Economic opportunities and trade-offs in collaborative forest landscape restoration

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Abstract

We modeled forest restoration scenarios to examine economic 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 trade-offs between generating revenue from forest restoration activities versus addressing ecological departure from fire resiliency, and wildfire risk to the 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 ecological departure from historic conditions. The analytical framework and results can inform ongoing collaborative restoration planning programs to help stakeholders

understand the opportunity cost of specific restoration objectives. This work represents the first spatially explicit, economic trade-off analysis 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.

Keywords: forest restoration, spatial optimization, restoration prioritization, production possibility frontiers, collaborative planning

1. Introduction

Restoration ecology has increasingly become a key component of many 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 the dry forest ecosystem (Noss et al., 2006; USDA Forest Service, 2012). The program encompasses 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 program was broadened with the Forest Landscape Restoration Act (Omnibus Public Land Management Act of 2009, Title IV) to establish a network of collaboratives to guide science-based planning efforts 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 increased employment in rural economies (USDA Forest Service, 2016b).

The science dialog around the program has been extensive, and included 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 Collaborative Forest Restoration Program (CFLRP, USDA Forest Service, 2015a) 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 increasingly 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 needs under national policy direction (Butler et al., 2015). 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 criteria (Neeson et al., 2016), including economics (Adame et al., 2015; Kimball et al., 2015), are non-existent. Nor are analyses conducted to evaluate trade-offs among economic services and the reduction of stressors (Allan et al., 2013; Bullock et al., 2011; Maron and Cockfield, 2008) that potentially impact the forests.

Trade-offs in restoration stem from finite budgets, capacity, and spatial variation and covariation across different restoration targets (Anderson et al., 2009; Martin et al., 2016;

Neeson et al., 2016). Thus, the opportunity cost of emphasizing one restoration objective over another is not known to stakeholders in the collaborative planning programs. Moreover, trade-offs are not exposed in regional 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 implementation scale (Barbour et al., 2008e; Rasmussen et al., 2012). The net result is that 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 due to scale effects on production functions (King et al., 2015) or scale mismatches (Cumming et al., 2006) between assessments and project implementation. Integrating economic and ecological trade-off analyses into current programs could have manifold effects on long-term effectiveness in terms of sustaining rural economies, and meeting fire resiliency objectives on large fire prone landscapes. For instance, economic analyses can help maximize the scale of treatments by subsidizing expensive fuels, mastication, and thinning treatments of small non-commercial trees with harvesting and sale of commercially viable materials elsewhere within planning areas.

In this paper we describe a detailed analysis of economic and ecological trade-offs within a fire prone ecoregion that includes 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 among the four national forests within the ecoregion. 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 a collaborative planning framework (Butler et al., 2015) via the use of production frontiers (Cavender-Bares et al., 2015; King et al., 2015) (Fig. 1).

Fig. 1. Examples of how production possibility frontiers (PPF) can be interpreted to facilitate decision trade-offs and collaborative planning in forest restoration programs. A) Convex to the original (black line) versus concave to the origin (red line) production relationships caused by spatial correlation, or lack thereof, of restoration targets determine the magnitude of trade-offs between competing restoration objectives and the need for collaborative planning; B) effect of increasing the scale from an accelerated restoration program where investment inputs are doubled and the ability to achieve multiple restoration goals is constrained because of differential scarcity in the restoration targets; and C) example where local collaborative groups select projects for implementation within national forests (red circles) but local preferences result in a non-optimal or non-preferred mix of outputs at the ecoregional scale. Panel C adapted from King et al. (2015).

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 forested areas are dominated by lodgepole pine (*Pinus contorta* Douglas ex Loudon) at higher elevations. About 50% of the study area is designated for

active forest management with mechanical treatments, with the remaining areas protected by wilderness and roadless legislation, and conservation. 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 cubic meters per year. 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. brevicornis* LaConte). However, over most of the area stand densification resulting from fire exclusion has resulted in densification.

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.

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).

Estimates of restoration need vary, with initial estimates suggesting that 34% (506,696 ha) of the forested area is in need of active restoration (USDA Forest Service, 2013). Downscaling a more recent analysis of ecological 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 significant ecological departure. Forest restoration treatments (mechanical treatments and fire) are completed on about 32,000 ha annually, or about 3.6% of the area (2005 – 2015). 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. The stand layer was obtained from the national forests and is used for project planning with stand boundaries following natural breaks in vegetation types and changes in stand structure from past management activities. Each stand was attributed with a land management designation from respective forest land management and resource plans, and protected areas within biodiversity and amenity reserves, including wilderness and inventoried roadless areas, were removed from consideration for restoration treatments. This resulted in a layer that 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). The following restoration objectives were defined: 1) forest departure from historic conditions, 2) potential basal area mortality from insects and disease, 3) potential wildfire hazard, 4) wildfire exposure to the wildland urban interface (WUI), 5) potential stemwood volume from mechanical thinning treatments, and 6) total net value of

171 treatments. The development of each objective is described below with more detailed
 172 descriptions in Appendix A and C. Finally, the study area was divided into 102 planning areas
 173 (average size = 15,102 ha) based on input from national forests (Fig. 3). The planning areas
 174 generally follow subwatershed boundaries.

175
 176 **Fig. 3.** Map of the 102 planning areas on the four national forests included in the study area.
 177 Planning area boundaries generally follow watershed boundaries in the study area. Small parcels
 178 created by discontinuities between national forest ownerships and watershed boundaries were
 179 merged with adjacent planning areas consistent with planning area delineation used by the
 180 national forests.

2.3. *Forest Departure from Historical Reference Conditions*

We used spatial data on ecological departure from Haugo et al. (2015) that quantifies departure of current vegetation conditions from historical vegetation conditions (Appendix A). The methodology builds on LANDFIRE (2013b) and fire regime-condition class score 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, however data we used from Haugo et al. (2015) measures stand densification and changes in species composition towards fire intolerant species. 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 opportunities due to 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). 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 average 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 an estimate of potential fire behavior for each stand assuming static weather conditions and fuel

moisture (Appendix A). Surface and canopy fuels flame length data were obtained from LANDFIRE (2013a) to estimate fire intensity. 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 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 fire intensity per stand (Fig. 4E).

Fig. 4. Spatial distribution of restoration objectives in the study area. (a) forest departure, (b) insect risk, (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, and (f) revenue from restoration treatments. See Section 2 for details on the estimation of each variable.

2.6. *Wildfire Exposure to the Urban Interface*

We measured wildfire transmission risk to 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. This included polygons with very low structure densities to correspond with current fire suppression efforts that aim to protect even individual structures in wildland areas. There were 52,202 WUI polygons containing 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 housing units 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 tree lists with the Forest Vegetation Simulator (FVS, Dixon, 2002). Tree lists for the study area were obtained from the LEMMA project that used an imputation process to extrapolate forest inventory data (FIA, Roesch and Reams, 1999) data to 30 m x 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, which were developed in detail in previous studies (Ager et al., 2007; Barbour et al., 2005). Thinning prescriptions prioritized removal of smaller trees to reduce ladder fuels, in order to prevent torching and crowing fire behavior. Additionally, prescriptions targeted removal of late-seral, fire-intolerant species (e.g., grand fir) and retention of fire tolerant early seral species (e.g., ponderosa pine). Stand conditions that resulted in a thinning treatment were based on thresholds for 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 high density of small trees. Specifically, if stand density was less than 45% of the maximum SDI but there were greater than 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 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 the expert opinion of local timber sale planning staff. Log “pond value” of harvested timber was calculated using the economics extension of 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 1) were collected from timber sales specialists on the Umatilla NF and used to calculate total value of delivered logs from each stand. Log pond values were only calculated for stands that generated greater than $28 \text{ m}^3 \text{ ha}^{-1}$ of merchantable timber, assuming stands producing less were not commercially viable. Material removed during restoration treatment of these 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). Ground-based harvesting system (Table 2) was assigned for stands having a slope less than or equal to 35%, and a cable harvesting system (Table 3) was assigned for all stands that exceed the 35% threshold. Average slope per stand was calculated from digital elevation data (30 x 30 m). Hauling distances were calculated using Euclidian distance to the nearest wood processing facility. Although this approach generated downward-biased estimates of 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 m³ per truck (Mason et al., 2008). Hauling costs were calculated as:

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

Note that hauling cost was only calculated for stands that generated greater than 28 m³ ha⁻¹ of merchantable timber, and considered the round trip distance to the nearest mill.

Stands that were treated to meet fire protection goals were assessed additional fixed costs to account for those treatments. For stands that generated greater than 28 m³ ha⁻¹, we assumed 40% of the stand would require additional fuel treatment at a cost of \$1,112 ha⁻¹. If a stand was triggered for treatment and did not generate 28 m³ ha⁻¹, we assumed a fixed cost of \$1,112.00 ha⁻¹ to treat residual fuel load. These cost parameters were derived from local transaction data on the national forests. This approach 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 prior to project implementation.

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

2.9. 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 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 stand polygon shapefile attributed with restoration objectives. The user enters a restoration scenario by specifying objectives (e.g., revenue), activity constraints (e.g., budget), and stand treatment thresholds (e.g., fire behavior), and the program identifies stands to treat such that:

$$\text{Max } \sum_{j=1}^k (Z_j * \sum (W_i N_{ij})) \quad (2)$$

Subject to:

$$\sum_{j=1}^k (Z_j A_j) \leq C \quad (3)$$

where C is a global constraint on activity per project area, 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 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, 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 in the study area based on restoration objective (Appendix A), and simulated treatment of 5000 ha in each planning area to address one or more of the restoration objectives described in section 2. Overstocked stands were assumed to generate wood volume from thinning treatments, and surface fuels reduction (mastication and

underburning) were also 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. Outputs were used to generate production possibility frontier relationships between harvest volume and each objective. For instance, weights of 10 and 0 for objective A and B, respectively, represent the maximum production for objective A, whereas weights of 20 and 20 for each objective represent a mixed production for both.

We used average per-area condition of each stand in the selection algorithm in order 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 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 (5000 ha in each planning area) ranged from a high of \$20.53 million to a low of \$-7.38 million with most profitable 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), and thus potential value of economic prioritization ranged from a high of \$26.7 million for the Umatilla NF and a low of \$8.9 million for the Ochoco NF. In general, variation among planning areas on the Ochoco NF was minimal compared to the Umatilla and Wallowa-Whitman. A similar pattern of 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 are a number of instances where low and high priority planning areas were adjacent, illustrating the magnitude of local variation of

368 restoration opportunity in the study area.

369
 370 **Fig. 5.** Results from optimization modeling showing prioritization of the 102 planning areas in
 371 the study area for each of the six restoration objectives. (a) forest departure, (b) insect risk, (c)
 372 merchantable timber volume generated from restoration thinning, (d) structure exposure in the
 373 wildland urban interface from national forest (NF) ignited wildfires, (e) wildfire hazard, and (f)
 374 revenue from restoration treatments. Stands selected for treatment within each planning area
 375 maximize attainment of the restoration targets assuming treatments on 5000 ha.

376 Production possibility frontiers (PPF) (Figs. 6, 7) exhibited three broad trade-off
 377 relationships between revenue and other restoration objectives. In some planning areas,
 378 attainment of one restoration objective was largely unaffected by the other (Fig. 6A, planning
 379 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 sub-merchantable material resulted in some differences in the PPFs for particular planning areas.

Fig. 6. Production possibility frontiers (PPF) for each of the 102 planning areas for revenue from treatment versus each of the other restoration objectives: A) forest departure, B) insect risk, C) structure exposure in the wildland urban interface from national forest (NF) ignited wildfires, and D) wildfire hazard. PPFs are generated by optimizing the selection of stands to address one or the other or both of the restoration objectives in each panel. Convex and symmetrical curves indicate opportunity for restoration to accomplish multiple objectives. Asymmetrical curves show that optimal joint production favors one objective over the other.

A number of forest-scale differences were apparent in 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 was feasible on the Wallowa-Whitman but not the Malheur (Fig. 6A).

Trade-offs among planning areas within the ecoregion as a whole can be visualized by tracing the outer (right) bounds of individual PPFs (Figs. 6, 7), thus providing insights into scale effects on restoration opportunity. It can be seen that an aggregate PPF among all planning areas was 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 forest departure and wildfire hazard had a more or less concave form, while structure exposure from national forest lands was convex, the latter suggesting sharper trade-offs among restoration opportunities at larger scales. By contrast, insect risk has a linear form as a result of correlation between risk and timber volume inherent in the calculations. Thus, increasing the scale of prioritization assessments presents different trade-offs to stakeholders and managers.

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 revenue and wildfire hazard (Fig. 8A). About 60% of the total structure 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.

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) forest departure, B) insect risk, C) structure exposure in the wildland urban interface from national forest (NF) ignited wildfires, and D) wildfire hazard. PPFs are generated by optimizing the selection of stands to address one or the other or both of the restoration objectives in each panel. Convex and symmetrical curves indicate opportunity for restoration to accomplish multiple objectives. Asymmetrical curves show that optimal joint production favors one objective over the other.

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. 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; and D) cumulative revenue as a function of total area treated. Attainment is measured as the total inventoried quantity of each of the restoration goals on the landscape.

To examine costs for achieving specific levels of each restoration objective we plotted revenue versus attainment for each optimization scenario (Fig. 8C). Comparing Figs. 8A and 8C it is possible to identify the economic cost or benefit of different restoration priorities.

Prioritizing forest departure was the costliest restoration objective where treating 50% of the departure in the study area would cost about \$450 million (Fig. 8C). By contrast, 60% of the available wood volume from thinning operations could be obtained at a net cost of \$0. About 50% of the total structure 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 the latter is optimized as in Fig. 8B. These data can be used to understand the economics of scale in prioritizing restoration objectives, i.e., the cost or revenue to treat specific levels of restoration targets as suggested in restoration assessments (Haugo et al., 2015). For instance, treating 500,000 ha under a revenue optimization scenario would break even financially (Fig. 8D, thinning volume) and treat 25% of the forest departure in the study area (Fig. 8B). By contrast, treating the same area under a scenario where forest departure (versus revenue) was prioritized (Fig. 8A) the attainment was increased by 15%. About 40% of the total forest departure would

require implementing projects on 365,000 ha (Fig. 8C) at a total cost of \$300 million.

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.

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 re-invested 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%).

Fig. 10. Percentage area treated by planning area under two scenarios showing the increased treatment area when revenue is re-invested within planning areas to maximize area treated, i.e. revenue from profitable stands is used to subsidize treatment of stands that do not generate revenue but require restoration treatment (mechanical thinning, fuels mastication). The increase in treated area averages 2478 ha per planning unit (17% increase). The average area treated per planning area is estimated at 26.7% when revenue is maximized, and 43.3% if revenue is re-invested to treat stands that do not generate sufficient revenue for harvesting and treatment costs.

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 of ecosystem services has 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 (Iftekhhar 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).

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 ecological departure from historic 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 (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. 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.

This work contributes to achieving long-term social and ecological goals for US national forest restoration programs, and communicating benefits of restoration efforts (Mentz et al. 2013). The importance of analyzing economic tradeoffs 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 tradeoffs from attaining this production level are not known. Our economic analyses corroborate earlier studies that show challenging economic opportunities for restoration programs (Barbour et al., 2008a; Beck Group, 2015; Rainville et al., 2008). Specifically, we showed diminishing returns for generating thinning volume as restoration activities were implemented at increasingly larger scales.

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 (e.g., riparian restoration, invasive plant control) to sustain ecosystem services from US national forests (Deal et al., 2012; 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 and fire hazard reduction outside of areas that affect WUIs. Thus, opportunity costs for increasing efforts for community wildfire protection impact progress towards reducing ecological departure and long-term reintroduction of fire, and generation of thinning volume. 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 in rural areas (USDA-USDI, 2014).

Our findings contribute in several ways to ongoing restoration programs in fire prone national forest landscapes. First, we present the first economic analyses of restoration where thinning activities were modeled at the stand-scale using actual planning area boundaries. These facsimile projects capture local spatial variation that is not represented in coarse-gridded forest inventories used in past economic studies (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 the production of provisional ecosystem services that can potentially perpetuate restoration programs and meet socioeconomic goals. Second, we show that decision trade-offs are scale-dependent and exist within planning areas, among planning areas, and among national forests, making prioritization of restoration activities a multi-scale problem. 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 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. 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 (Fig. 6, 7), versus linear or convex among planning areas (Fig. 8) and national forests (Fig. 9). Thus, local optimization of

restoration objectives presents a shallower trade-off compared to trade-offs among planning areas, among national forests, and at the bioregional 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 storylines 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, 2015b). However, detailed forecasts on the rate of treatment to achieve specific restoration goals and information on resources required to attain them are largely nonexistent, especially for activities such as thinning, burning, and economic value and raw wood materials. 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 ecological 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 begin to understand 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 among national forests to meet expectations of forest collaborative groups (Butler et al., 2015) and broad social expectations of restoration programs (Franklin et al., 2014). Moreover, articulating spatial priorities and trade-offs in US national forest restoration is of growing importance given that: 1) suppression expenditures are depleting restoration funds (USDA Forest Service, 2014), and 2) there is a growing importance to communicate priorities to stakeholders within rapidly expanding collaborative planning groups (Butler et al., 2015) that help set local priorities for restoration.

Biophysical restoration trade-offs stem 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 for some restoration targets but not others. Spatial correlation between ecological departure, provisional ecosystem services, and community wildfire protection issues is perhaps more diffuse than assumed by policymakers, 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. These trade-offs are the root cause of conflicts in local and regional planning, and thus understanding their spatial organization is key to resolving conflicts in participatory planning used in forest restoration (Butler et al., 2015). Understanding management trade-offs and how joint spatial distributions of both stressors and ecosystem services contribute to these trade-offs on large landscapes is arguably an important component of restoration planning (Allan et al., 2013; Bennett et al., 2009; Schroter et al., 2014). Studies on trade-offs and spatial prioritization in conservation biology and natural resource literature cover a 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: 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. 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) methodologies.

Specifically, 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. Differences in estimation may be a result of relatively few plot samples in Rainville et al. (2008) where for instance, county scale estimates were derived from a single plot. Moreover, the modeling 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 inch) DBH are not available to thin, and thus smaller trees must be thinned below SDI thresholds to achieve the target SDI. Excluding this correction would lead to underestimating potential volumes in all stands that contain larger tree size classes.

Differences in the modeled appraised value from our analysis and the price that a timber contractor will be willing to pay for a particular sale (timber receipts) are not only influenced by market value of timber minus costs but also desired 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 $\$347.20 \text{ ha}^{-1}$ and $\$63.77 \text{ m}^{-3}$ for all timber sale receipts within the study area from 2008 to 2015. From a simulation run of the same level of treated area (83,779 ha) we found an average value of $\$882.99 \text{ ha}^{-1}$ and $\$17.29 \text{ m}^{-3}$. 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 ecosystem services. Our hypothetical projects assumed that activities would fall within constraints posed by these factors. Modeling forest restoration programs can only include

a subset of these drivers, and furthermore, a number of assumptions were required to simulate implementation of restoration projects. These 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 issue at hand; and 2) treatments substantially reduced ecological departure, insect and disease issues, wildfire exposure to WUI, and generated raw wood material from thinning, much the same way that planners assume accomplishment towards restoration targets in operational settings (Brown et al., 2004). We recognize that modeled outputs and identification of stand-scale prioritization represents a general guideline for planners and that information on local conditions and constraints for biodiversity and amenity protection will be part of a downscaling process to adjust outputs to local conditions.

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. Current planning under the Collaborative Forest Landscape Planning Act (Butler et al., 2015) in general consists of participants expressing priorities to federal forest managers, and eventually decisions about priorities and projects lead to implementation. As such, collaborative restoration planning lacks analytical frameworks to examine important trade-offs among restoration priorities. Cavender-Bares et al. (2015) proposed an interesting 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 stem from dependencies on different mixes of social, economic, and ecosystem services generated on 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 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 adapted landscapes.

Fig. 11. Indifference curves superimposed on a production possibility frontier (PPF) illustrating different stakeholder preferences. Indifference curves describe preferences by individual stakeholders for the production of specific ecosystem services. In practice, indifference relationships could be determined from preference surveys and used along with PPFs in

collaborative planning 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.

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949 **Tables**

950 **Table 1.** Delivered log values (\$ m⁻³) by species and size class used in the estimation of revenues
 951 from thinning treatments.

Small-end diameter (cm)	Species			
	<i>Pseudotsuga</i>	<i>Tsuga</i> spp., <i>Abies</i>	<i>Pinus ponderosa</i>	<i>Pinus contorta</i> ,
	<i>menziesii</i> , <i>Larix</i> <i>occidentalis</i>	spp.		<i>Picea</i> spp.
12.7 - 20.1	49.05	42.04	44.14	38.54
20.2 - 30.2	66.43	55.95	59.44	52.45
30.3 - 45.5	80.78	70.42	79.74	68.42
45.6 +	98.78	88.90	119.77	68.42

952

953

954 **Table 2.** Ground-based logging activity cost (\$ m⁻³) by tree size class and number of cut trees per
 955 hectare used to estimate harvesting costs for each simulated project area. Ground-based logging
 956 activities were assumed in stands with an average slope <35%. Harvest cost estimates were
 957 adopted from Rainville et al. (2008).

Tree size, DBH (cm)	Number of cut trees per hectare					
	12	49	124	247	494	988
0.1 - 15.2					37	34
15.3 - 17.8				30	28	26
17.9 - 22.9			24	22	20	19
30 - 40.6		14	13	13	13	13
40.7 - 53.3	13	13	13	13	13	13

958

959

960 **Table 3.** Cable-based logging activity cost (\$ m⁻³) by tree size class and number of cut trees per
 961 hectare for each simulated project area. Cable-based logging activities were assumed in stands
 962 with an average slope >35%. Harvest cost estimates were adopted from Rainville et al. (2008).

Tree size, DBH (cm)	Number of cut trees per hectare					
	12	49	124	247	494	988
0.1 - 15.2					130	91
15.3 - 17.8				136	82	63
17.9 - 22.9			105	73	50	42
30 - 40.6		30	29	26	26	25
40.7 - 53.3	25	21	20	20	20	20

963