1 **Production possibility frontiers and socioecological tradeoffs for restoration of fire adapted** 2 3 4 5 6 7 **forests**

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10 Alan A. Ager¹, Michelle Day² and Kevin Vogler³

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13 ¹ USDA Forest Service, Rocky Mountain Research Station, Missoula Fire Sciences
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16 Laboratory, 5775 US Highway 10W, Missoula, MT 59808 USA, (541) 278-3740, e-mail:
17
18 aager@fs.fed.us (corresponding author)

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20
21 ² Oregon State University, College of Forestry, Forest Ecosystems & Society, 321 Richardson
22
23 Hall, Corvallis, OR 97331 USA, email: michelle.day@oregonstate.edu

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25 ³ Oregon State University, College of Forestry, Forest Engineering, Resources &
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27
28 Management, 043 Peavy Hall, Corvallis, OR 97331 USA, email: kevin.vogler@oregonstate.edu
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31 32 **Abstract**

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35 We used spatial optimization to analyze alternative restoration scenarios and quantify
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38 tradeoffs for a large, multifaceted restoration program to restore resiliency to forest landscapes in
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40 the western US. We specifically examined tradeoffs between provisional ecosystem services,
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42 fire protection, and the amelioration of key ecological stressors. The results revealed that
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44 attainment of multiple restoration objectives was constrained due to the joint spatial patterns of
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46 ecological conditions and socioeconomic values. We also found that current restoration projects
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48 are substantially suboptimal, perhaps the result of compromises in the collaborative planning
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50 process used by federal planners, or operational constraints on forest management activities. The
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52 juxtaposition of ecological settings with human values generated sharp tradeoffs, especially with
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54 respect to community wildfire protection versus generating revenue to support restoration and
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fire protection activities. The analysis and methods can be leveraged by ongoing restoration programs in many ways including: 1) integrated prioritization of restoration activities at multiple scales on public and adjoining private lands, 2) identification and mapping of conflicts between ecological restoration and socioeconomic objectives, 3) measuring the efficiency of ongoing restoration projects compared to the optimal production frontier, 4) consideration of fire transmission among public and private land parcels as a prioritization metric, and 5) finding socially optimal regions along the PPF as part of collaborative restoration planning.

Keywords: forest restoration, spatial optimization, wildfire, restoration prioritization, ecosystems services, restoration tradeoffs

1. Introduction

Restoration programs in many socioecological systems face substantial challenges prioritizing activities and balancing competing objectives (Maron and Cockfield 2008, Bullock et al. 2011, Allan et al. 2013). These challenges have inspired researchers to develop a wide range of decision support frameworks and tools to help disentangle the spatial and temporal dimensions of restoration goals, and prioritize landscapes for restoration projects (Moilanen et al. 2009, Noss et al. 2009, Watts et al. 2009). These frameworks include the use of production possibility frontiers (PPF) to analyze and communicate decision tradeoffs in the production of ecosystem services generated from restoration programs (Maron and Cockfield 2008, Cavender-Bares et al. 2015a). Tradeoff analyses reveal how the joint spatial organization of ecosystem stressors and services create conflicts and opportunities for restoration programs (Bennett et al. 2009, Allan et al. 2013, Schroter et al. 2014). For instance, spatially correlated restoration opportunities, i.e. co-located stressors and ecosystem services, create opportunities to achieve multiple restoration goals and sustain the production of various ecosystem services (Bennett et al. 2009). The use of PPFs and tradeoff analyses have been discussed as a useful framework for collaborative planning by providing the methods to quantify decision tradeoffs to stakeholders and find socially acceptable and ecologically optimal outcomes (Schroter et al. 2014, Cavender-Bares et al. 2015b, King et al. 2015).

A potential and largely unexplored application of PPFs and tradeoff analyses concerns the restoration of fire adapted forests in western North America. A century of selective logging, grazing, and fire suppression has led to widespread densification of forests and a reduction in fire resilient tree species (Noss et al. 2006, USDA Forest Service 2012), most notably ponderosa pine

(*Pinus ponderosa* Laws.). The result has been widespread expansion of forests that are prone to high intensity wildfires and bark beetle epidemics. Large scale restoration programs initiated on the US national forests have been addressing the problem using a number of management techniques including: 1) selective thinning to reduce stand density, increase average tree size, reduce surface and ladder fuels, and remove fire and drought intolerant species; and 2) mechanical treatments and prescribed fire to reduce surface and activity fuels generated from thinning operations (Brown et al. 2004, Agee and Skinner 2005)(Fig. 1). Forest restoration programs have been widely discussed in the literature including ecological aspects (Moore et al. 1999, Brown et al. 2004, Noss et al. 2006), planning frameworks (Franklin and Johnson 2012), implementation plans (Rieman et al. 2010), scientific guidelines (Franklin and Johnson 2012), social constraints (Franklin et al. 2014) and conflicts with biological conservation efforts (Myers 1995, Prather et al. 2008).

Despite the scrutiny of the program, the issue of prioritizing restoration investments across vast tracts of federal forests in the western US and quantifying associated tradeoffs among expected ecosystem services has received little attention. Restoration planning is inherently complex owing to the broad mix of underlying socioecological goals (USDA Forest Service 2006, 2013). For instance, restoring historical fire adapted structure in dry fire-prone forests (Noss et al. 2006) while meeting economic outputs expected from restoration programs (Rasmussen et al. 2012) may not result in acceptable levels of wildfire risk reduction for communities on adjacent private lands (Ager et al. 2015), and may adversely impact habitat conservation reserves (Gaines et al. 2010). Prioritization on US national forests in particular is further labored by collaborative planning processes enacted in US federal statutes (Schultz et al. 2012, Butler et al. 2015) where diverse stakeholder groups actively participate in the planning

process. Tradeoff analysis tools and frameworks (e.g., King et al. 2015) to support restoration planning, either in a collaborative venue or otherwise, do not exist at either policy or implementation scales, despite their potential to improve the chance of long-term success (Rappaport et al. 2015).

In this paper we describe the application of new analytical methods to analyze restoration associated tradeoffs on 3 million ha of fire-prone forests in the interior Pacific Northwest, USA. The study area was identified as a national priority to restore ecological resiliency to the diverse forest ecosystems, protect communities from wildfire, and provide economic opportunity to local wood processing mills (Rasmussen et al. 2012). However, the compatibility of these various socioecological objectives including both provisional ecosystem services (wood supply) and key socioecological stressors (wildfire risk, insects) and alternative prioritization schemes have yet to be examined. We asked three primary questions: 1) are there significant tradeoffs among socioecological restoration outcomes expected from the program; 2) are there benefits to a prioritization framework, i.e. can restoration goals be achieved more rapidly by focusing restoration investments on key areas, or are restoration targets evenly distributed; and 3) how efficient are current restoration activities relative to optimal as defined by optimal production possibility frontiers? The study provides both new methods and concepts for forest restoration planning and provides the first of socioecological tradeoff analyses of forest restoration using spatial optimization methods.

2. Methods

2.1 Study area

The four national forests (Malheur, Ochoco, Umatilla and Wallowa-Whitman) in the Blue Mountain ecoregion of eastern Oregon and southeastern Washington cover 2.5 million ha (Fig. 2). The area contains numerous small mountain ranges with steep canyons and large areas of plateau, and is steeply dissected by several rivers as part of the Columbia River basin. Elevations are mainly between 900 and 1500 m, although the highest peaks reach close to 3,000 m. Dry forests of largely ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) dominate the lower elevations, with dry mixed conifer (grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and moist conifer forests at the higher elevations. Cold dry forested areas are dominated by pure lodgepole pine (*Pinus contorta* Douglas ex Loudon) stands found throughout the area at mid to high elevations. In the managed portions of the study area the forests are a mosaic of stand ages, density, and species composition as a result of management and past disturbances. Wildfires and insect outbreaks in particular have impacted stand structure and composition over wide areas. About 22,000 ha (0.9%) is consumed annually by wildfires (1992-2013) (Short 2015), most of which are lightning caused. Major forest insect epidemics are a regular occurrence (Ager et al. 2004) with current outbreaks observed for mountain pine beetle (*Dendroctonus ponderosae* Hopkins) and western pine beetle (*D. brevicomis* LaConte). A number of studies in and around the study area have documented departure in stand structure and species composition from historical conditions due to fire exclusion. Most recently Hagmann et al. (2013) reported that stand densities have more than tripled over the past 90 years (68 ± 28 trees ha⁻¹ to 234 ± 122 trees ha⁻¹) while mean basal area increased by less than 20%. Most importantly, basal area of larger, fire resilient trees (>53 cm dbh) declined by >50%, and the abundance of large trees as a proportion of the total number of trees per hectare decreased by more than a factor of five.

The US Forest Service (USFS) plans forest restoration treatments on about 50,000 ha annually or about 1.4% of the total managed area (excluding wilderness and roadless areas). It is estimated that 34% (506,696 ha) of managed forests are in need of active restoration (USDA Forest Service 2013). Restoration objectives include protecting and retaining ecosystem services including clean air, clean water, biodiversity, recreational opportunities, and other services that are threatened by large scale disturbance. Specific treatments mirror management activities on other national forests (Agee and Skinner 2005, Roccaforte et al. 2008), where overstocked stands are thinned from below and surface fuels are treated to reduce potential wildfire behavior and improve forest health. Restoration projects specify on average treating about 5000 ha within a planning boundary of about 14,000 ha. Treatment thresholds (i.e., the selection of stands to treat) and the particular thinning intensities follow guidelines by Cochran et al. (1994; see section 2.2.4).

2.2 Modeled restoration objectives

We obtained individual stand polygon layers for each of the four national forests from USFS GIS spatial data libraries and combine these into a single layer. The stand layer delineates forest vegetation based on tree species composition, tree density, tree size, and management history (Appendix Fig. S1) according to standard inventory protocols. The original stand polygon layer was created through photo interpretation of 1:12,000 color photos in the late 1980s. The stand delineation is continually updated for management activities and disturbances. The resulting combined layer contained 200,637 polygons averaging 12 ha in area. Many stand boundaries follow natural breaks in vegetation resulting from topographic influences on growing conditions, particularly moisture (Appendix Fig. S1). The layer is used by the four national forests for a range of planning activities including design and implementation of restoration projects.

We then attributed each polygon with the following metrics describing restoration objectives (henceforth “objectives”): 1) potential fire hazard, 2) wildfire exposure to the wildland urban interface (WUI), 3) ecological departure from historic conditions, 4) potential stemwood volume from mechanical thinning treatments, and 5) potential mortality from insects and disease (Fig. 3). Data sources for each objective are described below with additional details in Appendix S1. Each polygon was attributed with land management designation based on respective forest plans, and only those areas whose primary management objectives included harvest activities to produce forest products were considered for treatment (Ager et al. 2015). This stratification removed wilderness and inventoried roadless areas, recreation sites, research natural areas, and a number of other protected or restricted areas. The remaining lands where treatments were allowed consisted of 145,395 polygons, ranging in size from <1 ha to 493 ha (mean = 10.6 ha), and covered 1,542,226 ha (64% of the study area).

2.2.1 Fire hazard

We used FlamMap (Finney 2006) to obtain an estimate of potential fire behavior for each stand assuming static weather conditions and fuel moisture (Appendix S1). FlamMap is a comprehensive fire simulation software package that is widely used in the US and elsewhere to model potential fire behavior. In the current application, surface and canopy fuels obtained from LANDFIRE (2013) data were used to estimate fire intensity as represented by flame length. Fire weather parameters were used from a previous study in the area and represented 97th percentile weather conditions for the central Blue Mountains (see Table 1 in Ager et al. 2007). The analysis methods parallel those used on many national forests to identify high fire hazard areas for fuel reduction activities. The simulations were performed at 90 x 90 m resolution and the

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4 166 resulting grid of flame lengths was overlaid with the stand map to calculate total potential fire
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6 167 intensity as represented by the sum of the flame length for all pixels in each stand (Fig. 3b).
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9 10 168 2.2.2 *Wildfire exposure to the wildland urban interface* 11

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13 169 We measured wildfire risk transmission to the wildland urban interface (WUI) adjacent to
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15 170 national forests using the methods of Ager et al. (2014). This approach used the SILVIS WUI
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17 171 data (Radeloff et al. 2005, SILVIS Lab 2012) that provide spatially explicit housing and
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20 172 population densities for the coterminous US based on US census blocks (Appendix S1). The
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22 173 number of housing units within each census block was derived from the 2010 US Census data
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25 174 and included seasonal residences. We modified the SILVIS WUI data by removing polygons
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27 175 that were 1) classified as uninhabited, 2) classified as water, 3) <0.1 ha in size, or 4) >10 km
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30 176 from the national forest boundary. We maintained polygons with low housing unit densities to
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32 177 align with fire suppression efforts that target even individual structures in wildland areas. Each
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35 178 polygon retained SILVIS housing unit (hereafter structure) values. There were a total of 52,202
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37 179 WUI polygons covering an area of over 1.6 million ha.
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39 180 We then used wildfire simulation outputs generated from the FSim model (Finney et al.
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42 181 2011) to quantify area of WUI burned by ignitions located on adjacent national forests. Detailed
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44 182 simulation methods can be found in Finney et al. (2011) and Appendix S2. FSim produces both
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47 183 polygon-based fire perimeters and ignition points for each simulated fire. Ignitions were filtered
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49 184 for those occurring within national forest boundaries and associated perimeters were intersected
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52 185 with WUI boundaries to determine WUI area burned annually by each ignition. Total WUI area
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54 186 burned per polygon was calculated by summing the contributions from all ignitions reaching that
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57 187 polygon. We then estimated structures affected by each national forest-ignited fire (each
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59 188 ignition) as the product of housing units in the polygon and the proportion of the polygon
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9 191 resulting raster was resampled to 10 m and both the mean WUI risk (structure loss) (Fig. 4) and
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12 192 stand-level structure loss (average WUI risk multiplied by the stand area) were attributed to each
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14 193 stand polygon.

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20 195 A national-scale map of vegetation departure was created by the LANDFIRE (2013)
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22 196 program, a US national scale forest and fuels conditions mapping project (Appendix S1). The
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25 197 departure layer (VDEP) describes departure between simulated current vegetation and historical
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27 198 vegetation conditions. Historical “reference” time periods are defined as prior to European-
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30 199 American settlement and vary across the US but are prior to 1850 in the Pacific Northwest
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32 200 (Barrett et al. 2010). Historical conditions were derived from the LANDSUM landscape
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35 201 succession and disturbance dynamics simulation model (Keane et al. 2006). The vegetation
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37 202 departure (VDEP) scores range from 0 to 100, the latter value indicating the maximum departure
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40 203 from historic conditions. Vegetation departure (hereafter forest departure) can stem from both
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44 205 inland Pacific Northwest are generally indicative of stand densification and changes in species
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47 206 composition towards fire intolerant species. VDEP replaces the earlier fire regime-condition
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49 207 class score, and both systems are widely used to prioritize stands and landscapes for treatments
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52 208 as specified in the National Fire Plan (USDA-USDI 2001). The 30 m resolution data were
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54 209 averaged for each polygon (Fig. 3d) and averaged values were used to calculate a stand-level
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57 210 metric which represented the total number of hectares in a departed condition.
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59 211 2.2.4 *Harvest volume from restoration thinning treatments* 60

Forest Inventory Data (FIA) and Current Vegetation Survey (CVS) data were derived from the LEMMA (Ohmann and Gregory 2002) project and consisted of gradient nearest neighbor (GNN) imputed inventory plot data of stand structure and tree species lists (with tree height and dbh) for each 30 x 30 m pixel in the study area (Appendix S1). The GNN grid of inventory plots was intersected with the stand polygon layer and the population of 30 m pixels that represented each stand was identified. We then simulated a restoration thinning in each stand using the Blue Mountains variant of the Forest Vegetation Simulator (FVS, Dixon 2002). Thinning prescriptions were adopted from operational practices by local Forest silviculturists developed in detail in previous studies (Ager et al. 2007). Stands where stand density index (SDI, Cochran et al. 1994) exceeded 65% of the maximum were thinned with tree removal ordered from smallest to largest, thus reducing ladder fuels that contribute to crown fire. Stands were thinned to 35% of the maximum SDI for the stand. Thinning prescriptions targeted removal of late-seral, fire-intolerant species (grand fir) in mixed-species stands, favoring early seral species such as ponderosa pine, western larch (*Larix occidentalis* Nutt.) and Douglas-fir. For each stand we averaged total thin volume reported by FVS (cubic feet per acre) and converted to $\text{m}^3 \text{ha}^{-1}$ (Fig. 3a) and then estimated a total merchantable volume from the thinning treatment by multiplying by the stand area.

2.2.5 *Insect and disease risk*

We used spatial data from the National Insect and Disease Risk Map (FHTET 2012) that incorporates 186 individual risk models to estimate basal area loss to major insects and diseases over a 15-year future period (Appendix S1). The process uses host tree species maps, and ancillary data such as climate, topography, soils, pest occurrence, etc., to model risk of mortality for individual pest agents. Native data are generated nationally at 240 m resolution. We

averaged total basal area loss grid values for each polygon to derive average estimated basal area mortality from insects and disease (Fig. 3c) as well as stand level total basal area which was the product of the average basal area mortality and stand area.

2.3 *Spatial optimization model*

We used the Landscape Treatment Designer (LTD, Ager et al. 2012, Ager et al. 2013) to model hypothetical restoration scenarios and identify tradeoffs among different restoration objectives. LTD has some similarity to the widely used Marxan with Zones (Watts et al. 2009), although the program has specific and unique features to solve the problem of treating landscapes to design forest restoration projects (versus designing conservation reserves) and is substantially less complex in terms of creating input files and processing outputs, thus enabling field application by planning specialists. LTD uses a stand polygon coverage attributed with landscape conditions relative to restoration objectives. The user supplies a restoration scenario in terms of objectives, activity constraints, and stand treatment thresholds, and the program identifies one or more project areas within the landscape and treatment areas that maximize the objective:

$$\text{Max } \sum_{j=1}^k (Z_j * \sum (W_i N_{ij})) \quad (1)$$

Subject to

$$\sum_{j=1}^k (Z_j A_j) \leq C \quad (2)$$

where C is a global constraint on investment level per project area (e.g., area treated), Z is a vector of binary variables indicating whether the j th stand is treated (e.g., $Z_j = 1$ for treated stands and 0 for untreated stands), N_{ij} is the contribution to objective i in stand j if treated, and A is the area of the j th treated stand. W_i is a weighting coefficient that can be used to emphasize one objective versus another. The LTD algorithm uses a relatively simple search heuristic (Fig. S2)

that tests each polygon as a seed to build a restoration project in the surrounding landscape, absorbing adjacent stands based on their potential contribution to the objective value, and treating those that exceed the treatment threshold. Stands within the project area that do not exceed the treatment threshold are absorbed into the project without contributing to the objective or activity (treated area) constraint. Thus simulated project areas resemble the composition of actual ones in terms of the mosaic of treated and untreated stands. Eventually the treatment area constraint (Eq. 2) is met and the objective value is recorded for the project area. The process is repeated until all polygons have been tested as a seed and the project area that maximized the objective is reported, along with objective values and details describing the stands that required treatment. The program also generates bitmap image files showing project areas and treatments, and shapefiles attributed with objective values.

We modeled a hypothetical five year restoration program based on historical management activities in the study area. The scenario called for 20 sequentially optimized projects, each treating 5000 ha to address one or more of the restoration objectives described above. Overstocked stands were assumed to generate wood volume from thinning treatments as described above, and surface fuels reduction (mastication and underburning) were also assumed to be part of treatments where appropriate, although these activities were not explicitly modeled. We assumed that each of the restoration objectives (e.g., insects and disease, departure, wildfire) would be addressed by these thinning and other treatments, consistent with operational planning of project areas and proposed treatments throughout the western US national forests.

We then examined tradeoffs between selected combinations of different objectives by changing the relative weights of each objective (Eq. 1). These comparisons focused on change in harvest volume (i.e., provisional ecosystem service) resulting from refocusing treatments to

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4 281 areas that required treatment to meet other objectives related to stressor reduction and fire
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6 282 protection. Integer weights were varied in all combinations from 0 to 4 in increments of 1 in a
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9 283 pairwise fashion. Outputs were used to generate production possibility frontier relationships
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11 284 between harvest volume and each objective. For instance, weights of 1 and 0 for objective A and
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13 285 B respectively, represent maximum production for objective A, whereas weights of 2 and 2 for
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16 286 each objective represent a mixed production for both.

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19 287 The algorithm to spatially aggregate polygons into project areas evaluated adjacent polygons
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21 288 for inclusion in the project area based on average per area (ha) condition of the polygon. We
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24 289 used average instead of total polygon value to remove potential bias from polygon size.
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26 290 However, overall optimality of the project was based on the total objective value calculated as
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29 291 the area weighted quantity (e.g., total thinning volume) to account for the differential
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31 292 contribution to the objective from polygons of different size. Thus, polygons were added to
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34 293 project area based on the mean value of the objective for that polygon, and when the area
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36 294 treatment constraint was met (5000 ha) the total objective value was summed for each polygon in
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39 295 the project. To standardize the reporting of the different metrics we calculated percentage
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41 296 contribution of each polygon to the study area and summed these values for each project, thus
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43 297 providing a metric that could be easily interpreted among the various objectives.

44 45 46 298 *2.4 Comparison of modeled with actual restoration projects*

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49 299 We compared optimal project areas derived from modeling with 14 restoration projects
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51 300 (Table S1) either implemented or near implementation since 2012. We overlaid treatment
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54 301 polygons for each of these projects on our restoration objective layers and calculated the
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56 302 attainment towards the goals using the same methods as in the simulation projects. Since actual
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59 303 project areas varied in size (1317 – 8894 ha), it was necessary to re-scale the attainment values to

match the 5000 ha treatment areas in the modeled projects. We then plotted the resulting values on the PPF to examine the optimality of actual restoration projects.

3. Results

Potential restoration progress for a sequence of 20 spatially optimized projects (five year restoration program) varied substantially among the five objectives analyzed (Fig. 5). Optimized projects were located on areas that had 3-31% of the total restoration need within the study area (mean = 11%). The different levels of attainment among the five restoration scenarios were the result of variable spatial grain in landscape conditions with respect to restoration objectives. In general, optimizing projects for a single restoration objective was associated with low attainment for the other objectives (reduction of 10-30%), with the exception of forest departure, where attainment of the non-optimized variables often increased 1-2% (Table 1, Fig. 5). The decline in attainment over the prioritized sequence of 20 projects also varied among restoration objectives, with WUI protection and harvest volume showing sharp declines between the most optimal (1st of 20) and lower priority projects (Fig. 5a,e). Forest departure and insect and disease risk exhibited relatively small differences among the sequence of optimized projects (Fig. 5c,d). Thus the value of spatial optimization and project prioritization for forest restoration programs varied among restoration objectives.

The spatial distribution of 20 optimal projects for each scenario (Fig. S2) was dispersed among all four national forests in the study area. Optimal projects for treating WUI risk transmission were scattered widely, reflecting the distribution of communities and surrounding urban interface (Fig. 6a) while optimal projects for forest departure tended to be clustered in the central portion of the study area (Fig. 6b). Among all restoration objectives priority projects did

not overlap spatially (Fig. S2). However there was an overlap of 2779 ha when the WUI protection scenario was not considered, or 0.2% of the treatable area. Insect risk, harvest volume and fire hazard had the most project overlap with 10,808 ha (0.7% of the treatable area).

Total harvest volume across all 20 projects was over 2.4 million m³ for the thinning scenario, and a low of 350,874 m³ when WUI risk was prioritized. Potential economic viability of the projects was assessed in more detail by calculating percentage area within each project that exceeded a threshold of 35 m³ ha⁻¹ (500 ft³ ac⁻¹). Harvest volumes at or above this level are generally considered to have positive economic benefits within the study area. In terms of thinning and harvest volume over five restoration scenarios, area of stands within a single 5000 ha project that exceeded a thin threshold of 35 m³ ha⁻¹ (500 ft³ ac⁻¹) was a high of 98% for the scenario where harvest volume was optimized, and a low of 0% for the WUI risk transmission scenario. For the WUI risk scenario, over half of the projects could not meet a 35 m³ ha⁻¹ harvest volume threshold and in the remaining projects less than 1% of the project area met the threshold (Fig. 7 and 8). The distribution of harvest volume within projects changed dramatically between the first and 20th project for the harvest volume scenario (Fig. 8).

Production possibility frontiers between harvest volume (Fig. 9) and the other four restoration metrics were convex for the first project and the mean for all projects, and roughly concave for the 20th project. While restoration goals were coincident with harvest volume outputs for response metrics such as forest departure and fire hazard, sharp tradeoffs were observed for WUI risk transmission (Fig. 9d). Moreover tradeoffs became sharper as more projects were implemented on the landscape. For instance the opportunity cost of increasing harvest volume in the highest priority project was generally lower than for the 20th project for all restoration objectives. Although the tradeoffs appear relatively small on a percentage basis, the

absolute values in terms of restoration attainment are substantial, given that each scenario treats over 5% of the study area that is available for active restoration.

The different potential project areas that formed the PPF were located in substantially different locations within the study area. For example, optimizing fire hazard versus harvest volume (Fig. 10) resulted in projects located in several locations including the southeastern portion of the Wallowa-Whitman (optimal fire hazard, Fig. 10, dark red project areas), northeastern corner of the Umatilla (optimal harvest volume, Fig. 10, dark blue project areas; and a mix of both objectives, orange project areas).

Comparison of optimized projects with a sample of 14 active projects from the four national forests showed that in general planned projects were located well inside of the PPF for the first and 20th projects, but in three cases were above the mean value (Fig. 9c and d). Moreover we did not observe projects at either extreme of the different PPFs, with most projects being relatively balanced with respect to the attainment of one objective versus the other. The attainment of restoration goals was lowest for WUI risk among the metrics analyzed.

4. Discussion

To our knowledge this work is the first to apply spatial optimization to quantify socioecological tradeoffs and production frontiers for a large-scale terrestrial restoration program. The model is relatively simple to use compared to other exact (e.g., integer programming) optimization systems (Appendix S3) and is now being applied as part of the restoration efforts in several regions in the western US to prioritize planning areas and analyze tradeoffs. In other recent work, we examined priorities and tradeoffs for a portion of the study area (Wallowa-Whitman NF) using similar restoration metrics and a scenario where planning

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4 371 areas were defined by watershed boundaries on the national forest (Vogler et al. 2015). Rather
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6 372 than using spatial optimization, restoration goals were maximized within each planning area by
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11 374 approach has several limitations, the foremost being that relatively few national forests expend
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13 375 the effort to build comprehensive planning area maps, and the use of pre-defined planning areas
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15 376 precludes the identification of spatially optimal planning areas.

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19 377 The value of prioritizing restoration projects was clearly shown for the study area for specific
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21 378 restoration metrics (Fig. 5). Our study area was representative of many other national forests in
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23 379 the western US, and thus this overall finding can likely be extrapolated beyond the study area.
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25 380 We believe that the model and methods can be utilized by the ongoing federal forest restoration
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27 381 program in many ways including: 1) prioritization of restoration projects at multiple scales on
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29 382 public and adjoining private lands, 2) identification and mapping of conflicts between ecological
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31 383 restoration and socioeconomic objectives, 3) measuring efficiency of ongoing restoration
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33 384 projects compared to optimal production frontiers, 4) assessment of fire transmission risk among
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35 385 public and private land parcels as a prioritization metric, and 5) finding socially acceptable
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37 386 regions along the PPF (King et al. 2015) as part of collaborative restoration planning (Schultz et
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39 387 al. 2012, Butler et al. 2015). Clear tradeoffs were observed among the various restoration goals
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41 388 and resulting in spatial “packaging” or covariance of different stressors and ecosystems services
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43 389 (Anderson et al. 2009, Allan et al. 2013). Federal forest restoration policy in the US has long
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45 390 assumed a strong spatial correlation among the various goals (USDA Forest Service 2006).
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47 391 However, linkages between forest departure, provisional ecosystem services, and community
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49 392 wildfire protection issues are weakened by past wildfires, harvesting activities, variation in
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which restoration projects are planned and executed on US national forests (e.g., 5000 – 25,000 ha). Specific ecological tradeoffs noted in the results stem from the diversity of existing forest structure and species composition within the study area. For instance, high insect-caused mortality is both observed and predicted in dense, mature lodgepole pine forests that grow in mid- to high-elevation cold forests. Conversely, many WUIs are located in lower elevation forest-grassland transition zones that have high forest departure from fire exclusion, and low potential thin volume due to historical harvesting activities. By contrast, mixed conifer forests in moist settings have high productivity and departure from fire exclusion and relatively high potential thin volumes.

Our comparison of actual forest restoration projects with production frontiers provided the first attempt to analyze the efficiency of widespread restoration and fuel management activities on federal lands in the US (Fig. 9). Evaluating restoration progress is generally recognized as a difficult problem and needs to consider both ecological and socioeconomic outcomes (Wortley et al. 2013). We demonstrated how production frontiers combined with empirical data on ongoing projects can be used to establish targets and monitoring benchmarks that are needed for restoration programs (Wortley et al. 2013). Our analyses suggested that recently implemented restoration projects were generally suboptimal (Fig. 9), which is not surprising since tools have heretofore not been available to identify landscape production frontiers for restoration projects.

Unlike many other terrestrial restoration programs, forest restoration has the potential to generate provisional ecosystem services (revenue from harvested wood) that can partially fund the program (Deal et al. 2012, USDA Forest Service 2012). However, both restoration and fire protection projects do not typically generate high-value materials (Rainville et al. 2008). The juxtaposition of ecological settings with human values generates sharp tradeoffs between

community wildfire protection and these provisional ecosystem services. These tradeoffs complicate prioritization of restoration programs for economic objectives, which aim to generate revenue to both support expanded restoration activities in areas that will not generate positive revenues, and improve community resilience in rural areas (USDA-USDI 2014). Our study specifically showed a sharp decline in the production of wood material with decreasing project priority, and the 4th project out of the top 20 had little commercial wood material (Fig. S3).

The application of spatial optimization for prioritizing restoration of fire-prone forests offers several important advances over existing methods used in US federal land management agencies. Current policy dictates a prioritization process based on a stand-scale fire regime-condition class rating that measures fire exclusion on a stand by stand basis (USDA-USDI 2001). The evaluation process ignores fuel contagion that can catalyze large fires on national forests and transmit risk to the WUI. Moreover, consideration of economic and social viability of projects is left to an ad hoc process of pondering large numbers of maps in collaborative settings without knowledge concerning tradeoffs and optimal project design. Understanding tradeoffs with spatial optimization and production possibility frontiers is arguably a key precursor to the development and prioritization of restoration and conservation plans (Allan et al. 2013). Studies on tradeoffs and spatial prioritization in conservation biology and natural resource literature cover a diverse range of ecosystems and associated services (Maron and Cockfield 2008, Chhatre and Agrawal 2009, Hauer et al. 2010, White et al. 2012, Schroter et al. 2014, Cattarino et al. 2015). For instance Allan et al. (2013) examined joint analysis of stressors and ecosystem services on restoration effectiveness and provisioning services in the Great Lakes region of the US, focusing on a wide range of stressors and their spatial distributions. Schroter et al. (2014) examined the problem of tradeoffs between creating forest protected areas and timber

production, and calculated the opportunity cost to inform conservation policy development. In this study the focus was on restoring ecological processes on a landscape rather than conservation of specific biodiversity values as in many previous studies discussed above.

It is well recognized that social barriers pose a substantial challenge to restoration programs on federal forests (Franklin et al. 2014). The enactment of the Collaborative Forest Landscape Program (USDA Forest Service 2016) provides for public participation in restoration planning to facilitate trust and conflict resolution among diverse stakeholders that derive ecosystem services from federal forests (Butler et al. 2015). However, without science-based decision support tools to prioritize and quantify tradeoffs, collaborative planning groups are missing strategic information about the opportunity cost associated with their particular values and interests in terms of long range progress towards socioecological restoration goals.

Modeling forest restoration programs poses many challenges and a number of assumptions concerning the impact of treatment on stressors were implicit in the study. For instance, we assumed that stands selected for active restoration would receive the appropriate suite of treatments including mechanical thinning, fuels mastication, and underburning, thereby addressing the particular issue at hand. We also assumed that treatments substantially reduced forest departure, insect and disease issues, and wildfire risk to the WUI. These assumptions are not inconsistent with current operational planning on the national forests and literature (Brown et al. 2004). Our modeled projects treated between 30% and 40% of the area within each project, compared to 35% on the four national forests in the study area, and thus result in significant landscape-scale change in forest resiliency to forest disturbances.

Landscape decision support tools to prioritize restoration management in the western US will likely play an increasingly important role in the development of restoration programs and

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4 463 contribute to the process of creating resilient forests while meeting concomitant demands for
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6 464 biodiversity, amenity values, and socioeconomic outputs (Noss et al. 2009). Optimization
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9 465 models can facilitate attainment of these goals by prioritizing management activities, identifying
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11 466 opportunities and conflicts, and identifying investment tradeoffs (Christensen and Walters 2004).
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14 467 Future work on production possibilities for restoration should also account for long-term
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16 468 landscape policy and disturbance feedbacks (fire, insects and disease) and their effects on
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19 469 tradeoffs over time (Spies et al. 2014, Rappaport et al. 2015). Understanding social constraints
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21 470 as part of restoration programs (Franklin et al. 2014) will also be an important step to improve
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24 471 collaborative restoration planning, and potentially can be accomplished by combining social
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26 472 preferences with PPFs as advocated in other studies (see Fig. 1 in King et al. 2015). Future work
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29 473 can help link prioritization across the multiple scales of restoration planning, from national
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31 474 forests to landscapes, stands, and tree neighborhoods (Larson and Churchill 2012). As in other
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33 475 restoration systems (Wilson et al. 2011, Rappaport et al. 2015), quantitative prioritization and
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36 476 tradeoff analyses are an important step in US forest restoration programs to achieve long-term
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38 477 goals of creating fire resilient landscapes and fire adapted communities, as well as sustaining the
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41 478 myriad of ecosystem services from public lands.

42 43 44 479 **Acknowledgments**

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Table 1. Restoration attainment values for five restoration objectives under a scenario where each variable is optimized within 20 projects with 5000 ha treated per project. Values represent percentage of total potential restoration attainment for the study area if all stands were treated.

Restoration outputs (% of total landscape)					
Restoration priority	Harvest volume	Forest departure	Fire hazard	Insect/disease risk	WUI risk transmission
Harvest volume	23.7	7.1	13.2	12.1	3.7
Forest departure	9.4	8.9	11.1	10.3	2.4
Fire hazard	10.5	6.4	26.7	9.7	11.1
Insect/disease risk	15.6	7.2	18.0	16.7	3.3
WUI risk transmission	3.4	5.9	10.0	4.1	30.6

Figure Captions

Fig. 1. Example restoration treatments within the Blue Mountain National Forests in eastern Oregon, US. (a) Pre-treatment stand of ponderosa pine and western larch with advanced regeneration and densification from fire exclusion; (b) same stand as in (a) after receiving thinning treatment to reduce stocking density and select fire resilient species; (c) stand that has received mechanical thinning treatments five years prior is targeted for a maintenance underburning treatment to reduce surface fuels, mimicking natural fire; (d) same stand as in (c) after underburning treatment.

Fig. 2. Map of the Blue Mountains National Forests in eastern Oregon and southeastern Washington, USA.

Fig. 3. Spatial distribution of restoration targets in the Blue Mountains National Forests of eastern Oregon and southeastern Washington attributed to stands. (a) Total live stemwood harvest volume generated from thinning treatments; (b) fire hazard as measured by average flame length from FlamMap (Finney 2006) wildfire simulations; (c) insect and disease risk as measured by expected forest stand basal area mortality (FHTET 2012); (d) forest departure based on vegetation departure from LANDFIRE (2013). See methods section for description and derivation of data sources and Appendix S1 for data source details.

Fig. 4. Map of the Blue Mountains National Forests in eastern Oregon and southeastern Washington showing areas that have potential to transmit wildfire to the wildland urban interface (WUI) and impact structures based on FSim simulation outputs (Finney et al. 2011) and SILVIS WUI data (Radeloff et al. 2005). See section 2.2.2 and Appendices S1 and S2 for additional details.

Fig. 5. Attainment of restoration objectives over a sequence of 20 projects on the Blue Mountains National Forests. Each panel shows outputs when one of the five restoration goals is optimized. (a) Total live stemwood harvest volume generated from thinning; (b) fire hazard as measured flame length from FlamMap (Finney 2006); (c) insect and disease risk as measured by expected basal area mortality (FHTET 2012); (d) forest departure based on LANDFIRE (2013); (e) wildfire risk transmission to the wildland urban interface estimated from simulation (Finney et al. 2011).

Fig. 6. Optimal restoration project locations for a 5-year restoration scenario on the Blue Mountains National Forests prioritizing (a) reduction in wildland urban interface wildfire risk transmission based on FSim simulation outputs (Finney et al. 2011) and (b) forest departure based on vegetation departure from LANDFIRE (2013). See section 2.3 for details regarding calculations. Each project area treated 5000 ha. Project area boundaries were enlarged in mapping to make them more visible at this scale.

Fig. 7. Percent of total project area (5000 ha) with a harvest volume yield of $\geq 35 \text{ m}^3 \text{ ha}^{-1}$ ($500 \text{ ft}^3 \text{ ac}^{-1}$) over a sequence of 20 projects on the Blue Mountains National Forests.

Fig. 8. Total treatable area in the Blue Mountains National Forests by harvest volume classes for the first (Project 1) and last (Project 20) priority project for optimizing harvest volume. Dotted vertical line indicates a reference harvest volume of $35 \text{ m}^3 \text{ ha}^{-1}$ ($500 \text{ ft}^3 \text{ ac}^{-1}$).

Fig. 9. Production possibility frontiers between selected restoration objectives for a sequence of 20 restoration projects. The first (highest priority), 20th (lowest priority), and mean attainment for all projects are shown. Axes show attainment as a percentage of the total possible in the

study area. Attainment of actual active projects planned by the Forest Service within the study area, as described in Table S1, are shown in red.

Fig. 10. Production possibility frontier for fire hazard and harvest volume, and associated project area location for the northeastern Wallowa-Whitman and Umatilla national forests. See Fig. 2.

Figure
Figure 1

Figure 2

Figure 3

(a) Harvest volume
($\text{m}^3 \text{ha}^{-1}$)

(b) Fire hazard (m)

(c) Insect/disease risk
($\text{m}^2 \text{ha}^{-1}$)

(d) Forest departure (%)

Figure 4

Figure 5

Figure 6

(a) WUI risk transmission

(b) Forest departure

Figure 7

Figure 8

Figure 9

Figure 10