvesting.

values indicating that heavily forested watersheds are more likely to experience timber harvests. Based on this FIA data, we assume an exponential relationship between the timber value in each watershed and the forest biomass:

$$Timber_i = \mu_{0,i} e^{\mu_{1,i} v_i} \tag{18}$$

We further assume that timber harvesting is allowed in watersheds 1 and 3 but prohibited in watershed 2 due to the presence of the wilderness area (see Fig. 4). For timber values in watershed 1, we fit an exponential curve to the FIA data: $\mu_{0,\ 1} = 10{,}208$; $\mu_{1,\ 1} = 0.074$. We assume timber values per unit of biomass are lower in watershed 3 due to a lack of roads for logging or the presence of lower-valued timber species: $\mu_{0,\ 2} = 6{,}000$; $\mu_{1,\ 2} = 0.06$.

3. Results

Fig. 5 illustrates the tradeoffs and complementarities involving water yield services implied by the relationships in Eqs. (10) through (18). Because water quality is increasing in the level of forest biomass, there is a tradeoff between water quality and water yield in watersheds 1 and 2. In fact, the provision of extremely large amounts of water for municipal and agricultural uses will cause local utilities to incur additional water treatment costs. Tradeoffs are also present when comparing water yield to timber values (bottom right panel in Fig. 4). Watersheds with low amounts of forest cover will provide relatively high water yield values but will not be areas conducive to high timber values. These tradeoffs are absent in watershed 2 since timber harvesting is prohibited in the wilderness area. Increases in water yield will also be accompanied by decreases in the value of recreation in watersheds 1 and 2. Since recreational visits increase with greater forest cover, efforts to increase water yield will be accompanied with losses in recreation values.

Complementarities arise when comparing water yield with hunting and grazing. In watershed 3, water yield and hunting are complements when water yield services are less than \$2 million. In this range the watersheds are characterized by high forest cover, and decreasing forest cover to increase water yield also attracts more hunting visits. Further decreases in forest cover increase water yield up to the maximum value of \$4 million per year, but hunting days decline with decreases in forest cover over this range. Removing all forest cover provides \$4 million per year in water yield values but completely eliminates hunting. Because grazing is decreasing in the level of forest biomass, grazing and water yield are complementary activities in all watersheds. These

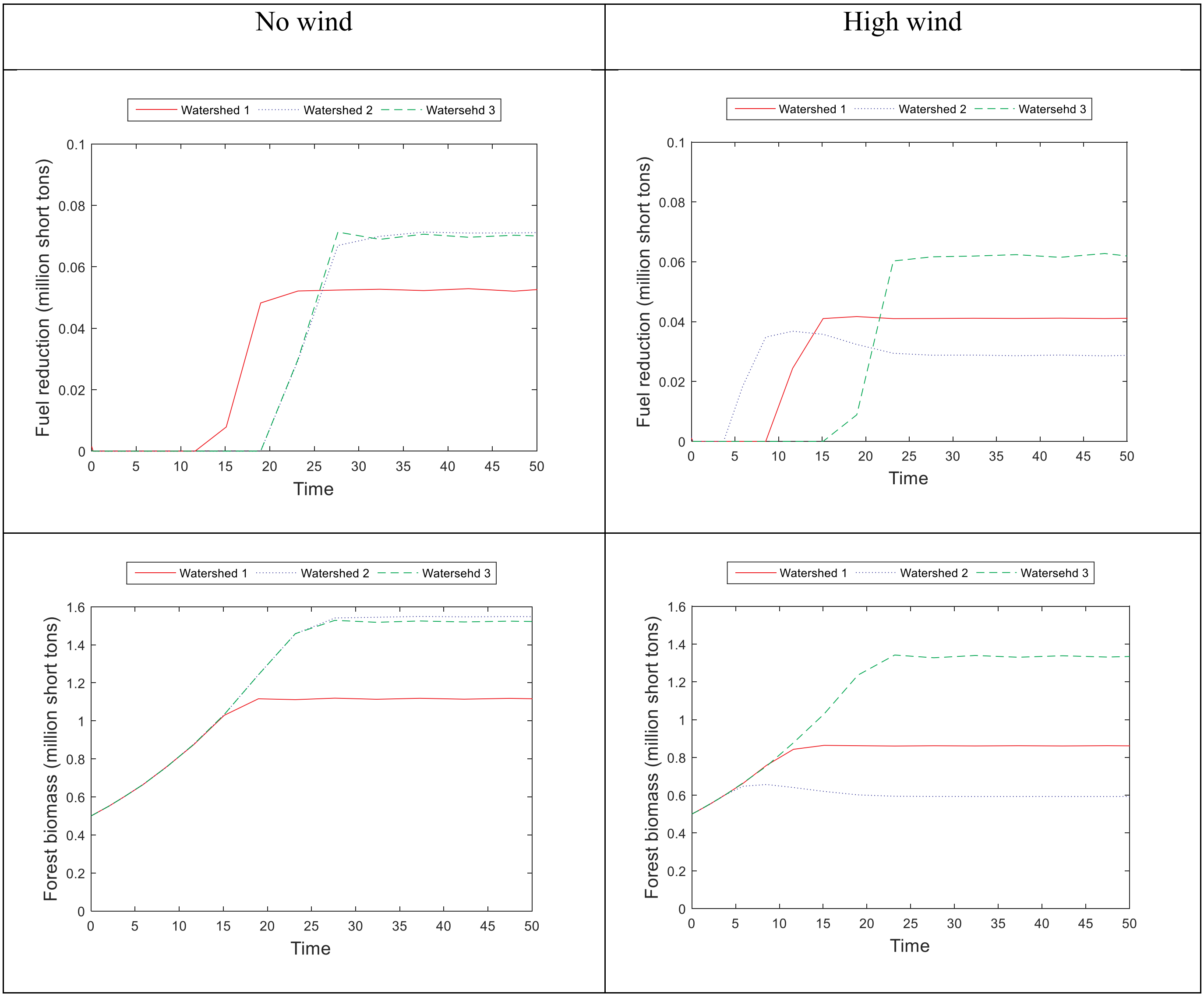

Fig. 6. Optimal fuel reduction strategies in three watersheds.

complementarities are most pronounced in watershed 3 where the value of water yield is lowest due to irrigation being the dominant water use and the value of grazing is highest due to the relatively high number of animal unit months that can be supported on grazing allotments in this watershed.

Assuming each watershed is initially in the same forest condition (v_{i0} = 0.5 = 500,00 short tons), Fig. 6 shows the optimal fuel management schedules and forest biomass that maximizes the present value of ecosystem services net of fuel management costs. These optimal fuel management strategies vary over time in response to the biomass growth dynamics, how biomass contributes to ecosystem services in Fig. 4, the implied tradeoffs and complementarities between ecosystem service values in Fig. 5, and the spatial dynamics of fire spread in Fig. 3.

Under the no wind scenario, no fuel management is optimal for at least ten years, which allows the biomass to increase. In eleven years, fuel management becomes optimal in watershed 1 but not in watersheds 2 and 3. Fuel management quickly plateaus at a level that removes 55,000 short tons of fuel per year and holds the biomass in watershed 1 at 1.1 million short tons. In nineteen years, fuel management becomes optimal in watersheds 2 and 3. Like watershed 1, the amount of fuel management quickly plateaus. However, the delay in fuel management allows the forest in watersheds 2 and 3 to increase to approximately 1.5 million short tons and requires a higher amount of fuel reduction (70,000 short tons of fuel removed annually) to offset the higher amount of forest growth and maintain this higher level of biomass. Even though the level of fuel management in watersheds 2 and 3 is higher than watershed 1, the delay in fuel management means a larger amount of forest biomass (and percent forest cover) in watersheds 2 and 3 compared to watershed 1.

The differences in fuel reduction strategies across watersheds are due to the different mixes of ecosystem service values in each watershed. Fuel reduction strategies impact ecosystem service values in the future by lowering the probability these ecosystem services will be lost to catastrophic fire. Fuel reduction strategies also impact *current* ecosystem service values through the reduction in biomass. To illustrate, Figs. 7 and 8 present the tradeoffs and complementarities involving fire hazard and ecosystem services implied by the relationships in Eqs. (10) through (18) and (2) through (6). These tradeoffs and complementarities illustrate how the mix of ecosystem service values in a watershed influence the incentives to engage in fuel reduction.

Using fuel reduction to lower the probability of a fire leads to a loss of ecosystem services that are increasing in biomass (i.e., water quality, access to outdoor recreation, provision of timber). For watersheds where these ecosystem service values are present, the loss of these

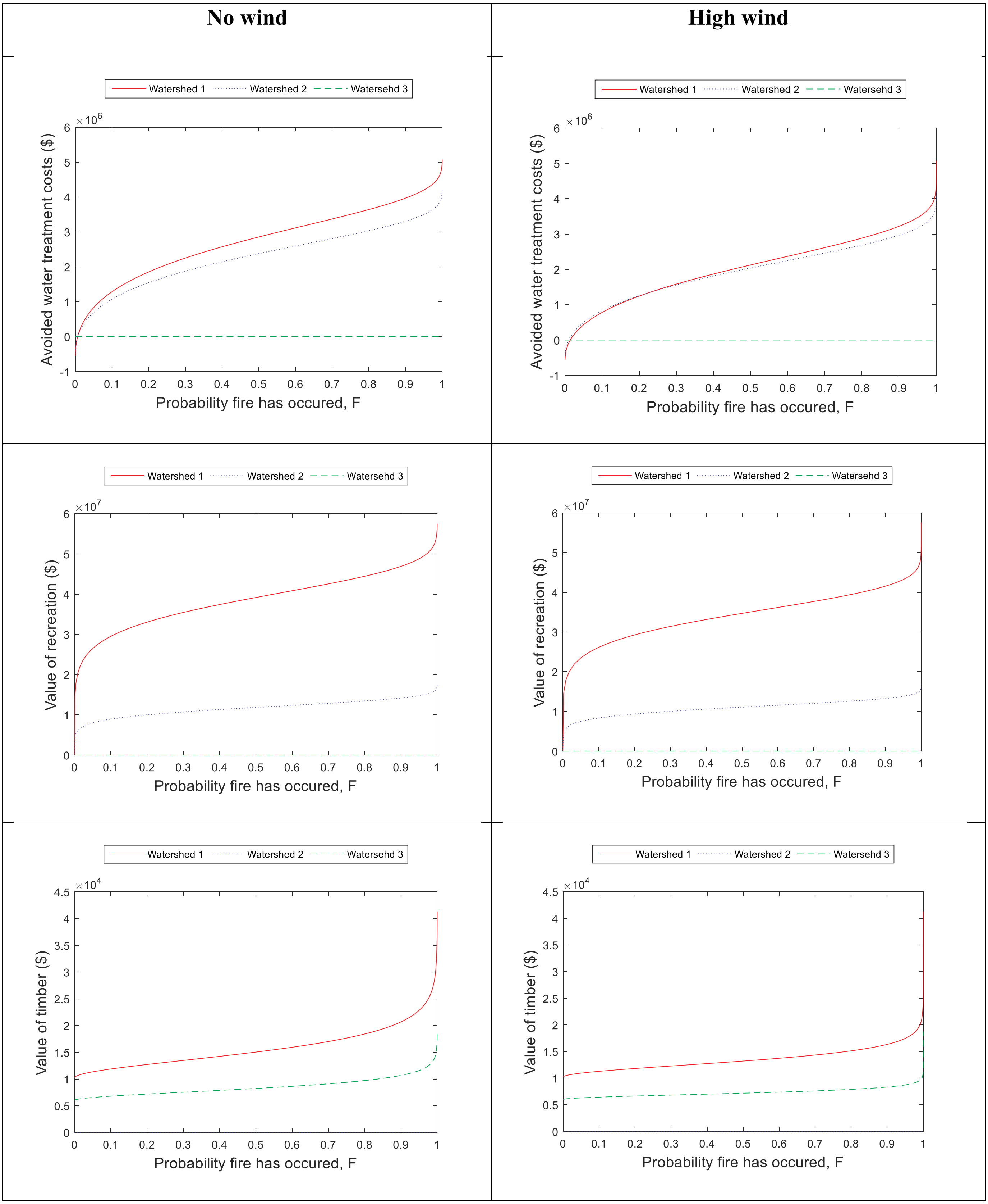

Fig. 7. Ecosystem service tradeoffs associated with reducing fire hazard.

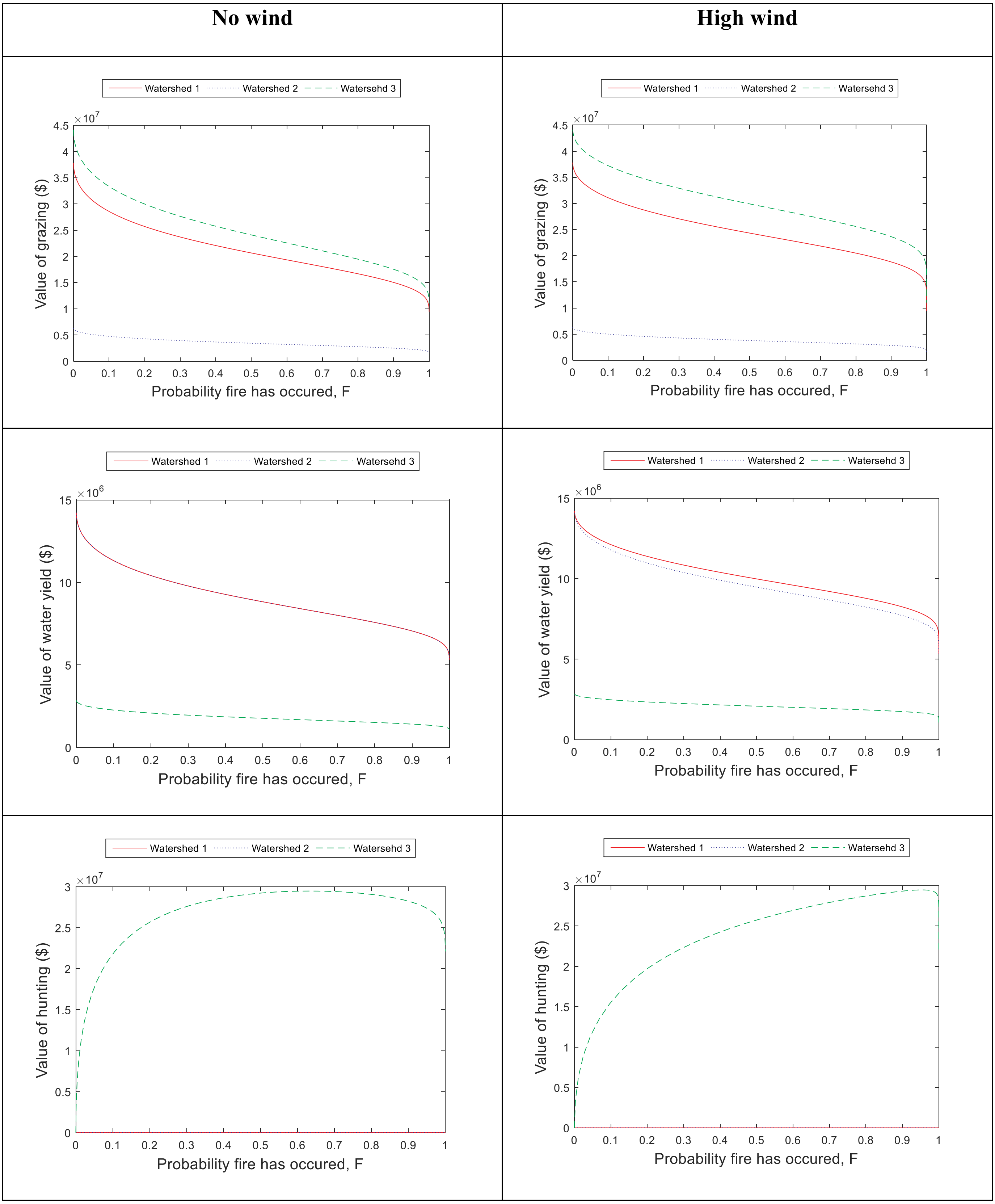

Fig. 8. Complementarities associated with reducing fire hazard and ecosystem service provision.

ecosystem service values represents an additional cost of fuel reduction. Thus, fuel management incurs the direct cost of removing fuel $C(m_1, m_2, m_3)$ and an indirect cost from immediate loss of ecosystem service values that are positively correlated with forest biomass. Likewise, fuel reduction leads to a gain of ecosystem services that are decreasing in biomass (i.e., provision of forage for grazing, water yield, and hunting). For watersheds where these ecosystem services are present, the gain in ecosystem service values represents an additional benefit of fuel reduction. Fuel management produces a future benefit by reducing the probability a fire occurs and an immediate benefit from gains in ecosystem service values that are positively correlated with forest biomass. For example, more fuel reduction will be optimal in watersheds that provide grazing opportunities than in watersheds that provide outdoor recreation opportunities, all other things equal.

The tradeoffs and complementarities between ecosystem services can exacerbate or mitigate these results. For example, water yield and provision of forage for grazing are complements (see Fig. 5) meaning that an increases the value of grazing also increases the value of water yield. Thus, a watershed that provides water yield services and forage for grazing provides a greater incentive for fuel reduction than a watershed that provides only forage for grazing. However, a watershed that provides forage for grazing and water quality may not provide a greater incentive for fuel reduction since these ecosystem service values imply a tradeoff.

In the high wind scenario, the overall fire hazard is higher which causes the optimal steady state level of forest biomass to be lower in all watersheds compared to the no wind scenario. However, the addition of wind further differentiates the optimal fuel management strategies across the three watersheds based on their location to one another. Biomass in watershed 2 contributes a larger share to the overall fire hazard compared to the other two watersheds. Due to the heightened fire risk originating from watershed 2, it is optimal to begin reducing fuel almost immediately in this watershed. The hastened fuel reduction and the lower levels of biomass that result mean that the long-run level of fuel reduction in watershed 2 is lower than in the no wind scenario. Early, aggressive fuel reduction allows for lower levels of fuel reduction in the future.

The biomass in watershed 1 also contributes to the fire hazard in watersheds 2 and 3, leading to earlier fuel reduction in watershed 1 compared to the no wind scenario. The earlier initiation of fuel reduction lowers the optimal steady state forest biomass from 1 million short tons in the no wind scenario to 0.8 million short tons in the high wind scenario. The biomass in watershed 3 does not contribute to fire hazard in any other watershed. However, the higher level of overall fire risk in the high wind scenario makes it optimal to begin fuel reduction slightly earlier than in the no wind scenario. This results in a slightly reduced level of long-run forest biomass in this watershed compared to the no wind scenario.

While fuel management decisions account for the spatial spillovers produced by high winds, decisions about the provision of ecosystem services typically do not account for these spatial spillovers. This becomes problematic since high wind does not have the same effect on the provision of all ecosystem services. Table 1 shows the long-run (steady state) levels of ecosystem services that result from optimal fuel management and the percent of these services that are produced in each watershed. Because spatial spillovers produced by high winds encourage lower levels of biomass in each watershed, the provision of ecosystem services that are enhanced by more biomass (water quality, outdoor recreation, hunting, and timber) will be lower while the provision of ecosystem services that are reduced by more biomass (water yield and grazing) will be higher. However, the spatial allocation of some ecosystem services also changes. With no wind, approximately equal amounts of water quality values are provided in watersheds 2 and 3. In the high wind scenario with spatial spillovers, almost all of the water quality services shift from watershed 2 to watershed 3. Likewise, wind causes some outdoor recreation values to shift from watershed 2 to watershed 1. Thus decisions about the location of ecosystem services may be reflecting forest conditions at distant locations.

To understand the effect of spatial spillovers in fire hazard on ecosystem service provision, return to the tradeoff and complementarity curves in Figs. 7 and 8. Spatial spillovers, in the form of high wind, lower the tradeoffs curves in Fig. 7. Compared to the no wind scenario, managers must allow for lower ecosystem service values to achieve the same fire hazard (or higher fire hazard to achieve the same level of these ecosystem services). Accounting for spatial spillovers in fire hazard will discourage the provision of these types of ecosystem services. Alternatively, spatial spillovers increase the complementarity curves in Fig. 8. In watersheds that generate large grazing and water yield values, managers receive higher ecosystem service values with the same fire hazard. The complementarities between fire hazard reduction and hunting are nearly eliminated in the high wind scenario. Accounting for spatial spillovers in fire hazard will encourage the provision of these types of ecosystem services. However, the effect of increased fire hazard due to wind is to flatten the tradeoff and complementarity curves in Figs. 7 and 8 which dampens the importance of accounting for complements and substitutes. That is, when fire risk is high, emphasis is placed on reducing vegetation to preserve future the future flow of ecosystem service values. Avoiding future losses in ecosystem services due to fire is more important than current losses due to fuel management.

4. Conclusion

Herein, we develop a spatially-explicit model of ecosystem service provision and wildfire hazard across several watersheds and parameterized it to an average watershed in Montana. The model captures several realistic features including the endogeneity of fire risk (i.e., fire hazard changes to reflect current forest conditions), the spread of fire over space, and tradeoffs and complementarities between ecosystem services. We use this model to highlight two primary implications for fuel management. First, some of the benefits of fuel management will be realized in distant locations. Wind, roads, and topography mean that fires that originate in one area will spread to neighboring areas implying that fire hazard in any area is influenced by the fire hazard in neighboring areas. Conversely, these spatial spillovers also mean that fuel reduction efforts lower fire risk in neighboring areas. Thus

Table 1
Total ecosystem services values and optimal allocation across watersheds

		Water quality	Outdoor recreation	Hunting	Water yield	Grazing	Timber
No wind	Total ($/year)	810	32,803,000	18,405,000	26,729,000	72,805,500	17,653
	Watershed 1	24%	75%	0%	47%	44%	63%
	Watershed 2	38%	25%	0%	44%	7%	0%
	Watershed 3	38%	0%	100%	9%	49%	37%
High wind	Total ($/year)	404	28,916,700	16,970,000	28,452,300	75,489,600	17,376
	Watershed 1	29%	79%	0%	45%	44%	63%
	Watershed 2	8%	21%	0%	46%	8%	0%
	Watershed 3	63%	0%	100%	9%	48%	37%

accounting for the way wind, roads, and topography spread fire over space will tend to increase the benefits of fuel management and enhance the economic case for fuel management. The magnitude of these spillover benefits of fuel management will depend on spatial connectivity of the fire hazard and the value of ecosystem services in neighboring areas.

Second, optimal fuel management will depend on both the total value of ecosystem services and the mix of different types of ecosystem services. Areas with larger ecosystem service values tend to be the focus of more extensive fuel management since fuel management lowers the probability of a catastrophic fire that would eliminate those services. However, the mix of ecosystem services and how those services are correlated with fuel reduction is also important. Ecosystem service values that are immediately enhanced by fuel reductions, such as forage for grazing, serve as an indirect benefit of fuel management. In contrast, ecosystem service values that are immediately harmed by fuel management, such as water quality, serve as an indirect cost of fuel management. Thus, tradeoffs and complementarities between ecosystem service values and fire hazard are critical determinants of optimal fuel management decisions. For example, a watershed that provides only grazing services will optimally receive more fuel reduction than a watershed that provides only water quality services even when the total value of ecosystem services prior to fuel management (the ecosystem service losses avoided through fuel management) are identical.

Communicating these values is an important component of the shared stewardship mission of federal agencies. This work has the potential to inform policy and planning related to reducing fire risk and providing a broad set of ecosystem services. Forests are tasked with moving from current to desired conditions for both fire risk and ecosystem service provisions. Achieving both of those at the same time may not always be possible. For example, our results illustrate how optimal fuel management may increase the probability that all ecosystem services will be provided into the future but it may also reduce the current amount of some ecosystem services. Furthermore, prevailing wisdom suggests human values on the land should dictate when and where fuels treatments take place. Our results illustrate how fuels treatments dictate when and where human values emerge.

Our work emphasizes that benefits from fuels treatments are not just about preserving values on the landscape; vegetation management itself affects ecosystem service values. Optimal management, therefore, will rely on non-market valuation studies and benefits transfer or ecosystem service values and elasticities with respect to forest vegetation. In most cases, such values are hard to come by, or are done with respect to a single ecosystem service. More studies are needed that take a comprehensive view of the myriad services natural systems provide (Warziniack et al., 2017a). While the ecological literature is mature in studies linking forest vegetation to sediment transport in a watershed, we are less informed about how hikers will respond to vegetation change. More work is needed in the social sciences on how human behavior reacts to both vegetation in the forest and risk of fire.

Our work, like much of the literature, relies on marginal values for ecosystem services. This approach may be fine for small changes on the forest (at least relative to the amount of forests in the region), but are not applicable at large scales or when a significant amount of the resource is affected. We acknowledge the need for more work with endogenous prices. Note that our work focuses on impacts from vegetation management, not impacts from fire. Impacts from fire are not likely to be marginal, but impacts from vegetation management are. Because managers historically have not had budgets available to affect landscape-scale vegetation, marginal values may be applicable more often than not.

Mostly, we see this work as a first step on a longer path of highlighting and, hopefully, addressing some of these issues. Future directions of research include developing case studies where valuation and management across multiple values is done accounting for the risk to these values. We also feel more work should be devoted toward appropriate scale for both management and ecosystem service valuation, particularly to address non-marginal changes in values that may result from landscape-scale management. In many cases, land managers are faced with cascading levels of management goals. That is, there may be landscape level goals that are managed at very local levels. Understanding how values interact and how management priorities interact across these spatial scales has not been adequately addressed in the literature.

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References

Aadland, D., Sims, C., Finnoff, D., 2015. Spatial dynamics of optimal management in bioeconomic systems. Comput. Econ. https://doi.org/10.1007/s10614-014-9442-y.

Albers, H.J., 1996. Modeling ecological constraints on tropical forest management: spatial interdependence, irreversibility, and uncertainty. J. Environ. Econ. Manage. https://doi.org/10.1006/jeem.1996.0006.

Amacher, G.S., 2004. Forest rotations and stand interdependency: ownership structure and timing of decisions. Nat. Resour. Model. 17, 1. https://doi.org/10.1111/j.1939-7445.2004.tb00127.x.

Arthaud, G.J., Rose, D.W., 1996. A methodology for estimating production possibility frontiers for wildlife habitat and timber value at the landscape level. Can. J. For. Res. https://doi.org/10.1139/x26-248.

Bagstad, K.J., Semmens, D.J., Waage, S., Winthrop, R., 2013. A comparative assessment of decision-support tools for ecosystem services quantification and valuation. Ecosyst. Serv. https://doi.org/10.1016/j.ecoser.2013.07.004.

Bekele, E.G., Lant, C.L., Soman, S., Misgna, G., 2013. The evolution and empirical estimation of ecological-economic production possibilities frontiers. Ecol. Econ. https://doi.org/10.1016/j.ecolecon.2013.02.012.

Binder, S., Haight, R.G., Polasky, S., Warziniack, T., Mockrin, M.H., Deal, R.L., Arthaud, G., 2017. Assessment and valuation of forest ecosystem services: state of the science review. Agric. For. Serv. North. Res. Station. 47, 1–47.

Bosch, J.M., Hewlett, J.D., 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. J. Hydrol. https://doi.org/10.1016/0022-1694(82)90117-2.

Bowes, M.D., Krutilla, J.V., 1985. Multiple use management of public forestlands. Handb. Nat. Resour. Energy Econ. 2, 531–569.

Calkin, D.E., Montgomery, C.A., Schumaker, N.H., Polasky, S., Arthur, J.L., Nalle, D.J., 2002. Developing a production possibility set of wildlife species persistence and timber harvest value. Can. J. For. Res. https://doi.org/10.1139/x02-056.

Duffield, J.W., Neher, C.J., Patterson, D.A., Deskins, A.M., 2013. Effects of wildfire on national park visitation and the regional economy: a natural experiment in the Northern Rockies. Int. J. Wildl. Fire 22, 1155–1166. https://doi.org/10.1071/WF12170.

Englin, J.E., Loomis, J.B., González-Cabán, A., 2001. The dynamic path of recreation values following a forest fire: a comparative analysis of states in the Intermountain West. Can. J. For. Res. 31, 1837–1944. https://doi.org/10.11369/cjfr-31-10-1837.

Hanula, J.L., Ulyshen, M.D., Horn, S., 2016. Conserving pollinators in North American forests: a review. Nat. Areas J. https://doi.org/10.3375/043.036.0409.

Hartman, R., 1976. The harvesting decision when a standing Forest has value. Econ. Inq. https://doi.org/10.1111/j.1465-7295.1976.tb00377.x.

Heines, B., Lenhart, S., Sims, C., 2018. Assessing the economic trade-offs between prevention and suppression of forest fires. Nat. Resour. Model. 31, e12159. https://doi.org/10.1111/nrm.12159.

Hessburg, P.F., Agee, J.K., Franklin, J.F., 2005. Dry forests and wildland fires of the inland Northwest USA: contrasting the landscape ecology of the pre-settlement and modern eras. For. Ecol. Manag. https://doi.org/10.1016/j.foreco.2005.02.016.

Hof, J.G., Bevers, M., Joyce, L., Kent, B., 1994. An integer programming approach for spatially and temporally optimizing wildlife populations. For. Sci. 40 (1), 177–191.

Johnson, K.N., Jones, D.B., Kent, B.M., 1980. Forest Planning Model (FORPLAN): User's Guide and Operations Manual. Land Management Planning Staff, USDA Forest Service.

Konoshima, M., Montgomery, C. a, Albers, H.J., Arthur, J.L., 2008. Spatial-endogenous fire risk and efficient fuel management and timber harvest. Land Econ. https://doi.org/10.3368/le.84.3.449.

Konoshima, M., Albers, H.J., Montgomery, C.A., Arthur, J.L., 2010. Optimal spatial patterns of fuel management and timber harvest with fire risk. Can. J. For. Res. https://doi.org/10.1139/X09-176.

Koskela, E., Ollikainen, M., 2001. Forest taxation and rotation age under private amenity valuation: new results. J. Environ. Econ. Manage. 42, 374–384. https://doi.org/10.1006/jeem.2000.1165.

Lee, C., Schlemme, C., Murray, J., Unsworth, R., 2015. The cost of climate change: ecosystem services and wildland fires. Ecol. Econ. https://doi.org/10.1016/j.ecolecon.

2015.04.020.
Minas, J.P., Hearne, J.W., Handmer, J.W., 2012. A review of operations research methods applicable to wildfire management. Int. J. Wildl. Fire. https://doi.org/10.1071/WF10129.
Nalle, D.J., Montgomery, C.A., Arthur, J.L., Polasky, S., Schumaker, N.H., 2004. Modeling joint production of wildlife and timber. J. Environ. Econ. Manage. https://doi.org/10.1016/j.jeem.2004.01.001.
Neary, D.G., Ryan, K.C., DeBano, L.F., 2005. Wildland Fire in Ecosystems, Effects of Fire on Soil and Water. USDA-FS Gen Tech. Rep. 4, 250. https://doi.org/10.1111/j.1467-7717.2009.01106.x.
Polasky, S., Nelson, E., Camm, J., Csuti, B., Fackler, P., Lonsdorf, E., Montgomery, C., White, D., Arthur, J., Garber-Yonts, B., Haight, R., Kagan, J., Starfield, A., Tobalske, C., 2008. Where to put things? Spatial land management to sustain biodiversity and economic returns. Biol. Conserv. 141, 1505–1524. https://doi.org/10.1016/j.biocon.2008.03.022.
Reed, W.J., 1984. The effects of the risk of fire on the optimal rotation of a forest. J. Environ. Econ. Manage. 11, 180–190. https://doi.org/10.1016/0095-0696(84)90016-0.
Rocca, M.E., Brown, P.M., Mac Donald, L.H., Carrico, C.M., 2014. Climate change impacts on fire regimes and key ecosystem services in Rocky Mountain forests. For. Ecol. Manag. https://doi.org/10.1016/j.foreco.2014.04.005.
Sahin, V., Hall, M.J., 1996. The effects of afforestation and deforestation on water yields. J. Hydrol. 178, 293–309. https://doi.org/10.1016/0022-1694(95)02825-0.
Sanchirico, J.N., Wilen, J.E., 1999. Bioeconomics of spatial exploitation in a patchy environment. J. Environ. Econ. Manage. https://doi.org/10.1006/jeem.1998.1060.
Schmidt, K.M., Menakis, J.P., Hardy, C.C., Hann, W.J., Bunnell, D.L., 2002. Development of Coarse-Scale Spatial Data for Wildland Fire and Fuel Management, General Technical Report RMRS-87.
Sills, E.O., Moore, S.E., Cubbage, F.W., McCarter, K.D., Holmes, T.P., Mercer, D.E., 2017. Trees at work: economic accounting for forest ecosystem services in the U.S. South. Agric. For. Serv. South. Res. Station. 226, 1–117.
Sims, C., Aadland, D., Powell, J., Finnoff, D.C., Crabb, B., 2014. Complementarity in the provision of ecosystem services reduces the cost of mitigating amplified natural disturbance events. Proc. Natl. Acad. Sci. 111, 16718–16723. https://doi.org/10.1073/pnas.1407381111.
Smith, M.D., Sanchirico, J.N., Wilen, J.E., 2009. The economics of spatial-dynamic processes: applications to renewable resources. J. Environ. Econ. Manage. https://doi.org/10.1016/j.jeem.2008.08.001.
Smith, H.G., Sheridan, G.J., Lane, P.N.J., Nyman, P., Haydon, S., 2011. Wildfire effects on water quality in forest catchments: a review with implications for water supply. J. Hydrol. 396, 170–192. https://doi.org/10.1016/j.jhydrol.2010.10.043.
Starbuck, C.M., Berrens, R.P., McKee, M., 2006. Simulating changes in forest recreation demand and associated economic impacts due to fire and fuels management activities. For. Policy Econ. 8, 52–66. https://doi.org/10.1016/j.forpol.2004.05.004.
Stednick, J.D., 1996. Monitoring the effects of timber harvest on annual water yield. J. Hydrol. 176, 79–95. https://doi.org/10.1016/0022-1694(95)02780-7.
Swallow, S.K., Wear, D.N., 1993. Spatial interactions in multiple-use forestry and substitution and wealth effects for the single stand. J. Environ. Econ. Manage. 25, 103–120. https://doi.org/10.1006/jeem.1993.1036.
Swallow, S.K., Talukdar, P., Wear, D.N., 1997. Spatial and temporal specialization in Forest ecosystem management under sole ownership. Am. J. Agric. Econ. 79, 311–326. https://doi.org/10.2307/1244132.
Thom, D., Seidl, R., 2016. Natural disturbance impacts on ecosystem services and biodiversity in temperate and boreal forests. Biol. Rev. Camb. Philos. Soc. https://doi.org/10.1111/brv.12193.
Tubbs, J.E., 2014. Montana State Water Plan.
Turner, R.K., Daily, G.C., 2008. The ecosystem services framework and natural capital conservation. Environ. Resour. Econ. https://doi.org/10.1007/s10640-007-9176-6.
Vogler, K.C., Ager, A.A., Day, M.A., Jennings, M., Bailey, J.D., 2015. Prioritization of forest restoration projects: Tradeoffs between wildfire protection, ecological restoration and economic objectives. Forests. https://doi.org/10.3390/f6124375.
Warziniack, T., Finnoff, D., Apriesnig, J., 2017a. Description of the general equilibrium model of ecosystem services (GEMES). Gen. Tech. Rep. RMRS-GTR-359. Fort Collins, CO: USDA For. Ser., Rocky Mountain Research Station. 14.
Warziniack, T., Sham, C.H., Morgan, R., Feferholtz, Y., 2017b. Effect of Forest cover on water treatment costs. Water Econ. Policy 03, 1750006. https://doi.org/10.1142/S2382624X17500060.
Zhang, L., Dawes, W.R., Walker, G.R., 2001. Response of mean annual evapotranspiration to vegetation changes at catchment scale. Water Resour. Res. 37, 701–708. https://doi.org/10.1029/2000WR900325.