FUNDAMENTAL RESEARCH

For. Sci. 61(■):000–000
http://dx.doi.org/10.5849/forsci.14-030

economics

The Economics of Ecological Restoration and Hazardous Fuel Reduction Treatments in the Ponderosa Pine Forest Ecosystem

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In this article, we develop a simulation model of the benefits and costs of managing the ponderosa pine forest ecosystem in the southwestern United States. Using the model, we evaluate and compare the economic benefits and costs of ecological restoration and hazardous fuel reduction treatments. Both treatment approaches increase the expected number of low-severity wildfires, which can promote postfire rehabilitation. Hazardous fuel reduction treatments are likely to reduce expected wildfire suppression costs, but not enough to offset the costs of implementing treatments. Conversely, ecological restoration treatments do not necessarily reduce expected wildfire suppression costs but fully restore the ecosystem in more than half of the simulation runs, which lowers the need for future fire suppression and reduces the chance of conversion to nonforest, alternative stable states. We find that the choice between hazardous fuel reduction and ecological treatments will depend on the management objective being pursued, as well as on site-specific factors such as the wildfire return interval and the economic value of biomass removed.

Keywords: wildfire, simulation model, state-and-transition model, ecological thresholds, policy

One of the primary drivers of increasing wildfire hazard in the United States and throughout the world is forest overcrowding that is attributed, in part, to historical suppression of low-severity wildfires in frequent fire regimes (Covington et al. 1994, Fulé et al. 2004, Ohlson et al. 2006, Wang et al. 2007). This overcrowding has increased the frequency of severe, stand-replacing crown fires that are often expensive and difficult to suppress, damage property and infrastructure, endanger human life, and lead to undesirable, and sometimes irreversible, changes in ecosystem function (Covington and Moore 1994, Hessburg et al. 1994, Finkral and Evans 2008, Benayas et al. 2009, Evans et al. 2011). Fuels and fire managers seeking to reduce expected wildfire costs and damages in this context have two treatment options: hazardous fuel reduction treatments (HFRTs), which focus on reducing fuel loading and changing fuel characteristics to achieve short-term reductions in wildfire hazard; or ecological restoration treatments (ERTs), which are a composite of treatments targeting a more long-term reduction in wildfire suppression costs and damages by restoring historical stand composition and density (Hunter et al. 2007, Evans et al. 2011). Despite the fact that escalating wildfire activity related to forest overcrowding is a pervasive issue throughout the world, there is no empirical research that has directly compared the economic benefits and costs of ERTs and HFRTs in this setting (Kline 2004, Mercer and Prestemon 2008).

In this article, we use simulation methods to evaluate and compare the economic benefits and costs of ERTs and HFRTs in the ponderosa pine (*Pinus ponderosa* P.&C. Lawson) forest ecosystem of the Colorado Plateau in the western United States (henceforth the "PIPO ecosystem"). We focus on the PIPO ecosystem because recent escalations in wildfire extent and severity related to forest overcrowding along with the corresponding increases in wildfire suppression costs and damages are a major public policy concern in the PIPO ecosystem, as is the role that ERTs and HFRTs should play in addressing this issue (Hunter et al. 2007, Evans et al. 2011, Ecological Restoration Institute 2013). Our approach simulates long-run wildfire activity and suppression costs with and without ERTs and HFRTs and allows us to analyze how the two treatment approaches, both of which reduce fuel accumulations, influence the expected

Manuscript received February 21, 2014; accepted June 9, 2015; published online July 16, 2015.

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Acknowledgments: *We acknowledge support from the Ecological Restoration Institute of Northern Arizona University, the USDA Forest Service, and the USDA Agricultural Research Service's "Areawide Pest Management Program for Annual Grasses in the Great Basin Ecosystem." These sponsors did not play any role in the design of this study, in the article preparation, or in the decision to submit this article for publication. We thank Krista M. Gebert and Michael Hand from the USDA Forest Service Rocky Mountain Research Station for providing us with the wildfire suppression cost data used in this study. The authors retain sole responsibility for the results and views presented in this article. Northern Arizona University and University of Nevada are equal opportunity providers.*

total number of wildfires, the instance of beneficial wildfires (which can facilitate ecosystem recovery postfire) relative to degrading wildfires (which can damage ecosystem health), and expected wildfire suppression costs. These reductions in expected wildfire suppression costs are compared with treatment costs to determine whether either treatment approach can be justified on the basis of wildfire suppression cost savings alone (i.e., whether the net benefits of treatment are positive). In addition, we consider how our conclusions change when the economic value of biomass removed during implementation of ERTs, which are more likely to generate revenue from biomass removal than HFRTs, is included in the analysis.

The analysis in this article illustrates how the expected outcomes of ERTs and HFRTs are influenced by site-specific ecological and economic conditions. In particular, the analysis considers how expected outcomes are influence by factors that vary with ecological conditions, such as wildfire frequency, expected wildfire suppression costs, and expected treatment longevity, as well as with economic factors such as treatments costs and the market price of biomass removed while performing ERTs. This analysis can help set realistic expectations for the impacts of ERT or HFRT at a particular site. Previous authors have noted that stakeholders often form unrealistic expectations about what can be achieved through treatment (Finney and Cohen 2003) and that these unrealistic expectations can "lead to polarization of what should be a nondivisive issue" (Reinhardt et al. 2008). In addition, the analysis allows us to consider broader policy questions, such as when, if ever, will both ERTs and HFRTs simultaneously be used as part of an optimal management strategy at the landscape scale, where to perform treatment on a heterogeneous landscape given a limited budget for management, and how anticipated changes in wildfire regimes, market conditions (e.g., treatment costs and biofuel prices) and other factors are likely to change the relative attractiveness of ERTs and HFRTs.

To the best of our knowledge, this article represents the first attempt in the literature to both analyze the long-run economic benefits and costs of ERTs in a forested ecosystem and evaluate and compare the economic benefits and costs of ERTs and HFRTs. There have been several studies that have analyzed the economic benefits and costs of fuel treatments in forest ecosystems. The analytical frameworks developed in these previous studies, however, are not suited to analyzing and comparing ERTs and HFRTs. For example, many previous studies begin their analysis by assuming the impacts of fuel treatments on wildfire activity rather than modeling these impacts (Snider et al. 2006, Prestemon et al. 2012, Huang et al. 2013), so that any comparison of the economic benefits and costs of alternative treatment strategies in these frameworks would be tautological. Similarly, several studies that do model the effect of fuel treatments on wildfire behavior do not incorporate dynamic ecological models into their analysis (Mercer et al. 2007, Thompson et al. 2013), and, as such, cannot evaluate how the HFRTs and, in particular, the ERTs influence the dynamic trajectory of the ecosystem and thus long-run wildfire activity and suppression costs. In a recent study, Houtman et al. (2013) incorporated ecosystem dynamics into their economic framework using a state-and-transition (STM) model based on wildfire condition classes. Rather than evaluating the economic benefits and costs of fuel treatments (either ERTs or HFRTs), however, their analysis focuses on whether letting certain wildfires burn rather than being actively suppressed results in net wildfire suppression cost savings over 100 years.

The simulation model developed and parameterized in this article captures the ecological dynamics in the PIPO ecosystem using an approach based on the STM ecological framework (sensu Westoby et al. 1989). In the STM framework, an ecosystem is described as being in one of several ecological states that are separated by ecological thresholds. The STM framework has been used to model (both mathematically and conceptually) the PIPO ecosystem in the Southern Colorado Plateau (Fischer and Bradley 1987, Moir and Dieterich 1988, Sesnie and Bailey 2003, Savage and Mast 2005), as well as many other ecosystems in the United State and throughout the world (Allen-Diaz and Bartolome 1998). The STM framework allows us to incorporate ecosystem dynamics in our simulation, including the role of wildfire as a catalyst for transitions between ecological states, which is necessary to analyze the long-run impact of ERTs and HFRTs on ecological condition, wildfire behavior, and suppression costs. In addition, the STM framework allows us to readily incorporate stochastic wildfire, suppression costs, and ecological transitions in our simulation model. Incorporating these stochastic elements in the model is particularly important for evaluation of ERTs and HFRTs, for which policymakers may be more interested in avoiding instances of the least desired outcome, such as large, high-severity wildfires, than in maximizing the expected net benefits (benefits minus costs) of the average outcome.

It is important to recognize that this article considers the financial benefits of ERTs and HFRTs only in terms of wildfire suppression cost savings and the revenues from biomass removed during treatment. In considering only this narrow set of financial benefits, our analysis ignores potential reductions in the other market (e.g., damage to housing and other infrastructure) and nonmarket (e.g., postfire erosion) costs of wildfire from implementing treatments, as well as the potential benefits of fuel treatments in terms of enhanced ecosystem services. Ignoring these additional benefits implies that our analysis understates the total economic benefits of ERTs and HFRTs. This shortcoming of the analysis, however, is in keeping with the approach taken throughout this article, in which parameters and assumptions are chosen deliberately to avoid overstating the benefits or understating the costs of fuel treatment. As a result of this conservative approach, our results provide a lower bound for the economic benefits of fuel treatment and mean that we are able to reach definite conclusions about the circumstances under which the expected benefits of fuel treatment are likely to outweigh the costs but not vice versa.

Additionally, there are several other limitations to the modeling approach taken in this article. For example, the analysis does not consider the benefits of fuel treatments in terms of reductions in wildfire hazard and expected suppression costs on land adjacent to the treatment site. Similarly, by analyzing the benefits and costs of fuel treatment on a representative parcel of land, the approach ignores the influence of landscape features that can influence the wildfire suppression cost savings from fuel treatment (e.g., the proximity of the treatment site to wildland-urban interface communities), as well as treatment costs. These and other limitations to our modeling approach are discussed in greater detail below.

Materials and Methods

Stylized STM

We capture ecosystem dynamics for the PIPO ecosystem using a STM model. STMs describe an ecosystem as being in one of several ecological states that are separated by ecological thresholds. The PIPO ecosystem analyzed in this article is a broad ecological classification that refers to several different ecological sites, each of which

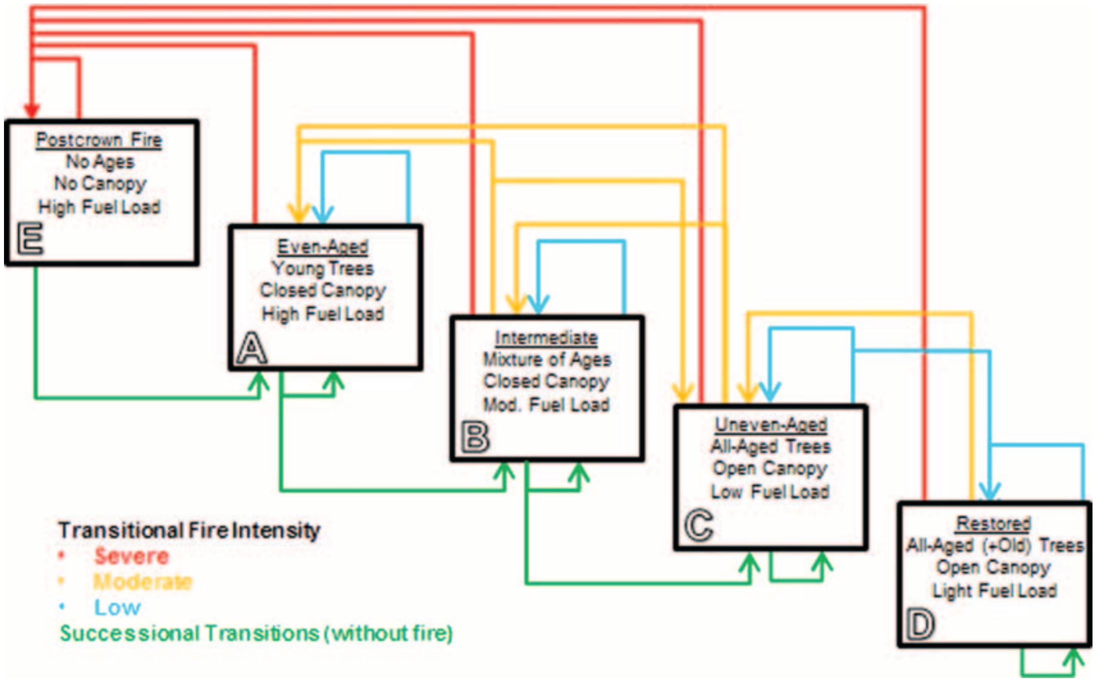

Figure 1. PIPO: stylized STM.

can be represented by its own STM (Graham and Jain 2005). Because the economic data (wildfire suppression costs, treatment costs, and others) are organized according to these broad classifications, we develop a stylized STM that is intended to be broadly representative of ecological sites found in PIPO ecosystem.

Our stylized STM for the PIPO system consists of five ecological states (Figure 1). PIPO-A is characterized by a closed-canopy forest with even-aged trees and a high fuel load. Because of the high fuel loads, wildfires in PIPO-A are often intense crown fires that cause the system to transition to the postcrown state, PIPO-E. If the system remains in PIPO-A for several decades without fire or a fuel treatment, it will eventually transition to PIPO-B, which is characterized by a closed-canopy forest with a mixture of tree ages and moderate fuel loads. Wildfires in PIPO-B are less likely to be intense crown fires than in PIPO-A. Without wildfire or treatment, the system will eventually transition to PIPO-C, which is characterized by an open-canopy forest with trees of all ages and a low fuel load.[1] If the system remains in PIPO-C for a number of years and experiences low-severity wildfires and/or fuel treatments at regular intervals, it will transition to PIPO-D, the restored state. PIPO-D is characterized by an open canopy and light fuel load. Regular low-severity wildfires maintain the system in PIPO-D. The final state, PIPO-E, is the postcrown fire state, which is characterized by high fuel loads comprised of grasses and shrubs. After several decades in PIPO-E without wildfire, the system may return along successional pathways to PIPO-A, although it has been suggested that PIPO-E may transition to an alternative stable state of grass and shrub land (Savage and Mast 2005).

Simulation Methods

Simulation Model Description

The simulation model considers the progression of the PIPO ecosystem with and without fuel treatment. The analysis focuses on differences between the with- and without-treatment scenarios in terms of wildfire occurrence, wildfire suppression costs, and other factors. The initial state of the PIPO ecosystem is varied across simulation runs to evaluate how the outcomes of ERTs and HFRTs change, depending on the ecological state of the land being treated. In the discussion of the results, when we state that the initial state of the model is, for example, PIPO-A, we are assuming that the initial state of the system is PIPO-A with the maximum number of years before the system transitions to PIPO-B via ecological succession (e.g., 100 years under our "baseline" assumptions; see below for further detail). All results are reported on a per-acre basis over a 200-year time horizon, with a 3% discount rate used to calculate present values.[2] All monetary results are presented in constant 2011 dollars.

The strength of the model is that we can treat as stochastic parameters wildfire occurrences, wildfire severity (given that a wildfire has occurred), per-acre wildfire suppression costs in each year given wildfire occurrence and severity, and postwildfire transitions between ecological states in each year. Each run of the model considers the progression of the system in the with- and without-treatment scenarios with different randomly generated realizations of these stochastic parameters in each year.[3] The stochastic parameters lead to substantial variations in key variables, including wildfire suppression cost savings, between model runs. For this reason, results in this article are reported for 10,000 model runs unless otherwise noted, and the discussion focuses on the expected values of key variables, which are calculated as the mean of each resampled variable over 10,000 model runs.

Figure 2 explains the sequence of events in year t in the mth run of the simulation model. The state of the system in year t is described by two state variables: $S_{t,m}^{V}$ is the ecological state in year t ($S_{t,m}^{V}$ can be either PIPO-A, -B, -C, -D, or –E) and $s_{t,m}^{V}$ is the number of years that the system has been in $S_{t,m}^{V}$ in year t. The variable $s_{t,m}^{V}$ is necessary because the system can only remain in a given state for a finite amount of time in the absence of management treatment or wildfire before transitioning to a new state. The superscript V indicates the treatment scenario; $V = T$ for a "treatment" scenario (T = ERT or HFRT) and $V = NT$ for a "no treatment" scenario.

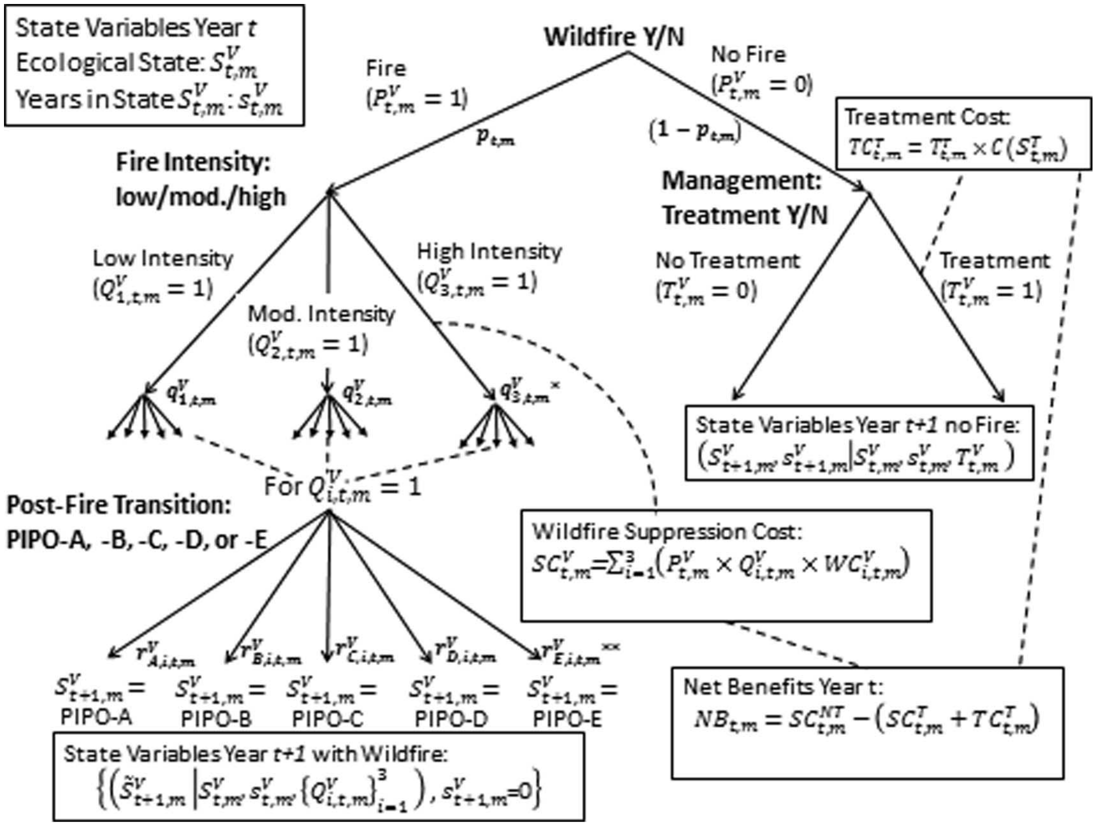

Figure 2. Simulation model: *m*th model run, year *t*, treatment scenario $V = T$ (ERT or HFRT), NT (no treatment). *$q^V_{1,t,m} + q^V_{2,t,m} + q^V_{3,t,m} = 1$. **$r^V_{A,i,t,m} + r^V_{B,i,t,m} + r^V_{C,i,t,m} + r^V_{D,i,t,m} + r^V_{E,i,t,m}) = 1$.

In the treatment scenarios, ERTs or HFRTs may take place in years in which wildfire does not occur. The model assumes that the year begins before the wildfire season and that wildfire occurs or does no not occur before treatments take place. Each model run considers a treatment schedule that determines whether a treatment occurs in year t given $S^V_{t,m}$ and $s^V_{t,m}$. The variable $T^T_{t,m}$ is equal to 1 if a treatment occurs in year t and 0 otherwise. In the no treatment scenario, $T^{NT}_{t,m} = 0$ for all $S^{NT}_{t,m}$ and $s^{NT}_{t,m}$. If treatment is performed in year t, a state-specific treatment cost, $C(S^V_{t,m})$, is incurred. The treatment schedules for ERTs and HFRTs that are assumed in the model and the state-specific treatment costs for ERTs and HFRTs used in the model are described below.

Figure 2 also illustrates how wildfire is included in the model. The random variable $\tilde{P}^V_{t,m}$ is equal to 1 if a wildfire occurs in year t and 0 otherwise. The probability that a wildfire occurs in year t (i.e., the probability that $P^R_{t,m} = 1$ in year t) is $p^V_{t,m}$, which depends on the ecological state in year t, $S^V_{t,m}$. In years when a wildfire occurs, the random variable $\tilde{Q}^V_{1,t,m}$ ($\tilde{Q}^V_{2,t,m}$, $\tilde{Q}^V_{3,t,m}$) is equal to 1 if the wildfire is low (moderate, high) severity and 0 otherwise. The probability that $\tilde{Q}^V_{1,t,m}$ ($\tilde{Q}^V_{2,t,m}$, $\tilde{Q}^V_{3,t,m}$) is equal to 1 is $q^V_{1,t,m}$ ($q^V_{2,t,m}$, $q^V_{3,t,m}$), which depends on the ecological state in year t, $S^V_{t,m}$, and, in the case of HFRTs, on whether an HFRT has been performed in the previous 15 years, $\{T^{\text{HFRT}}_{u,m}\}^{t-1}_{u=t-15}$.[4] When a wildfire occurs in year t, wildfire suppression cost is a random variable, $WC^V_{t,m}$, that is drawn from a state- and fire severity-specific distribution of per-acre wildfire suppression costs.

In years when a wildfire occurs, the postwildfire transitions between ecological states are stochastic. In particular, the probability that the random variable $\tilde{S}^V_{t+1,m}$ is equal to PIPO-A (-B, -C, -D, -E) in year $t + 1$ is $r^V_{A,i,t,m}$ ($r^V_{B,i,t,m}$, $r^V_{C,i,t,m}$, $r^V_{D,i,t,m}$, $r^V_{E,i,t,m}$), which depends on the ecological state at time t, $S^V_{t,m}$, and whether the wildfire was low ($Q^V_{1,t,m} = 1$), moderate ($Q^V_{2,t,m} = 1$), or high ($Q^V_{3,t,m} = 1$) severity.[5] In the year after a wildfire occurs, $s^V_{t+1,m}$ is always reset to 0. In years when wildfire does not occur, $S^V_{t+1,m}$ and $s^V_{t+1,m}$ are deterministic and depend on the state of the system at time t, $S^V_{t,m}$ and $s^V_{t,m}$, as well as on whether or not an ERT or HFRT was performed in year t. The assumptions concerning transitions between ecological states in the model are described in detail below.

The net benefits of fuel treatment are calculated as the present value of the reduction in cumulative wildfire suppression costs resulting from treatment less the present value of total treatment costs over 200 years. The net benefits from fuel treatment for the mth run of the model are given by

$$\text{NPV}_m = \sum_{t=1}^{200} \frac{1}{(1 + r)^t}\left[\sum_{i=1}^{3}(P^{NT}_{t,m} \times Q^{NT}_{i,t,m} \times WC^{NT}_{t,m})\right] - \sum_{t=1}^{200} \frac{1}{(1 + r)^t}\left[\sum_{i=1}^{3}(P^{T}_{t,m} \times Q^{T}_{i,t,m} \times WC^{T}_{t,m}) + T^T_{t,m} \times C(S^T_{t,m})\right]$$

where r is the discount rate ($r = 3\%$ for the results presented in this article), $P^V_{t,m}$, $Q^V_{t,m}$, $i = 1, 2, 3$, and $WC^V_{t,m}$, $R = T$, NT, are the realizations of the random variables $\tilde{P}^V_{t,m}$, $\tilde{Q}^V_{t,m}$, $i = 1, 2, 3$, and $WC^V_{t,m}$ in year t in the treatment and no treatment scenarios. The expected value of net benefits is calculated as the mean of net benefits for 10,000 model runs.

The simulation model presented in this article shares several features with the simulation model presented in Taylor et al. (2013).

Most notably, both models use the STM framework. As is argued in the Introduction, among other advantages, the STM framework allows ecosystem dynamics and stochastic elements to be incorporated into the simulation model and facilitates model parameterization because the required ecological and economic data are collected to be consistent with the STM framework. There are two important differences, however, between the simulation model developed in this article and the model in Taylor et al. (2013). First, the salient features of wildfire activity in the PIPO ecosystem are different from those of two rangeland ecosystems analyzed in Taylor et al. (2013). These differences require that the simulation model presented in this article expand on Taylor et al. (2013) to include new features such as stochastic postwildfire transitions between ecological states and variable wildfire severities (low, moderate, and high).

Second, there are differences between the fuel management issues relevant to the PIPO ecosystem addressed in this article and those addressed in Taylor et al. (2013). In particular, this article expands on the analysis in Taylor et al. (2013) to consider both ERTs and HFRTs (Taylor et al. only consider ERTs) and considers the multistage ERTs that are necessary to rehabilitate a forest ecosystem that has been degraded by forest overcrowding due to historical suppression of low-severity frequent fires. Analyzing these additional fuel management issues requires that the simulation model presented in this article expand on Taylor et al. (2013) to include new features such as HFRTs that change the conditional probability that a wildfire is low, moderate, or high severity, decay in the effectiveness of HFRTs over time, and reapplication intervals for both HFRTs and multistage ERTs.

ERT Scenarios

ERTs influence wildfire behavior and wildfire suppression costs in the simulation by moving the system from the less desirable states of PIPO-A and PIPO-B to the more desirable states of PIPO-C and PIPO-D. As is explained below, wildfire behavior and expected suppression costs are influenced by transitions between ecological states because of differences between states in wildfire return intervals, in the likelihood that when a wildfire does occur it is of low, moderate, or high severity, in the expected wildfire suppression costs associated with a wildfire of given severity, and in expected postwildfire ecological conditions.

Whereas ERTs influence wildfire behavior and expected suppression costs by causing the ecosystem to transition between ecological states, it is assumed that ERTs do not do the following: change the probability a wildfire of a given severity (low, moderate, or high severity) will occur in a given ecological state; or change the expected suppression costs associated with a wildfire of a given severity in a given ecological state. Alternatively, if we assumed that for a period after treatment, ERTs changed the probability of a wildfire of a given severity occurring (e.g., increased the probability that when a wildfire occurs it is low severity) or made wildfires easier and therefore less expensive to suppress, this would increase the wildfire suppression cost savings from ERTs.

The typical treatment method used to perform ERTs in each state in the PIPO ecosystem and the influence of ERTs on transition between ecological states in the model are as follows. In even-aged forest, PIPO-A, we assume that ERTs involve mechanical and/or hand thinning (to open-up the canopy and restore historical spatial composition) coupled with pile and burn of removed biomass (prescribed fire carries too high a risk/cost in this state) are used to move the systems to intermediate-aged forest, PIPO-B. We assume that after an ERT moves the system from PIPO-A to PIPO-B, the system must recover for 20 years before a thinning treatment (thinning smaller trees from below) followed by a prescribed burn (to remove fine fuels) can be used to move the system from PIPO-B to PIPO-C, all-aged forest. Once the system has moved to PIPO-C, we assume that regular prescribed burns (every 12 years in the absence of naturally occurring wildfire) are performed for a period of 40 years to control the regeneration of trees in open spaces and move the system to the restored state, PIPO-D. This fully restored state can be maintained by regular prescribed burning and/or wildfire managed for resource benefits. Wildfire managed for resource benefits (henceforth "wildland fire use") refers to the management of naturally-occurring wildfires (i.e., not prescribed burns) as an HFRT. The cost of wildland fire use is limited to the cost of monitoring, which is included in the calculation of wildfire suppression costs, rather than as treatment costs. In addition, we do not consider any strategies to rehabilitate the system from the postcrown fire state, PIPO-E.

HFRT Scenarios

HFRTs influence wildfire behavior and wildfire suppression costs in the simulation by increasing the conditional probability that when a wildfire occurs it is of low severity for 15 years after treatment.[6] Fifteen years corresponds to the typical length of the period that HFRTs mitigate wildfire severity reported in Biswell et al. (1973).[7] In this article, we define the "effectiveness" of an HFRT as the conditional probability that a wildfire is of low severity in the 15-year period after treatment. For example, without an HFRT, the conditional probability that a wildfire in PIPO-A is low severity, given that a wildfire occurs, is 5%, whereas the conditional probability that a wildfire is high severity is 95% (in PIPO A, moderate-severity fires are deemed unlikely; see Table 2 for details on the conditional probability that a wildfire is of low, moderate, or high severity in each state). We refer to an HFRT treatment as being 90% effective in PIPO-A if it changes the conditional probability of low-severity wildfire to 90% and the conditional probability of a high-severity fire to 10% during the duration of the 15-year period after treatment. Whereas HFRTs change the distribution of wildfire severity for a period after treatment, it is assumed that HFRTs do not (1) change the overall probability that a wildfire will occur in any given ecological state, (2) cause the system to move between ecological states, (3) change the successional trajectory from PIPO-A to PIPO-B or from PIPO-B to PIPO-C, or (4) change the expected suppression costs associated with a wildfire of a given severity (low, moderate, or high severity) in a given ecological state. It is also important to note that although both ERTS and HFRTs make severe wildfire less likely, neither treatment removes the possibility of a severe wildfire completely.

We assumed that HFRTs are only applied in PIPO-A and PIPO-B. The typical treatment method used to perform HFRTs in PIPO-A and PIPO-B includes a single or decreasingly effective prescribed fire and/or mechanical or hand thinning treatment(s). Although HFRTs are generally successful in meeting short-term fuel-reduction objectives such that a treated stand is probably more resilient to high-severity wildfire (Stephens et al. 2012), they differ from ERTs in that the primary objectives usually focus on the reduction of surface, ladder, and canopy fuel loads and usually do not target historical composition, structural and function. For our simulations, the major difference between HFRTs and ERTs is that HFRTs do not include the reintroduction of frequent fire or moving

Table 1. Wildfire suppression costs by ecological state ($000 in 2011 dollars).

Ecological state	NFDRS fuel model[a]	Fire severity	n	Average expenditure per fire ($000/fire)[b]	Total expenditure ($000)	Average acres burned per fire (acres/fire)	Total acres burned (acres)	Average expenditure per acre ($/acre)
PIPO-A	F, G, H, or K	Low	13	1,389	18,051	5,532	71,916	393.9
		Moderate	23	9,588	220,533	41,593	956,639	441.3
		High	17	3,546	60,280	10,926	185,742	919.2
PIPO-B	U	Low	5	294	1,472	7,530	37,650	80.3
		Moderate	5	6,057	30,286	17,403	87,015	886.2
		High[c]	0	N/A	N/A	N/A	N/A	N/A
PIPO-C	C or J	Low	22	1,149	25,284	7,683	169,026	276.6
		Moderate	11	1,374	15,109	4,078	44,858	667.7
		High	5	10,700	53,500	31,171	155,855	857.7
PIPO-D	A	Low[d]	2	69	138	2,203	4,406	27.3
		Moderate	2	553	1,106	23,230	46,460	38.3
		High[c]	0	N/A	N/A	N/A	N/A	N/A
PIPO-E	B, L, or T	Low	7	3,990	27,930	24,656	172,592	204.8
		Moderate	1	1,700	1,700	5,143	5,143	368.4
		High	1	5,700	5,700	9,629	9,629	625.0

NA, not applicable.
[a] See Endnote 9 for a discussion of NFDRS fuel models.
[b] Wildfire suppression costs are reported in constant 2011 dollars, using the "Government Consumption Expenditures and Gross Investment—Non-Defense" price index from the US Department of Commerce's Bureau of Economic Analysis as part of the National Income and Product Accounts. This price index captures the change in prices relevant for wildfire suppression costs (e.g., labor, fuel, and mechanical equipment costs).
[c] In the simulation model, the cost of wildfire suppression for low-severity wildfires in PIPO-D is assumed to be $20 per acre. This corresponds to the cost of "wildland fire use."
[d] The cost of high-severity fires for PIPO-A is also used for PIPO-B and PIPO–D because of the lack of data on the cost of high-severity fires in these two states.

the age structure of the stand to an all-aged state as a primary objective and thus require more, decreasingly effective, mechanical treatments to maintain fuel loads in PIPO-A and PIPO-B and cannot facilitate transitions to PIPO-C. In model runs in which HFRTs are pursued, the treatment regimes in PIPO-C and PIPO-D are the same as the treatment regimes in PIPO-C and PIPO-D for the case of ERTs. In addition, we do not consider the use of HFRTs in PIPO-E.

Fuel Treatments and Wildfire Suppression Costs

As is explained above, ERTs and HFRTs can reduce cumulative wildfire suppression costs over the 200-year time horizon by reducing the instances of severe wildfires in a given ecological state (HFRTs), promoting ecological restoration to ecological states with lower expected wildfire suppression costs (ERTs) and by reducing the likelihood of the system transitioning to a degraded ecological state with higher expected suppression costs because of a severe wildfire (both ERTs and HFRTs). The analysis also assumes, however, that neither ERTs nor HFRTs change the expected costs associated with suppressing a wildfire of a given severity in a given ecological state. This assumption is counter to the evidence that fuel treatments make wildfire suppression easier and therefore less costly (Moghaddas and Craggs 2007, Murphy et al. 2007, Rogers et al. 2008, Bostwick et al. 2011).[8] We do not, however, have information about how ERTs and HFRTs reduce expected suppression costs in the PIPO ecosystem. We are able to consider in the results, however, how the expected wildfire suppression cost savings from ERTs and HFRTs change if we assume that both treatment approaches change fuel conditions so that certain low- and moderate-severity wildfires can be managed through wildland fire use rather than being actively suppressed.

Data and Parameters

The stylized STM for the PIPO ecosystem depicted in Figure 1 is numerically implemented to simulate wildfire activity, ecological transitions between states, and wildfire suppression cost with and without treatment (ERT or HFRT). This section describes the parameters and data used in our model. Tables 1–4 summarize all model parameters and the data described in this section, including treatment costs, suppression costs, wildfire frequencies, and the transitions between ecological states in the PIPO ecosystem.

Wildfire Suppression Costs

Wildfire suppression costs data for US Department of Agriculture (USDA) Forest Service Regions 2, 3, and 4 from 1995 to 2011 were obtained from the USDA Forest Service Rocky Mountain Research Station. The data on wildfire suppression costs from the USDA Forest Service were merged with data from the Monitoring Trends in Burn Severity (MTBS) data set (Eidenshink et al. 2007) to distinguish between acres burned in low-, moderate-, and high-severity wildfires in each ecological state.[9] In brief, we first queried the MTBS data set for all fires that occurred in the Four Corners states (Arizona, Colorado, New Mexico, and Utah) and then filtered for fires that fell within the ponderosa pine forest type. This reduced sample ($n = 2{,}477$) was then combined with the wildfire suppression cost data ($n = 1{,}184$) by fire identification, name, acreage, and spatial location, paying close attention to fires with the same name or those that spatially overlapped. Last, estimates of proportions of low, moderate, and high severity were summarized for each fire to obtain a final sample ($n = 324$) spanning the range of costs/acreage/severity proportions of wildfires on USDA Forest Service lands in the Four Corners region between 1995 and 2011. The analysis assumes that all wildfires are suppressed regardless of their characteristics, with the exception of low-severity wildfires in PIPO-D, which are managed through wildland fire use.[10] Table 1 summarizes suppression costs for each wildfire severity for the five states in the PIPO ecosystem.

In the simulation, a random draw from a state- and severity-specific (low-, moderate-, or high-severity) sample of per-acre wildfire suppression expenditures is taken each time a wildfire occurs. In order for our per-acre suppression cost distributions to reflect the fact that a given acre is more likely to burn in a large fire than in a

Table 2. Transitions between ecological states.

	PIPO-A: even-aged forest	PIPO-B: intermediate-aged forest	PIPO-C: uneven-aged forest	PIPO-D: restored	PIPO-E: postcrown fire
Description	Young trees, closed canopy, high fuel load	Mixture of ages, closed canopy, moderate fuel load	All-aged trees, open canopy, low fuel load	All-aged (+old) trees, open canopy, light fuel load	No ages, no canopy, high fuel load
Time to transition without wildfire	50–150 years → PIPO-B	60–100 years → PIPO-C	20–60 years → PIPO-D	60–500 years → PIPO-C	60–100 years → PIPO-E
Baseline model assumption	100 years	80 years	40 years	200 years	80 years
Wildfire return interval	100+ years	24–100 years	0–50 years	2–20 years	75–125 years
Baseline model assumption (annual probability)	100 years (0.01)	70 years (0.014286)	40 years (0.025)	10 years (0.1)	80 years (0.0125)
Conditional probability of low-severity fire	0.05	0.05	0.20	0.996	NA (0.00)
Transition with low-severity fire	→ PIPO-A	→ PIPO-B	0.90 →No change 0.10 → PIPO-D	→ PIPO-D	NA
Conditional probability of moderate-severity fire	NA (0.00)	0.45	0.50	0.003	NA (0.00)
Transition with moderate-severity fire	NA	0.78 → PIPO-A 0.22 → PIPO-C	0.60 → PIPO-A 0.40 → PIPO-B	→ PIPO-C	NA
Conditional probability of high-severity fire	0.95	0.50	0.30	0.001	1.00
Transition with high-severity fire	→ PIPO-E	→ PIPO-E	→ PIPO-E	→ PIPO-E	→ PIPO-E

NA, not applicable.

small fire, we draw from a weighted distribution of per-acre wildfire suppression costs, with wildfire size used as weights. This approach of using data on past wildfires to populate the state- and severity-specific distribution of wildfire suppression costs assumes the following two factors: that the costs of suppressing future wildfires of given size and severity in a given ecological state will be the same as the costs of suppressing similar wildfires in the USDA Forest Service data; and that the size distribution of future wildfires of a given severity in a given ecological state will be the same as in the USDA Forest Service data.[11]

The wildfire suppression cost data used in this article probably understate actual per-acre wildfire suppression costs and, hence, the expected wildfire suppression cost savings from implementing HFRTs and ERTs, for two reasons. First, the data include only wildfires of more than 100 acres (300 acres after 2003) that "escaped" initial suppression efforts by local and state agencies. Because smaller wildfires tend to have larger per-acre suppression costs than larger wildfires, their exclusion implies that fires with higher per-acre costs may be underrepresented in the distributions of per-acre wildfire suppression costs that we draw from in our simulation. Second, because of the difficulty of obtaining wildfire suppression expenditure information from local and state agencies, these suppression costs are excluded from our analysis. The data used in our analysis include wildfire suppression expenditures incurred by the USDA Forest Service for FY 1995–2003 and suppression expenditures incurred by both the USDA Forest Service and the US Department of Interior, which houses the Bureau of Land Management, for FY 2004 onward.

Wildfire Frequency

To simulate stochastic wildfire occurrences, annual wildfire probabilities and conditional probabilities of a low-, moderate-, or high-severity wildfire were chosen by the authors using Schmidt et al. (2002) and in consultation with two experts on wildfire behavior for the PIPO ecosystem.[12,13] The wildfire probabilities were chosen to represent current conditions in the PIPO ecosystem, with the exception of the annual wildfire probability in PIPO-D, which was chosen to capture historical wildfire behavior. We assume a historical annual wildfire probability in PIPO-D because the goal of ERTs in this article is to restore the system to PIPO-D, in which it can be inexpensively maintained through a regime of frequent, low-intensity wildfires. Table 2 reports annual wildfire probabilities in each ecological state used in the simulation.

It is important to emphasize that the wildfire return interval on a given acre captures both wildfire ignitions that occur on that acre and ignitions that occur elsewhere on the landscape and spread to that acre. As such, wildfire return intervals implicitly account for the spatial spread of wildfire. Indeed, differences in wildfire return intervals between ecological states in the PIPO ecosystem reported in Table 2 are attributed to the influence of differences in vegetation and stand densities between states on the wildfire hazard and the spatial spread of wildfire rather than to differences in the rates of wildfire ignitions (wildfire risk) between states.[14] In addition, as the wildfire return intervals were chosen to represent current conditions in the PIPO ecosystem, they implicitly account for the influence of wildfire suppression responses on wildfire sizes, given current suppression resources and objectives in the study area. It is the case, however, that the wildfire return intervals used in this study may not capture future wildfire return intervals if factors such as climate change (e.g., longer and more frequent drought) cause wildfires to become more frequent. To investigate this possibility, we analyze the sensitivity of our results to increases in wildfire frequency from our baseline assumptions in the Results and Discussion section below.

Wildfire return intervals are computed as the average interval between two successive wildfires for a collection of points on a landscape. Annual wildfire probabilities are the reciprocal of wildfire return intervals under the assumption that wildfires occur according to a geometric distribution (i.e., the probability of a wildfire is constant and independent across years). An important advantage of fire-return intervals is the equivalence between the average fire-return interval on a landscape and the fire rotation interval, which is

Table 3. Transitions between ecological states after wildfire.

State before wildfire	State after wildfire					Total probability
	PIPO-A	PIPO-B	PIPO-C	PIPO-D	PIPO-E	
PIPO-A	0.05 (low) Beneficial				0.95 (severe) Degrading	1.0
PIPO-B	0.35 (moderate) Degrading	0.05 (low) Beneficial	0.10 (moderate) Beneficial		0.50 (severe) Degrading	1.0
PIPO-C	0.30 (moderate) Degrading	0.20 (moderate) Degrading	0.18 (low) Beneficial	0.02 (low) Beneficial	0.30 (severe) Degrading	1.0
PIPO-D			0.003 (moderate) Degrading	0.996 (low) Beneficial	0.001 (severe) Degrading	1.0
PIPO-E					1.00 (severe) Degrading	1.0

Fire severity is in parentheses. Wildfires are classified as being either "Beneficial" or "Degrading," depending on the state the ecosystem transitions into after a wildfire, given the initial state.

the length of time required to burn the equivalent of a specific area (Baker 2009).[15] This equivalence implies that the average number of acres burned predicted by the simulation model over the 200 years will hold for any size area. For example, if the simulation model predicts an average of 1.6 wildfires over 200 years in 10,000 model runs, this implies than an area of 1,000 acres in the same initial ecological state is predicted to experience an average of 1,600 burned acres over 200 years, with some areas burning multiple times and others not burning at all. This argument also implies that the per-acre expected net present value (NPV) of wildfire suppression cost savings from ERTs and HFRTs reported in the results can be "scaled-up" linearly to apply to larger treatment areas. It is not reasonable, however, to scale the per-acre results reported in this study to the landscape scale, as typically only a small fraction of a landscape is treated at any point in time.

Transitions between Ecological States

We assume that the PIPO ecosystem will transition between ecological states eventually without either treatment or wildfire. Times for ecological transition used in the simulation are described in Table 2 and were chosen by the authors and vetted by an expert on wildfire in the PIPO ecosystem.[16] In brief, we started with the same subset of wildfires in the MTBS data set that were within the PIPO ecosystem type located in the Four Corners states (Arizona, Colorado, New Mexico, and Utah) that we used to obtain our wildfire suppression cost information. The resulting fire landscapes (individual fires classified by low, moderate, and high severity from the MTBS data set) were further classified based on pre- and postfire vegetation states. Transition probabilities were assigned based on the observed proportions of land moving between states for the observation period (2002–2012) within the entire Four Corners area. Although our sample is limited to fires with sufficient data reported, we assume that there is no systematic underreporting and the sampled fires are representative of typical transitions of PIPO ecosystem.

We evaluate the sensitivity of our results to our assumptions concerning the number of years to transition between ecological states without wildfire or treatment in Appendix A. The analysis in these appendixes demonstrates that the results from the baseline model are invariant to substantial changes in these parameters from baseline values.

Wildfire is a catalyst for transition between ecological states in the PIPO ecosystem. Further, postwildfire transitions between states in the PIPO ecosystem are influenced by wildfire severity. In Figure 1, the postwildfire transition paths between states are color-coded for low-severity (blue), moderate-severity (yellow), and high-severity (red) wildfires. Postwildfire transition probabilities between states after low-, moderate-, and high-severity wildfire for each state in the PIPO ecosystem were chosen by the authors and vetted by an expert on wildfire in the PIPO ecosystem and are reported in Table 3.[17]

Treatment Costs

As is explained above, we assume that the appropriate suite of treatment methods for ERTs and HFRTs and, hence, per-acre treatment costs, varies by ecological state in the PIPO ecosystem. Information on treatment costs were obtained by contacting program officers from four national forests to gather the most recent transaction evidence (FY 2008), expert opinion, and planning estimates. Based on these estimates and previously published articles in this region (Larson and Mirth 2004, Hjerpe and Kim 2008, Kim 2010), average costs per acre are calculated for the proposed treatments in each state. Table 4 reports the per-acre costs for ERTs and HFRTs for each state in the PIPO ecosystem used in the simulation. Details on how per-acre treatment costs for the PIPO ecosystem were calculated are given in Appendix B.

Biomass Removal

The analysis also considers how including the revenue from the sale of the biomass removed while performing ERTs in PIPO-A and PIPO-B can influence the net economic benefits from ERTs. We only consider the revenue from ERTs in PIPO-A and PIPO-B because it is only in these states that fuel treatment involves mechanical/hand thinning. Revenues from forest biomass vary significantly across regions in the United States (Galik et al. 2009), including across sites in the PIPO ecosystems in the Southern Colorado Plateau, because of factors such as quality road access (Becker et al. 2011), transportation costs (Han et al. 2004), and the absence of biomass utilization facilities in many regions (Barbour et al. 2008, Ince et al. 2008, Prestemon et al. 2008). Because of this variability in revenues, we consider a range of per-acre revenue from biomass removal in the analysis. We assume that revenue from the sale of biomass is determined by two factors: the number of "bone-dry" tons (short tons; 2,000 lb) of biomass removed per acre through treatment and (ii) the market price per bone dry ton of biomass. We assume removal of 11.2 bone-dry tons of biomass per acre (Rummer et al. 2005) and a range of market prices drawn from existing regional market prices for softwood biomass (Department of Energy 2011).

Table 4. Treatment costs (total costs of treatments, including operation and administration costs; 2011 dollars).

	PIPO-A: even-aged forest	PIPO-B: intermediate-aged forest	PIPO-C: uneven-aged forest	PIPO-D: restored	PIPO-E: postcrown fire
Management focus	Mechanical or hand thinning to open up canopy; pile/burn[a]	Pole size timber; thin from below followed by prescribed burn	Controlling regeneration in opening; fine fuels removed via prescribed burn	Prescribed burn or wildland fire use	Reforestation
Ecological state after treatment	PIPO-B (after 20 years)	PIPO-C (after 20 years)	PIPO-D (after 40 years)	PIPO-D	PIPO-A
Wildland fire use	NA	NA	\$10–\$30	\$8	NA
Prescribed fire	\$400–\$600	\$75–\$300	\$25–125	\$12–\$75	NA
Precommercial thinning	\$1,200-\$2,000	NA	NA	NA	NA
Thinning	NA	\$471–\$766	NA	NA	NA
Reforestation	NA	NA	NA	NA	\$800–\$2,000
ERT total	\$2,100	\$500	\$60	\$45	\$1,400
Treatment schedule	1 year	1 year	After 12 years without treatment or wildfire	After 12 years without treatment or wildfire	5 years after wildfire
HFRT[b]	\$60	\$60	NA	NA	NA

NA, not applicable.
[a] Any type of burn in PIPO-A will carry high risk/high expected cost.
[b] A single-entry or decreasingly effective prescribed fire or mechanical/hand thinning treatment(s) is not intended to reintroduce frequent fire or alter the stand's age structure, thus only effective and occurring in PIPO-A and PIPO-B.

Table 5. No treatment results (\$ per acre; 2011 dollars).

	PIPO-A: even-aged forest	PIPO-B: intermediate-aged forest	PIPO-C: uneven-aged forest
Mean number of low-severity wildfires	0.07 (0, 1)[a]	0.24 (0, 1)	0.69 (0, 2)
Mean number of moderate-severity wildfires	0.16 (0, 1)	0.61 (0, 2)	0.76 (0, 2)
Mean number of high-severity wildfires	2.1 (0, 5)	1.97 (0, 5)	1.94 (0, 5)
Mean total number of wildfires	2.34 (0, 5)	2.83 (1, 6)	3.39 (1, 6)
Mean total number of beneficial wildfires	0.10 (0, 1)	0.33 (0, 1)	0.71 (0, 2)
Mean total number of degrading wildfires	2.25 (0, 5)	2.5 (1, 5)	2.68 (1, 5)
Mean total suppression costs (NPV)[b]	\$165 (\$0, \$655)	\$200 (\$3, \$763)	\$275 (\$10, \$807)
Final State (A, B, C, D, E)[c]	3,286, 401, 861, 16, 5,436	3,064, 828, 550, 103, 5,455	3,071, 845, 552, 279, 5,253

[a] 5th and 95th percentiles.
[b] Sum over 200 years using a 3% discount rate (following Loomis 2002).
[c] "Final state" is the final state of the system (PIPO-A, PIPO-B, PIPO-C, PIPO-D, or PIPO-E) after 200 years.

Table 6. ERTs: baseline results (\$ per acre; 2011 dollars).

	PIPO-A: even-aged forest	PIPO-B: intermediate-aged forest	PIPO-C: uneven-aged forest
Mean number of low-severity wildfires	6.14 (0, 18)[a]	8.48 (0, 21)	8.64 (0, 21)
Mean number of moderate-severity wildfires	0.64 (0, 2)	0.63 (0, 2)	0.61 (0, 2)
Mean number of high-severity wildfires	1.4 (0, 5)	1.17 (0, 5)	1.13 (0, 5)
Mean total number of wildfires	8.2 (2, 18)	10.3 (2, 21)	10.4 (2, 21)
Mean total number of beneficial wildfires	6.2 (0, 18)	8.5 (0, 21)	8.7 (0, 21)
Mean total number of degrading wildfires	1.99 (1, 5)	1.78 (0, 5)	1.73 (0, 5)
Mean total suppression costs (NPV)[b]	\$217 (\$10, \$728)	\$256 (\$16, \$800)	\$259 (\$15, \$811)
Mean number of treatments	4.9 (1, 9)	4.4 (1, 8)	4.5 (1, 8)
Mean treatment costs (NPV)[b]	\$2,460 (\$2,100, \$3,038)	\$730 (\$500, \$1,503)	\$298 (\$60, \$1,093)
Final state (A, B, C, D, E)[c]	623, 567, 727, 5,126, 2,957	511, 500, 630, 5,960, 2,399	506, 503, 571, 6,090, 2,330

[a] 5th and 95th percentiles.
[b] Sum over 200 years using a 3% discount rate (following Loomis 2002).
[c] "Final state" is the final state of the system (PIPO-A, PIPO-B, PIPO-C, PIPO-D, or PIPO-E) after 200 years.

Results and Discussion

Tables 5–7 report results for the no treatment case, ERT, and HFRT, respectively, under the default parameter assumptions reported in Tables 1–4 (henceforth the "baseline" results). The results for ERTs are reported for three different initial states: PIPO-A, PIPO-B, and PIPO-C; the results for HFRTs are reported for two initial states: PIPO-A and PIPO-B, as the treatment regime in PIPO-C is the same for ERTs and HFRTs.

Wildfire Activity

The baseline results for ERTs and HFRTs reported in Tables 6 and 7 indicate that, given our assumptions and default parameters, both treatment approaches probably increase the expected total number of wildfires in the PIPO ecosystem over the 200-year time horizon relative to the no treatment case reported in Table 5. ERTs increase the total number of wildfires because they accelerate the transitions from PIPO-A and PIPO-B, which have relatively long wildfire return intervals (100 and 70 years, respectively), to PIPO-C and PIPO-D, which have shorter wildfire return interval (40 and 10 years, respectively). In addition, transitions from PIPO-A and PIPO-B to PIPO-C and PIPO-D reduce the number of high-severity crown fires that cause the system to transition to PIPO-E, which also has a relatively long wildfire return interval (80 years). The

Table 7. HFRTs: baseline results ($ per acre; 2011 dollars).

	PIPO-A: even-aged forest	PIPO-B: intermediate-aged forest
Mean number of low-severity wildfires	1.72 (0, 4)[a]	4.42 (0, 12)
Mean number of moderate-severity wildfires	0.23 (0, 1)	0.43 (0, 1)
Mean number of high-severity wildfires	0.60 (0, 3)	0.70 (0, 3)
Mean total number of wildfires	2.54 (0, 5)	5.56 (1, 12)
Mean total number of beneficial wildfires	1.73 (0, 4)	4.44 (0, 13)
Mean total number of degrading wildfires	0.81 (0, 3)	1.11 (0, 4)
Mean total suppression costs (NPV)[b]	$113 ($0, $487)	$77 ($2, $284)
Mean number of treatments	12 (5, 14)	9 (5, 14)
Mean treatment costs (NPV)[b]	$154 ($103, $203)	$148 ($243, $991)
Final State (A, B, C, D, E)[c]	2,250, 699, 4,783, 103, 2,165	2,219, 1,408, 786, 3,725, 1,862

[a] 5th and 95th percentiles.
[b] Sum over 200 years using a 3% discount rate (following Loomis 2002).
[c] "Final state" is the final state of the system (PIPO-A, PIPO-B, PIPO-C, PIPO-D, or PIPO-E) after 200 years.

results in Tables 5 and 6 also demonstrate, however, that by transitioning the system from PIPO-A and PIPO-B to PIPO-C and, eventually, to PIPO-D, ERTs are successful in reducing the number of high-severity crown fires (2.1 to 1.4 when the initial state is PIPO-A) and increasing the number of moderate-severity wildfires (0.2 to 0.6 when the initial state is PIPO-A) and low-severity wildfires (0.1 to 6.1 when the initial state is PIPO-A). In the subsequent sections, we discuss the implications of this change in the distribution of low-, moderate-, and high-severity wildfires for the instance of ecologically beneficial and degrading wildfire.

Because we assume that HFRTS do not change the time it takes the ecosystem to transition from PIPO-A to PIPO-B (100 years) or from PIPO-B to PIPO-C (80 years), HFRTs do not significantly change the total number of wildfires when the initial state is PIPO-A, as the system remains in PIPO-A and PIPO-B for the majority of the 200-year time horizon with and without HFRTs. HFRTs do, however, increase the total number of wildfires from 2.8 to 5.6 when the initial state is PIPO-B because HFRTS facilitate the transition from PIPO-C to PIPO-D in 40 years in the absence of wildfire. Table 7 also indicates that although HFRTS do not reduce the mean total number of wildfires, they do, as one would expect, increase the number of low-severity wildfires from 0.1 to 1.7 over 200 years and reduce the number of high-severity wildfires from 2.1 to 0.5 over 200 years, while resulting in very little change in the number of moderate-severity wildfires (0.2 to 0.1 over 200 years).

Beneficial and Degrading Wildfires

One outcome of ERTs and HFRTs of particular interest to fire managers is how they affect the instances of "beneficial" and "degrading" wildfires. Beneficial wildfires are low- or moderate-severity fires that facilitate postfire ecosystem recovery, and, ultimately, are beneficial to ecosystem health and function. Conversely, degrading wildfires provide little ecosystem benefits and can damage ecosystem health. Table 3 describes how beneficial and degrading wildfires are classified in our simulation. From Tables 6 and 7, although both fuel treatment approaches increase the expected total number of wildfires, they are both successful in reducing the number of degrading wildfires and increasing the number of beneficial wildfires. For example, when the initial state is PIPO-A, ERTs decrease the number of degrading wildfires over the 200-year time horizon from 2.3 to 2.0 and increase the number of beneficial wildfires from 0.1 to 6.2. Similarly, when the initial state is PIPO-A, HFRTs reduce the number of degrading wildfires from 2.3 to 0.8 and increase the number of beneficial wildfires from 0.1 to 1.7. The increase in the number of beneficial wildfires and the decrease in the number of degrading wildfires reduce the likelihood that the system will transition to PIPO-E, the postcrown fire state. For example, when the initial state is PIPO-A, the model predicts that at the end of 200 years, the systems will be PIPO-E 54.4% of time without ERTs compared with 29.6% of the time with ERTs.

Wildfire Suppression Costs

Tables 6 and 7 report the expected NPVs of wildfire suppression costs for ERTs and HFRTs. Table 6 indicates that ERTs are not effective in reducing wildfire suppression costs relative to the no treatment case reported in Table 5 because the reduction in the number of high-severity wildfires is more than offset by the increase in suppression costs from more frequent low- and moderate-severity wildfires. Conversely, HFRTs reduce the expected NPVs of wildfire suppression costs from $170 to $121 per acre when the initial state is PIPO-A and from $199 to $137 per acre when the initial state is PIPO-B. The wildfire suppression cost savings associated with HFRTs, however, are not enough to offset the costs of implementing fuel treatments. For example, for HFRTs when the initial state is PIPO-A, the expected NPV of treatment costs is $154 per acre and the expected NPV of wildfire suppression cost savings is $51 ($164–$113) per acre. Our finding that neither ERTs nor HFRTs are economically justified in the PIPO system on the basis of wildfire suppression cost savings alone is consistent with the results of Prestemon et al. (2012), who provide ranges of the long-run expected economic benefits of mechanical fuel treatments for all nonreserved timberlands (both public and private) in the contiguous western United States; this is the only other study we are aware of that considers the wildfire suppression cost savings resulting from fuel treatments for forested ecosystems in the western United States.

ERT: Wildfire Return Intervals

As we explain above, ERTs are not effective in reducing wildfire suppression costs in our simulation model despite the facts that they reduce the number of high-severity crown fires and are effective at rehabilitating the system to PIPO-D and avoiding transitions to PIPO-E. When the system starts in PIPO-A, high-severity wildfires in PIPO-A can occur early in the planning horizon, where, due to discounting, they have a large influence on the calculation of the expected value of net benefits. These facts suggest that avoiding high-severity wildfire in PIPO-A could be an important benefit of performing ERTs and that our baseline assumption of a 1% annual wildfire probability in PIPO-A is likely to have a large influence on the magnitude of expected net benefits from ERTs. For this reason, we consider in Table 8 how the expected net benefits from ERTs

Table 8. Expected net benefits from ERT with changes in annual wildfire probabilities in PIPO-A and PIPO-E; initial state = PIPO-A ($ per acre; 2011 dollars).

	Annual wildfire probability in PIPO-A				
Annual wildfire probability in PIPO-E	1%	5%	10%	15%	20%
1.25%	−$2,507	−$2,098	−$1,851	−$1,706	−$1,591
5%	−$2,444	−$1,711	−$1,456	−$1,209	−$1,088
10%	−$2,366	−$1,237	−$795	−$555	−$523
15%	−$2,312	−$779	−$259	−$28	$122
20%	−$2,199	−$282	$398	$642	$819

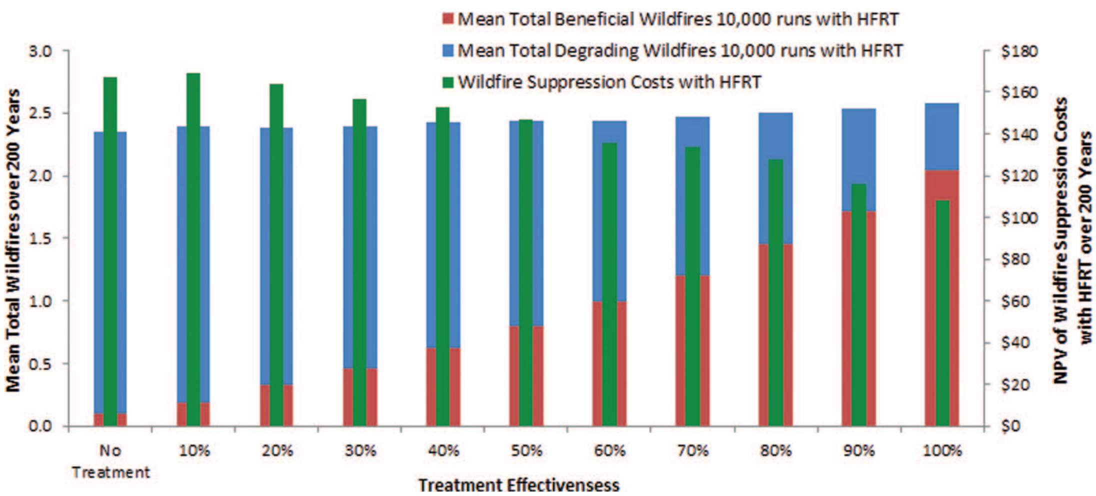

Figure 3. HFRTs: treatment effectiveness.

change with the annual probability of wildfire in PIPO-A. In addition, whereas ERTs are an effective tool to keep the system from transitioning to PIPO-E, under our baseline assumptions the expected wildfire suppression costs for PIPO-E are relatively low, which is another reason why fuel treatments are not effective in reducing wildfire suppression costs. We also consider how the expected benefits of fuel treatments change with increases in the annual probability of wildfire in PIPO-E.

The results in Table 8 confirm that the expected net benefits from treatment are indeed sensitive to our assumptions about the annual probability of wildfire in PIPO-A and PIPO-E. Holding all other parameters constant, an increase in the annual probability of a wildfire in PIPO-A from our baseline of 1% increases the expected wildfire suppression cost savings from treatment and, hence, the expected net benefits from fuel treatment, to $409 per acre at 5% ($2,507 − $2,098 = $409) and $656 per acre ($2,507 − $1,851 = $656) at 10%. Increasing the annual wildfire probability in PIPO-E also increases the expected net benefits from ERTs, albeit by a smaller amount than the increases in the wildfire probability for PIPO-A. The analysis in this section focuses on changes in wildfire return intervals in PIPO-A and PIPO-E because these two parameters were likely to (and do) have a large impact on the model's predictions. It has been suggested that wildfire return intervals in the PIPO ecosystems are likely to be reduced as a result of climate change, invasive plants, and other factors (Westerling et al. 2006, Millar et al. 2007, Abatzoglou and Kolden 2011). The analysis in this section suggests that to the extent that climate change shortens wildfire return intervals in PIPO-A and PIPO-E, ERTs will result in significantly larger wildfire suppression cost savings in the future than they do at present.

HFRT: Treatment Effectiveness

Figure 3 considers how changes in the effectiveness of HFRTs influence three fire management outcomes: the expected number of wildfires, the expected number of beneficial and degrading wildfires, and the expected wildfire suppression costs. Recall that we define the "effectiveness" of an HFRT as the probability that the treatment will ensure that a wildfire that occurs during the duration of the treatment (15 years) will be low severity. Figure 3 illustrates how increasing treatment effectiveness reduces wildfire suppression costs by changing the distribution of wildfires, reducing the number of degrading wildfires (which generally have high suppression costs), and increasing the number of beneficial wildfires (which generally have lower suppression costs). These results imply that the benefits of HFRTs depend in large measure on whether the treatment is executed in a manner that allows for a high degree of treatment effectiveness. That is, the benefits of HFRTs depend significantly on the ability and diligence of the fuels and fire manager performing the treatments.

Posttreatment Wildland Fire Use

So far the analysis has assumed that neither ERTs nor HFRTs change the expected suppression costs for a given wildfire. In this section, we relax these assumptions and analyze the possibility that ERTs and HFRTs change fuel conditions so that certain low- and moderate-severity wildfires can be managed through "wildland fire use." In particular, we consider the sensitivity of our results to different assumptions about the proportions of low- and moderate-severity wildfires in PIPO-C that can be managed through wildland fire use after application of ERT or HFRT. We focus on PIPO-C

Table 9. Wildland fire use for low- and moderate-severity wildfires in PIPO-C: mean total wildfire suppression costs (NPV) (2011 dollars).

	Low- and moderate-severity wildfires in PIPO-C managed via wildland fire use						
Initial state	0%[a]	20%	40%	60%	80%	100%	No treatment
ERTs							
PIPO-A	$217 ($10, $728)[b]	$209 ($9, $715)	$206 ($8, $700)	$195 ($9, $698)	$191 ($8, $768)	$171 ($7, $754)	$165 ($0, $655)
PIPO-B	$256 ($16, $800)	$227 ($14, $719)	$212 ($13, $756)	$186 ($14, $648)	$171 ($12, $662)	$141 ($12, $596)	$200 ($3, $763)
PIPO-C[c]	$259 ($15, $811)	$235 ($16, $765)	$215 ($15, $741)	$192 ($15, 663)	$165 ($13, $599)	$152 ($14, $648)	$275 ($10, $807)
HFRTs							
PIPO-A	$113 ($0, $487)	$104 ($0, $447)	$118 ($0, $512)	$114 ($0, $507)	$123 ($0, $502)	$110 ($0, $446)	$164 ($0, $687)
PIPO-B	$77 ($2, $284)	$70 ($2, $322)	$68 ($2, $279)	$73 ($2, $247)	$69 ($2, $288)	$60 ($1, $234)	$202 ($3, $766)

[a] Baseline scenario reported on Tables 6 and 7.
[b] 5th and 95th percentiles.
[c] As we explain in the text above, we do not report results for HFRTs when the initial state is PIPO-C because the treatment regime in PIPO-C is the same for ERTs and HFRTs, and, as such, there is little difference between the two approaches when the initial state is PIPO-C.

Table 10. ERTs: biomass removal (2011 dollars).

	Market price bone dry ton (11.2 bdt/acre)[a]							
	$25	$50	$75	$100	$125	$150	$175	$200
Initial state = PIPO-A								
Mean revenue from biomass removal (NPV)	$533	$1,063	$1,594	$2,121	$2,646	$3,189	$3,743	$4,294
Mean revenue from biomass removal net of treatment costs (NPV)	−$2,039	−$1,617	−$1,202	−$785	−$376	$44	$474	$895
Mean wildfire suppression costs savings and revenue from biomass removal net of treatment costs (NPV)	−$2,095	−$1,662	−$1,253	−$835	−$420	−$3	$424	$843
Initial state = PIPO-B								
Mean revenue from biomass removal (NPV)	$578	$1,148	$1,741	$2,297	$2,892	$3,501	$4,056	$4,636
Mean revenue from biomass removal net of treatment costs (NPV)	−$421	−$102	$210	$527	$834	$1,159	$1,468	$1,784
Mean wildfire suppression costs savings and revenue from biomass removal net of treatment costs (NPV)	−$479	−$153	$155	$474	$783	$1,101	$1,413	$1,722

[a] The bone-dry tons refers to short tons (2,000 lb).

because both ERTs and HFRTs can reasonably be expected to reduce the probability of tree mortality in the event of a low- or moderate-severity wildfire in PIPO-C; thus, both treatment options increase the potential to manage certain low- or moderate-severity wildfires through wildland fire use.

Table 9 summarizes results from six assumptions concerning the proportion of low- and moderate-severity wildfires in PIPO-C managed via wildland fire use after application of ERT or HFRT: 0% (the baseline assumption), 20, 40, 60, 80, and 100% for three initial ecological states: PIPO-A, PIPO-B, and PIPO-C. Table 9 reports only how posttreatment wildland fire use in PIPO-C influences wildfire suppression costs, not changes in expected future wildfire occurrence or severity. These results illustrate that allowing certain low- and moderate-severity wildfires in PIPO-C to be treated through wildland fire use posttreatment leads to large reductions in expected wildfire suppression costs for ERTs and smaller reductions in expected wildfire suppression costs for HFRTs. For ERTs, the large reductions are due to the fact that when the initial state is PIPO-A or PIPO-B, ERTs effectuate a relatively rapid transition to PIPO-C (21 years when the initial state is PIPO-A; 1 year when the initial state is PIPO-B). This rapid transition to PIPO-C means that the wildfire suppression cost savings associated with managing a portion of low- and moderate-severity wildfires in PIPO-C through wildland fire occurs early in the planning horizon for ERTs, and, as a result, has a strong impact on the calculation of the NPV of wildfire suppression costs. In contrast, HFRTs do not reduce the time required for the systems to transition from PIPO-A and PIPO-B to PIPO-C, so that the wildfire suppression cost savings associated with posttreatment wildland fire use in PIPO-C are only realized late in the planning horizon. These results indicate that our assumption that neither ERTs nor HFRTs change the expected wildfire suppression cost associated with a given wildfire leads our baseline results to understate the wildfire suppression cost savings associated with both ERTs and HFRTs and that the magnitude of this understatement will depend on how early in the planning horizon the reduced suppression costs are realized.

ERT: Biomass Removal

The results reported in Table 10 demonstrate that including revenue from the sale of the biomass removed while performing ERTs in PIPO-A and PIPO-B can have a large effect on the net economic benefits from ERTs. For example, at a price of $25 per bone dry ton, inclusion of biomass in the analysis increases the expected present value of the economic benefits of ERTs when the initial state is PIPO-A by $533 per acre. The results in Table 10 also demonstrate, however, that the net economic benefits of ERTs (wildfire suppression cost savings plus the economic value of biomass less the costs of treatment) are only positive for prices of biomass that are several times the typical price of biomass per bone dry ton in the study area (Department of Energy 2011).

Modeling Limitations

In this section, we discuss the limitations to our modeling approach and the implications of these limitations for our results. The limitations to our modeling approach can be divided into three categories. First, there are limitations related to elements that we omit from our analysis. These include the costs of wildfire other than wildfire suppression costs and the benefits of fuel treatment in terms of enhanced ecosystem services. Second, there are limitations related to the fact that our approach does not consider spatial issues related to the spread of wildfire and the impact of location-specific characteristics on the benefits and costs of fuel treatments. Third, there are limitations on the implications of our results related to the assumptions about how treatment costs are included in the model. The discussion in this section explains why the limitations to our modeling approach will cause our analysis to understate the economic benefits of ERTs and HFRTs. As we stated in the Introduction, an important strength of our analysis is that model parameters and assumptions are chosen so that our results provide a lower-bound estimate of the net economic benefits from implementing either fuel treatment option.

In addition to these three limitations, another important limitation of our analysis is that we do not fully explore the potential benefits of ERTs and HFRTs in terms of improving the efficacy of the wildfire suppression response, thereby reducing suppression costs. As this limitation was discussed above when we discuss the data and parameters, we do not address it again in this section.

Additional Benefits from Fuel Treatment

As we discuss in the Introduction, by focusing on a narrow set of financial benefits from fuel treatment, the analysis in this article does not consider several additional benefits from fuel treatment. First, it does not consider how fuel treatments can reduce the financial, or market, costs of wildfire other than wildfire suppression costs. Important financial costs of wildfire not considered in the analysis include damage to housing and other infrastructure and the costs of postfire emergency rehabilitation. Second, the analysis also does not consider potential reductions in the nonmarket cost to wildfire as a result of treatment. These nonfinancial costs include postfire soil loss and flooding and the costs to human health from smoke and particulate matter released during the wildfire. Third, the analysis does not consider the potential benefits of fuel treatments from enhanced ecosystem services. Fuel treatments can enhance ecosystem services by reducing the likelihood of ecologically degrading wildfires, and, in the case of ERTs, by directly promoting ecological restoration. Ecosystem services that can potentially be improved by fuel treatment include wildlife habitat, erosion control, recreational opportunities, and the aesthetic beauty of the landscape.[18] Not considering these additional benefits of fuel treatments is not, strictly speaking, a limitation of our modeling approach. Indeed, a strength of our framework is that it can readily incorporate additional benefits from fuel treatment as this information becomes available.

In addition to not considering the additional benefits from fuel treatments listed in the previous paragraph, another limitation of our analysis is that it cannot offer insight into the benefits and costs of ERTs of HFRTs under different assumptions about the expected wildfire suppression response. This is an important limitation because it implies that it is not possible to use the results reported in this article to infer the likely benefits and costs of ERTs and HFRTs in the wildland-urban interface where the wildfire suppression response is likely to be more aggressive than in the wildland setting considered in this article.[19] The reason for this limitation is that the wildfire suppression response influences both per-acre wildfire suppression costs and wildfire size (and, hence, the wildfire return interval), and these two effects have an offsetting influence on expected total suppression costs. As such, although previous authors have found that wildfires in the wildland-urban interface are, on average, smaller and have higher per-acre suppression costs than fires on wildlands (Cohen 2000, Winter and Fried 2000), the overall impact of these changes on expected total suppression costs, and, hence, on the benefits and costs of ERTs and HFRTs, depends on the magnitude of these two effects. For this reason, we need specific information on both per-acre suppression costs and wildfire return intervals from wildland-urban interface communities in the study region to comment on the benefits and costs of ERTs and HFRTs in this setting.[20]

Spatial Wildfire Spread and Treatment Location

An important limitation of our approach is that we do not consider the spatial spread of wildfire. This has three important limitations for our results. First, not considering issues related to the spatial spread of wildfire implies that we are ignoring the impact of fuel treatments on wildfire behavior and suppression costs outside of the treatment zone. This is potentially a significant shortcoming given the experimental evidence that fuel treatments can reduce the wildfire hazard in large regions surrounding the treatment zone (Finney 2001, Stratton 2004). It means that our per-acre analysis will necessarily understate the expected wildfire suppression cost savings from fuel treatments. Previous studies have suggested, however, that in the PIPO ecosystem, the majority of wildfire suppression cost savings from treatment occur on treated acres, with only small reductions in acres burned and suppression costs occurring outside the treated acreage due to the effect of the fuel treatments on the spatial spread of wildfire (Thompson et al. 2013).

Second, the fact that the wildfire risk on one stand is dependent on the fuel conditions on surrounding stands because of the spatial spread of wildfire implies that fuel treatments must be performed at or above a minimum spatial scale for the results in this article to hold. In particular, fuel treatments must be performed on a sufficiently large scale for the movements in ecological state to influence wildfire return intervals in the case of ERTs and to change the instance of low-, moderate-, and high-severity wildfires in the case of HFRTs. Although the minimum spatial scale for treatments is certainly larger than an acre, it is not possible for us to state definitively the minimum scale for ERTs or HFRTs. We do not, however, view our inability to define a minimum spatial scale for our results as a significant limitation of our analysis. This is because, in practice, managers will only perform fuel treatments at a spatial scale that they believe is sufficient to influence wildfire behavior on the treated acres, as well as on land adjacent to the treated area.

Third, the fact that the wildfire risk on one stand is dependent on the fuel conditions on surrounding stands because of the spatial spread of wildfire means that the fuel conditions (and, hence, wildfire hazard) on land adjacent to the treatment site are likely to influence the annual wildfire probability, the instance of low-, moderate-, and high-severity wildfires, and even the per-acre suppression costs on the treatment site. An additional limitation of our analysis is that we do not consider the potential effects of wildfire hazard on land adjacent to the treatment site on these model parameters. Ignoring these potential effects is equivalent to assuming that the conditions on land adjacent to the treatment site are such that the per-acre wildfire suppression costs reported in Table 1 and the wildfire probabilities reported in Table 2, which were chosen to represent average conditions of land in each of the five ecological states in the PIPO ecosystem, hold for the treatment site. This implicit assumption is consistent with the focus in this article on the benefits and costs of ERTs and HFRTs on an average acre.

Another limitation of our analysis is that, by focusing on ERTs and HFRTs on an average acre, the analysis ignores the role of managerial discretion in choosing locations of fuel treatments and in designing fuel treatments. Landscape features such as topography (slope, aspect, and others), distance to housing and other infrastructure, and proximity to riparian areas imply that the spatial placement and design of fuel treatments will have a large influence on their impact on wildfire hazard and hence on their expected benefits, as well as on treatment costs (Finney 2001, Agee and Skinner 2005). All else being equal, managers will target areas for fuel treatments where the wildfire suppression cost savings and other benefits from treatment are larger than average (e.g., areas near the wildland-urban interface) and/or where the costs of treatment are lower than average (e.g., areas that are easily accessible by heavy equipment). Given these facts, not taking into account managerial discretion is likely to cause our analysis to further understate the net economic benefits of fuel treatments.

Treatment Cost Assumptions

This section discusses three assumptions about how treatment costs are included in the simulation model. First, our analysis considers only variable costs of implementing a fuel treatment and does not consider fixed costs.[21] Our focus on variable costs implies that when using the results of our analysis to evaluate a specific fuel treatment project, the sum of the per-acre net expected benefits must be larger than the fixed costs of the project for the net benefits for the project as a whole to be positive.

Second, the treatment cost estimates used in this article do not consider how treatment costs vary with the size of treated area. For a specific fuel treatment project, the previous literature has found that the average per-acre treatment costs decrease with the size of the treated area (Rummer 2008) and that this decline can be partially attributable to declining variable costs (and partially attributable to indivisible fixed costs). In addition, at the regional scale, limits to the number of trained foresters and the availability of specialized equipment can constrain the number of treatment projects that can be completed in a season and may also drive up variable treatment costs in years where many projects take place.

Third, the analysis in this article does not consider third-party damages from implementation of fuel treatments. Relevant third-party damages include the smoke created by prescribed fires, damages that occur if prescribed fires escape the intended boundaries (and the psychological harm this possibility inflicts on nearby residents), and soil compaction and increased erosion caused by heavy equipment used for mechanical fuel thinning and removal. The analysis in this article does not consider these potential additional costs of implementing fuel treatments and, therefore, may understate the costs of ERTs and HFRTs in circumstances where these additional costs are likely to be significant.

Conclusions

The model developed here simulates management of a PIPO ecosystem and evaluates and compares the economic benefits and costs of ERTs and HFRTs. More often than not, ERTs are successful at restoring the ecosystem to PIPO-D, which is a fire-resistant state characterized by an open canopy and frequent, low-severity wildfires. However, because restoration involves the system transitioning to ecological states where wildfire is more frequent, ERTs also increase the total expected number of wildfires. Further, ERTs reduce the number of ecologically degrading wildfires that lead the system to transition to the degraded, postcrown fire state, which reinforces the benefits of ERTs in terms of ecological restoration. ERTs increase expected wildfire suppression costs in our baseline specification, but this result is reversed if we shorten wildfire return intervals in ecologically degraded states from the baseline values. Conversely, HFRTs decrease expected wildfire suppression costs, but these savings depend on the ability of the land manager to achieve a high level of treatment effectiveness. We expanded our analysis to consider how additional benefits from ERTs and HFRTs affect the results. Assuming that certain wildfires in healthy ecological states are to be managed through wildland fire use, rather than being actively suppressed, leads to lower expected wildfire suppression costs for ERTs, which facilitate transitions to healthy ecological states, and, similar, although quantitatively smaller, reductions for HFRTs. Further, including revenues from the sale of biomass removed when ERTs are performed significantly increased the economic benefits from ERTs, but the net economic benefits from ERTs are only positive for biomass prices that are several times those currently observed in the study area.

Given that we are not able to consider the full slate of economic benefits associated with the two treatment options, the analysis in this article is not sufficiently comprehensive to allow us to conclude categorically whether ERTs are preferable to HFRTs, or vice versa, or whether it is more economically efficient to direct treatment effort toward degraded ecological states, such as PIPO-A, PIPO-B, or PIPO-E, or toward PIPO-C, which is closer to restored conditions. Neither ERTs nor HFRTs can be justified on the basis of wildfire suppression cost savings alone, which is important for setting up realistic expectations for both treatment options. Fuel treatments are intended to modify fire behavior, making wildfires more acceptable in a given landscape, not necessarily to limit the extent of wildland fires (Reinhardt et al. 2008). The primary factors that explained the majority of variation in wildland fire suppression costs were fire size, proximity to the wildland-urban interface, and the position of the private land burned (Gebert et al. 2007, Liang et al. 2008, Yoder and Gebert 2012). Thus, the choice between ERTs and HFRTs for a particular site should depend on site-specific factors such as the wildfire return interval and the economic value of biomass removed, as well as the importance managers place on other economic benefits from fuel treatments not considered in the analysis (e.g., damage to housing and infrastructure and ecosystem services).

More broadly, this is the first study to evaluate and compare the economic benefits and costs of ERTs and HFRTs in a forested ecosystem in which increasing wildfire hazard is driven by forest overcrowding largely attributed to historical suppression of low-severity wildfires in frequent fire regimes. This innovation is important because escalating wildfire activity related to forest overcrowding is a pervasive problem in the United States and throughout the world, and the choice between ERTs and HFRTs is a prevalent management concern in this context. We address this issue by developing a simulation model of the PIPO ecosystem based on the STM ecological framework. This approach allows us to incorporate ecological dynamics into the model, including the role of wildfire as a catalyst for transitions between ecological states, as well as provisions for treatment effectiveness and longevity. Incorporating these elements is necessary to evaluate the long-run impact of ERTs and HFRTS on ecological condition, wildfire behavior, and suppression costs. In addition to analyzing the PIPO ecosystem, the simulation model developed in this article can readily be adapted to evaluate and compare the economic benefits and costs of ERTs and HFRTs in other forested ecosystems whose ecological dynamics can be described in the STM framework.

Endnotes

1. Given the slow growth rate of ponderosa pines, a plausible alternative to our assumption in the baseline model that it takes the ecosystem 80 years to transition from PIPO-B to PIPO-C through ecological succession is that there is no successional pathway from PIPO-B to PIPO-C. We reconsidered the baseline results reported in Tables 6 and 7 under the alternative assumption that there is no successional pathway between PIPO-B and PIPO-C and found that there is no appreciable difference between the alternative parameterization and the baseline model's predictions.
2. Given the discounting assumption, the 200-year time horizon is appropriate for considering the long-run economic benefits and costs of fuel treatments because any benefits and costs that occur more than 200 years in the future will have a negligible influence on the present value of net benefits. The 3% discount rate used in this study is the discount rate used in benefit-cost analysis by US federal agencies such as the National Oceanic and Atmospheric Administration, the US Department of the Interior, and the US Environmental Protection Agency (Loomis 2002).
3. The realizations of the stochastic parameters in each year are generated using the random number generator in Matlab.
4. $q^V_{1,t,m} + q^V_{2,t,m} + q^V_{3,t,m} = 1$ for all t, m, and V.
5. $r^V_{A,i,t,m} + r^V_{B,i,t,m} + r^V_{C,i,t,m} + r^V_{D,i,t,m} + r^V_{E,i,t,m} = 1$ for all t, m, i, and V.
6. There are three additional assumptions related to how HFRTs are included in the model. First, it is assumed that in model runs where HFRTs are pursued, they are performed every 15 years (year 1, year 16, …) in PIPO-A (PIPO-B) until the system transitions to PIPO-B (PIPO-C). Second, if a wildfire occurs in PIPO-A (PIPO-B) in the 15 years after the implementation of an HFRT and the wildfire is low-intensity, then the treatment remains effective (i.e., continues to increase the probability that a wildfire is low intensity) for the reminder of the 15-year life of the treatment. That is, a low-intensity wildfire does not undo the effect of an HFRT. Third, we assume that when a low-intensity wildfire occurs in the 15 years after the implementation of an HFRT, the fire does not reset the 100-year "time to transition without wildfire" between PIPO-A to PIPO-B (80-year time to transition without wildfire between PIPO-B to PIPO-C). In the absence of an HFRT, a low-intensity wildfire in PIPO-A (PIPO-B) resets the time to transition without wildfire so that it takes 100 years (80 years) to transition to PIPO-B (PIPO-C) after the low-intensity wildfire.
7. Biswell et al. (1973), Fernandes et al. (2004), and Finney (2005) report HFRT mitigation of wildfire severity out to 15, 13, and 9 years, respectively. We elected to use the 15 years reported in Biswell et al. (1973) because it is the only one of the three studies focused on the southwestern United States.
8. It is also possible that by improving the effectiveness of the wildfire suppression response, fuel treatments can lower the size of the wildfires that do occur. If this is the case, then fuel treatments would lower the effective wildfire return interval in and around the treatment area. This additional benefits of fuel treatments is also not considered in the analysis in this article, thus causing the results to further understate the wildfire suppression cost savings associated with treatment.
9. The available data for wildfire suppression expenditures do not include information that directly identifies ecological state or states at the site of each fire. The data do include, however, the National Fire Danger Rating System (NFDRS) fuel model category at the point of ignition. Correspondence between the ecological states in our stylized STM for the PIPO ecosystem and the NFDRS fuel models is made based on the vegetation composition descriptions in Anderson (1982), as summarized in Table 1.
10. The simulation assumes that wildfire fire use is employed for all low-intensity wildfires in PIPO-D. This assumption is unrealistic in light of previous studies finding that social pressure from residents in nearby communities and other stakeholders can limit the use of many fuel management techniques, including wildland fire use (McCaffrey 2006, 2009, Winter et al. 2006). It is unlikely, however, that relaxing this assumption so that only a portion of the low-intensity wildfires in PIPO-D are managed wildland fire use would meaningfully influence our results. This is the case because the average costs of suppressing low-intensity wildfires in PIPO-D is not much greater than the assumed cost of wildland fire use ($60 versus $20 per acre; see Table 1) and because it generally takes a long time (if ever) for the ecosystem to be rehabilitated to PIPO-D through ecological succession or as a result of ERTs or HFRTs, so that changes in suppression costs in PIPO-D will only have a small effect on NPVs because of discounting.
11. Concerning suppression costs, previous studies such as that of Gebert et al. (2007) have developed models to analyze the determinants of wildfire suppression costs and to predict future wildfire suppression costs. The models developed in these studies could conceivably be used to predict changes in future suppression costs in response to anticipated changes in factors such as climate change, frequency of drought conditions, and growth of the housing stock in the wildland-urban interface. The distributions of state- and intensity-specific per-acre wildfire suppression costs used in the simulation model could then be updated to incorporate these predicted changes. Modeling these predicted changes, however, is beyond the scope of this article.
12. The wildfire return intervals and the conditional probabilities of low-, moderate-, and high-severity wildfire were chosen in consultation with Tessa Nicolet (Regional Fire Ecologist at the USDA Forest Service) and Bruce Greco (retired USDA Forest Service Fire Management Officer, Coconino National Forest).
13. Schmidt et al. (2002) report wildfire return intervals, rather than annual wildfire probabilities. Annual wildfire probabilities are the reciprocal of wildfire return intervals under the assumption that wildfires occur according to a geometric distribution (i.e., the probability of a wildfire is constant and independent across years).
14. There is an apparent inconsistency between the wildfire return intervals reported in Table 2 and the "Average acres burned per fire" reported in Table 1. The apparent inconsistency is that the differences in "Average acres burned per fire" between states reported in Table 1 do not support the extent of the variation in wildfire return intervals (annual burn probabilities) between states (100 years in PIPO-A, 70 years in PIPO-B, 40 years in PIPO-C, 10 years in PIPO-D, and 80 years in PIPO-E) reported in Table 2. This apparent inconsistency underlines the shortcomings of using the data reported in Table 1 to calculate wildfire return intervals. Although the data from Table 1 provide us with the best information available on wildfire suppression costs in each ecological state in the study region given current suppression resources and priorities, the data cover a relatively short period (1995–2011) and only information on wildfires of $>$100 acres (300 acres after 2003) that "escaped" initial suppression efforts by local and state agencies is included, thus representing just a small sample of wildfires from the region. As is pointed out in Baker (2009), wildfire return intervals calculated over short time spans may be substantially influenced by one or two extreme fire years and as such may not be reliable indicators of future wildfire behavior. This is the reason why we do not use the data reported in Table 1 to calculate annual wildfire probabilities and why it is not surprising that "Average acres burned per fire" reported in Table 1 do not agree with the wildfire return intervals reported in Table 2.
15. The definition of the fire rotation interval does not imply that the entire area will burn during a cycle; rather, some sites will burn several time and others not at all.
16. The times for ecological transition were chosen in consultation with Dave Huffman of the Ecological Restoration Institute at Northern Arizona University.
17. The postwildfire transition probabilities between states were chosen in consultation with Dave Huffman of the Ecological Restoration Institute at Northern Arizona University.
18. Several studies have quantified ecosystem services in western forests, including the benefits from conservation of critical wildlife habitat (Loomis and González-Cabán 2010), from reductions in wildfire-related losses (Butry et al. 2001, Lynch 2004, Snider et al. 2006, Combrink et al. 2013), and on the impact of fuel treatments, wildfire, and forest health on real estate values (Kim and Wells 2005, Mueller et al. 2009, Stetler et al. 2010). Unfortunately, these studies do not quantified how these ecosystem services vary between ecological states in the PIPO ecosystem, and, as such, the estimates of the value of ecosystem services from these studies cannot be readily integrated into the simulation model.
19. The wildfire suppression cost data used to construct our distribution of per-acre wildfire suppression costs only contain information on wildfire of $>$100 acres (300 acres after 2003) that "escaped" initial suppression efforts by local and state agencies. As a result, the data does not contain any information on wildfires in

the wildland-urban interface, as these wildfire are generally suppressed before reaching the minimum size threshold. Similarly, our wildfire return intervals were chosen to represent current conditions on wildlands in the PIPO ecosystem and not in the wildland-urban interface.

20. There are two additional reasons why it is not possible to use the results reported in this article to infer the likely benefits and costs of ERTs and HFRTs in the wildland-urban interface. First, human-caused wildfire ignitions are more common in the wildland-urban interface than in wildlands (Cardille et al. 2001), which will cause the wildfire return interval in the wildland-urban interface to differ from those on wildlands independent of any differences in the wildfire suppression response. Second, fire managers in the wildland-urban interface may be obliged to aggressively suppress all wildfires to reduce the possibility of damage to property, infrastructure, and human life and to minimize the risk to human health related to released smoke and particulate matter. This concern for "resources at risk" in the wildland-urban interface may limit wildland fire use in ecologically restored states such as PIPO-D.
21. The variable costs of fuel treatment include labor and materials required to perform treatment on an additional acre (or any unit of land) after the fixed costs of performing fuel treatment at a given site have been committed. The fixed costs of fuel treatment include the costs of transporting equipment to and from the treatment site, the costs of equipment (including maintenance and depreciation), and the administrative costs of project planning and compliance.

Literature Cited

ABATZOGLOU, J.T., AND C.A. KOLDEN. 2011. Climate change in western US deserts: Potential for increased wildfire and invasive annual grasses. *Rangel. Ecol. Manage.* 64(5):471–478.

AGEE, J.K., AND C.N. SKINNER. 2005. Basic principles of forest fuel reduction treatments. *For. Ecol. Manage.* 211(1–2):83–96.

ALLEN-DIAZ, B., AND J.W. BARTOLOME. 1998. Sagebrush-grass vegetation dynamics: Comparing classical and state-transition models. *Ecol. Applic.* 8(3):795–804.

ANDERSON, H.E. 1982. *Aid to determining fuel models for estimating fire behavior*. USDA For. Serv., Gen. Tech. Rep. INT-122, Intermountain Forest and Range, Ogden, UT. 28 p.

BAKER, W.L. 2009. *Fire ecology in Rocky Mountain landscapes*. Island Press, Washington, DC. 628 p.

BARBOUR, R.J., X. ZHOU, AND J.P. PRESTEMON. 2008. Timber product output implications of a program of mechanical fuel treatments applied on public timberland in the western United States. *For. Policy Econ.* 10(6):373–385.

BENAYAS, J.M.R., A.C. NEWTON, A. DIAZ, AND J.M. BULLOCK. 2009. Enhancement of biodiversity and ecosystem services by ecological restoration: A meta-analysis. *Science* 325:1121–1124.

BECKER, D.R., S.M. MCCAFFREY, D. ABBAS, K.E. HALVORSEN, P. JAKES, AND C. MOSELEY. 2011. Conventional wisdoms of woody biomass utilization on federal public lands. *J. For.* 109(4):208–218.

BERRY, A.H., AND H. HESSELN. 2004. The effect of the wildland-urban interface on prescribed burning costs in the Pacific Northwestern United States. *J. For.* 102(6):33–37.

BISWELL, H.H., H.R. KALLANDER, R. KOMAREK, R.J. VOGL, AND H. WEAVER. 1973. *Ponderosa fire management: A task force evaluation of controlled burning in ponderosa pine forest of central Arizona*. Tall Timbers Research Station, Misc. Publ. No. 2, Tallahassee, FL. 49 p.

BOSTWICK, P., J. MENAKIS, AND T. SEXTON. 2011. *How fuel treatments saved homes from the Wallow Fire*. USDA For. Serv., Unpublished report, Washington, DC. 14 p.

BUTRY, D.T., E. MERCER, J.P. PRESTEMON, J.M. PYE, AND T.P. HOLMES. 2001. What is the price of catastrophic wildfire? *J. For.* 99(11):9–17.

CALKIN, D.E., AND K. GEBERT. 2006. Modeling fuel treatment costs on forest service lands in the western United States. *West. J. Appl. For.* 21(4):217–221.

CARDILLE, J.A., S.J. VENTURA, AND M.G. TURNER. 2001. Environmental and social factors influencing wildfires in the Upper Midwest, United States. *Ecol. Applic.* 11:111–127.

COHEN, J.D. 2000. Preventing disaster: Home ignitability in the wildland-urban interface. *J. For.* 98:15–21.

COMBRINK, T., C. COTHRAN, W. FOX, J. PETERSON, AND G. SNIDER. 2013. *Full cost accounting of the 2010 Schultz Fire.* ERI—Issues in Forest Restoration, Ecological Restoration Institute, Northern Arizona University, Flagstaff, AZ. 44 p.

COVINGTON, W.W., R.L. EVERETT, R. STEELE, L.L. IRWIN, T.A. DAER, AND A.N.D. AUCLAIR. 1994. Historical and anticipated changes in forest ecosystems of the Inland West of the United States. *J. Sustain. For.* 2:13–63.

COVINGTON, W.W., AND M.M. MOORE. 1994. Southwestern ponderosa pine forest structure: Changes since Euro-American settlement. *J. For.* 92(1):39–47.

ECOLOGICAL RESTORATION INSTITUTE. 2013. *The efficacy of hazardous fuel treatments: A rapid assessment of the economic and ecological consequences of alternative hazardous fuel treatments: A summary document for policy makers*. Ecological Restoration Institute, Northern Arizona University, Flagstaff, AZ. 28 p.

EIDENSHINK, J., B. SCHWIND, K. BREWER, Z. ZHU, B. QUAYLE, AND S. HOWARD. 2007. A project for monitoring trends in burn severity. *Fire Ecol.* 3(1):3–21.

EVANS, A.M., R.G. EVERETT, S.L. STEPHENS, AND J.A. YOUTZ. 2011. *Comprehensive fuels treatment practices guide for mixed conifer forests: California, Central and Southern Rockies, and the Southwest.* USDA For. Serv., Southwestern Region and The Forest Guild, Albuquerque, NM. 106 p.

FERNANDES, P.A.M., C.A. LOUREIRO, AND H.S. BOTELHO. 2004. Fire behavior severity in a maritime pine stand under differing fuel conditions. *Ann. For. Sci.* 61:537–544.

FINKRAL, A.J., AND A.M. EVANS. 2008. Effects of thinning treatment on carbon stocks in a northern Arizona ponderosa pine forest. *For. Ecol. Manage.* 255:2743–2750.

FINNEY, M.A. 2001. Design of regular landscape fuel treatment patterns for modifying fire growth and behavior. *For. Sci.* 47(2):219–228.

FINNEY, M.A. 2005. The challenge of quantitative risk assessment for wildland fire. *For. Ecol. Manage.* 211:97–108.

FINNEY, M.A., AND J.D. COHEN. 2003. Expectation and evaluation of fuel management objectives. P. 353–366 in *Fire, fuel treatments, and ecological restoration: Conference proceedings: 2002 16–18 April, Fort Collins, CO.* USDA For. Serv., Proc. RMRS-P-29, Rocky Mountain Research Station, Fort Collins, CO.

FISCHER, W.C., AND A.F. BRADLEY. 1987. *Fire ecology of western Montana forest habitat types*. USDA For. Serv., Gen. Tech. Rep. INT-223, Intermountain Forest and Range Experiment Station, Ogden, UT. 95 p.

FULÉ, P.Z., J.E. CROUSE, A.E. COCKE, M.M. MOORE, AND W.W. COVINGTON. 2004. Changes in canopy fuels and potential fire behavior 1880–2040. *Ecol. Model.* 175:231–248.

GALIK, C.S., R. ABT, AND Y. WU. 2009. Forest biomass supply in the southeastern United States implications for industrial roundwood and bioenergy production. *J. For.* 107(2):69–77.

GEBERT, K.A., D.E. CALKIN, AND J. YODER. 2007. Estimating suppression expenditures for individual large wildland fires. *West. J. Appl. For.* 22(3):188–196.

GRAHAM, R.T., AND T.B. JAIN. 2005. Ponderosa pine ecosystems. In *Proc. of the symposium on ponderosa pine: Issues, trends, and management, 2004 October 18–21, Klamath Falls, OR*, Ritchie, M.W., D.A. Maguire, and A. Youngblood (tech. coords.), USDA For. Serv., Gen. Tech. Rep PSW-GTR-198, Pacific Southwest Research Station, Albany, CA. 281 p.

HAN, H., H.W. LEE, AND L.R. JOHNSON. 2004. Economic feasibility of an integrated harvesting system for small-diameter trees in southwest Idaho. *For. Prod. J.* 54(2):21–27.

HESSBURG, P.F., R.G. MITCHELL, AND G.M. FILIP. 1994. *Historical and current roles of insects and pathogens in eastern Oregon and Washington forested landscapes*. USDA For. Serv., Gen. Tech. Rep. PNW-GTR-327, Pacific Northwest Research Station, Portland, OR. 72 p.

HJERPE, E.E., AND Y.S. KIM. 2008. Economic impacts of national forest fuels reduction programs in the Southwest. *J. For.* 106(6):311–316.

Houtman, R.M., C.A. Montgomery, A.R. Gagnon, D.E. Calkin, T.G. Dietterich, S. McGregor, and M. Crowley. 2013. Allowing a wildfire to burn: Estimating the effect on future wildfire suppression costs. *Int. J. Wildl. Fire* 22(7):871–882.

Huang, C.-H., A. Finkral, C. Sorensen, and T. Kolb. 2013. Toward full economic valuation for forest fuels reduction treatments. *J. Environ. Manage.* 130:221–231.

Hunter, M.E., W.D. Shepperd, L.B. Lentile, J.E. Lundquist, M.G. Andreu, J.L. Butler, and F.W. Smith. 2007. *A comprehensive guide to fuels treatment practices for ponderosa pine in the Black Hills, Colorado Front Range, and Southwest*. USDA For. Serv., Gen. Tech. Rep. RMRS-GTR-198, Rocky Mountain Research Station, Fort Collins, CO. 99 p.

Ince, P.J., H. Spelter, K.E. Skog, A. Kramp, and D.P. Dykstra. 2008. Market impacts of hypothetical fuel treatment thinning programs on federal lands in the western United States. *For. Policy Econ.* 10(6):363–372.

Kim, Y.-S. 2010. *Ecological restoration as economic stimulus: A regional analysis*. ERI—Issues in Forest Restoration, Ecological Restoration Institute, Northern Arizona University, Flagstaff, AZ. 20 p.

Kim, Y.-S., and A. Wells. 2005. The impacts of forest density on property values. *J. For.* 103(3):146–151.

Kline, J. 2004. *Issues in evaluating the costs and benefits of fuel treatments to reduce wildfire in the nation's forests*. USDA For. Serv., Res. Note PNW-RN-542, Pacific Northwest Research Station, Portland, OR. 49 p.

Larson, D., and R. Mirth. 2004. A case study on the economics of thinning in the wildland urban interface. *West. J. Appl. For.* 19(1):60–65.

Liang, J., D.E. Calkin, K.M. Gebert, T.J. Venn, and R.P. Silverstein. 2008. Factors influencing large wildland fire suppression expenditures. *Int. J. Wildl. Fire* 17:650–659.

Loomis, J.B. 2002. *Integrated public lands management: Principles and applications to national forests, parks, wildlife refuges, and BLM lands*. Columbia University Press, New York. 544 p.

Loomis, J., and A. González-Cabán. 2010. Forest service use of nonmarket valuation in fire economics: Past, present, and future. *J. For.* 108(8):389–396.

Lynch, D.L. 2004. What do forest fires really cost? *J. For.* 102(6):42–49.

McCaffrey, S.M. 2006. *The public and wildland fire management: Social science findings for managers.* USDA For. Serv., Gen. Tech. Rep. NRS-1, Northern Research Station, Newtown Square, PA. 202 p.

McCaffrey, S.M. 2009. Crucial factors influencing public acceptance of fuel treatments. *Fire Manage. Today* 69(1):9–12.

Mercer, D.E., and J.P. Prestemon. 2008. Economic analysis of fuel treatment. P. 294–307 in *Cumulative watershed effects of fuels management in the Eastern United States*, LaFayette, R., M.T. Brooks, J.P. Potyondy, L. Audin, S.L. Krieger, and C.C. Trettin (eds.). USDA For. Serv., Gen. Tech. Rep. SRS-161, Southern Research Station, Asheville, NC.

Mercer, D.E., J.P. Prestemon, D.T. Butry, and J.M. Pye. 2007. Evaluating alternative prescribed burning policies to reduce net economic damages from wildfire. *Am. J. Agric. Econ.* 89(1):63–77.

Millar, C.I., N.L. Stephenson, and S.L. Stephens. 2007. Climate change and forests of the future: Managing in the face of uncertainty. *Ecol. Applic.* 17(8):2145–2151.

Moghaddas, J.J., and L. Craggs. 2007. A fuel treatment reduces fire severity and increases suppression efficiency in a mixed conifer forest. *Int. J. Wildl. Fire* 16(6):673–678.

Moir, W.H., and J.H. Dieterich. 1988. Old-growth ponderosa pine from succession in pine bunchgrass forests in Arizona and New Mexico. *Natur. Areas J.* 8(1):17–24.

Mueller, J.M., J.B. Loomis, and A. González-Cabán. 2009. Do repeated wildfires change homebuyers' demand for homes in high-risk areas? A hedonic analysis of the short and long-term effects of repeated wildfires on house prices in Southern California. *J. Real Estate Finan. Econ.* 38(2):155–117.

Murphy, K., T. Rich, and T. Sexton. 2007. An assessment of fuel treatment effects on fire behavior, suppression effectiveness, and structure ignition on the Angora Fire. USDA For. Serv., R5-TP 025, Pacific Southwest Region, Vallejo, CA. 32 p.

Ohlson, D.W., T.M. Berry, R.W. Gray, B.A. Blackwell, and B.C. Hawkes. 2006. Multi-attribute evaluation of landscape-level fuel management to reduce wildfire risk. *For. Policy Econ.* 8(1):824–837.

Prestemon, J.P., K.L. Abt, and R.J. Barbour. 2012. Quantifying the net economic benefits of mechanical wildfire hazard treatments on timberlands of the western United States. *For. Policy Econ.* 21:44–53.

Prestemon, J.P., K.L. Abt, and R.J. Huggett Jr. 2008. Market impacts of a multiyear mechanical fuel treatment program in the US. *For. Policy Econ.* 10(6):386–399.

Reinhardt, E.D., R.E. Keane, D.E. Calkin, and J.D. Cohen. 2008. Review: Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. *For. Ecol. Manage.* 256:1997–2006.

Rogers, G., W. Hann, C. Martin, T. Nicolet, and M. Pence. 2008. *Fuel treatment effects on fire behavior, suppression effectiveness, and structure ignition, Grass Valley fire, San Bernadino National Forest*. USDA For. Serv., R5-TP-026a, Region 5, Vallejo, CA. 35 p.

Rummer, B. 2008. Assessing the cost of fuel reduction treatments: A critical review. *For. Policy Econ.* 10(6):355–362.

Rummer, B., J. Prestemon, D. May, P. Miles, J. Vissage, R. McRoberts, G. Liknes, et al. 2005. *A strategic assessment of forest biomass and fuel reduction treatments in western States*. USDA For. Serv., Gen. Tech. Rep. RMRS-GTR-149, Rocky Mountain Research Station, Fort Collins, CO. 17 p.

Savage, M., and J.N. Mast. 2005. How resilient are southwestern ponderosa pine forests after crown fires? *Can. J. For. Res.* 35:967–977.

Schmidt, K.M., J.P. Menakis, C.C. Hardy, J. Wendel, and D.L. Bunnell. 2002. *Development of coarse-scale spatial data for wildland fire and fuel management.* USDA For. Serv., Gen. Tech. Rep. RMRS-GTR-87, Rocky Mountain Research Station, Fort Collins, CO. 41 p.

Sesnie, S., and J.D. Bailey. 2003. Using history to plan the future of old-growth ponderosa pine. *J. For.* 101(7):40–47.

Stephens, L.S., J.D. McIver, R.E.J. Boerner, C.J. Fettig, J.B. Fontaine, B.R. Hartsough, P. Kennedy, and D.W. Schwilk. 2012. Effects of forest fuel reduction treatments in the United States. *BioScience* 62(6):549–560.

Stetler, K.M., T.J. Venn, and D.E. Calkin. 2010. The effects of wildfire and environmental amenities on property values in northwest Montana, USA. *Ecol. Econ.* 69:2233–2243.

Stratton, R.D. 2004. Assessing the effectiveness of landscape fuel treatments on fire growth and behavior. *J. For.* 102(7):32–41.

Snider, G., P.J. Daugherty, and D. Wood. 2006. The irrationality of continued fire suppression: An avoided cost analysis of fire hazard reduction treatments versus no treatment. *J. For.* (104):431–437.

Taylor, M.H., K. Rollins, M. Kobayashi, and R.J. Tausch. 2013. The economics of fuel management: Wildfire, invasive plants, and the evolution of sagebrush rangelands in the western United States. *J. Environ. Manage.* 126:157–173.

Thompson, M.P., N.M. Vaillant, J.R. Haas, K.M. Gebert, and K.D. Stockmann. 2013. Quantifying the potential impacts of fuel treatments on wildfire suppression costs. *J. For.* 111(1):49–58.

US Department of Energy. 2011. *US billion-ton update: Biomass supply for a bioenergy and bioproducts industry*. ORNL/TM-2011/224, Oak Ridge National Laboratory, Oak Ridge, TN. 235 p.

US Government Accounting Office. 2006. *Update on federal agency efforts to develop a cohesive strategy to address wildland fire threats*. US Government, GAO-06–671R, Washington, DC. 19 p.

Wang, X., H.S. He, and X. Li. 2007. The long-term effects of fire suppression and reforestation on a forest landscape in northeastern China after a catastrophic wildfire. *Landsc. Urban Plan.* 79:84–95.

WESTERLING, A.L., H.G. HIDALGO, D.R. CAYAN, AND T.W. SWETNAM. 2006. Warming and earlier spring increases western US forest wildfire activity. *Science* 313:940–943.

WESTOBY, M., B.H. WALKER, AND N. MEIR. 1989. Opportunistic management for rangelands at disequilibrium. *J. Range Manage.* 42:266–274.

WINTER, G., AND J.S. FRIED. 2000. Homeowner perspectives on fire hazard, responsibility, and management strategies at the wildland-urban interface. *Soc. Natur. Resour.* 13(1):33–49.

WINTER, G., C. VOGT, AND S. MCCAFFREY. 2006. Residents warming up to fuels management: Homeowners? Acceptance of wildfire and fuels management in the wildland-urban interface. P. 19–32 in *The public and wildland fire management: Social science findings for managers*, McCaffrey, S.M. (tech. ed.). USDA For. Serv., Gen. Tech. Rep. NRS-1, Northern Research Station, Newtown Square, PA.

YODER, J., AND K. GEBERT. 2012. An econometric model for ex ante prediction wildfire suppression costs. *J. For. Econ.* 18:76–89.

Appendix A: Sensitivity of the Results to the Number of Years to Transition between Ecological States without Wildfire

In this appendix, we examine the sensitivity of our results to our assumption concerning the number of years to transition between ecological states in the PIPO ecosystem without wildfire. We perform sensitivity analysis on these parameters because they have the least empirical support in the ecology literature. Tables A1 and A2 report results for both ERTs and HFRTs when the initial state is PIPO-A, and the times to transition between PIPO-A and PIPO-B without wildfire are 50, 150, and 200% of the baseline value of 100 years. Similarly, Tables A3 and A4 (Table A5) report results for both ERTs and HFRTs when the initial state is PIPO-B (PIPO-C) and the times to transition in PIPO-B (PIPO-C) are 50, 150, and 200% of the baseline value of 80 years (40 years).

We do not perform sensitivity analysis on our baseline assumptions on the time to transition between ecological states without wildfire in PIPO-D or PIPO-E. We do not perform sensitivity analysis on these parameters because, in the case of PIPO-D, the time to transition without wildfire is long (200 years), so that varying it, even substantially, is unlikely to influence the results. Similarly, as the ecosystem only transitions to PIPO-E after a crown fire, which is a relatively rare occurrence, varying our assumption about the number of years to transition back to PIPO-A from PIPO-E without a wildfire has little effect on the results reported in this article.

The results reported in Tables A1–A5 indicate that our results are not sensitive to the number of years to transition without wildfire and that our conclusions would largely be intact if we were to change these parameters, even substantially, from the baseline values. Our results are not sensitive to these assumptions because the times to transition without treatment and wildfire between PIPO-A and PIPO-B and PIPO-B and PIPO-C are relatively long (80 and 100 years, respectively), so the changes in expected wildfire suppression costs associated with changes in these parameters occurs far in the planning horizon, and, as a result, has little influence on the NPV of wildfire suppression costs. On the other hand, whereas shortening (lengthening) the time to transition between PIPO-C and PIPO-D leads to faster (slower) rehabilitation to PIPO-D with ERT or HFRT, the increase in the number of wildfires in PIPO-D is balanced by the reduced per-fire suppression costs in PIPO-D (the majority of wildfires in PIPO-D are low-severity wildfires that can be managed via wildland fire use), so that on balanced shortening (lengthening) the time to transition between PIPO-C and PIPO-D has only a small influence on the NPV of wildfire suppression costs associated with treatment.

Table A1. Varying the time to transition without wildfire in PIPO-A: initial state PIPO-A, ERTs.

	Number of years to transition without wildfire in PIPO-A			
ERTs = PIPO-A	50 years	100 years[a]	150 years	200 years
Mean total number of wildfires: no treatment	2.45 (1, 5)[b]	2.34 (0, 5)	2.23 (0, 5)	2.18 (0, 5)
Mean total number of wildfires: with treatment	7.98 (2, 18)	8.2 (2, 18)	8.06 (2, 18)	8.06 (2, 18)
Mean total number of good wildfires: no treatment	0.15 (0, 1)	0.10 (0, 1)	0.08 (0, 1)	0.06 (0, 1)
Mean total number of good wildfires: treatment	6.07 (0, 18)	6.2 (0, 18)	6.1 (0, 18)	6.10 (0, 18)
Mean total number of bad wildfires: no treatment	2.29 (0, 5)	2.25 (0, 5)	2.15 (0, 5)	2.12 (0, 5)
Mean total number of bad wildfires: treatment	1.91 (0, 5)	1.99 (1, 5)	1.96 (0, 5)	1.96 (0, 5)
Mean total suppression costs (NPV): no treatment[c]	$174 ($0, $696)	$165 ($0, $655)	$154 ($0, $598)	$154 ($0, $598)
Mean total suppression costs (NPV): with treatment[c]	$217 ($10, $717)	$217 ($10, $728)	$219 ($9, $737)	$219 ($9, $737)
Mean number of treatments	4.9 (1, 8)	4.9 (1, 9)	4.9 (1, 9)	4.9 (1, 9)
Mean treatment costs (NPV)[c]	$2,443 ($2,100, $3,034)	$2,460 ($2,100, $3,038)	$2,464 ($2,100, $3,076)	$2,464 ($2,100, $3,076)
Final state: no treatment (A, B, C, D, E)[d]	209, 183, 79, 4, 525	3,286, 401, 861, 16, 5,436	325, 112, 8, 0, 555	437, 0, 0, 0, 563
Final state: with treatment (A, B, C, D, E)	66, 59, 73, 514, 288	623, 567, 727, 5,126, 2,957	68, 52, 74, 509, 297	68, 52, 74, 509, 297

[a] Baseline scenario reported in Tables 6 and 7.
[b] 5th and 95th percentiles.
[c] Sum over 200 years using a 3% discount rate (following Loomis 2002).
[d] "Final state" is the final state of the system (PIPO-A, PIPO-B, PIPO-C, PIPO-D, or PIPO-E) after 200 years.

Table A2. Varying the time to transition without wildfire in PIPO-A: initial state PIPO-A; HFRTs.

HFRTs (initial state = PIPO-A)	Number of years to transition without wildfire in PIPO-A			
	50 years	100 years[a]	150 years	200 years
Mean total number of wildfires: no treatment	2.52 (1, 5)[b]	2.35 (0, 5)	2.2 (0, 5)	2.21 (0, 5)
Mean total number of wildfires: with treatment	3.77 (1, 8)	2.54 (0, 5)	2.14 (0, 5)	2.05 (0, 5)
Mean total number of good wildfires: no treatment	0.19 (0, 1)	0.10 (0, 1)	0.06 (0, 1)	0.05 (0, 0)
Mean total number of good wildfires: treatment	2.75 (0, 7)	1.73 (0, 4)	1.60 (0, 4)	1.47 (0, 4)
Mean total number of bad wildfires: no treatment	2.33 (0, 5)	2.25 (0, 5)	2.14 (0, 5)	2.16 (0, 5)
Mean total number of bad wildfires: treatment	1.02 (0, 3)	0.81 (0, 3)	0.54 (0, 3)	0.58 (0, 3)
Mean total suppression costs (NPV): no treatment[c]	$158 ($1, $598)	$164 ($0, $687)	$171 ($0, $696)	$171 ($0, $652)
Mean total suppression costs (NPV): with treatment[c]	$99 ($1, $436)	$113 ($0, $487)	$122 (0, $522)	$120 ($0, $512)
Mean number of treatments	11.22 (5, 15)	12.0 (5, 14)	12.4 (5, 14)	11.5 (5, 13)
Mean treatment costs (NPV)[c]	$163 ($102, $171)	$154 ($103, $203)	$151 ($99, $159)	$152 ($104, $159)
Final state: no treatment (A, B, C, D, E)[d]	203, 192, 67, 6, 532	3,283, 398, 859, 13, 5,447	324, 119, 6, 0, 551	424, 0, 0, 0, 576
Final state: with treatment A, B, C, D, E)	119, 247, 11, 367, 256	2,250, 699, 4,783, 103, 2,165	136, 713, 6, 0, 145	823, 0, 0, 0, 177

[a] Baseline scenario reported in Tables 6 and 7.
[b] 5th and 95th percentiles.
[c] Sum over 200 years using a 3% discount rate (following Loomis 2002).
[d] "Final state" is the final state of the system (PIPO-A, PIPO-B, PIPO-C, PIPO-D, or PIPO-E) after 200 years.

Table A3. Varying the time to transition without wildfire in PIPO-B: initial state PIPO-B; ERTs.

ERTs (initial state = PIPO-B)	Number of years to transition without wildfire in PIPO-B			
	40 years	80 years[a]	120 years	160 years
Mean total number of wildfires: no treatment	3 (1, 6)[b]	2.83 (1, 6)	2.69 (1, 5)	2.56 (1, 5)
Mean total number of wildfires: with treatment	10.12 (2, 20)	10.3 (2, 21)	10.41 (2, 21)	10.29 (3, 21)
Mean total number of good wildfires: no treatment	0.52 (0, 2)	0.33 (0, 1)	0.27 (0, 1)	0.18 (0, 1)
Mean total number of good wildfires: treatment	8.40 (0, 20)	8.5 (0, 21)	8.70 (0, 21)	8.51 (0, 21)
Mean total number of bad wildfires: no treatment	2.48 (1, 5)	2.5 (1, 5)	2.43 (1, 5)	2.38 (0, 5)
Mean total number of bad wildfires: treatment	1.72 (0, 5)	1.78 (0, 5)	1.71 (0, 5)	1.78 (0, 5)
Mean total suppression costs (NPV): no treatment[c]	$201 ($6, $689)	$200 ($3, $763)	$209 ($3, $763)	$192 ($0, $711)
Mean total suppression costs (NPV): with treatment[c]	$253 ($16, $803)	$256 ($16, $800)	$245 ($16, $753)	$253 ($16, $774)
Mean number of treatments	4.4 (1, 8)	4.4 (1, 8)	4.4 (1, 8)	4.5 (2, 8)
Mean treatment costs (NPV)[c]	$744 ($500, $1,545)	$730 ($500, $1,503)	$720 ($500, $1,549)	$727 ($508, $1,438)
Final state: no treatment (A, B, C, D, E)[d]	295, 62, 73, 25, 545	3,064, 828, 550, 103, 5,455	326, 89, 38, 7, 540	303, 95, 55, 1, 546
Final state: with treatment (A, B, C, D, E)	56, 42, 61, 608, 233	511, 500, 630, 5,960, 2,399	61, 44, 51, 608, 236	53, 56, 64, 591, 236

[a] Baseline scenario reported in Tables 6 and 7.
[b] 5th and 95th percentiles.
[c] Sum over 200 years using a 3% discount rate (following Loomis 2002).
[d] "Final state" is the final state of the system (PIPO-A, PIPO-B, PIPO-C, PIPO-D, or PIPO-E) after 200 years.

Table A4. Varying the time to transition without wildfire in PIPO-B: initial state PIPO-B; HFRTs.

HFRTs (initial state = PIPO-B)	Number of years to transition without wildfire in PIPO-B			
	40 years	80 years[a]	120 years	160 years
Mean total number of wildfires: no treatment	3.06 (1, 6)[b]	2.80 (1, 5)	2.71 (1, 6)	2.58 (1, 5)
Mean total number of wildfires: with treatment	7.48 (1, 17)	5.56 (1, 12)	4.23 (1, 8)	3.03 (1, 6)
Mean total number of good wildfires: no treatment	0.51 (0, 2)	0.32 (0, 1)	0.25 (0, 1)	0.25 (0, 1)
Mean total number of good wildfires: treatment	6.27 (0, 17)	4.44 (0, 13)	3.21 (0, 8)	2.08 (0, 5)
Mean total number of bad wildfires: no treatment	2.54 (1, 5)	2.48 (1, 5)	2.45 (1, 5)	2.33 (0, 5)
Mean total number of bad wildfires: treatment	1.22 (0, 4)	1.11 (0, 4)	1.02 (0, 3)	0.94 (0, 3)
Mean total suppression costs (NPV): no treatment[c]	$204 ($6, $676)	$202 ($3, $766)	$208 ($2, $721)	$193 ($1, $793)
Mean total suppression costs (NPV): with treatment[c]	$107 ($6, $335)	$77 ($2, $284)	$65 ($1, $239)	$65 ($1, $247)
Mean number of treatments	8.0 (3, 14)	9.0 (5, 14)	10.7 (6, 14)	11.5 (6, 14)
Mean treatment costs (NPV)[c]	$145 ($121, $165)	$148 ($243, $991)	$153 ($122, $160)	$153 ($121, $159)
Final state: no treatment (A, B, C, D, E)[d]	307, 74, 69, 21, 529	3,073, 844, 567, 98, 5,418	326, 83, 50, 6, 535	301, 98, 59, 3, 539
Final state: with treatment (A, B, C, D, E)	198, 90, 87, 464, 161	2,219, 1,408, 786, 3,725, 1,862	237, 162, 12, 343, 246	257, 186, 294, 20, 243

[a] Baseline scenario reported in Tables 6 and 7.
[b] 5th and 95th percentiles.
[c] Sum over 200 years using a 3% discount rate (following Loomis 2002).
[d] "Final state" is the final state of the system (PIPO-A, PIPO-B, PIPO-C, PIPO-D, or PIPO-E) after 200 years.

Table A5. Varying the time to transition without wildfire in PIPO-C: initial state PIPO-C; both ERTs and HFRTs.

Initial state = PIPO-C	Number of years to transition without wildfire in PIPO-C			
	20 years	40 years[a]	60 years	80 years
Mean total number of wildfires: no treatment	4.88 (1, 18)[b]	3.39 (1, 6)	3.43 (1, 6)	3.44 (1, 6)
Mean total number of wildfires: with treatment	14.67 (3, 24)	10.4 (2, 21)	7.85 (2, 18)	5.87 (2, 15)
Mean total number of good wildfires: no treatment	2.43 (0, 18)	0.71 (0, 2)	0.83 (0, 2)	0.71 (0, 2)
Mean total number of good wildfires: treatment	13.62 (0, 24)	8.7 (0, 21)	5.73 (0, 18)	3.33 (0, 15)
Mean total number of bad wildfires: no treatment	2.45 (0, 6)	2.68 (1, 5)	2.6 (1, 5)	2.73 (1, 6)
Mean total number of bad wildfires: treatment	1.05 (0, 5)	1.73 (0, 5)	2.12 (0, 5)	2.54 (0, 6)
Mean total suppression costs (NPV): no treatment[c]	$265 ($10, $784)	$275 ($10, $807)	$263 ($9, $775)	$277 ($12, $809)
Mean total suppression costs (NPV): with treatment[c]	$219 ($25, $774)	$259 ($15, $811)	$263 ($10, $787)	$279 ($9, $806)
Mean number of treatments	2.48 (1, 5)	4.5 (1, 8)	5.39 (1, 10)	6.22 (1, 12)
Mean treatment costs (NPV)[c]	$223 ($60, $1,126)	$298 ($60, $1,093)	$317 ($60, $1,072)	$332 ($60, $1,145)
Final State: no treatment (A, B, C, D, E)[d]	270, 73, 36, 150, 471	3,071, 845, 552, 279, 5,253	299, 87, 56, 34, 524	296, 99, 48, 24, 533
Final state: with treatment (A, B, C, D, E)	16, 16, 25, 801, 142	506, 503, 571, 6,090, 2,330	70, 60, 112, 436, 322	79, 92, 143, 284, 402

[a] Baseline scenario reported in Tables 6 and 7.
[b] 5th and 95th percentiles.
[c] Sum over 200 years using a 3% discount rate (following Loomis 2002).
[d] "Final state" is the final state of the system (PIPO-A, PIPO-B, PIPO-C, PIPO-D, or PIPO-E) after 200 years.

Appendix B: Treatment Costs in Ponderosa Pine Forest Ecosystems

The cost of the proposed treatments includes many different components, which can be classified into two categories: operation costs directly related to application of the treatment itself (e.g., equipment and labor); and administrative costs (e.g., site preparation, monitoring, and project planning). In addition, the proposed treatments need to go through the process for National Environmental Policy Act of 1969 (NEPA) compliance, and the costs associated with the process (e.g., archeological survey and documentation) should also be included in the administrative costs. Four national forests were contacted to gather the most recent transaction evidence (FY 2008), expert opinions, and planning estimates at each. Based on these estimates and previously published articles in this region (Larson and Mirth 2004, Hjerpe and Kim 2008), an average cost per acre for the proposed treatments in each national forest was calculated. Although the study is based on the most recent and reliable estimates, some assumptions were necessary to generate estimates of average treatment costs per acre. The following sources of variation were not incorporated into treatment cost calculations but should be acknowledged.

- Treatment costs for the Forest Service consist of fixed (e.g., administrative costs) and variable costs (e.g., operation costs). Fixed costs per acre decrease significantly as the project size increases. In most cases, national forests reported average costs by dividing the total expenditure by the treated acres. This is most evident in cost variations for prescribed burning, for which cost per acre varies a great deal depending on project sizes and locations.
- The administrative costs incurred by public agencies are mostly fixed costs often spread out over many years and involve multiple branches within the agency, which make it difficult to estimate a per-acre base. The cost estimates also vary depending on how much planning, preparation, operations, and monitoring are done by agency personnel, rather than contracted out.
- Treatments within the wildland-urban interface are likely to be more expensive because the treatments are smaller in size and require more personnel and equipment as well as more intensive monitoring (see Berry and Hesseln 2004, Calkin and Gebert 2006).
- Treatment costs probably vary depending on site conditions and the severity of treatments. However, engineering cost estimation models that can account for those variations are often designed to minimize cost per harvested volume of merchantable timber and are not directly applicable to ERTs (Rummer 2008), especially in the Southwest where the majority of removed volume is probably submerchantable in the conventional sense. ERTs are different from commercial harvesting in terms of objectives, methods, and outcomes. To achieve full restoration goals, the treatments should be designed to emulate the structure of presettlement forests (e.g., grouped/clumpy stand structure, rather than even-spaced), treat fuel loads, and restore fire in prescription as well as reducing stand densities (Fulé et al. 2004). Without these additional operations, logging itself can actually exacerbate the fire risk by deteriorating site conditions (Rummer 2008).

There are also inherent limitations to estimating costs based on expert opinion, transaction evidence, and planning estimates. As Rummer (2008) points out, these estimates document costs ex post and may not accurately reflect changing treatment conditions and technology. In fact, the Office of Management and Budget believes that the agencies cannot produce credible, long-term funding estimates for wildland fire risk reduction treatments at this point and allows the agencies to publish the estimates only with sufficiently reliable data (US Government Accounting Office 2006).