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Forest Ecology and Management 211 (2005) 117–139
Forest Ecology and Management
www.elsevier.com/locate/foreco

Dry forests and wildland fires of the inland Northwest USA: Contrasting the landscape ecology of the pre-settlement and modern eras

Paul F. Hessburg [a,*], James K. Agee [b,1], Jerry F. Franklin [b,2]

[a] USDA Forest Service, Pacific Northwest Research Station, 1133 N. Western Avenue, Wenatchee, WA 98801-1229, USA
[b] College of Forest Resources, P.O. Box 352100, University of Washington, Seattle, WA 98195-2100, USA

Abstract

Prior to Euro–American settlement, dry ponderosa pine and mixed conifer forests (hereafter, the ‘‘dry forests’’) of the Inland Northwest were burned by frequent low- or mixed-severity fires. These mostly surface fires maintained low and variable tree densities, light and patchy ground fuels, simplified forest structure, and favored fire-tolerant trees, such as ponderosa pine, and a low and patchy cover of associated fire-tolerant shrubs and herbs.

Low- and mixed-severity fires provided other important feedbacks and effects to ponderosa pine-dominated stands and landscapes. For example, in stands, frequent surface fires favored an ongoing yet piecemeal regeneration of fire-tolerant trees by periodically exposing patches of mineral soil. They maintained fire-tolerant forest structures by elevating tree crown bases and scorching or consuming many seedlings, saplings, and pole-sized trees. They cycled nutrients from branches and foliage to the soil, where they could be used by other plants, and promoted the growth and development of low and patchy understory shrub and herb vegetation. Finally, surface fires reduced the long-term threat of running crown fires by reducing the fuel bed and metering out individual tree and group torching, and they reduced competition for site resources among surviving trees, shrubs, and herbs. In landscapes, the patterns of dry forest structure and composition that resulted from frequent fires reinforced the occurrence of low- or mixed-severity fires, because frequent burning spatially isolated conditions that supported high-severity fires. These spatial patterns reduced the likelihood of severe fire behavior and effects at each episode of fire. Rarely, dry forest landscapes were affected by more severe climate-driven events.

Extant dry forests no longer appear or function as they once did. Large landscapes are homogeneous in their composition and structure, and the regional landscape is set up for severe, large fire and insect disturbance events. Among ecologists, there is also a high degree of concern about how future dry forests will develop, if fires continue to be large and severe. In this paper, we describe the key landscape pattern and process changes wrought by the sum of the settlement and management influences to date, and we point to an uncertain future for ecosystem management. Widespread selection cutting of the largest and oldest ponderosa pine and Douglas-fir in the 20th century has reduced much of the economic opportunity that might have been associated with restoration, and long-term investment will likely be needed, if large-scale restoration activities are attempted. An

* Corresponding author. Tel.: +1 509 664 1722; fax: +1 509 664 1780.
E-mail addresses: phessburg@fs.fed.us (P.F. Hessburg), jagee@u.washington.edu (J.K. Agee), jff@u.washington.edu (J.F. Franklin).
[1] Tel.: +1 206 543 2688; fax: +1 206 543 3254.
[2] Tel.: +1 206 543 2138; fax: +1 206 543 3254.

0378-1127/$ – see front matter
doi:10.1016/j.foreco.2005.02.016

uncertain future for ecosystem management is based on the lack of current and improbable future social consensus concerning desired outcomes for public forestlands, the need for significant financial investment in ecosystem restoration, a lack of integrated planning and decision tools, and mismatches between the existing planning process, Congressional appropriations, and complex management and restoration problems.

Keywords: *Pinus ponderosa*; *Abies grandis*; *Abies concolor*; *Pseudostuga menziesii*; Landscape ecology; Mixed conifer forests; Fire ecology; Fire history; European settlement; Historical range of variability

1. Introduction

Ponderosa pine (*Pinus ponderosa*) forests became the poster child for unhealthy forests of the western United States (US) around the time, when Gast et al. (1991) presented their expert panel-based evaluation of declining forest conditions in the Blue Mountains. Thereafter, a number of additional qualitative and quantitative assessments were undertaken to detail changes that had occurred in the Inland West (Everett et al., 1994; Hann et al., 1997; Hessburg et al., 1999a; Huff et al., 1995; Lehmkuhl et al., 1994; O'Laughlin et al., 1993; Perry et al., 1994; SNEP, 1996; USDA, 1993). But forest ecologists and fire scientists were well aware of the importance of historical fires to these environments decades before this time (e.g., see Arno, 1976, 1980; Arno and Allison-Bunnell, 2002; Cooper, 1961a,b; Daubenmire and Daubenmire, 1968; Gruell et al., 1982; Hall, 1976; Morris, 1934a,b; Pyne, 1982; Soeriaatmadja, 1966; Weaver, 1959, 1961).

Usually when people speak of "unhealthy" forests, those prone to uncharacteristically intense or large-scale fires, insect outbreaks, and epidemics of forest diseases, they think of relatively dry ponderosa pine forests that have experienced fire exclusion and heavy selection cutting over several entries. The dry pine forests of the *Pinus ponderosa* zone have certainly been affected in this way, but dry mixed conifer forests, primarily in the grand fir (*Abies grandis*), white fir (*Abies concolor*), and Douglas-fir (*Pseudotsuga menziesii*) zones (*sensu* Franklin and Dyrness, 1988), have also been similarly affected by 200 years of settlement and management. We refer to forests in these four zones collectively as the dry forest, and they are represented by a range of cover types. Each cover type in the dry forest was once dominated by ponderosa pine cover under the influence of low- and mixed-severity fire regimes of the last several centuries. Without frequent low-severity or surface fires, Douglas-fir, grand fir, white fir, and on the driest sites ponderosa pine regenerated and released in the understory, and have been growing there for more than three-quarters of a century. To understand what makes these forests currently "unhealthy", it is insightful to examine the changes that have occurred to them over the 200-year period of settlement and management.

2. Pre-settlement-era dry forests

Prior to the settlement of the West by European immigrants, most dry forest environments were burned by relatively frequent low and mixed-severity fires (Arno and Allison-Bunnell, 2002). When we refer to low-severity fires, we are describing fires that occurred frequently, usually every 1–25 years, and where less than 20% of the basal area was killed (Agee, 1990, 1993). When we refer to mixed-severity fires, we refer to fires that occurred with moderate frequency, usually every 25–100 years, and where 20–70% of the basal area may have been fire-killed. In the context of dry forests, mixed-severity fires tended to be at the lower end of this 20–70% overstory mortality range. Such low- and mixed-severity fires favored relatively low tree density and clumped tree distribution, light and patchy fuel beds, simple canopy layering, and fire-tolerant tree and associated species compositions (Fig. 1).

In stands, low-severity fires favored fire-tolerant forest structures by removing the lower crown classes (Fig. 2). These fires also cycled nutrients from foliage and branches into the soil, promoted the growth of a low and patchy shrub and herb cover, reduced the threat of running crown fires by continually thinning stands, eliminating fuel ladders, elevating crown bases, and reduced competition for site resources

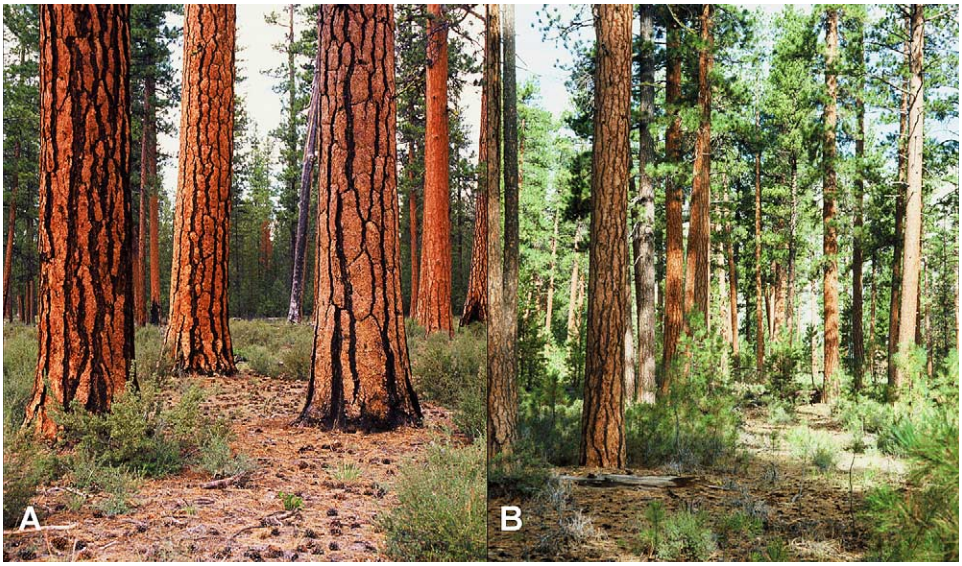

Fig. 1. (A) Example of a typical single-layered, fire-tolerant, open ponderosa pine stand one would have seen under a low-severity fire regime in the dry *Pinus ponderosa* zone (Franklin and Dyrness, 1988). Note the abundant bitterbrush (*Purshia tridentata*) shrub cover, the elevated tree crown bases, the variably spaced trees, and the fire scars at the base of many trees. In stands, such as these, so-called ''healthy'' conditions would have included some ongoing bark beetle mortality, especially in the tree clumps with higher stocking levels, and occasionally root disease mortality. These processes contributed high-quality large snags and down wood. (B) Example of a ponderosa pine stand from the *Pseudotsuga menziesii* zone. Note the advance regeneration of pine and Douglas-fir in the absence of recent fire.

Fig. 2. Example of a typical surface fire showing flame lengths and scorch heights less than 2 m. Above-ground portions of shrubs are scorched and partially consumed, while root systems remain intact, thereby maintaining the edibility of many browse species.

among surviving trees, shrubs, and herbs (Kauffman, 1990). Frequent surface fires tended to favor the largest trees with the thickest bark, producing an even-aged appearance, but stands themselves exhibited patchy age class distribution owing to the continuous regeneration of trees by fires (Harrod et al., 1999).

In landscapes, the natural patterns of dry forest structure and composition favored low- or mixed-severity fires by maintaining a semi-predictable mosaic, which spatially isolated conditions that supported high-severity fires (Fig. 3). Hence, severe fire behavior and fire effects were uncharacteristic of dry forest-dominated landscapes (Hessburg and Agee, 2003; Hessburg et al., 1999a). Rarely, dry forest landscapes were relatively more synchronized in their vegetation and fuels conditions and affected by climate-driven, high-severity fire events (Agee, 1997, 1998; Swetnam and Lynch, 1993; Whitlock and Knox, 2002).

3. Modern-era dry forests

The most influential change agents of the settlement and management periods can be grouped according to their primary ecological effects. Change agents acted to exclude fires, directly advance secondary succession, suppress fires, or some combination of these effects.

Domestic livestock and wild ungulate grazing, road and rail construction, grassland conversion to agriculture, urbanization, and rural development all contributed to the direct or indirect exclusion of fires. For example, domestic livestock grazing beginning in the 1870s rather abruptly reduced the abundance and distribution of flashy fuels that can rapidly carry surface fires across the landscape (Belsky and Blumenthal, 1997; Belsky et al., 1999; Irwin et al., 1994). In addition, grazing by large populations of deer during the 1940s–1960s, and more recently elk, combined with domestic livestock grazing to remove much of edible shrub cover as well as the flashy fuels (Irwin et al., 1994; Riggs et al., 2000).

Road and rail construction fragmented broad forest landscapes into smaller more isolated pieces, especially where road and rail beds functioned as effective fuel breaks to potentially expanding low flame length surface fires. Additionally, lands allotted to railroad companies by Congress were sold to settlers or harvested by the railroaders to pay the cost of building the new rail lines. These changes in land allocation and land use further fragmented the historical landscape with respect to historical fires (see Fig. 8 in Hessburg and Agee, 2003).

Similarly, grassland conversion to agriculture excluded fires, because many historical surface fires in dry forests actually began on grassy benches, ridge tops, or valley bottoms adjacent to dry forests and woodlands, or in nearby shrub steppe communities, and then migrated into dry forests. These fires were the result of natural lightning ignitions or intentional aboriginal burning (Agee, 1993, and references therein; Barrett and Arno, 1982; Boyd, 1999, and references therein; Robbins, 1999, and references therein; Whitlock and Knox, 2002, and references therein). Urbanization and rural development fragmented fire-prone dry forests at a fine scale by substituting highly combustible vegetation types with bare ground, concrete or asphalt, and grazed, irrigated, or cultivated spaces.

Repeated selection cutting had the direct effect of advancing secondary succession (Hessburg et al., 1999a, 2000; Hessburg and Agee, 2003). The preferred commercial species by land area and timber volume affected were ponderosa pine and Douglas-fir, and to a lesser extent western larch (*Larix occidentalis*); these species were also important early seral components that seeded in after fires (note that Douglas-fir has broad ecological amplitude; it can seed in after fire and tolerate shade). In addition, large-diameter ponderosa pine and Douglas-fir were quite tolerant of surface fires owing to their thick bark. As these larger trees were removed by selective logging, small canopy gaps were created and later filled by small diameter Douglas-fir, grand fir, and white fir. Fire exclusion and fire suppression also indirectly contributed to advancing secondary succession by preventing fire disturbances of a frequency and spatial scale that would favor the dominance of the early seral cover species.

Fire prevention and suppression still persist to this day. While well intentioned, such suppression compounds problems of advancing secondary succession and the extreme fire intolerance and high contagion of large expanses of dry forest. Small fires, if they had been allowed to burn in the early 20th century, or were

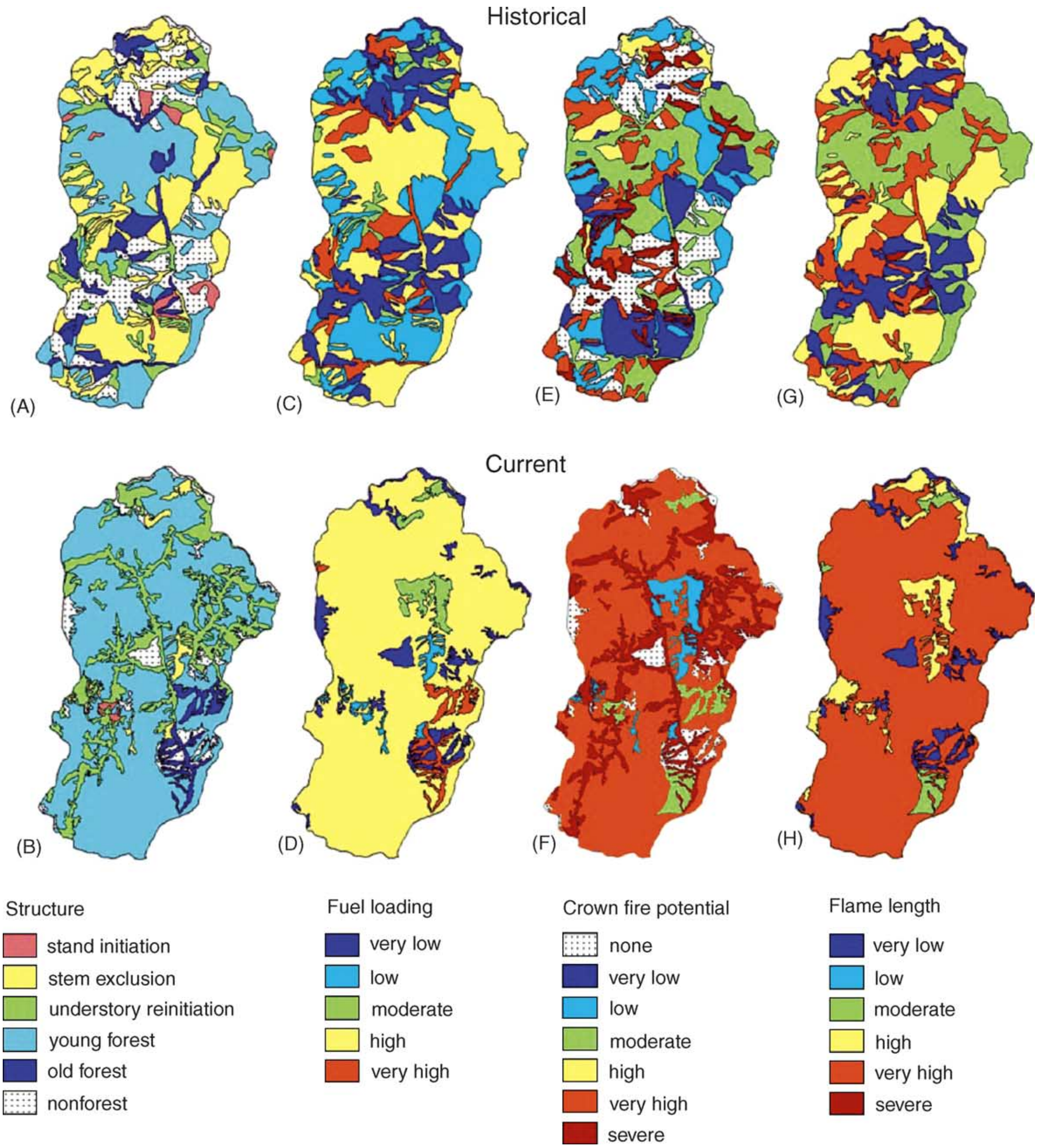

Fig. 3. Reconstructed historical (1900s) and current (1990s) maps of the Peavine Creek drainage, a dry forest subwatershed of the Lower Grand Ronde subbasin in the Blue Mountains province displaying historical and current structural classes (A and B), fuel loading (C and D), crown fire potential under average wildfire conditions (E and F), and flame length under average wildfire conditions (G and H), respectively (from Hessburg and Agee, 2003). Structural class abbreviations are: si, stand initiation; se, stem exclusion (both open and closed canopy conditions); ur, understory reinitiation; yfms, young multi-story forest; of, old multi-story and single story forest; nf, non-forest. Fuel loading classes are: very low < 22.5 Mg/ha; low = 22.5–44.9 Mg/ha; moderate = 45–56.1 Mg/ha; high = 56.2–67.3 Mg/ha; very high > 67.3 Mg/ha. Crown fire potential classes were a relativized index. Flame length classes were: very low < 0.6 m; low = 0.7–1.2 m; moderate = 1.3–1.8 m; high = 1.9–2.4 m; very high = 2.5–3.4 m; severe > 3.4 m.

intentionally lit, would have broken up the dry forest, thereby reducing the size of the area influenced by uncontrolled wildfires in the modern era. By virtue of their political and social inertia, it is unlikely that current fire prevention and suppression strategies will change in a substantive way in the short term.

3.1. Ecologically important changes

What were some of the most important changes to historical dry forest landscapes? The unintended chain of settlement and management influences produced ecologically unprecedented changes in dry forest vegetation and its susceptibility to previously more benign natural disturbances (Table 1). For example, absence of the frequent surface fires and influenced by grazing (Wissmar et al., 1994a), dry forests encroached on nearby grasslands, shrublands, and meadows (Fig. 4). Open stands of ponderosa pine or pine mixed with Douglas-fir developed dense understories of Douglas-fir, or Douglas-fir mixed with grand or white fir (Fig. 5). Old forest patches were selectively logged or clearcut leading to reduced old forest area and increased fragmentation of the remaining patches (Fig. 6).

Except for the first quarter of the 20th century, most wildfires were suppressed, when fires were smaller than 4 ha (Agee, 1993, 1998). While seldom discussed, this led to significantly reduced area and connectivity of newly regenerated forest (i.e., early seral shrub and grass areas in forest environmental settings, and areas of new forest stand initiation structure, Fig. 6, Hessburg et al., 1999a,c, 2000).

Fire-tolerant cover types, such as ponderosa pine and western larch were harvested and widely replaced by shade-tolerant Douglas-fir, grand fir, and white fir cover types, most of which are also fire-intolerant due to low crown bases, heat-trapping crowns, and thin bark (Fig. 7). The fire-tolerant cover types were harvested, because they contained the largest and oldest high grade trees, hence the term ''high grade'' logging. Before settlement, remnant large trees (>63.5 cm d.b.h.) of these same early seral species were broadly distributed in other forest cover types, and in forest structures that would not be considered old forest, under most definitions (Hessburg et al., 1999a, 2000). These scattered large trees occurred as a lasting remnant after stand replacement disturbances providing an important structural and habitat legacy that would typically last for many centuries, first as a standing live tree, then as a snag, and ultimately as large down wood. These live remnant large trees were widely eliminated by selection cutting (Fig. 8).

Table 1
Key changes in the historical dry forest landscapes of the Inland West brought about by settlement and management influences (from Hessburg et al., 2000; Hessburg and Agee, 2003)

Change	Effect
(1) Reduced grassland and shrubland area in forest potential vegetation settings, and expanded woodland and forest area (Fig. 4)	(1) Increased homogeneity of the landscape vegetation and fuels mosaic
(2) Reduced old and new forest area and connectivity (Fig. 6)	(2) Increased homogeneity of the landscape vegetation and fuels mosaic, reduced spatial isolation of areas prone to high-severity fires
(3) Reduced area of fire-tolerant forest cover types, especially ponderosa pine and western larch, and concordant increased area of fire-intolerant forest cover types, especially Douglas-fir, grand fir, and white fir (Fig. 7)*	(3) Reduced fire tolerance, which favored: thin-barked species, foliage that trapped more heat, and species that typically have lower crown bases, thereby maintaining continuity with fuel ladders
(4) Loss of grass and shrub understories, and the concordant addition of shade-tolerant conifer understories*	(4) Reduced likelihood of low-severity fires with increasing flame length, fireline intensity, rate of spread, increased fuel ladders and likelihood of crown fires
(5) Reduced forest area with large, early seral trees (Fig. 8)*	(5) Reduced fire tolerance (same as 3 above)
(6) Increased tree canopy cover, and canopy layers*	(6) Increased fuel ladders, potential flame lengths, fireline intensity, rate of spread, and likelihood of crown fires
(7) Increased young multi-story forest area (Fig. 9)*	(7) Increased landscape homogeneity, reduced fire tolerance, increased fuel ladders, potential flame lengths, fireline intensity, rate of spread, and likelihood of crown fires

The asterisk (*) indicates a strong correlation with current severe fire behavior.

3.2. Changes in disturbance processes

There were also many changes in disturbance processes that were directly related to changes in forest landscape attributes (Hessburg et al., 1999a, 2000; Hessburg and Agee, 2003) (Fig. 9). These included: (1) elevated fuel loadings and increased connectivity of high fuel loading; (2) increased potential for running crowning fires; (3) increased vulnerability to many insect and disease disturbances of fire-intolerant tree species; (4) increased likelihood of severe fire behavior in forest stands or patches with respect to flame length, rate of spread, and fireline intensity; (5) increased contagion or spatial aggregation of vulnerability to severe fire and insect and disease disturbances (e.g., Figs. 3 and 10). Table 2

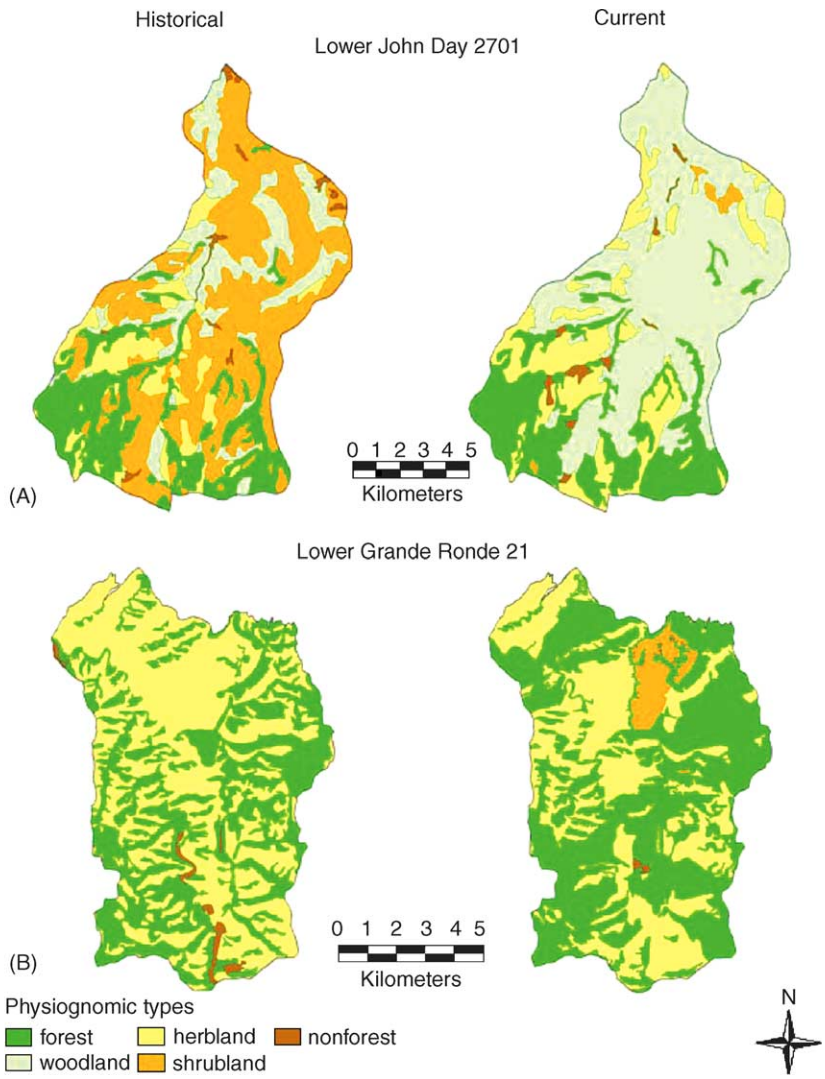

Fig. 4. Reconstructed historical (1900s) and current (1990s) maps of forest and woodland cover type changes in (A) subwatershed 2701 in the Lower John Day subbasin of the Columbia Plateau province and (B) subwatershed 21 in the Lower Grande Ronde subbasin of the Blue Mountains province (from Hessburg and Agee, 2003). Photograph (C) shows active encroachment of ponderosa pine (foreground) on a forest meadow formerly maintained in grass and herb cover by frequent low-intensity fires.

Fig. 4. (*Continued*).

Table 2
Transitions from historical (ca. 1800) to current fire-severity in forested environments of the Interior Columbia River Basin (adapted from Hann et al., 1997, and Quigley et al., 1996)

Historical to current transitions		
Fire-severity class transition (historical to current)	Percentage of historical forested area affected	Hectares of historical forest affected
Low[a] → low	17.8	5,067,100
Low → mixed	16.1	4,557,500
Low → high	11.1	3,150,900
Subtotal (historical)	45.0	12,775,500
Mixed → low	0.8	216,900
Mixed → mixed	11.5	3,250,800
Mixed → high	18.9	5,358,400
Subtotal (historical)	31.1	8,826,100
High → low	2.4	681,900
High → mixed	4.3	1,224,500
High → high	17.2	4,879,900
Subtotal (historical)	23.9	6,786,300
Total	100.0	28,387,900

Historical and current fire-severity base maps are available at http://www.icbemp.gov.

[a] Fire-severity classes used in Hann et al. (1997) were: ‘‘non-lethal’’, which is represented by ‘‘low’’, above; ‘‘mixed’’, which is represented by ‘‘mixed’’ and ‘‘lethal’’, which is represented by ‘‘high’’. Hann et al. (1997) defined lethal fire-severity as ‘‘stand replacement fires [that] leave less than 20% of the basal area or less than 10% of the overstory vegetation [cover] that was living prior to the fire.’’ The latter clause applied to lethal fires of shrub and herb physiognomies that are not reported here. Hann et al. (1997) defined non-lethal fire-severity as ‘‘fires [that] leave more than 70% of the basal area or more than 90% of the canopy cover that was living prior to the fire.’’ The latter clause applied to non-lethal fires of shrub and herb physiognomies that are not reported here. They defined mixed fire-severity to ‘‘include all fires of intermediate effects’’, that is, intermediate to lethal and non-lethal fire-severity. Intermediate effects would come from fires having both lethal and non-lethal components, and fires would leave 20–70% of the basal area or 10–90% of the overstory vegetation cover. These definitions are consistent with those used in this paper.

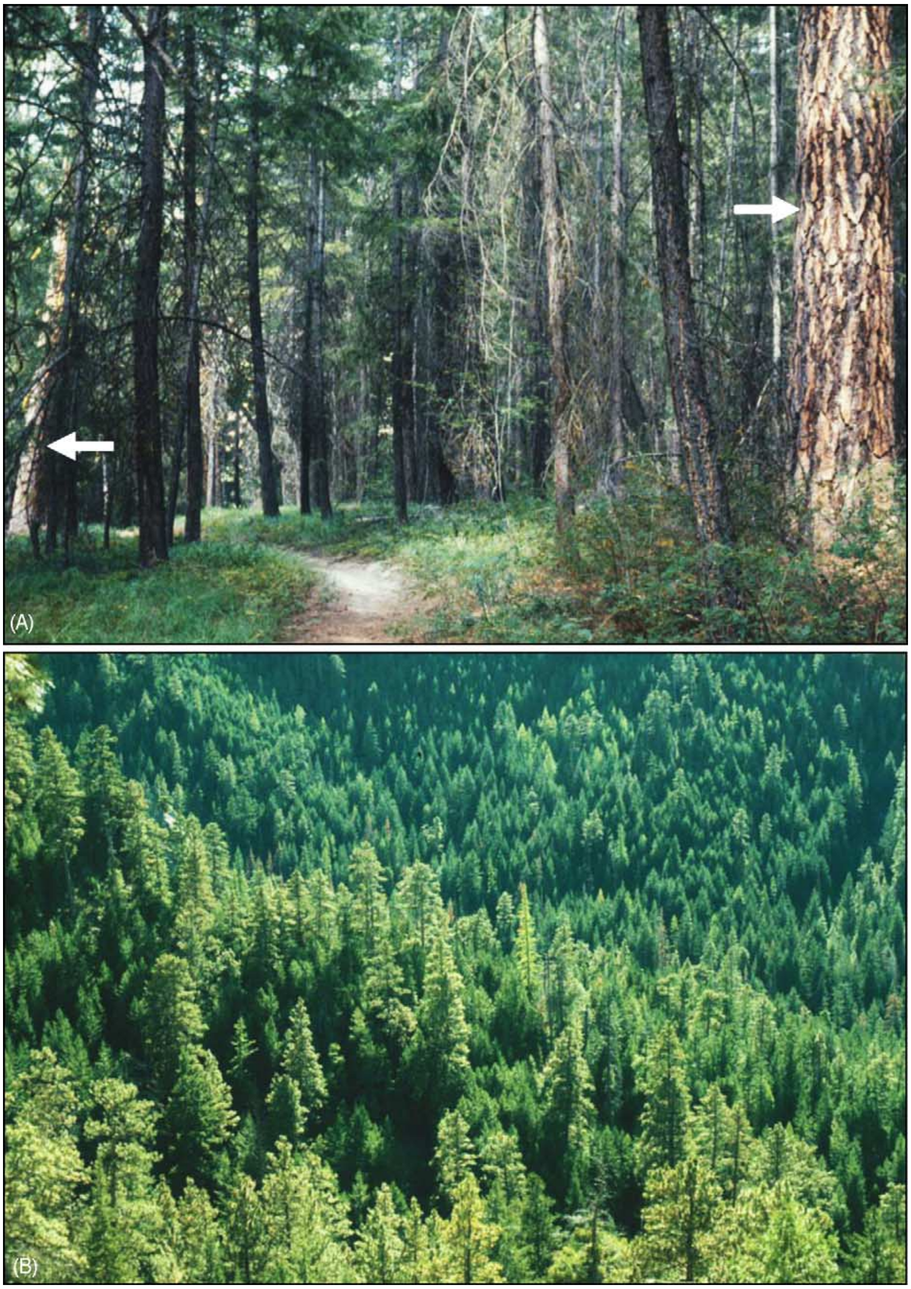

Fig. 5. Example of how an open ponderosa pine stand appears after heavy selection cutting and decades of in-growth by Douglas-fir (A). White arrows show two remaining large ponderosa pine after selection cutting. From above (B), one can readily see the filling in of the landscape with shade-tolerant, pole- to medium-sized trees.

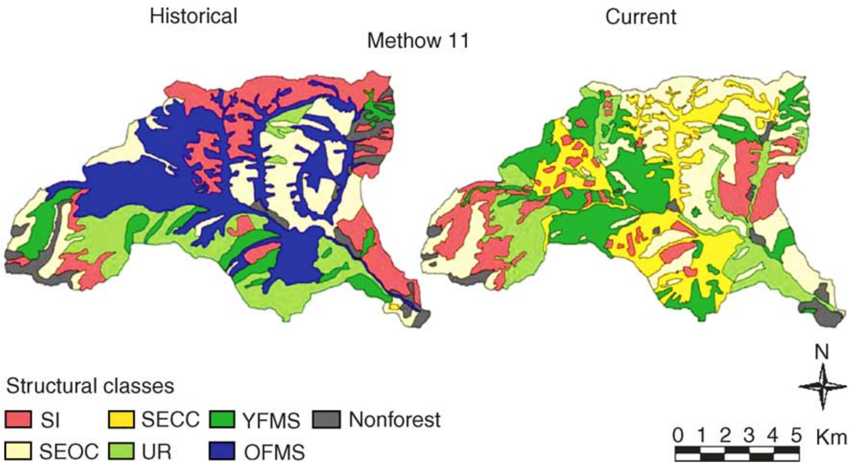

Fig. 6. Reconstructed historical (1900s) and current (1990s) maps of the Libby Creek drainage (Methow 11), a dry forest subwatershed of Methow subbasin in the Northern Glaciated Mountains province displaying structural classes (from Hessburg et al., 1999a). Structural classes are: si, stand initiation; seoc, stem exclusion, open canopy; secc, stem exclusion, closed canopy; ur, understory reinitiation; yfms, young multi-story forest; ofms, old forest, multi-story, and non-forest.

shows the transitions in fire-severity from historical (ca. 1800) to existing conditions in the Interior Columbia River Basin (adapted from Hann et al., 1997, Quigley et al., 1996). Table 3 summarizes the change in hectares and change from historical percentage area of the Interior Columbia Basin by fire-severity class. By examining both tables, one may quickly see that in the Inland Northwest, the scale of fire regime shifting has been enormous. There has been a net reduction of low-severity regime area in the Basin of 53%, the area of mixed-severity fire is roughly the same (although the area has moved from characteristic to uncharacteristic sites, Table 2), and the area of high-severity fire has nearly doubled (Table 3).

Surface fuels became elevated primarily in two ways: fuels created by repeated selection cutting were often left untreated, and stands that resulted from selection cutting became densely stocked, layered, and composed of tree species that were vulnerable to a broad array of tree-killing insects and pathogens. The dense stocking and layering was due to the ongoing regeneration and release of shade-tolerant trees in the gaps left by each successive cutting, and it created highly effective fuel ladders. Decades of ongoing forest insect and disease mortality also contributed to present-day fuel loads. High connectivity of expanded fuel loads occurred, because frequently surface fired historical ponderosa pine forests tended to be spatially well connected. Repeated selection cutting of these same patterns left well-connected fuel beds.

The likelihood of running crown fires increased, because selection cutting and fire exclusion favored increased tree cover, canopy layering, and tree density. The increased tree cover consisted of shade-tolerant trees with low crown bases and heat-trapping foliage. The increase in canopy layers ensured adequate fuel

Table 3
Change in actual and percentage area of fire-severity classes from the historical (ca. 1800) to the existing condition in the Interior Columbia River Basin (adapted from Hann et al., 1997, and Quigley et al., 1996)

Fire-severity class	Historical area (ha)	Current area (ha)	Change in area (ha)	Percentage of historical area
Low[a]	12,775,500	5,965,900	−6,809,600	47
Mixed	8,826,100	9,032,800	206,700	102
High	6,786,300	13,389,200	6,602,900	197
Total	28,387,900	28,387,900		

Historical and current fire-severity base maps are available at http://www.icbemp.gov.
[a] See footnote (a) in Table 2.

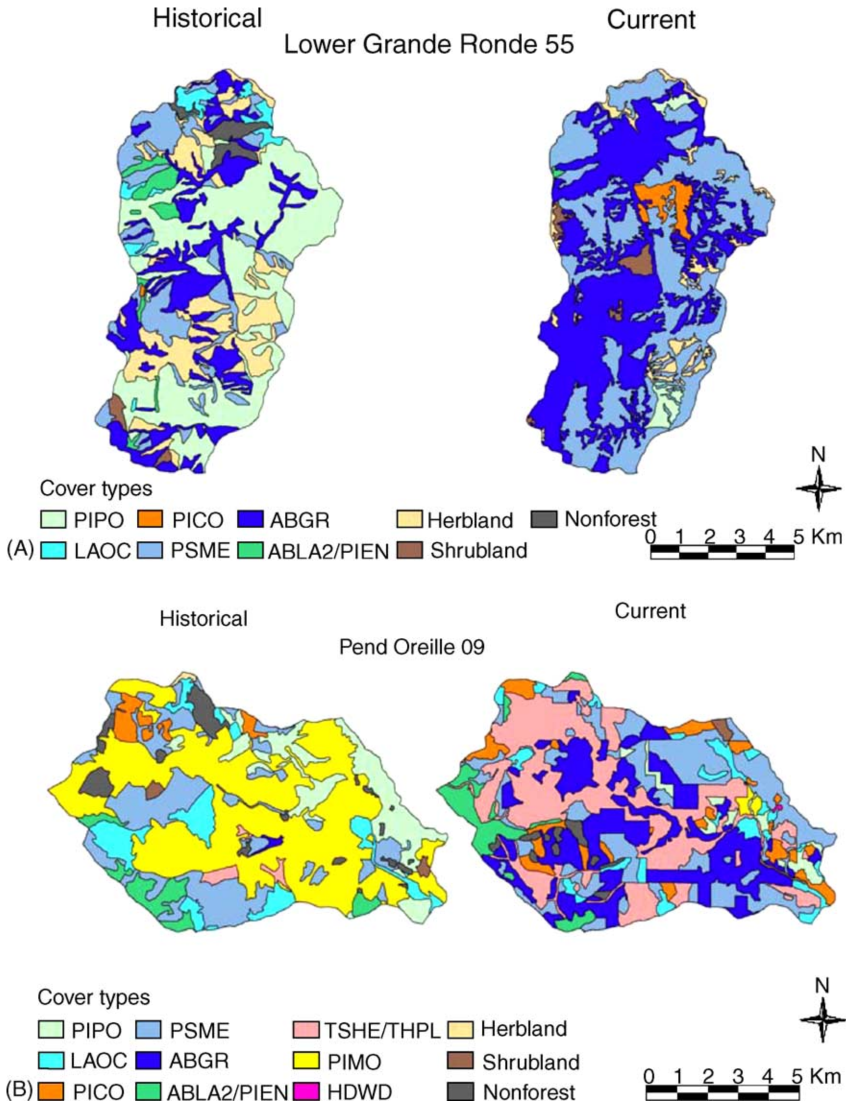

Fig. 7. Reconstructed historical (1900s) and current (1990s) maps of forest cover types in: (A) Peavine Creek drainage, a dry forest subwatershed of the Lower Grand Ronde subbasin in the Blue Mountains province and (B) subwatershed 09 in the Pend Oreille subbasin of the Northern Glaciated Mountains province (from Hessburg et al., 1999a). Cover type abbreviations are: PIPO, ponderosa pine; LAOC, western larch; PICO, lodgepole pine; PSME, Douglas-fir; ABGR, grand fir; ABLA2/PIEN, subalpine fir and Engelmann spruce; TSHE/THPL, western hemlock and western redcedar; PIMO, western white pine; HDWD, hardwoods.

ladders for conveying fire from the forest floor to the overstory canopy. Similarly, fireline intensity, fire rate of spread, and flame length attributes increased in severity because of the expanded coverage of highly combustible surface fuels, and the increased layering and spatial extent of understory fire-intolerant trees (Huff et al., 1995).

4. Implications for future management

Much of the increased area in present-day mixed and high-severity fire regimes developed in the dry forests of the Douglas-fir, grand fir, and white fir zones, where ponderosa pine was the primary early seral species. As an alternative to blanket prescriptions

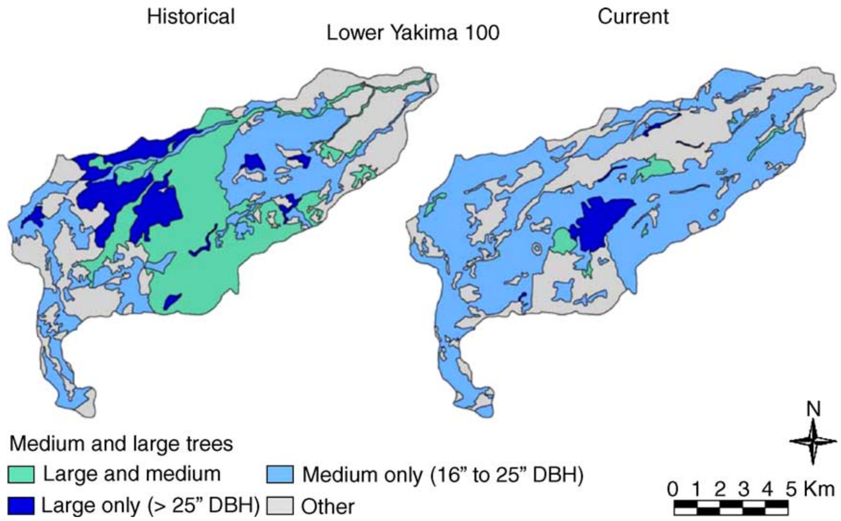

Fig. 8. Reconstructed historical (1900s) and current (1990s) maps showing forest patches with remnant large trees in Lower Yakima 100, a dry forest subwatershed of the Lower Yakima subbasin in the Northern Cascade Mountains province (from Hessburg et al., 1999a).

that simply reduce fuels and thin dry forests, we suggest a more ecological approach is warranted; one that would restore more natural spatial and temporal patterns of dry forest structure, composition, snags, and down wood (e.g., Hessburg et al., 1999b,c). It was the broad natural (including aboriginal inputs) range of spatial patterns and temporal variation in those patterns of living and dead forest, grassland, shrubland, and woodland structure that historically supported the native species and processes that were inherited at the time of settlement (Lindenmayer and Franklin, 2003; Thompson and Harestad, 2004; Turner and Romme, 1994).

The suggested approach would involve reconstructing, using empirical, simulation, or some combination of methods, a representative portrayal of the broad natural variation in vegetation patterns that existed prior to management and settlement (e.g., see Hessburg et al., 2004; Keane et al., 2002; Reynolds and Hessburg, 2004). Landscape management of the dry forests based on the knowledge of natural patterns of structure and composition would inevitably lead to more characteristic fire regimes, and widespread reductions in dead wood in the form of snags and down wood, as levels of these components are currently elevated in present-day dry forests. These more ecologically motivated management actions would enable the restoration of vegetation pattern and disturbance process interactions that support the fire regimes, forest resources and values, and other ecosystem processes that society is apparently interested in (H.R., 1904, U.S. Government, 2003).

An evaluation of changes in fire regimes of the Interior Columbia River basin and the Central Sierra Nevada ecosystems over the last 200 years reveals that many dry forest landscapes will be burned by mixed or high-severity fires in the coming years (Hann et al., 1997; SNEP, 1996). At watershed to subregional scales, most dry forest landscapes in their current condition are set up for large, severe fire events (Figs. 3 and 11). It is apparent that the choice available to decision makers and citizens alike is the type of fire and smoke they will choose—that associated with wildfires or that which is actively prescribed and managed.

The effects of wildland fires of the last 10–15 years alone suggest that some sort of active management is needed to increase the likelihood that managers can produce the desired amount, timing, and spatial arrangement of treatment effects on current dry forest landscapes. To date, wildland fires alone have not created ecological outcomes that are desired by society or that are consistent with natural ecosystem functioning. Even with active management, there is no guarantee that managers will always achieve desired results, because environments and ecosystem structures and processes all have strongly associated stochastic

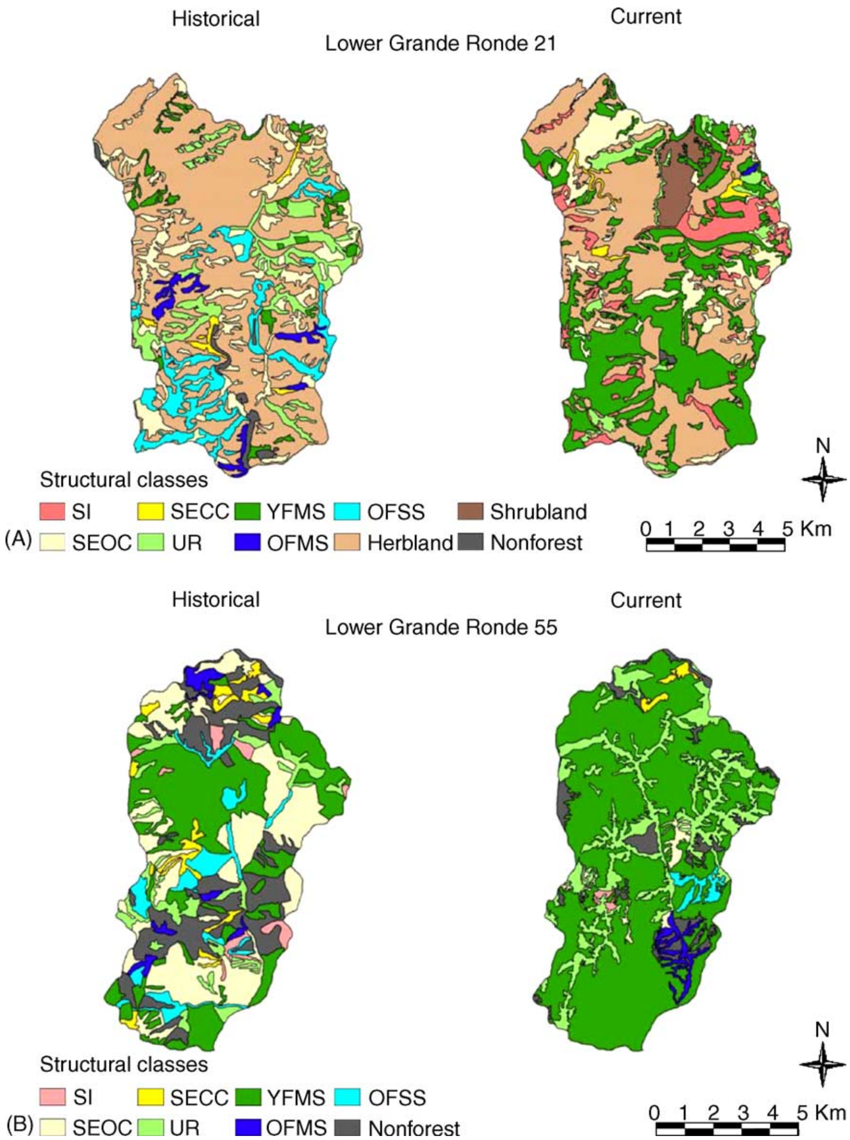

Fig. 9. Reconstructed historical (1900s) and current (1990s) maps of: (A) Lower Grande Ronde 21 and (B) the Peavine Creek drainage (Lower Grande Ronde 55), dry forest subwatersheds of the Grande Ronde subbasin in the Blue Mountains province displaying structural classes (from Hessburg et al., 1999a). Structural classes are: si, stand initiation; seoc, stem exclusion, open canopy; secc, stem exclusion, closed canopy; ur, understory reinitiation; yfms, young multi-story forest; ofms, old forest, multi-story; ofss, old forest, single story, and non-forest.

features (Lewontin, 1966; Reckhow, 1994; Walters and Hollings, 1990).

It is likely that various combinations of pre-commercial and commercial thinning, mechanical slash treatments, and prescribed burning will be used at a stand-level to produce the desired effects on fuels and vegetation, because that is what managers are skilled at doing. Furthermore, several sequential applications of prescribed burning may be needed, because the first burn treatments may temporarily increase surface fuels (Agee et al., 2000; Agee, 2002).

At a landscape-level, recent fires and those of the last several centuries have taught us that managers may have more success in planning and managing for

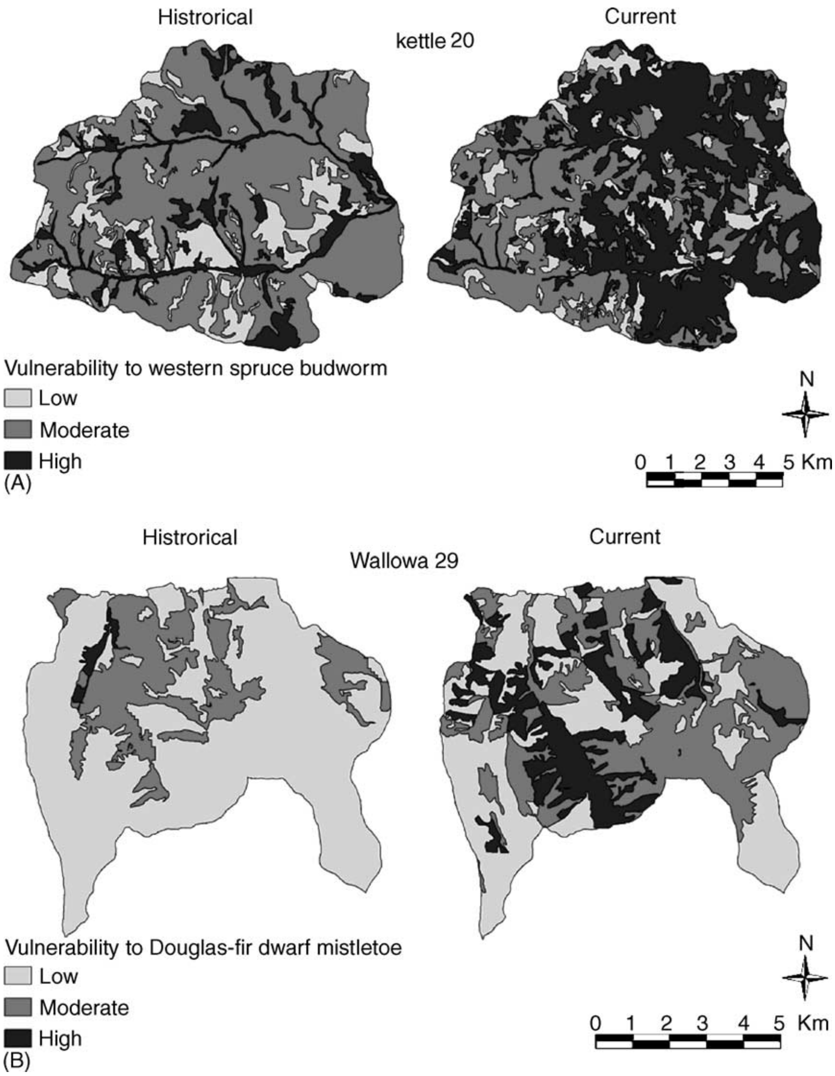

Fig. 10. Reconstructed historical (1900s) and current (1990s) maps of: (A) Kettle 20 and (B) Wallowa 29, dry forest subwatersheds of the Kettle and Wallowa subbasins, in the Northern Glaciated Mountains and Blue Mountains provinces, respectively. Maps show changes in vulnerability to western spruce budworm (A) and Douglas-fir dwarf mistletoe (B). Changes in vulnerability are related to changes in vegetation structure and composition (see Section 2 for estimating vulnerabilities in Hessburg et al., 1999a,d).

dynamic landscapes rather than predetermined static or reserved conditions. Inability to control chance events, climatic regime shifting, and the stochastic features of environments and ecosystem processes compel one to manage for desired future dynamics rather than desired future conditions. That is, one will likely be more successful managing forests within a range of conditions, rather than in specific states.

5. Planning priorities for altering fuels and fire behavior

When considering treatment priorities in restoration planning, several simple guidelines emerge. The stylized concept of historical fire regime areas can be used as a starting basis or template (Cissel et al., 1999; Agee, 2003), with local alterations for overlapping

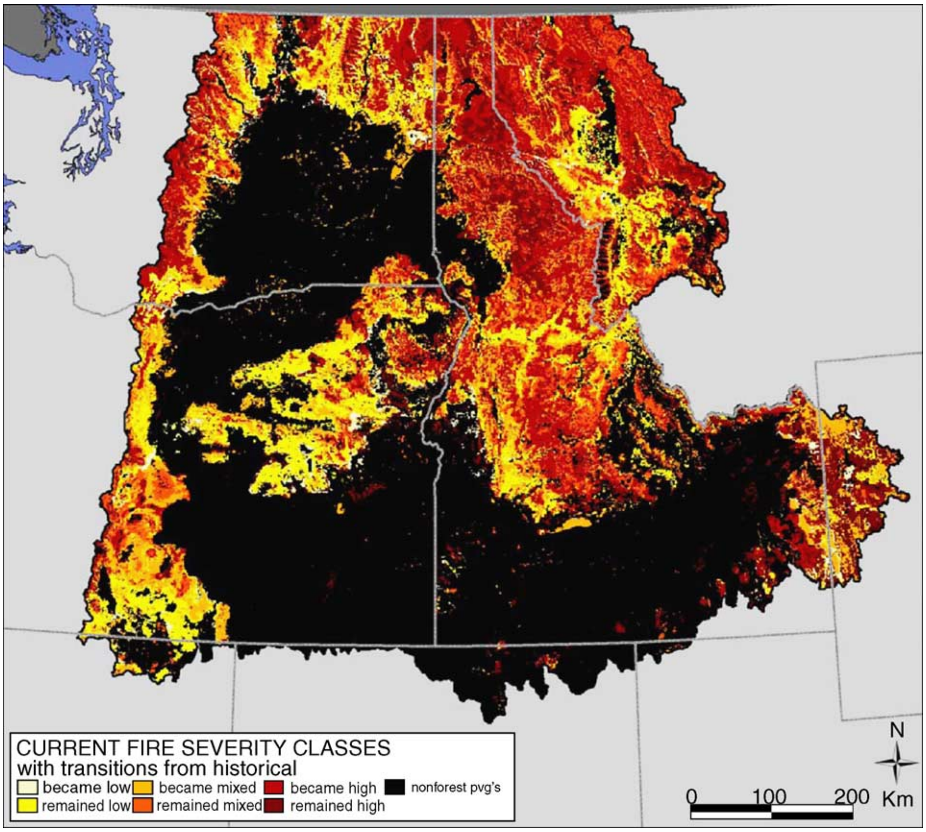

Fig. 11. Broadscale (1-km^2 pixels) map of current fire-severity classes with transitions from historical in the Interior Columbia Basin and vicinity (adapted from Hann et al., 1997). High severity = stand-replacing fire that kills > 70% of the overstory tree basal area; low severity = fire that kills < 20% of the overstory tree basal area; mixed severity = fire that kills 20–70% of the overstory tree basal area.

regime areas as is appropriate. We describe current priorities for restoration planning in terms of the land area of the historical fire regime, because it is spatially the most accurate and succinct way of describing the actual places that could most benefit from restoration. The problem is that the current area of the high and mixed-severity fire regimes is a mixture of places that have changed very little, in terms of fire regime parameters, and of places that have changed a great deal. Tables 2 and 3 illustrate this point.

5.1. Historical low-severity regime areas

Land areas that historically supported low-severity surface fires are, from an ecological perspective, the highest priority for fuel reduction and vegetation pattern restoration treatments (Hessburg and Agee, 2003). These are the landscapes dominated by the dry forests. The structure, composition, and spatial patterns of historical dry forest landscapes tolerated fires well, and these conditions and their variability can be used as a basis for restoring more characteristic fires. The principles of fire-tolerant or fire-resilient (‘‘FireSafe’’) forests of Agee and others (Agee, 2002; Agee et al., 2000; Hessburg and Agee, 2003) can be applied through prescribed burning and with the use of silviculture at the stand-level, and these are relatively well-accepted principles. But as one scales up from individual patches to landscapes, the priority of where to treat and how much to treat becomes much less clear, and it will be essential to define spatial and temporal patterns in upcoming management decisions.

One of the challenges faced by research and management is to quantify the spatial patterns, temporal variation, and amounts of vegetation and fuel treatments that are needed to make current landscapes more fire-resilient and at the same time supportive of native species, their habitats, and related processes through time. Modeling efforts are underway to begin to address the fire resilience component of this need (e.g., see Finney, 2001), but practical landscape experiments are also needed on the ground to evaluate and authenticate model outputs, and understand controlling factors and variation. Reference conditions provide useful insights into the ranges of vegetation and fuels spatial patterns that have supported low-severity fire regimes and native species habitats in the past (Agee, 2003; Fulé et al., 1997; Hessburg et al., 1999b,c; Landres et al., 1999; Swetnam et al., 1999).

5.2. Historical mixed-severity regime areas

Also from an ecological perspective, land areas that historically supported mixed-severity fires are the second priority for fuel reduction and vegetation pattern restoration treatments (Hessburg and Agee, 2003). These are the landscapes dominated by both dry and mesic mixed conifer forests. In the past, the importance of these mixed-severity landscapes has been understated, because managers and scientists, trying to convey a simple and concise story about change in dry forests, focused their attention on forests that were burned by frequent surface fires. But it is increasingly clear that throughout the Inland Northwest, many mixed conifer forests were burned by both low- and mixed-severity fires (Agee, 1993, 1998; Hessburg and Agee, 2003). This variation in fire regime provided additional variation in resulting patterns of living and dead vegetation, habitats, and related ecological processes.

At the present time, the least is known about restoring patterns and functionality to forests of the historical mixed-severity regime, because mixed-severity fires represent the broadest range of fire effects. Reasonable hypotheses can be derived from reconstructions of historical landscapes about how much area to treat, what kind of mosaic to leave, and which treatments may be functional fire surrogates, but they should be tested. Again, estimates of historical range and variation in vegetation and fuels patterns can provide valuable insights into the spatial patterns and temporal variation in patterns that supported the full range of mixed-severity fires and their effects.

6. Landscape restoration considerations

6.1. Buying time for forest restoration

It has taken more than a century for the conditions to develop that exist today, and at best, it may take several decades to make substantive and widespread improvements. Currently, dry forest landscapes of the Inland Northwest exhibit high landscape connectivity of conditions that support large and severe fires. To buy time for more thoughtful and carefully planned forest restoration, it makes sense to begin restoration by designing and developing networks of shaded fuelbreaks throughout the dry forests (Agee et al., 2000; Arno and Allison-Bunnell, 2002). These networks would provide the advantage of breaking large fire-prone landscapes into smaller and more manageable pieces, which would be of significant benefit, both for restoration and fire suppression efforts. It would be useful to position fuelbreaks adjacent to existing roads so that the fuelbreaks could be revisited at regular intervals, and re-treated to maintain a widely scattered cover of medium and large-sized ponderosa pine cover (where available) with only light fuels.

Fuelbreaks could also make wise use of topography so that fire managers could use them as boundaries to implement broader fuel treatments, and fire fighters could use them to burn up against in fire suppression. For example, ridgetop topographies and valley bottoms adjacent to major stream courses and stream confluences (but well outside of the riparian zone) may be suitable locations (Agee et al., 2000; Camp et al., 1997). The intent with shaded fuelbreaks is to allow fire to burn through the fuelbreak area, but to constrain fire behavior to that of a low-intensity surface fire (Agee et al., 2000; Anderson and Brown, 1988; Schmidt and Wakimoto, 1988).

Many fuelbreaks can be created by thinning. To be effective, it is postulated that fuelbreaks should be at least 400-m wide to improve chances of inhibiting the spread of a running crown fire (Agee et al., 2000; Anderson and Brown, 1988; Finney, 2001). Wildfires can move through fuelbreaks, but usually as surface fires, except when fire weather and fire behavior are extreme. In these cases, even broad fuelbreaks can be jumped for great distances by blowing fire brands and spotting fire. Thinning should remove the smaller trees which make up the fuel ladders. Crown fire hazard can be reduced by removing enough of the taller trees to reduce forest crown cover to less than 35% and maintain a minimum open space between the crowns of the taller trees of at least 10 feet (Schmidt and Wakimoto, 1988). However, such removal can also encourage regeneration of shrubs and trees, such that the required maintenance interval shortens significantly. While removal of surface fuels and fuel ladders is important, treatment of the overstory involves more tradeoffs and may not always be part of a successful treatment strategy (Agee and Skinner, this volume). Post-treatment fuels may be piled and burned, or broadcast burned, where feasible to increase treatment efficacy, and fuelbreaks may be retreated often enough that only a low intensity surface fire could be supported by the fuelbed.

6.2. Improving fire resilience with silviculture and prescribed burning

Added to fuelbreaks, the fire resilience of dry forest landscapes can be improved by thinning from below, free thinning, applying shelterwood regeneration harvest with reserves, and by coupling prescribed burning treatments with silvicultural treatments over large areas (Agee et al., 2000; Graham et al., 1999, 2004). Initially, small drainages or subwatersheds could be considered for applying area-wide treatments (Arno and Allison-Bunnell, 2002). Usually, fuelbed and tree density conditions have changed the most on the southerly dry aspects, and these can be the first areas designated for treatment. By treating southerly aspects initially, large landscapes composed of contagious fuelbeds and dense and layered tree cover can be broken up in a highly intuitive manner.

The effectiveness of treatments, whether thinning or shelterwood regeneration harvests, may be enhanced by emphasizing opening up the canopy to relatively wide spacing, reducing canopy layering, removing of the most fire-intolerant species, usually grand and white fir, but also Douglas-fir, and removing the smaller size classes. These are the ordinary effects of surface fires. Arno and Allison-Bunnell (2002) suggest that historical surface fire regimes perpetuated ponderosa pine-dominated stands with 30–100 trees per acre. Harrod et al. (1999) suggest similar wide spacing. Trees that are left behind may typically be those that are the most vigorous, are largest, and have the thickest bark. Generally, ponderosa pine may be favored over other species in the mix of leave trees, but Douglas-fir, another natural component, can be left as well, especially where trees are large and thickly barked, vigorous, and well-spaced. In general, silvicultural systems that are applied to manage dry forest stands can be designed to reduce the potential for a running crown fire. Selection thinning and crown thinning (thinning from above) that maintain multiple layers typically may not reduce the risk of crown fires and tree torching nearly as much as low thinning, free thinning, and shelterwood harvests with reserves (Graham et al., 1999).

Obviously, if only a few of the best methods were applied everywhere to improve the fire resilience of Inland Northwest dry forests, the structure and composition of dry forests could become overly simplified, and biotic diversity and ecosystem functioning could be adversely impacted. No single thinning or management prescription, therefore, may be expected to achieve all objectives across any given landscape. By understanding the stand and landscape characteristics that affect prescribed or wildfire behavior, and that are necessary to support native species and processes, forest landscapes may be designed through the use of silviculture and prescribed burning that minimize the potential for severe fire effects or crown fire behavior.

6.3. Using historical spatio-temporal variation as a basis for laying out treatments

To a large extent, what drives uncharacteristic fires is pronounced change in landscape patterns of fuels, forest structure, and composition (Hessburg and Agee, 2003) coupled with ordinarily high potential for severe fire weather in the West (Agee, 1997, 1998; Franklin and Agee, 2003). Over long-time frames, fires, insect outbreaks, disease epidemics, and weather events historically created and maintained patterns of dry forest structure and composition that supported an exceptional variety of plant and animal species, and a host of critical processes. The interplay between patterns and processes created a metastable patch dynamic (Ahl and Allen, 1996; Allen and Starr, 1982; Wu and Loucks, 1995), a broad set of conditions that strongly tended to support surface fires over other kinds. Native species and processes are at best adapted to and at worst showed they could persist through these patch dynamics, and perhaps more importantly, native species and processes showed that they could persist within the context of the rates of change that were characteristic of dry forest ecosystems.

Characteristic patterns of forest structure and composition have been replaced by new and untested sets of patterns that resulted from an unintended chain of influences. Furthermore, rates of ecosystem and habitat change in the last two centuries appear to be much more rapid than was formerly the case. This may be particularly important, when considering native species viabilities in the long term, and it may explain why there are so many native plant and animal species currently listed as sensitive, threatened, or endangered.

Current patterns of dry forest landscapes primarily support mixed and high-severity fires, and these fires are occurring over much broader landscapes than was formerly the case. For example, patches of isolated stand replacement fire were common in historical dry forest landscapes, but today, entire landscapes are claimed by severe fires (Fig. 3). Furthermore, present-day large wildfires synchronize landscapes by creating very large patches with corresponding forest regeneration, species composition, structure, fuel beds, and size and age class distribution, thereby facilitating very large future wildfires.

To create fire regimes that are more predictable and more consistent with environmental settings under the current climatic regime, we suggest that landscape patterns of fuel, forest structure, and composition will need to be created that are characteristically associated with those regimes. We further suggest, that to improve assurances that native species and processes will persist, it will also be important to restore forest landscapes that reflect some semblance of the spatial and temporal variation in patterns that species evolved with. Dramatic and rapid departures in fire regimes and landscape patterns portend losses to species and uncharacteristic changes to vital processes.

6.4. *Dealing with the past while heading into the future*

Many dry forests may be in dire need of pattern restoration, but there is a legacy associated with past management actions, and restorative management treatments may not be possible in the short-term everywhere they may be needed. For example, the current code of federal laws requires that remaining strongholds for native anadromous and cold water fishes listed under the Endangered Species Act must be maintained as landscape patterns are restored (Lee et al., 1997). Hence, when considering both listed fish and landscape restoration needs, it may be important to prioritize and sequence landscape restoration needs at continental, regional, and subregional scales, and to consider a corresponding watershed hierarchy (see Rieman et al., 2000).

Similarly, timber harvests have so reduced the natural stores of large trees that it seems important to conserve many of the remaining populations for their biological and ecological values alone (Lindenmayer and Franklin, 2003; Franklin et al., 1981). Characteristic late-successional and old forests are in very short supply as is landscape area with remnant large trees (Hann et al., 1997; Hessburg et al., 1999a; SNEP, 1996). If motivated for ecological restoration, future thinning and prescribed burning, therefore, could primarily target removal of the small and medium-sized fire-intolerant trees. Such ecologically motivated restoration in the Inland Northwest could focus on managing for more and better connected areas of characteristic late-successional and old forests and areas with remnant large trees, and adding back these elements in characteristic patterns.

Large-scale restoration will be costly in many places. Widespread selection cutting of the largest and oldest ponderosa pine and Douglas-fir has reduced the economic opportunity that might have otherwise been associated with dry forest restoration. Large-scale and long-term investment in restoration treatments will be needed in some landscapes to return a semblance of characteristic patterns and processes.

7. Conclusions

Dry forests of the present-day no longer appear or function as they once did. Current patterns of forest structure and composition do not resemble even recent historical conditions, neither do they represent what we would expect to see under or more natural or characteristic disturbance regimes and the current climate. There is little evidence that current patterns are sustainable and this has important ecological consequences.

Large landscapes are increasingly homogeneous in their composition and structure, and the regional landscape is set up for severe, large fire and insect disturbance events. Among ecologists, there is a high degree of concern about how future dry forests will develop and what they will become, if fires continue to be large and severe. Restoration of forest patterns with high ecological functionality will not be easy and solutions will not be clear cut. Furthermore, the possibility that certain changes in ecosystems may be chronic (i.e., irreversible, from a practical standpoint) may make full restoration of patterns and processes impossible.

7.1. What can be done?

Much can be done to make progress in restoring pattern and process of dry forest ecosystems. Managers may make great headway in modifying the structure and composition of dry forests using various combinations of commercial and pre-commercial thinning and prescribed burning treatments. They can favor fire-tolerant species like ponderosa pine and western larch in their treatments and discriminate against fire-intolerant species, such as Douglas-fir, grand fir, and white fir. They can favor leaving the large and very large trees, and remove the small and medium-sized trees. Managers can treat large areas, but first they must be mindful of the consequences of the past and shape their future courses of action with that understanding. These considerations can influence, where they treat, how they treat, and when they will treat.

Managers can study the natural vegetation and fire patterns of areas they intend to manage and other areas like it, and apply their knowledge of those patterns to their management (Agee, 2003; Hessburg et al., 1999b,c). And they can continue preventing and suppressing fires, at least until more characteristic patterns of vegetation and fire regime are restored. Finally, managers may need to apply relatively frequent follow up treatments to already treated stands and landscapes to maintain their fire-tolerance. Ideally, to control costs and provide desirable fire effects, this would include a program of regular underburning.

The current planning environment for forest management in the Inland West is exceedingly complex in terms of both issues and information available for issue resolution. From recent bioregional assessment projects (e.g., Columbia River Basin and Sierra Nevada Ecosystem Projects), we now know more about existing forest and rangeland conditions, and the state of ecosystems and their inhabitants than ever before. Restoration and management planning requires adequate incorporation and integration of this information to address interactions between landscape spatial and temporal patterns of vegetation, habitats, fuels, and potential fire behavior. Planning further should consider the relations between these conditions and a host of issues surrounding terrestrial habitats and associated aquatic ecosystem conditions. Before management alternatives can be selected and implemented, they must be adequately evaluated for their effects on many terrestrial and aquatic ecosystem components.

In light of the above planning requisites, we believe there is a critical need for modeling and simulation tools that not only incorporate multidimensional data and examine the effects of management activities on key facets of ecosystems, but that also serve as platforms for integrating and evaluating the simultaneous effects of alternatives on a host of ecosystem conditions. While many modeling and simulation, scenario planning and decision support tools have been developed in relative isolation in the last decade (see discussion in Hessburg and Agee, 2003), their use in planning is poorly established. Increased emphasis and investment in the further development, acceptance and application of these tools in the planning environment would seem extremely important in the future.

A major challenge for restoring pattern and process in Interior Northwest forests relates to the complex and sometimes contentious sociopolitical context within which forest management is planned and implemented. It may be unlikely that effective long-term management options can be developed through social consensus due to variation in values among different sectors of society, fluctuating social values over time, and a tendency of consensus-based options to favor little change over the status quo (McCool et al., 1997). From a public policy standpoint, however, the recently enacted public law H.R., 1904, also known as the Healthy Forests Restoration Act (HFRA, U.S. Government, 2003) has emerged as a major influence on future forest restoration planning and implementation.

HFRA will result in vastly expanded efforts by land managers in coming years in support of fuel reduction and forest-thinning projects within the dry forests of the Inland West, concurrent with limited judicial review. Recognizing that re-establishment of healthy forests is a dynamic, long-term (decades to centuries) process (Hessburg and Agee, 2003), we believe that the effectiveness of HFRA activities may be greatly enhanced, if due attention is given to ongoing monitoring of both the ecological effects of treatments/projects, and their effectiveness in meeting prescribed objectives. In further recognition that restoration is a long-term investment, addressing and emphasizing the needs and long-term financial support for repeated (over time) thinning, fuel reduction and other vegetation management efforts would also seem extremely important in HFRA implementation. Lastly, we believe that consideration of two critically important premises could greatly motivate and improve the implementation of HFRA: (1) an upfront admission that neither scientists nor managers have all, or even many, of the answers to questions of how to restore more healthy forest conditions and (2) acknowledgement that once restored, dry forests should not only support the fire regime of interest, but also viable populations of native species in functional habitat networks across space and through time. If embraced, the first premise would call for an experimental, adaptive approach to management (sensu Bormann et al., 1999; Franklin, 1993; Thomas and Huke, 1996), operationally uniting research and management in planning, implementation and monitoring of treatments. The second premise would call for formulating fuels reduction, thinning and other vegetation treatments in the context of improved landscape functionality for species and vital ecosystem processes.

Acknowledgments

This research was supported in part by the Managing Disturbances Program of the Pacific Northwest Research Station (RWU-4577), USDA Forest Service. Ed Depuit and three anonymous reviewers provided helpful reviews of an earlier draft. We thank Brion Salter for his excellent assistance in developing the figures and tables.

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