Ecological Economics 136 (2017) 226–239

Contents lists available at ScienceDirect

Ecological Economics

journal homepage: www.elsevier.com/locate/ecolecon

Economic Opportunities and Trade-Offs in Collaborative Forest Landscape Restoration

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ARTICLE INFO

Article history:
Received 15 August 2016
Received in revised form 23 November 2016
Accepted 9 January 2017
Available online xxxx

Keywords:
Forest Restoration
Spatial Optimization
Restoration Prioritization
Production Possibility Frontiers
Collaborative Planning

ABSTRACT

We modeled forest restoration scenarios to examine socioeconomic and ecological trade-offs associated with alternative prioritization scenarios. The study examined four US national forests designated as priorities for investments to restore fire resiliency and generate economic opportunities to support local industry. We were particularly interested in economic trade-offs that would result from prioritization of management activities to address forest departure and wildfire risk to the adjacent urban interface. The results showed strong trade-offs and scale effects on production possibility frontiers, and substantial variation among planning areas and national forests. The results pointed to spatially explicit priorities and opportunities to achieve restoration goals within the study area. However, optimizing revenue to help finance restoration projects led to a sharp reduction in the attainment of other socioecological objectives, especially reducing forest departure from historical conditions. The analytical framework and results can inform ongoing collaborative restoration planning to help stakeholders understand the opportunity cost of specific restoration objectives. This work represents one of the first spatially explicit, economic trade-off analyses of national forest restoration programs, and reveals the relative cost of different restoration strategies, as well scale-related changes in production frontiers associated with restoration investments.

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1. Introduction

Restoration ecology has increasingly become a key component of land management programs on both public and private lands in many regions of the world (Adame et al., 2015; BenDor et al., 2015; Wilson et al., 2012). A case in point are the large scale forest restoration programs initiated on western US national forests under the Healthy Forest Restoration Act (HFRA, 2003) to improve the health and fire resiliency of dry forest ecosystems (Noss et al., 2006; USDA Forest Service, 2012). The programs encompass a multitude of ecosystems and services with focal points on resiliency of landscapes to fire, watershed condition, invasive species, and wildlife habitat. Fire resiliency objectives are achieved through fuel management projects that use forest thinning, prescribed fire, and a range of other techniques aimed at returning fire frequent forests to pre-settlement conditions (Agee and Skinner, 2005; Brown et al., 2004). The HFRA was broadened with the Omnibus Public Land Management Act of 2009 (Title IV) which established the Collaborative Forest Landscape Restoration Program (CFLRP, USDA Forest Service, 2016b) to encourage science-based planning and promote diverse restoration approaches to meet broad ecological, economic, and resource protection objectives (Butler et al., 2015). Key outputs from the restoration program include commercial wood supply to private entities to offset restoration treatment costs and employment opportunities in rural economies (USDA Forest Service, 2016b). The science dialog around the program has been extensive, and includes discussions of ecological goals (Brown et al., 2004; Haugo et al., 2015; Moore et al., 1999; Noss et al., 2006), planning frameworks (Butler et al., 2015; Franklin and Johnson, 2012; Schultz et al., 2012; USDA Forest Service, 2016b), implementation strategies (Rieman et al., 2010), economic assessments (Rasmussen et al., 2012; Rummer, 2008), and human dimensions (Franklin et al., 2014; Payne, 2013). A recent five-year review of the CFLRP (USDA Forest Service, 2015) and a national conference of managers and stakeholders highlighted local implementation of the program and results from specific restoration projects.

Ongoing implementation of the restoration programs and inclusion of diverse stakeholder groups in the planning process has challenged federal land managers to better articulate priorities and desired outcomes from the program (Butler et al., 2015). Under the current process, local forest managers in concert with stakeholder groups attempt to blend local values with broad regional assessments of restoration

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http://dx.doi.org/10.1016/j.ecolecon.2017.01.001
0921-8009/Published by Elsevier B.V.

needs under national policy direction. The analytical component of current collaborative planning efforts largely consists of ad hoc analysis of spatial data from regional and local assessments coupled with field observations to determine site specific projects and planning areas (Butler et al., 2015). Guidelines and analytical protocols to prioritize restoration planning areas based on singular or multiple goals (Neeson et al., 2016), including economics (Adame et al., 2015; Kimball et al., 2015), are nonexistent. Nor are analyses conducted to evaluate trade-offs among economic aspects and the reduction of stressors (Allan et al., 2013; Bullock et al., 2011; Maron and Cockfield, 2008) that potentially adversely impact forest health and resiliency. The trade-offs in restoration activities stem from finite budgets, operational capacity, and spatial variation and covariation across different restoration targets (Anderson et al., 2009; Martin et al., 2016; Neeson et al., 2016). The net result is that stakeholders participating in collaborative restoration planning are not fully informed about the opportunity cost of emphasizing one restoration objective over another. Moreover, trade-offs are not considered in strategic assessments of restoration need because they either generally have a singular objective (Haugo et al., 2015; Rasmussen et al., 2012; USDA Forest Service, 2011) or the coarse scale of assessment inputs precludes analysis at the project implementation scale (Barbour et al., 2008b; Rasmussen et al., 2012). Thus spatial priorities and targets established by regional assessments to address specific socioeconomic and ecological issues, including wood supply (Barbour et al., 2008a), fire protection to the wildland urban interface (WUI, Bailey, 2013), and ecological departure from historical conditions (Haugo et al., 2015) ignore trade-offs, and may well suggest unobtainable or non-optimal outcomes. Scale effects on production functions (King et al., 2015) and scale mismatches (Cumming et al., 2006) between assessments and project implementation can also contribute to a decoupling of restoration policy goals with actual implementation in the field. Clearly, integrating economic and ecological trade-off analyses could provide manifold improvements to the current planning efforts, especially with respect to the primary goals of sustaining rural economies and meeting fire resiliency objectives in fire prone, forested areas. For instance, economic analyses can pinpoint locations where treatments can generate revenue that in turn can be used to subsidize non-economic fuels mastication and thinning treatments elsewhere within planning areas, thereby maximizing the total area restored for a given level of financial investment.

In this paper we describe a detailed analysis of economic and ecological trade-offs within four US national forests (NF) designated as a national priority for restoration (USDA Forest Service, 2016a). We first examined how generating revenue from restoration affected opportunities to address social and ecological goals within 102 individual planning areas. We then examined cumulative net revenue realized from specific restoration targets over increasing scales of implementation. Of specific interest was the idea that maximizing revenue could help facilitate building large fire resilient landscapes by subsidizing treatment of forest stands that cannot produce economic benefits, but require fuels treatment for fire resiliency objectives. We use the study to stimulate a discussion about ways to improve stakeholder engagement in the prioritization of restoration projects as part of collaborative planning (Butler et al., 2015) via the use of production frontiers (Cavender-Bares et al., 2015; King et al., 2015) (Fig. 1).

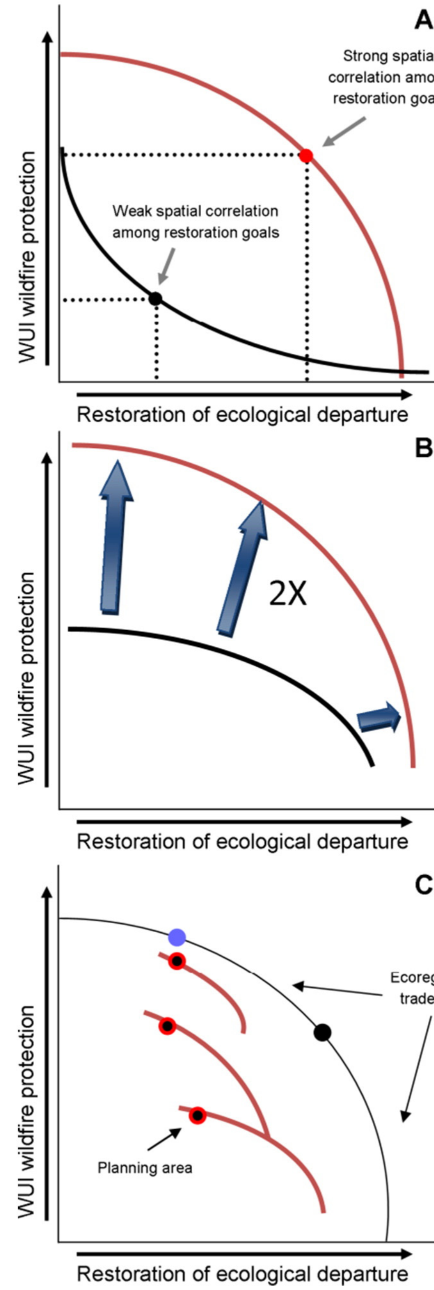

Fig. 1. Example production possibility frontiers (PPFs) for US federal forest restoration programs. A) PPFs showing convex to the origin (black line) versus concave to the origin (red line) production relationships between two restoration goals achieved through forest management activities. Strong spatial correlation among different restoration treatment goals makes it possible for joint, optimal attainment (red dot). When spatial correlation of restoration targets is weak, joint attainment (black dot) results in significantly less progress towards multiple restoration goals, and sharp declines in the potential treatment of each goal individually. B) Possible change in PPF from additional investments in restoration where the production frontier becomes asymmetrical due to the scarcity of stands requiring treatment for one of the objectives (ecological departure) but not the other; C) example where local collaborative groups select projects for implementation for planning areas within individual national forests (red circles) but local preferences result in suboptimal production at the ecoregional scale (blue circle) or are not preferred by policymakers (versus optimal production, black circle). WUI = wildland urban interface. Panel C adapted from King et al. (2015). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

2. Methods

2.1. Study Area

The study area encompassed four national forests (Malheur, Ochoco, Umatilla and Wallowa-Whitman) in the Blue Mountain ecoregion (USDA Forest Service, 1994) of eastern Oregon and southeastern Washington and includes 2.5 million ha of forest and rangelands (Fig. 2). The area is interspersed with small mountain ranges, canyons, and plateaus. Elevations generally range from 900 m to 1500 m, with higher peaks close to 3000 m. Dry forests of ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) dominate lower elevations, with dry mixed conifer (grand fir (*Abies grandis* (Douglas ex D. Don) Lindl) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco)) at higher elevations. Cold dry

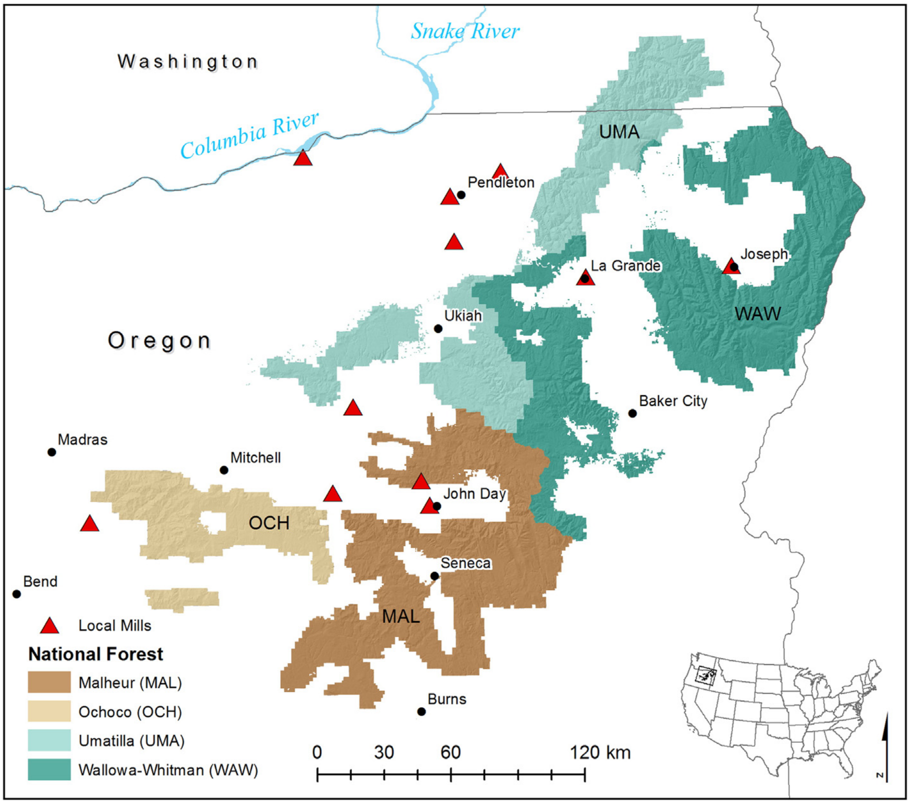

Fig. 2. Map of the study area showing the four national forests in the Blue Mountains ecoregion of eastern Oregon and southeastern Washington, USA, and the locations of 22 wood processing facilities used in the haul cost calculation. Wood processing facilities included mills that consume logs to generate dimensional lumber. Facilities that generated specialty wood products from saw logs or consumed chips for pulp or energy were not included. Figure adapted from Ager et al. (2016). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

forested found at mid to high elevation areas are dominated by lodgepole pine (*Pinus contorta* Douglas ex Loudon). About 50% of the study area is designated for active forest management with mechanical treatments, with the remaining areas restricted by wilderness and roadless legislation. The forests were heavily managed for timber production until the mid-1990s. A combination of factors, including ecological and amenity values, led to a severe decline in commercial harvest from about 1.85 to 0.56 million m^3 per year (USDA Forest Service, 2016c). Wildfires and insect outbreaks have impacted stand structure and composition in some areas. About 22,000 ha (0.9%) are burned annually by predominantly lightning caused wildfires (1992–2013) (Short, 2015). 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). Large areas of forests have undergone dramatic changes in stand structure and species composition since pre-settlement periods as a result of fire exclusion policies. In a recent study, Hagmann et al. (2013) reported a tripling of stand densities on adjacent national forests over the past 90 years (68 ± 28 trees ha^{-1} to 234 ± 122 trees ha^{-1}), although the proportion of large, fire resilient trees (>53 cm DBH) decreased by more than a factor of five. At the same time, Hagmann et al. (2013) reported less than a 20% increase in mean basal area, with the basal area of large trees declining by >50%, leaving large areas susceptible to future disturbance from insects and wildfire.

Estimates of restoration need suggest that 34% (506,696 ha) of the forested area could benefit from restoration treatments (USDA Forest Service, 2013). Downscaling a more recent analysis of forest structure departure for the Pacific Northwest (Haugo et al., 2015), we estimated that roughly 890,000 ha (58%) of the study area within managed forests (excluding wilderness and other conservation reserves) are in a state of structural departure (henceforth forest departure) from historical conditions due primarily to development of dense, multistory forest stands. This trend has been directly related to fire suppression practices that excluded natural fires for much of the area. Currently, forest restoration treatments (mechanical treatments and prescribed fire) are completed on about 32,000 ha annually, or about 3.6% of the area (2005–2015)(USDA Forest Service, 2016d). Specific forest 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 the severity of potential wildfire behavior. Large scale restoration projects within the study area specify treating an average of about 5000 ha within planning areas that range in size from 20,000 to 40,000 ha. Mechanical thinning thresholds (i.e., the selection of stands to treat) and particular thinning intensities follow guidelines by Cochran et al. (1994; see Section 2.2.4).

2.2. Modeled Restoration Objectives

We used restoration objectives described in Ager et al. (2016) and further outlined in Appendix A. Individual forest stands ($n = 204{,}610$) were defined using corporate USDA Forest Service spatial databases. Stand boundaries follow natural breaks in vegetation types and changes in stand structure from past management activities. Each stand was attributed with a land management designation from forest land management and resource plans. Wilderness and inventoried roadless areas were removed from consideration for restoration treatments. The resulting layer consisted of 145,395 stands ranging in size from <1 ha to 493 ha (mean = 10.6 ha), and totaled 1,542,226 ha (64% of the study area).

We then attributed each stand with the current condition relative to six primary goals of the restoration program in the Blue Mountains ecoregion. Four of the objectives pertained to the restoration of forest health and fire resiliency using stand treatments to reduce: 1) departure in forest structure from historical conditions (henceforth forest departure), 2) potential basal area mortality from insects, 3) wildfire hazard, and 4) wildfire exposure to the adjacent wildland urban interface (WUI). The remaining two objectives quantified economic objectives as measured by the production of stemwood volume from thinning treatments, and the resulting net revenue. The development of each objective is described below with more detailed descriptions in Appendix A and C. Finally, the study area was divided into 102 planning areas (average size = 15,102 ha) based on input from local planners (Fig. 3). The planning areas generally followed subwatershed boundaries and are used for project development on the forests.

2.3. Forest Departure From Historical Reference Conditions

We used spatial data from Haugo et al. (2015) that quantifies departure of current forest vegetation conditions from historical conditions (Appendix A). The methodology builds on LANDFIRE (2013b) fire regime-condition class scores which have been widely used to prioritize stands and landscapes for treatments as specified in the National Fire Plan (USDA-USDI, 2001). Ecological departure in forests (hereafter forest departure) can stem from both surplus and deficiencies of species and size structure distributions, however data we used from Haugo et al. (2015) are primarily an identifier of areas where fire exclusion has led to structural and compositional changes including densification, development of multistory structures with high levels of ladder fuels, and higher prevalence of fire intolerant species (e.g., grand-fir). The 30 m gridded data were averaged for each stand (Fig. 4A).

2.4. Insect and Disease Risk

Insect epidemics influence management objectives and restoration treatment to address forest health concerns. We used spatial data from the National Insect and Disease Risk Map (FHTET, 2014) to estimate basal area loss due to major insects and diseases over a 15 year future time period (2013–2027; Appendix A). The process incorporates 186 individual risk models using host tree species maps and ancillary data such as climate, topography, soils, and pest occurrence. Data for the assessment are generated nationally at a resolution of 240 m. We averaged grid values for basal area loss in each stand to estimate basal area mortality from insects and disease (Fig. 4B).

2.5. Wildfire Hazard

We used FlamMap (Finney, 2006), a widely used fire simulation software package, to obtain potential fire behavior for each stand assuming static weather conditions and fuel moisture (Appendix A). Surface and canopy fuels data were obtained from LANDFIRE (2013a). Fire weather parameters represented 97th percentile weather conditions for the central Blue Mountains and were derived from a previous study (see Table 1 in Ager et al., 2007). These methods are similar to those used on

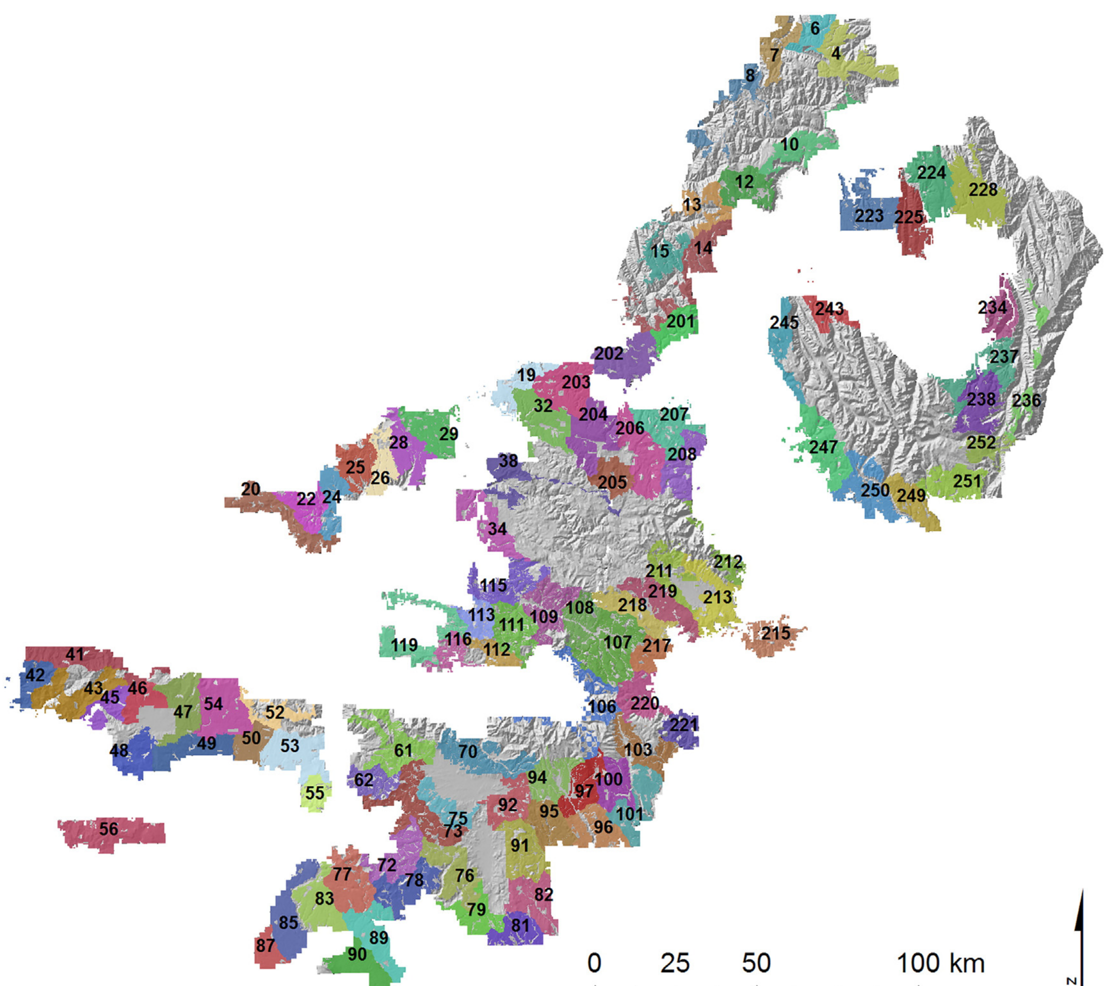

Fig. 3. Map of the 102 planning areas on the four national forests included in the study area. Planning area boundaries generally follow watershed boundaries. Small parcels created by discontinuities between national forest ownerships and watershed boundaries were merged with adjacent planning areas consistent with planning area delineation used by the national forests. Gray areas indicate wilderness and roadless areas where mechanical thinning and fuels mastication is either not allowed or not practiced under current forest planning regulations.

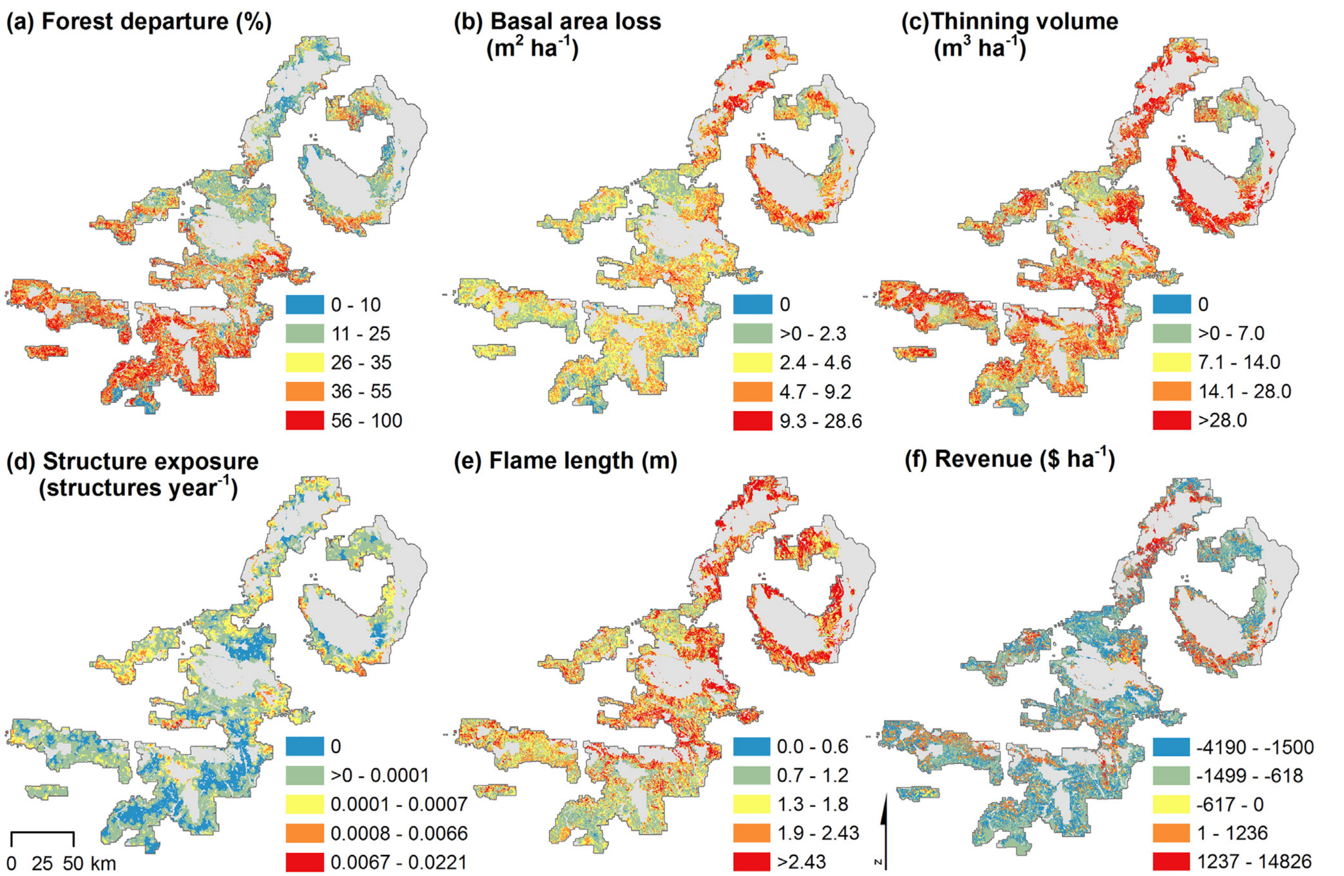

Fig. 4. Spatial distribution of restoration objectives in the study area. (a) forest departure, (b) insect risk as measured by basal area mortality, (c) merchantable timber volume generated from restoration thinning, (d) structure exposure in the wildland urban interface from national forest (NF) ignited wildfires, (e) wildfire hazard represented by flame length, and (f) net revenue from restoration treatments. See Section 2 for details on the estimation of each variable. Gray areas indicate wilderness and roadless areas where mechanical thinning and fuels mastication are either not allowed or not practiced under current forest planning regulations.

many national forests to identify high fire hazard areas for fuel reduction activities. Simulations used 90 m resolution and the resulting grid of flame lengths was overlaid with the stand map to calculate average flame length per stand (Fig. 4E).

2.6. Wildfire Exposure to the Urban Interface

We measured wildfire transmission to the WUI adjacent to national forests in the study area using methods found in Ager et al. (2014) and as described in Appendix A. We used SILVIS WUI data (Radeloff et al., 2005) for the study area, but removed SILVIS polygons that were 1) classified as uninhabited, 2) classified as water, 3) <0.1 ha in size, or 4) >10 km from the national forest boundary. There were 52,202 WUI polygons containing housing unit (hereafter structure) density data covering an area of over 1.6 million ha.

We used wildfire simulation outputs generated from FSim (Finney et al., 2011) to quantify area of WUI burned by ignitions located on adjacent national forests. Detailed simulation methods can be found in Finney et al. (2011) and Appendix B. FSim produces both polygon-based fire perimeters and ignition points for each simulated fire. Ignitions were filtered for those occurring within national forest boundaries and associated perimeters were intersected with WUI boundaries to determine WUI area burned annually by each ignition. Total WUI area burned per stand was calculated by summing contributions from all ignitions reaching that stand. We estimated annual structures affected by each national forest-ignited fire as the product of the structures within each stand and the proportion of the polygon burned. The resulting point data were smoothed using an inverse distance weighting model to generate a continuous 0.5 km raster grid using a 5 km fixed search radius for the entire study area. The resulting raster was resampled to 10 m and mean structure exposure (Fig. 4D) was attributed to each stand.

2.7. Thinning Volume

Thinning volume was estimated for each forested stand by processing forest inventory data with the Forest Vegetation Simulator (FVS, Dixon, 2002). Tree lists were obtained from the LEMMA project that employed an imputation process to extrapolate forest inventory data (FIA, Roesch and Reams, 1999) to 30 m × 30 m pixels using a gradient nearest neighbor (GNN) procedure (Ohmann and Gregory, 2002)(Appendix A). We simulated a restoration thinning in each stand using the Blue Mountains variant of FVS (Keyser and Dixon, 2015). Thinning prescriptions were adopted from operational practices by local national forest silviculturists as developed in previous studies (Ager et al., 2007; Barbour et al., 2005). Thinning prescriptions prioritized removal of smaller trees to reduce ladder fuels, and late-seral, fire-intolerant species (e.g., grand fir), while retaining fire tolerant, early seral species (e.g., ponderosa pine). Thinning thresholds were based on stand density index (SDI, Cochran et al., 1994) and density of small trees. Maximum SDI values were assumed for each plant association group as described by Cochran et al. (1994). If SDI exceeded 45% of the maximum SDI for plots in the plant association group, we simulated a thinning from below to achieve a post-thin stocking of 35% of the maximum SDI. Maximum tree harvest size was 53.3 cm as specified in local harvest guidelines (USDA and USDI, 1994). A second filter was applied to treat stands that did not meet the SDI threshold but had a high density of small trees. Specifically, if stand density was <45% of the maximum SDI and there were >741 trees per hectare (TPH), with diameters ranging from 0.25 cm to 17.78 cm at breast height (DBH), the stand was thinned to 333 TPH (approximately 5.5 m between trees) with a 53.3 cm maximum harvest size. FVS outputs were assigned to each GNN pixel and the total thinning volume per stand was calculated by summing the individual pixel values.

2.8. Financial Valuation of Restoration Treatments

We estimated potential revenue from restoration treatments using a residual value appraisal approach (Rummer, 2008). Residual value was estimated based on log values after subtracting costs associated with harvesting, hauling, and ancillary expenses. Parameters for the costs are tabulated in Appendix C and were derived from expert opinion of local timber sale planning staff. Log "pond value" of harvested timber was calculated using the economics extension in FVS by converting modeled harvest volume outputs into logs of specific size and species (Martin, 2013). Average pond values by species and size class for all dimensional timber mills within the study area (Table C1) were obtained from timber sales specialists and used to calculate total value of delivered logs from each stand. Log pond values were only calculated for stands that generated >28 m^3 ha^{-1} of merchantable timber, assuming stands producing less were not commercially viable. Material removed during restoration treatment of the latter stands would be masticated or burned on site. Harvesting costs were calculated based on slope and tree size class consistent with methods used in previous studies (Rainville et al., 2008; Rummer, 2008). A ground-based harvesting system (Table C2) was assigned for stands having a slope less than or equal to 35%, and a cable harvesting system (Table C3) was assigned for all stands that exceeded the 35% threshold. Average slope per stand was calculated from digital elevation data (30 × 30 m). Hauling distances were calculated using Euclidian distance to the nearest wood processing facility. Although Euclidean distance generated downward-biased estimates of haul distance, the approach was deemed adequate for the current study. We assumed an average transportation speed of 70 kph, an operational trucking cost of $85.00 per hour, and an average log load of 17 m^3 per truck (Mason et al., 2008). Hauling costs were calculated as:

$$\frac{\text{Distance to nearest mill (k)}}{70\text{ kph}} \times \frac{\$85.00}{\text{hr}} \times \frac{\text{Total merch.volume }(\text{m}^3)}{17\text{ m}^3\text{ per truck}} \times 2 \tag{1}$$

Note that hauling cost was only calculated for stands that generated >28 m^3 ha^{-1} of merchantable timber, and considered the round trip distance to the nearest mill.

Treated stands were assessed additional fixed costs to account for surface fuel mastication. For stands that generated >28 m^3 ha^{-1}, we assumed 40% of the stand would require additional fuel treatment at a cost of \$1112 ha^{-1}. If a stand was triggered for treatment and did not generate 28 m^3 ha^{-1}, we assumed a fixed cost of \$1112 ha^{-1} to treat residual fuel. These cost parameters were derived from local transaction data on the national forests. Our calculations ignored: 1) planning and contracting costs, 2) cost of road maintenance and construction, 3) removal of non-merchantable volume generated from thinned stands and marginally merchantable pulpwood material, and 4) underburning. These additional costs were omitted because most are budgeted outside of the restoration planning process (i.e., Forest Service employee salaries), and activities such as road construction and re-construction are not possible at the scale of the study.

To calculate residual value from treatments, harvesting, hauling, and fixed costs were subtracted from total log pond value. Average and net revenue per stand were calculated (Fig. 4F) and used as restoration objective values in the trade-off analysis along with other restoration objectives previously described. Validation included a comparison of outputs with other published and unpublished sources as tabulated in Appendix C (Tables C4–C8).

2.9. Spatial Optimization Model

We used the Landscape Treatment Designer (LTD, Ager et al., 2012; Ager et al., 2013) to prioritize restoration and identify trade-offs among different restoration objectives (Appendix B). LTD has specific design features for prioritizing and optimizing spatial fuel treatment patterns as part of restoration planning. LTD uses a shapefile of forest stand polygons attributed with restoration objectives. The user enters a restoration scenario by specifying one or more objectives (e.g., revenue), activity constraints (e.g., budget, area treated), and stand treatment thresholds (e.g., fire behavior). The program then selects stands for treatment to maximize the objective value calculated as:

$$\text{Max}\sum\nolimits_{j=1}^{k}\left(Z_j * \sum\left(W_i N_{ij}\right)\right) \tag{2}$$

Subject to:

$$\sum\nolimits_{j=1}^{k}\left(Z_j A_j\right) \le C \tag{3}$$

where C is a global constraint on activity per project area, Z is a vector of binary variables indicating whether the jth 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 jth treated stand. W_i is a weighting coefficient that can be used to emphasize one objective versus another. The LTD program can be used to solve optimization problems with or without adjacency constraints. In the current application, we used existing planning areas obtained from national forests and used the program to identify stands to treat that maximized the objective value, while meeting constraints. Given multiple planning areas, the program iterates through each one, then reports the maximum objective value and the selected treatment stands.

We prioritized each of the 102 planning areas to simulate treatment of 5000 ha in each planning area to address restoration objectives described in Section 2. Surface fuels reduction (mastication and underburning) were assumed to be part of the treatments where conditions met treatment thresholds as described in Section 2.9. It was assumed that each restoration objective (e.g., insect risk, forest departure, wildfire hazard) would be addressed by these thinning and other treatments, consistent with operational analyses of proposed restoration projects on US national forests.

Trade-offs were analyzed between selected combinations of different objectives by changing the relative weights of each objective (Eq. 2). These comparisons focused on trade-offs in revenue from different restoration objectives. Here, integer weights were varied in all combinations from 0 to 50 in increments of 10 in a pairwise fashion, excluding redundant weight combinations (e.g., weights of 10, 10 equals 20, 20, etc.). For instance, weights of 50 and 0 for objective A and B, respectively, generated the maximum production for objective A, whereas equal weights for each objective generated a balanced production for both. Outputs were used to generate production possibility frontiers between the different objectives.

We used average per-area condition of each stand in the selection algorithm to remove potential bias resulting from stand size. However, overall optimality of projects was based on the total objective value calculated as the area weighted quantity (e.g., total thinning volume) to account for differential contribution to the objective from stands of different sizes. Thus, stands were added to project areas based on mean objective values for that stand, and once the treatment constraint was met (5000 ha) the total objective value was summed for each stand in the project. To standardize reporting of different metrics we calculated the percentage contribution of attainment of each stand to the study area and summed these values for each project, thus providing a standardized metric that could be compared among different objectives.

We performed additional sensitivity analyses by analyzing cumulative attainment of restoration objectives with increasing number of project areas under different restoration objectives. We calculated net revenue under each optimization scenario as a function of the increasing scale of restoration. Finally, to explore how maximizing revenue could increase the scale of restoration activities we performed a sensitivity analysis whereby planning areas were treated until revenues from profitable stands were expended to treat unprofitable areas up to the breakeven point. In this analysis, we incrementally added stands to each planning area in order of decreasing profitability until net revenue was zero, and then examined the extent to which restoration treatments could be expanded by reinvesting revenue from profitable to unprofitable stands.

3. Results

Net revenue from restoration treatments varied among the planning areas from a high of $20.53 million to a low of $−7.38 million, with the most profitable planning areas on the Umatilla and Wallowa-Whitman NFs. High revenue areas had a slight concentration in the northern portions of the Umatilla and central Wallowa-Whitman NFs (Figs. 2, 5F). Variation among planning areas on the Ochoco NF was minimal compared to the Umatilla and Wallowa-Whitman. Similar variation among planning areas within national forests was observed for insect risk and thinning volume, but not for forest departure (Fig. 5). For all objectives, there were a number of instances where low and high priority planning areas were adjacent, illustrating the magnitude of local variation of restoration opportunity in the study area.

Production possibility frontiers (PPF) exhibited three broad trade-off relationships between revenue and other restoration objectives (Fig. 6). In some planning areas, attainment of one restoration objective was largely unaffected by the other (Fig. 6A, planning areas 4, 12 and 13). In others, the trade-offs were symmetrical and concave to the origin (convex to the outer surface; Fig. 6B, planning areas 251, 250, 247) or asymmetrical with sharp trade-offs in one direction and not the other. In many planning areas, PPFs indicated sharp decision trade-offs, where the marginal benefit from improving the attainment of one restoration objective exceeded twice the loss in the other objective. By contrast, trade-offs between revenue and insect risk (Fig. 6B) showed relatively small trade-offs within planning areas, and complementary production among planning areas. Trade-off relationships were generally similar for revenue and thinning volume (Figs. 6, 7), although economic factors such as transportation costs and processing of submerchantable material resulted in some differences in the PPFs for particular planning areas. One notable trade-off relationship concerned treating stands to reduce forest departure versus generating revenue (Fig 6A). For many of the planning areas, attempts to optimize stands to generate revenue resulted in a sharp reduction in the treatment of areas with high forest departure. Planning areas that had less overall forest departure tended to have more flexibility in terms of increasing revenue while still meeting the same level of attainment for treating forest departure (e.g. planning areas 14, 13, 12, 4).

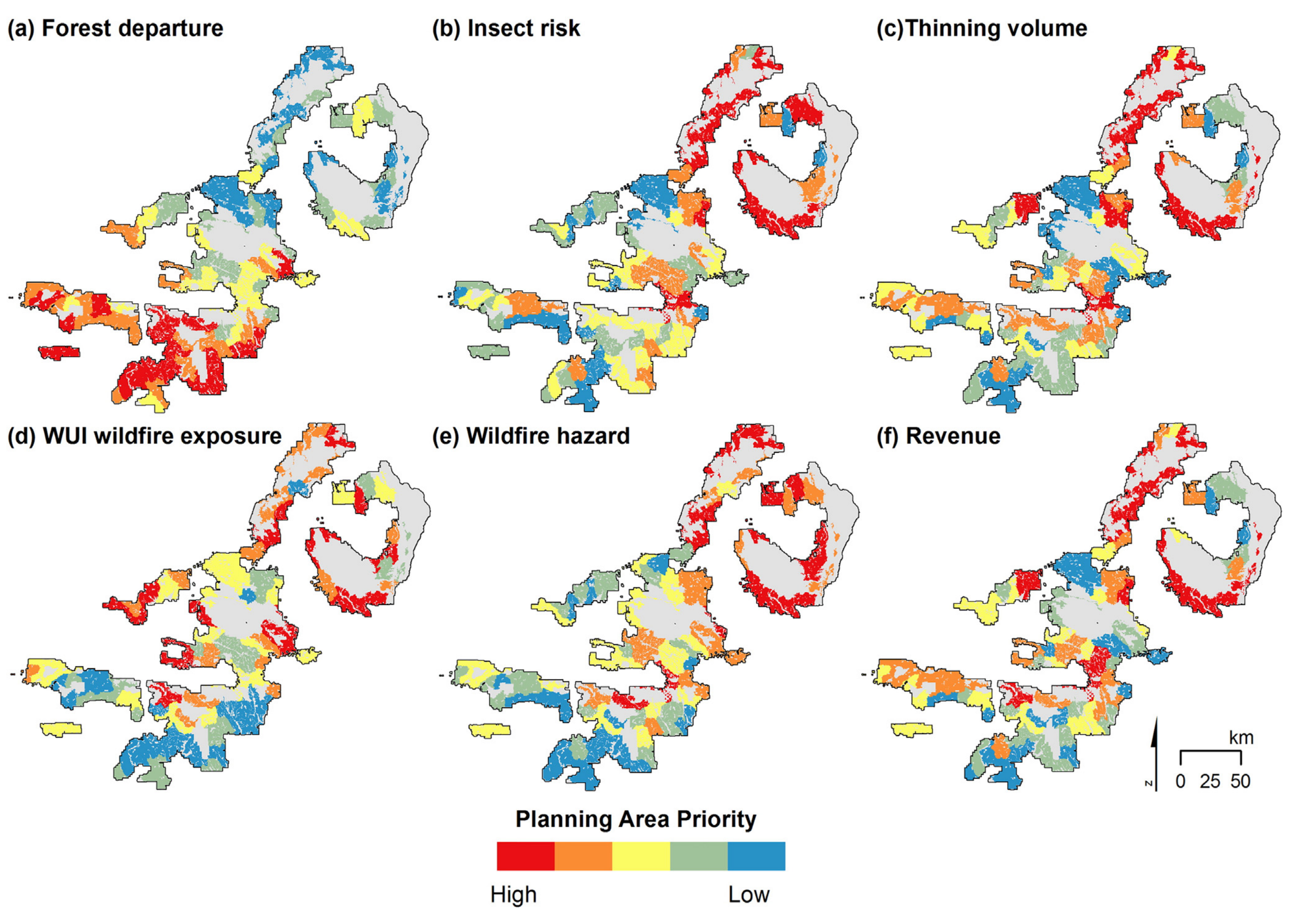

Fig. 5. Results from optimization modeling showing prioritization of the 102 planning areas in the study area for each of the six restoration objectives. (a) forest departure, (b) insect risk, (c) merchantable timber volume generated from restoration thinning, (d) wildland urban interface (WUI) wildfire exposure from national forest (NF) ignited wildfires, (e) wildfire hazard, and (f) revenue from restoration treatments. Stands selected for treatment within each planning area maximize attainment of the restoration targets assuming treatments on 5000 ha. Gray areas indicate wilderness and roadless areas where mechanical thinning and fuels mastication are either not allowed or not practiced under current forest planning regulations.

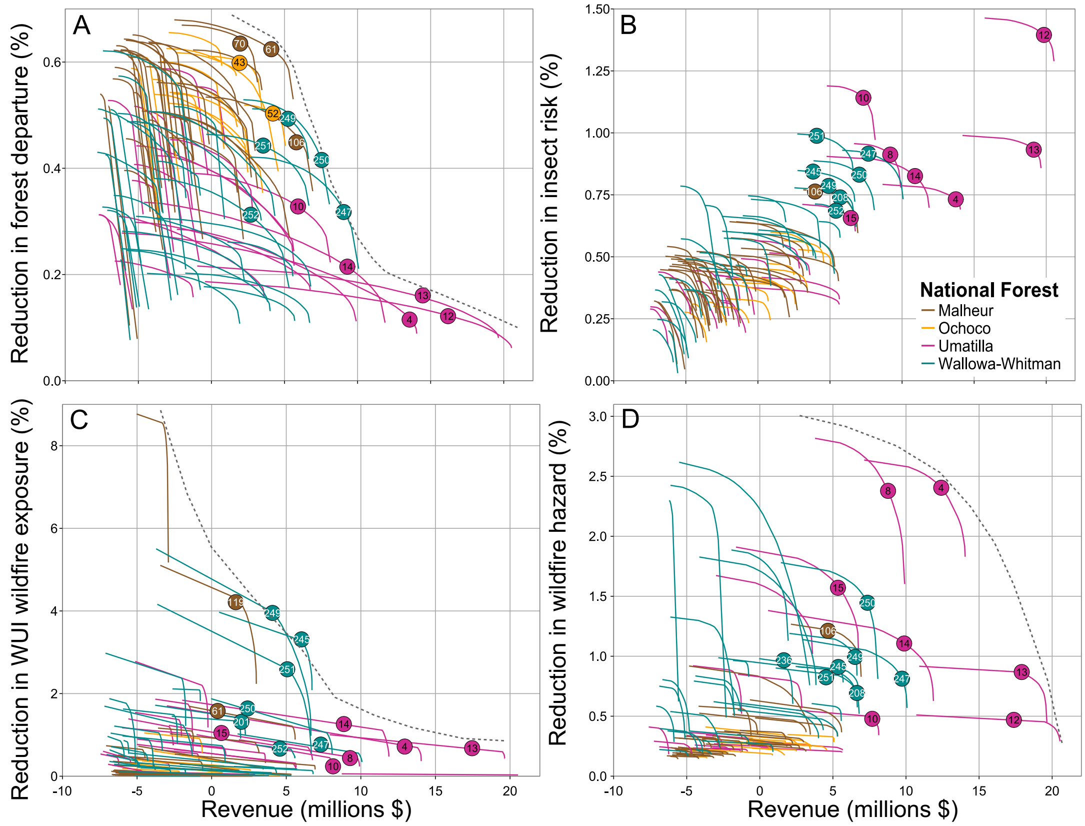

Fig. 6. Production possibility frontiers (PPF) for each of the 102 planning areas in the study area. Each panel shows trade-offs between selecting stands to treat to maximize revenue versus address non-economic restoration objectives: A) reduction in forest departure, B) reduction in insect risk, C) reduction in wildland urban interface (WUI) wildfire exposure from national forest (NF) ignited wildfires, and D) reduction in wildfire hazard. PPFs are generated by optimizing the selection of stands to address a mix of the joint production of two restoration objectives in each panel. Convex PPFs for specific planning areas indicate a wide range and relatively high potential for the joint production of revenue and one of the other four objectives. PPFs for planning areas distant from the origin (e.g., 4 in panel D) indicate high potential joint production and thus are higher priority for implementation. Asymmetrical PPFs (e.g., 12 in A) suggest that a wide range of production is possible for revenue without an impact on treating forest departure. Dashed gray lines portray the larger scale trade-offs associated with selecting a single project for each of the planning areas on the production frontier. This latter trade-off describes implementation choices and potential attainment outcomes for forest supervisors at the ecoregional scale.

A number of forest-scale differences were apparent in the PPF relationships. Specifically, there was substantially more opportunity on the Umatilla and Wallowa-Whitman to produce revenue and treat wildfire transmission to WUIs (Fig. 6C) compared to the Ochoco and Malheur. Likewise, generating revenue and treating forest departure were feasible on the Wallowa-Whitman but not the Malheur (Fig. 6A).

Trade-offs in the optimal joint production *among* planning areas within the ecoregion can be examined by tracing the outer bounds of the PPFs for all the planning areas (Figs. 6, 7, dashed gray lines). These trade-offs represent a regional perspective on prioritizing a project among the 102 planning areas for implementation, versus the trade-offs associated with selecting stands to treat within an individual planning area selected for implementation (Fig. 1C). The point of this example is to show that there are multiple scales of trade-offs in restoration planning, including within a single planning area (one line in Fig. 1C) versus among a large number of planning areas. It can be seen that ecoregional PPFs among all planning areas were either linear (Figs. 6B, 7B), convex to the origin (Figs. 6C, 7C), or concave to the origin (Figs. 6D, 7D), compared to concave for individual planning areas. Trade-offs at the ecoregion scale for wildfire hazard had a more or less concave form, while forest departure and WUI exposure from national forest lands were convex, the latter suggesting sharper trade-offs among planning areas (larger scales) for restoration opportunities. By contrast, insect risk had a linear form meaning that the production of these two objectives at the regional scale were highly correlated, which results from the fact that the insect and disease risk metric is derived from stand basal area. Overall, the shape (convex versus concave) of the PPF and associated trade-off relationships at the ecoregional scale (choosing a project in each of the planning areas on the frontier) was substantially different compared to that observed within individual planning areas.

Optimal rates of attainment for each objective were examined by first determining the optimum schedule of planning areas and then calculating the cumulative attainment as treatments were implemented in each planning area, from highest to lowest priority (Fig. 8A). The rate of attainment under different optimization scenarios was non-linear, especially for WUI wildfire exposure and revenue (Fig. 8A, B). About 60% of the total WUI exposure could be treated on 100,000 ha (20 planning areas) (Fig. 8A). By contrast, treating the same area under a scenario where treating forest departure was the primary objective addressed only 10% of the total departure in the study area. The differential rates of attainment resulted from spatial patterns of restoration targets within and between planning areas.

We plotted cumulative attainment for each of the restoration objectives under a revenue optimization scenario to understand the economics of ecological objectives (Fig. 8B). Here, planning areas were implemented based on net revenue (high to low) and the cumulative attainment of the other objectives was measured (Fig. 8B). As expected, attainment was substantially less (40–60% reduction depending on the objective) compared to scenarios where they were individually optimized (Fig. 8A), the exception being thinning volume where reduction in attainment was relatively minor (5–10% reduction). Cumulative revenue initially increased with increasing treated area until 250,000 ha of treatment, and then declined as planning areas with negative revenues were added to the scenario.

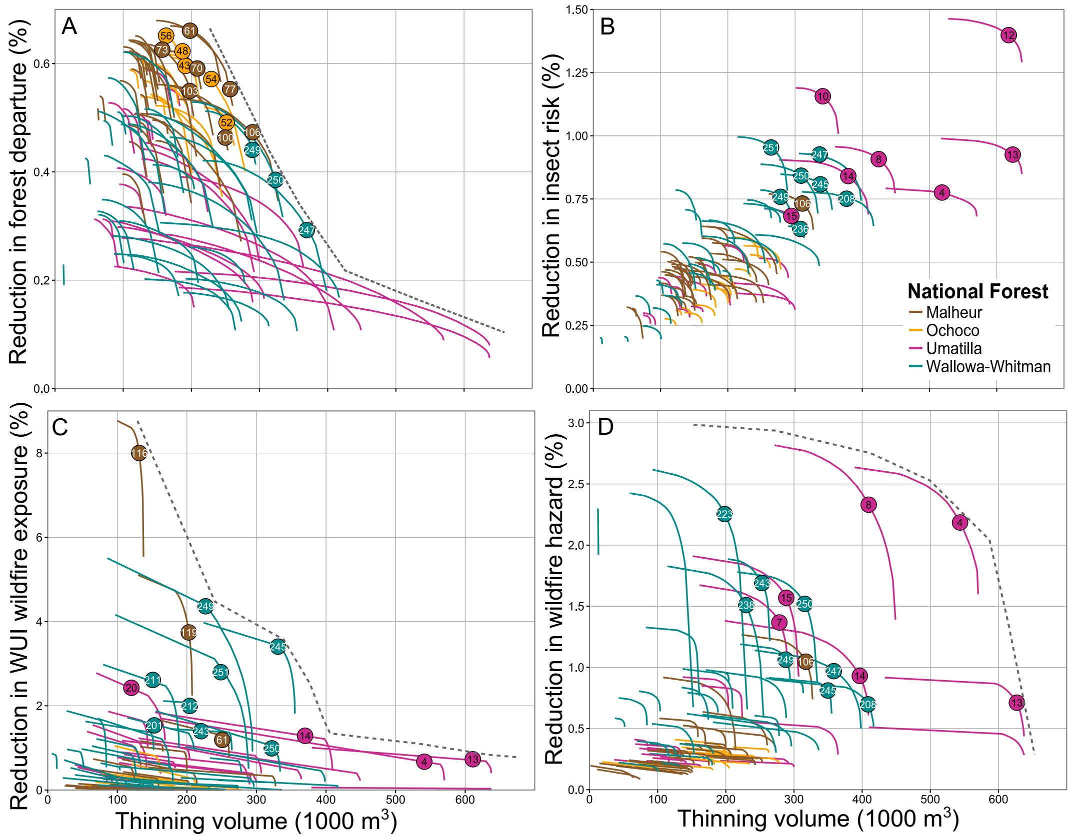

Fig. 7. Production possibility frontiers (PPF) for each of the 102 planning areas for thinning volume generated from treatment versus each of the other restoration objectives: A) reduction in forest departure, B) reduction in insect risk, C) reduction in wildland urban interface (WUI) wildfire exposure from national forest (NF) ignited wildfires, and D) reduction in wildfire hazard. PPFs are generated by optimizing the selection of stands to address a mix of the joint production of two restoration objectives in each panel. Convex PPFs for specific planning areas indicate a wide range and relatively high potential for the joint production of revenue and one of the other four objectives. PPFs for planning areas distant from the origin (e.g., 4 in panel D) indicate high potential joint production and thus are higher priority for implementation. Asymmetrical PPFs (e.g., 13 in B) suggest that a wide range of production is possible for thinning volume without an impact on treating WUI wildfire exposure. Dashed gray lines portray the larger scale trade-offs associated with selecting a single project for each of the planning areas on the production frontier. This latter trade-off describes implementation choices and potential attainment outcomes for forest supervisors at the ecoregional scale.

To examine costs for achieving specific levels of each restoration objective we plotted revenue versus attainment for each optimization scenario (Fig. 8C). Comparing Fig. 8A and C it is possible to identify the economic cost or benefit of different restoration priorities. For example, prioritizing forest departure was the costliest restoration objective where treating 40% of the departure in the study area (ca. 350,000 ha, Fig. 8A) would cost about $300 million (Fig. 8C). By contrast, 60% of the available wood volume from thinning operations (500,000 ha, Fig. 8A) could be obtained at a net cost of $0 (Fig. 8C). About 50% of the total WUI wildfire exposure can be treated at a cost of $58 million. Prioritizing restoration treatments on areas of high insect risk resulted in positive revenue until cumulative attainment reached 35% (Fig. 8C) at which point treatments would be implemented on 265,000 ha. In Fig. 8D we show the relationship between treated area and revenue when each restoration objective is optimized individually as in Fig 8A and C. These figures can be used to directly examine the economics of scale in prioritizing restoration objectives, i.e., the cost or revenue to treat specific levels of restoration targets (area), as discussed in restoration assessments (Haugo et al., 2015). For instance, treating 500,000 ha to reduce WUI wildfire exposure or forest departure would cost over $400 million (Fig. 8D). Under a scenario that optimized the reduction in forest departure, treating 300,000 ha (Fig. 8D) would cost $250 million (Fig. 8D) and treat about 35% of the forest departure in the study area (Fig. 8A).

We analyzed the relationship between restoration effort (treated area per planning area) and economic viability to examine variability in optimal treatment intensities. We performed this economic sensitivity analysis by incrementally adding stands to the treated pool in each planning area in order of decreasing profitability. The results revealed economic optima at a range of treatment intensities, and that many of the planning areas do not contain stands that have the potential to generate positive revenue (Fig. 9). Mean treated area at the revenue optima was about 26.7% (Fig. 10), compared to 43.3% at the break-even point (net revenue = 0). This latter treatment intensity represents a scenario where revenue from profitable stands is reinvested to maximize the area treated and post-treatment fire resiliency per planning area. The increase in final area treated under the two different scenarios, maximizing revenue versus maximizing area treated without investments (Fig. 10), varied among the planning areas and averaged 2478 ha (17%).

4. Discussion

Two major challenges in forest restoration planning are balancing the broad mix of socioeconomic and ecological goals, and funding restoration management where economic returns are not possible. In the case of restoration on national forests in the US, both challenges are amplified by diverse stakeholder interests that are vetted through a legislated collaborative planning process. The importance of economic prioritization and the identification of cost efficient areas for restoration have been widely discussed for a number of ecological restoration problems (Adame et al., 2015; Kimball et al., 2015; Wilson et al., 2012), including implications of inefficient prioritization (Iftekhar et al., 2016). However, only recently have restoration scientists and practitioners begun to include economic aspects in the design of restoration projects (Blignaut et al., 2014).

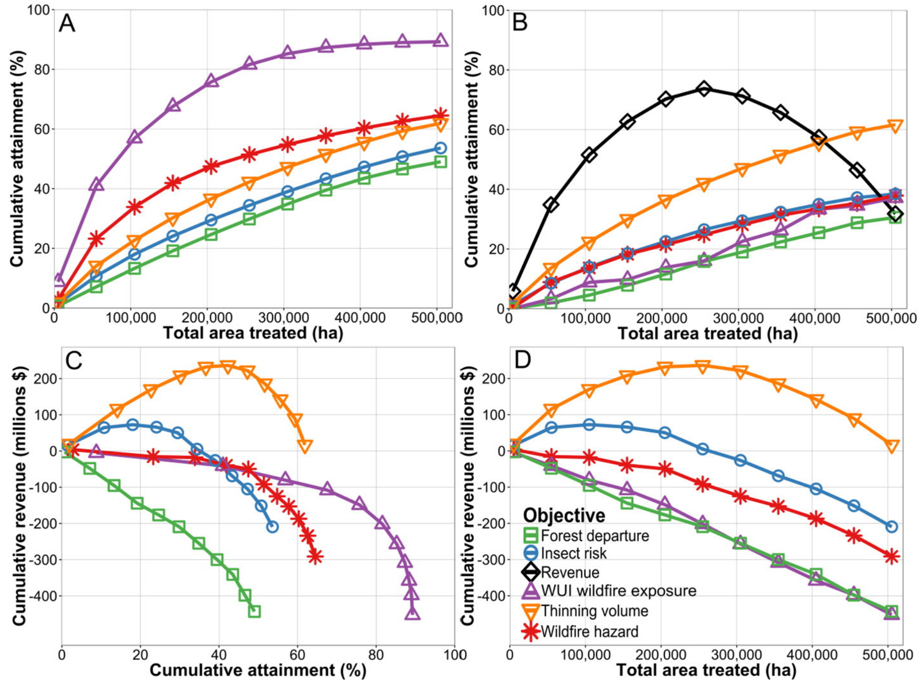

Fig. 8. Comparison of net revenue and restoration attainment among planning areas under alternative prioritization scenarios. A) Cumulative attainment of the five objectives when each of them are individually prioritized; B) same as A when revenue is prioritized; C) the revenue associated with levels of attainment when each of the objectives are prioritized in A; and D) cumulative revenue as a function of total area treated when each objective is individually prioritized as in A. Attainment is measured as the percentage of the total restoration objective in the study area that is treated in a particular scenario. WUI = wildland urban interface.

In our study, we found the economic viability of restoration projects is tenuous under current practices, as noted in earlier studies (Barbour et al., 2008a; Haynes et al., 2001), and that optimizing revenue to help finance restoration projects leads to a sharp reduction in the attainment of other socioecological objectives, especially reducing forest departure from historical conditions (Haugo et al., 2015). However, production possibility frontiers were highly variable among planning areas and national forests, and point to specific spatial priorities and opportunities to achieve restoration goals within the study area. Under assumptions of static landscape conditions, restoration attainment shows non-linear, diminishing returns in terms of restoration objectives and revenue as projects are implemented over increasingly larger areas.

The current study builds on our previous research (Ager et al., 2016; Vogler et al., 2015) by adding an economic analysis that shows the cost of specific restoration objectives and levels of attainment in terms of percentage of the restoration objective or area treated. We expanded the scale and scope compared to earlier work (Vogler et al., 2015) to quantify variability in production possibilities and decision trade-offs at the ecoregion level. Optimization modeling was simplified compared to Ager et al. (2016) concerning the same study area, by predefining planning areas rather than using spatial optimization algorithms to build them. This latter modification simplifies field application of our methods by restoration planners to prioritize projects on national forests.

The importance of analyzing economic trade-offs from Forest Service restoration programs is underscored by the fact that agency budgets specify annual wood volume targets (3.2 billion board feet in FY 2016), and the ecological trade-offs from attaining this production level are not known. Our economic analyses corroborate earlier studies that show scarce economic opportunities for restoration programs (Barbour et al., 2008a; Beck Group, 2015; Rainville et al., 2008), but also show that economic viability is highly dependent on the prioritization scheme used by managers to select both planning areas and stands within them for management.

Identifying economic opportunities from national restoration programs would seem to be a key step in collaborative planning, since revenue from harvested wood can support non-economic restoration activities (e.g., riparian restoration, invasive plant control) (USDA Forest Service, 2012). The juxtaposition of ecological settings with human values generates sharp trade-offs, especially with respect to community wildfire protection versus generating revenue to support restoration. This complicates 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 to wildfire in rural areas (USDA-USDI, 2014).

Our methods provide a number of improvements to current practices in collaborative restoration planning. First we incorporated a detailed economic analysis of potential restoration treatments that included modeling of costs and revenue at the stand-scale using operational planning boundaries. Modeling facsimile projects within the planning areas using stand-scale data captured local spatial variation in economic parameters that are potentially not represented in gridded forest inventory plots that are typically conducted at the scale of several kilometers per plot (Hartsough et al., 2008). Thus our financial analysis of forest restoration incorporated contagion, or lack thereof, of restoration opportunity not considered in earlier studies. Scale-related issues caused spatial packaging of thinning volume and other restoration objectives that can be minimized by using finer-scale modeling approaches than those typically used for restoration assessments. This modeling approach can provide more realistic assessments of potential revenue generated from restoration activities. Second, we show that decision trade-offs are scale-dependent and exist both within planning areas in terms of specific stands to treat, as well among planning areas and national forests. Thus the trade-off in prioritization analyses for restoration activities is a multiscale problem, with potentially different trade-off relationships at different scales. Scale-dependent trade-offs are not articulated in regional or national assessments (Barbour et al., 2005), nor to our knowledge quantitated and exposed in collaborative planning efforts. We believe that scale-dependent variations in PPFs

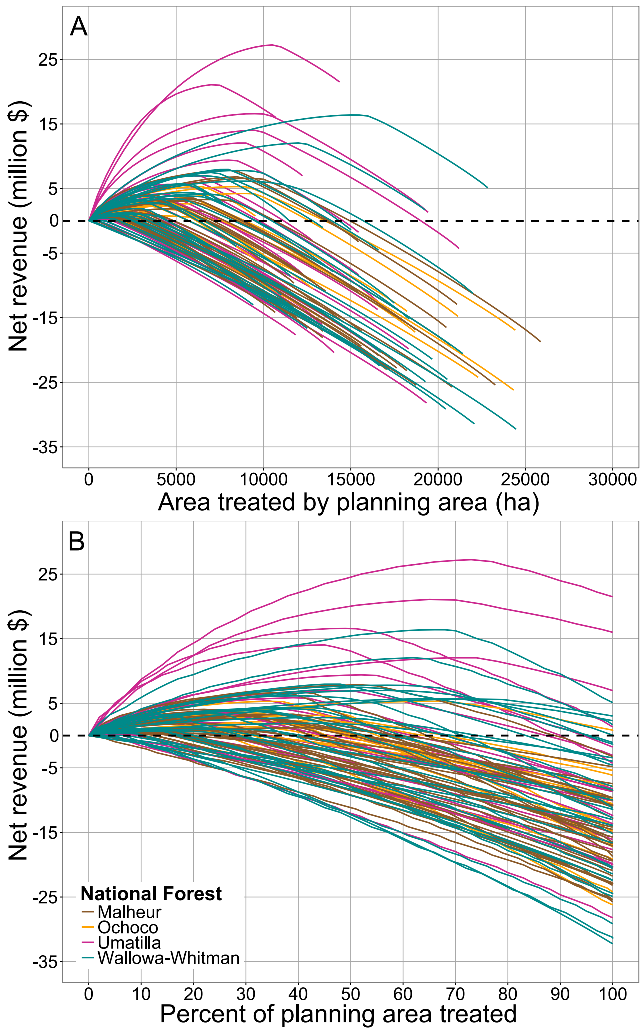

Fig. 9. Effect of treatment area on net revenue for each of the 102 planning areas in the study area under a revenue maximization scenario. (A) Revenue generated with increasing area treated by planning area; and (B) revenue versus percent of the planning area treated. The point of 0 revenue (Y = 0) shows the maximum area that can be treated per planning area at no cost, thus maximizing area restored per project.

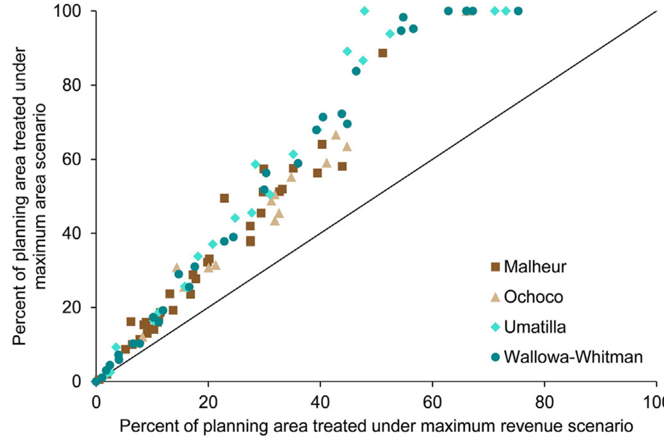

Fig. 10. Comparison of the percent area treated within each planning area for two different investment scenarios. X-axis shows the area treated when revenue was maximized by treating only stands where net revenue was positive. Y axis shows the area treated if net revenue was maximized and then re-invested in the planning areas to expand the total area treated to both stands that generated positive revenue and stands that did not but required restoration treatment (mechanical thinning, fuels mastication). The difference in the diagonal line and the individual symbols for each planning area is the increased area treated by re-investing revenue generated from treating profitable stands to expand the total area treated.

are important to recognize in agency initiatives to accelerate restoration initiatives. For instance, locally optimal PPFs, as determined by collaborative groups or by optimal joint production on a PPF, may not lead to optimal outcomes at larger (e.g., agency wide) scales (Fig. 1C). The former maximize the utility of stakeholder preferences, but may lead to inefficient outcomes at larger scales.

The effect of scale on PPFs was evident among all the variables tested. For instance, we noted production frontiers within a planning area were concave (Figs. 6, 7), versus linear or convex among planning areas (Figs. 6, 7 dotted lines) and national forests. Thus, local optimization of restoration objectives presents a shallower trade-off compared to trade-offs among planning areas, among national forests, and at the ecoregional scale of planning. Change in production functions with geographic scale in terms of shape (convex versus concave) and magnitude has heretofore not been explored. While our finding that larger scales present more choices for restoration programs is not surprising, we note that the change in trade-offs was not consistent among all restoration objectives studied. The hierarchy of trade-offs creates unique restoration opportunities for individual national forests and planning areas within them.

A number of management guidelines, frameworks, and strategic visions have been written about restoration and contemporary forest management (Franklin and Johnson, 2012; Franklin et al., 2013; Stine et al., 2014; USDA Forest Service, Pacific Northwest Region, 2015b). However, schedules that forecast how specific rates of treatment will generate outputs from the program, including raw wood materials and revenue, and simultaneously achieve goals for restoring fire resiliency, are nonexistent. In fire prone forests of the Pacific Northwest, forest restoration efforts were initially guided by watershed scale ratings of restoration need (Rollins, 2009), and later replaced with more comprehensive work by Haugo et al. (2015) on forest structure departure at subwatershed scales. By contrast, our work clearly shows variation among and within planning areas and national forests in terms of potential restoration attainment and trade-offs among multiple objectives for restoration programs. These trade-offs were particularly sharp between ecological restoration and socioeconomic objectives, including fire transmission among public and private land parcels. The relatively large geographic scale of the study allowed us to identify restoration storylines for each national forest, and demonstrate a prioritization process that can be used at multiple scales (stands, planning areas, national forests, ecoregions). National forests throughout the western US are surrounded by unique timber-dependent communities and socioecological settings (Paveglio et al., 2009), and restoration efforts must be balanced to meet expectations of forest collaborative groups (Butler et al., 2015) and broad social expectations of restoration programs (Franklin et al., 2014). Articulating spatial priorities and trade-offs in US national forest restoration would seem useful given that: 1) fire suppression expenditures are depleting restoration funds (USDA Forest Service, 2014), and 2) communicating priorities to stakeholders within collaborative planning groups is a key part of restoration planning (Butler et al., 2015).

Biophysical restoration trade-offs result from the diversity of forest conditions that evolved from past management, biophysical setting, and disturbances such as wildfire. The net result of these anthropogenic and natural factors is a reduction in the spatial covariation of some restoration targets but not others. The spatial correlation between areas with high forest departure, community wildfire protection issues, and areas that generate revenue is perhaps weaker than assumed in restoration policy documents, especially those related to spatial patterns of socioeconomic values. We observed and quantified trade-offs at the scale of national forests, planning areas, and among stands within planning areas, all of which are useful outputs to examine alternative restoration priorities. Considering the spatial organization of these trade-offs is key to resolving conflicts in participatory planning used in forest restoration. Understanding management trade-offs and how the joint spatial distributions of both stressors and economic restoration goals contribute to these trade-offs on large landscapes is an important component of restoration planning (Allan et al., 2013; Bennett et al., 2009; Schroter et al., 2014). Trade-off analyses are being conducted in an increasingly diverse range of ecosystems and associated services (Cattarino et al., 2015; Chhatre and Agrawal, 2009; Hauer et al., 2010; Maron and Cockfield, 2008; Schroter et al., 2014; White et al., 2012).

We note several limitations in our assumptions and modeling. In particular, our approach for assessing revenues from restoration programs should be considered in its relative context rather than absolute values. Treatment costs are affected by a wide range of factors. Our methodology estimated costs associated with logging and hauling operations but ignored other relevant factors as outlined in Section 2.8. In addition, it is not possible to estimate acceptable profits, risk, and overhead rates for industrial entities that perform fuel management operations (Rummer, 2008). Estimating cost of fuel treatment programs is complicated by the fact that operations typically involve many different types of equipment and treatment methods applied to a range of stand conditions in terms of both physical setting and stand structure (Rummer, 2008). Our analysis found similar results for cost and net revenue of forest restoration operations as previous studies that used both engineering cost analysis (Rainville et al., 2008) and transaction evidence (Beck Group, 2015). We compared our results to several independent data sources and in general found reasonable agreement. These comparisons focused on individual analyses of the four economic subsystems (harvest costs, overstocked forested area, total volume production, and net project revenue). In terms of harvest costs, our average of \$27.90 m^{-3} for all harvested stands was close to that obtained in a recent five-year cash flow survey of all timber sales in the study area (\$26.50 m^{-3}, Robert Schatz, Ochoco National Forest, personal comm.). The area of overstocked stands was similar to that reported by Barbour et al. (2008a) for Morrow, Umatilla, and Grant counties for both number of overstocked hectares and potential merchantable volume from thinning treatments. However, we found large discrepancies for both metrics in Baker, Harney, Union, Wallowa, Crook, and Wheeler counties (Tables C6–C7). Differences in estimation may be a result of relatively few plot samples in Rainville et al. (2008) where in some instances, county scale estimates were derived from a single plot. Moreover, the modeling in Rainville et al. (2008) contained a technical error in thinning prescription where the calculation of post-thin stand density index was not adjusted to account for the fact that trees >53.3 cm (21 in.) DBH are not available to thin, and thus additional smaller trees must be thinned below SDI thresholds to achieve the target SDI. By not accounting for the SDI in large trees, the desired target will potentially not be met, underestimating thinning volume.

Differences in the modeled appraised value and the price that a timber contractor will be willing to pay for a particular sale (timber receipts) are influenced by market value of timber and costs as well as profit margin, risk preferences, overhead rates and bidding competition. Forest Service reports on total volume and value (timber sale receipts) show an average value of \$347.20 ha^{-1} and \$7.95 m^{-3} for all timber sale receipts within the study area from 2008 to 2015 (USDA Forest Service, 2016c). From a simulation run of the same level of treated area (83,779 ha) we found an average value of \$882.99 ha^{-1} and \$17.29 m^{-3}. Our estimates of timber value should therefore be viewed as the maximum potential value a contractor would be willing to pay for a timber sale and not an estimate of future potential harvesting receipts.

In addition to economic limitations, our simulation of projects within planning areas required a number of assumptions. The implementation of restoration projects on national forests is driven by many factors including legislation, case law, budget appropriations, and conflicts over the production of goods and services. Our hypothetical projects assumed that activities would fall within constraints posed by these factors and thus be implemented according to our assumptions. Modeling forest restoration programs can only include a subset of the myriad of details that are part of restoration planning. The major assumptions we made in the modeling of production possibility frontiers included: 1) stands selected for active restoration would receive an appropriate suite of treatments including mechanical thinning, fuels mastication, and underburning, thereby addressing the particular restoration objective at hand; and 2) the post-treatment conditions of the stand would meet restoration goals for the various objectives , i.e. a reduction in insect and disease risk, reduce transmission of wildfires to the WUI, production of wood products, and reduction in forest departure. The same assumptions are made in the reporting of accomplishments from restoration programs in operational settings (Brown et al., 2004).

Restoration planning on national forests is a multiscale process where policy must be downscaled to forests, landscapes, stands, and tree neighborhoods (Larson and Churchill, 2012), creating a cascading series of decision trade-offs at each step. A major part of participatory planning under the Collaborative Forest Landscape Restoration Program (Butler et al., 2015) is deciding on landscape priorities that meet both the interests of stakeholders and the restoration goals of the Forest Service. As such, the process lacks analytical frameworks to examine important trade-offs among restoration priorities. Cavender-Bares et al. (2015) proposed an analytical protocol as part of a sustainability framework that leveraged production possibility frontiers in collaborative planning where utility functions for different stakeholder groups are superimposed on PPFs to maximize utility for each stakeholder group (Fig. 11). Utility functions integrate values and services that stakeholders associate with different levels of services in the production possibility frontier. Stakeholder groups can be characterized by utility functions that represent their core resource values in relation to national forests as they affect their quality of life, economic livelihood, and risk from natural disturbance on adjacent national forest lands. Divergence among stakeholders in restoration planning stems from dependencies on different mixes of social, economic, and ecological outputs generated from national forests, such as recreation and wood products. The challenge with applying this approach for restoration programs on national forests is that large decentralized agencies will have utility functions that vary by scale, and thus national policy objectives as represented in utility functions would likely vary for stakeholder groups involved in local planning. Future refinements in restoration planning, including

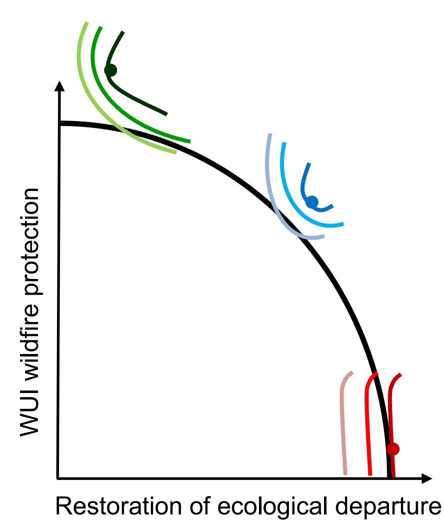

Fig. 11. Indifference curves superimposed on a production possibility frontier (PPF) illustrating different stakeholder preferences for restoration of one goal versus another. Indifference curves describe preferences by individual stakeholders for the production of specific mixes of restoration objectives. In practice, indifference relationships could be determined from preference surveys and used along with PPFs to apply resource economic principles as part of the collaborative forest restoration planning process. Figure adapted from King et al. (2015) for the problem of restoration.

application of methods we present here and socioeconomic modeling of the collaborative planning process, will in the long-run facilitate implementation of restoration policies on US public lands and adjacent fire prone landscapes.

Acknowledgements

This work was funded by the Natural Resources Staff Group at the Pacific Northwest Region, USDA Forest Service. We are grateful to Stu Brittain, Alturas Solutions, for development of the Landscape Treatment Designer and to Ken Bunzel for help with GIS processing. Edwin Brown and Sarah Crim provided useful guidance on several aspects of the study. We thank Rob Schantz, Kone Hancock, and Neil McClusker of the Forest Service for help with parameterizing the economics model. Rachel Houtman assisted with technical editing.

Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.ecolecon.2017.01.001.

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