



Landscape-level habitat supply modelling to develop and evaluate management practices that maintain diverse forest values in a dry forest ecosystem in southern British Columbia

Walt Klenner^{*}, Russ Walton

British Columbia Ministry of Forests and Range, Southern Interior Forest Region, 515 Columbia Street, Kamloops, British Columbia, Canada V2C 2T7

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ABSTRACT

We used the TELSAs forest landscape model to examine the long-term consequences of applying different forest management scenarios on indicators of wildlife habitat, understory productivity, crown fuel hazard, timber yield and treatment costs. The study area was a dry forest dominated landscape of approximately 100 000 ha near Kamloops, BC where fire safe communities, timber harvest and the maintenance of biodiversity are important social expectations. Conventional partial cut harvesting treatments maintained timber yields but strongly reduced the area supporting high snag densities, provided low levels of understory productivity and diminished the area that was undisturbed for extended periods. In contrast, the application of extensive fuel management treatments designed to prevent extreme fire behaviour diminished timber flow, but, as with partial cut harvesting, reduced the area with high snag densities and extended periods without disturbance. The application and maintenance of extensive fuel management treatments was also the most costly treatment. Alternatives, such as strategically placed fuel management treatments in critical zones around communities or areas with high investments, diminished costs and somewhat mitigated the reduction in snag densities. More complex land use zoning approaches that incorporate specific practices (e.g. protected areas) to maintain high quality wildlife habitat, and treatments that promote productive understories, appear to hold promise if multiple forest values and objectives are to be maintained in the long-term. We conclude that forest landscape modelling can and should be used to assess the consequences of current or new management initiatives prior to their application to ensure expectations regarding costs and achievement of objectives are realistic.

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1. Introduction

Over the last two decades forest management planning has evolved to address increasingly complex resource allocation issues. Initially, maintaining a sustainable timber supply was a key objective, but concerns about wildlife habitat (Baskett and Jordan, 1991; Hansen et al., 1993), visual resources (Sheppard and Harshaw, 2001) and evolving issues such as climate change (He et al., 2002) have created the need to consider multiple objectives in forest management planning. Management (Mladenoff et al., 1993; Wallin et al., 1994; Gustafson and Crow, 1996; Radeloff et al., 2006; Zollner et al., 2008), natural disturbances (Pickett and White, 1985) and interactions between natural and managed disturbances (Klenner et al., 2000; Fall et al., 2004; Shifley et al., 2006) shape landscape pattern and associated ecological processes.

Landscape ecology has contributed much to the understanding of pattern and forest ecological process across a range of spatial scales (Leitão and Ahern, 2002) and, together with the development of forest disturbance and succession models (e.g. Kurz et al., 2000; Schneider et al., 2003; Mladenoff, 2004; He, 2008), has facilitated the evaluation of management options in a comparative, multiple scenario framework. Although the application and utility of forest landscape models in a research environment has been clearly established, their application in the development and testing of operational forest management practices to achieve multiple resource objectives is not well established.

The dry forests of southern British Columbia generally occur at lower elevations (under 1200 m), with ponderosa pine (*Pinus ponderosa* Laws) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) being the dominant tree species (the Bunchgrass, Ponderosa pine and interior Douglas-fir biogeoclimatic zones; Meidinger and Pojar, 1991). Because of their extent (over 5 million ha) and location, they provide a number of socially important commodities including timber, wildlife habitat,

^{*} Corresponding author. Tel.: +1 250 828 4158; fax: +1 250 828 4154.
E-mail address: Walt.Klenner@gov.bc.ca (W. Klenner).

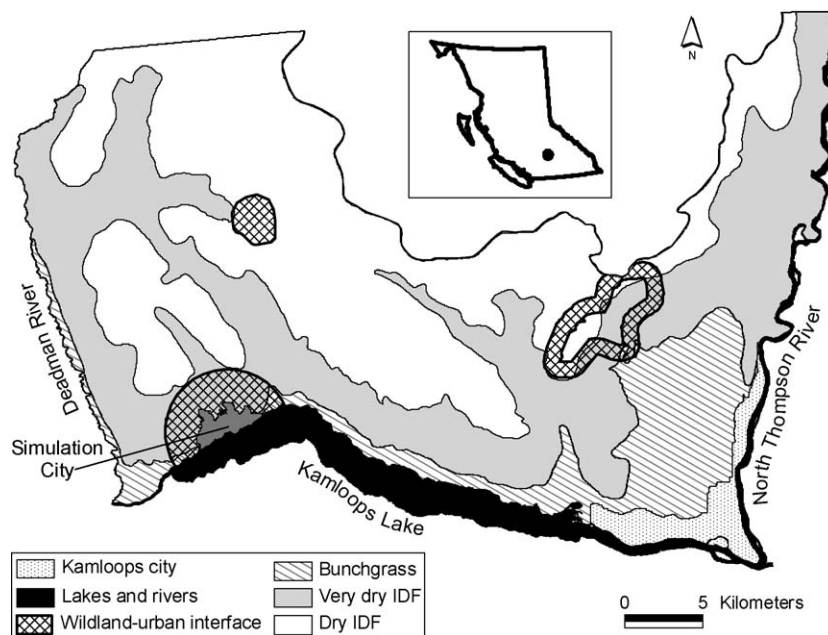


Fig. 1. The study area in south-central British Columbia. Black dot in the insert diagram indicates the location of the study area adjacent to the city of Kamloops (50° 41'N; 120° 20'W). Light grey fill indicates the very dry IDF used for extensive fuel management, while cross-hatching indicates the three smaller wildland–urban interface fuel management zones.

livestock forage and recreation opportunities. In addition, since many urban centers in the southern interior of British Columbia are located at lower elevations, dry forests provide an important viewshed surrounding communities. Recent large wildfire events and insect pest outbreaks in British Columbia (Filmon, 2004; Westfall, 2006) and the western United States (Romme et al., 2006) have demonstrated the challenges associated with managing these forests for current social expectations and have initiated a review of fire management policy in some jurisdictions (e.g. Stephens and Ruth, 2005). They have also spurred a reassessment of what constitutes a healthy forest, with some arguing that fire suppression has led to unnaturally dense stands and dangerously high fuel accumulations, prompting calls for large scale “ecosystem restoration” (Allen et al., 2002) and fuel management programs (Stephens and Ruth, 2005; Snider et al., 2006). However, across much of North America, the need for maintaining non-timber values in these forests, including wildlife habitat, recreation and livestock grazing, has increased concurrent with the need to protect communities from wildfire. While the debate about historic, natural conditions continues (Klenner et al., 2008), managers are being encouraged to apply fuel management treatments (Filmon, 2004; Snider et al., 2006) to address concerns about wildfire risk to communities. Recent work has explored the application of fuel management treatments at the landscape scale (Agee et al., 2000; Bevers et al., 2004; Finney, 2006), but applications remain untested and the costs and benefits of implementing different fuel management approaches and their effects on other forest management objectives have not been demonstrated.

Forest landscape modelling can be used to evaluate the likely outcomes of a wide range of forest management options on indicators that reflect a range of forest values. We used the TELSA (Tool for Exploratory Landscape Scenario Analyses) forest landscape model (Kurz et al., 2000; Beukema et al., 2003) to assess the implications of several dry forest management options on a range of indicators representing wildlife, livestock forage, crown fuel hazard, timber yield and treatment costs. This approach allows for an *a priori* evaluation of management options to identify potentially unacceptable impacts or costs before programs are

implemented. As with all modelling exercises, the scenarios we evaluated represent simplified, conceptual applications of complex programs that have fiscal (e.g. funding uncertainty) and social (e.g. smoke and air quality) dimensions that can only be reconciled during program implementation. In this study we apply management scenarios in a simulation environment to a 106 724 ha landscape composed primarily of dry ponderosa pine, Douglas-fir and lodgepole pine forests (*P. contorta* var. *latifolia* Dougl.) in the interior of British Columbia, and project these scenarios over a 100-year period. We chose five management strategies to simulate a range of options, from a “hands-off”, no management approach, to scenarios with fuel management and protected area strategies.

2. Study area

The study area is located immediately northwest of Kamloops, British Columbia and is bounded on three sides by the North Thompson River, the Deadman River and Kamloops Lake (Fig. 1). Elevation ranges between 334 and 1556 m and the site is predominantly a southern exposure, with variable slopes averaging 25%. Approximately 74% of the study area is forested (78 690 ha), with most of the remainder classified as open grassland. Forested land falls within the Ponderosa Pine or Interior Douglas-fir biogeoclimatic zones (Meidinger and Pojar, 1991), and is dominated by Douglas-fir (79%), lodgepole pine (11%) and ponderosa pine (6%). Common understory vegetation includes bluebunch wheatgrass (*Pseudoroegneria spicata* Pursh), rough fescue (*Festuca campestris* Rydberg), arrow-leaved balsamroot (*Balsamorhiza sagittata* (Pursh.) Nutt.), pinegrass (*Calamagrostis rubescens* Buckley) and saskatoon berry (*Amelanchier alnifolia* Nutt.). Open grasslands are dominated by bluebunch wheatgrass, rough fescue and big sagebrush (*Artemisia tridentata* Nutt.). Approximately 78% of forested stands (61 437 ha) are considered part of the timber harvesting land base (THLB), except in some model scenarios where other land use designations such as protected areas or old-growth management areas are applied. Previous management and natural disturbance histories are similar to many areas in south-central BC and represent the scale at which landscape planning is likely to be addressed.

3. The TELSA model

The TELSA forest landscape model (see Kurz et al., 2000; Beukema et al., 2003; ESSA Technologies, 2008) was designed to help resource managers explore the interactions between vegetation succession, natural disturbances and management decisions over large spatial (e.g. 10 000–300 000 ha or larger) and temporal (e.g. up to 500 years or more) scales. The model is spatially explicit and incorporates a detailed forest succession simulator (Vegetation Dynamics Development Tool: ESSA Technologies, 2007), the ability to impose deterministic or stochastic natural disturbances, user-specified management and a range of spatial and non-spatial indicators to evaluate future habitat conditions that would likely develop under competing scenarios. Previous applications of the model and a more detailed description of TELSA model components are presented in Klenner et al. (2000), Kurz et al. (2000), Beukema et al. (2003) and ESSA Technologies (2008).

4. Model initialization

We created polygons of approximately 1 ha (0.005–1.049 ha) using a tessellation routine and assigned a vegetation class based on species and age information derived from the BC Forests and Range Vegetation Resource Inventory (VRI; see <http://www.for.gov.bc.ca/hts/vri/intro/index.html>). The forested area was grouped into two subzones of the Interior Douglas-fir (IDF) biogeoclimatic zone (Meidinger and Pojar, 1991): very dry IDF forest (IDF xh2; 41 508 ha) and dry IDF forest (IDF dk1; 43 387 ha). In very dry IDF forest, stands were assigned to four main vegetation lineages: pure ponderosa pine stands, mixed ponderosa pine/Douglas-fir stands, pure Douglas-fir stands, and mixed spruce/Douglas-fir stands, representing a gradient of increasing soil moisture. In dry IDF forest, stands were also assigned to four main vegetation lineages: pure Douglas-fir stands, mixed Douglas-fir/lodgepole pine, pure lodgepole pine and mixed spruce/Douglas-fir stands. Vegetation succession involved changes in the overstory tree species composition in response to stand age, management and natural disturbances. The nature and rate of successional change was calibrated from the input of several plant ecologists familiar with the ecosystems being modelled. Stands remained within a forested vegetation class unless affected by a stand-replacing disturbance. If this occurred, stands changed to grass/forb, shrub, or herbaceous vegetation classes, and remained in these classes for 20 years before transitioning back to a forested vegetation class.

Two kinds of harvest-stand management treatments were applied to stands in our study area: partial cut harvesting and fuel management. Partial cutting is a widely accepted and practised harvesting prescription in dry forests of this region (Vyse et al., 1991). Partial cutting removed 45% of the stand volume on sites greater than 80 and 100 years of age for dry and very dry IDF stands, respectively, with re-entry prohibited for at least 50 years following harvest. Conifer stem densities in these forests are approximately 700 stems ha⁻¹ prior to harvest, excluding stems <7.5 cm diameter at breast height (dbh), and this was reduced to approximately 400 stems ha⁻¹ following partial cutting. Fuel management treatments involved several interventions to manipulate forest structure including harvest and silvicultural treatments to reduce stem density to ≤150 stems ha⁻¹ (8 m spacing, for stems >7.5 cm dbh) and crown closure to ≤20%, pruning to diminish ladder fuels, and ground fuel reductions using prescribed fire or pile and burn treatments. Such treatments, when maintained with periodic entries, are thought to diminish the likelihood and severity of crown fire under a broad range of fire weather conditions (Agee et al., 2000; Agee and Skinner, 2005). Complementary stand-level modelling (unpublished data) using the TASS (Tree and Stand Simulator) model (Mitchell, 1975) was used to

determine the stem density required to maintain a crown closure of ≤20%, and fuel management treatments were applied every 15 years to maintain this density and to remove ladder fuels and fuel accumulations in the understory (Battaglia et al., 2008; Busse and Riegel, 2009). Non-forested polygons such as grassland and non-productive brush within fuel management areas were treated by brush removal and prescribed fire.

We modelled the effects of insect attack and wildfire as natural disturbances on our landscape. Although other insect pests such as western spruce budworm (*Choristoneura occidentalis* Freeman) and Douglas-fir tussock moth (*Orgyia pseudotsugata* McDunnough) are common in our study area, we only included insects responsible for most of the mature tree mortality. We simulated attack by Douglas-fir beetle (*Dendroctonus pseudotsugae* Hopkins) and mountain pine beetle (*D. ponderosae* Hopkins) that attack Douglas-fir and ponderosa pine–lodgepole pine, respectively. Douglas-fir beetles are endemic but increased mature tree mortality is observed approximately every 20–25 years (L. MacLauchlan, BC Forest Service entomologist, pers. comm.). To simulate outbreaks, we created 10 possible patterns of Douglas-fir beetle disturbances that were then assigned to each of the 10 replicate simulations for each management scenario. For each run we randomly chose a year between 1 and 25 for the first Douglas-fir beetle attack, and then randomly chose between 20 and 30 years for subsequent attacks until 100 years was exceeded. We set Douglas-fir beetles to attack 15% of Douglas-fir stands on the landscape >120 years old for very dry forests and >80 years old for dry forest types. Within a stand, attacks resulted in the mortality of the largest Douglas-fir trees, killing 10% of trees in pure stands and 5% of overall trees in mixed-species stands (L. MacLauchlan, BC Forest Service entomologist, pers. comm.).

Two types of mountain pine beetle attacks occur in this landscape: severe and extensive attacks that occur approximately every 100 years (e.g. Westfall, 2006), and less extensive and less severe attacks that recur approximately every 20 years. In both attacks the largest diameter trees are most severely affected. To simulate mountain pine beetle attacks, we created 10 possible patterns for model runs and, as with Douglas-fir beetle attack, assigned one to each of the 10 replicate simulations for each management scenario. For severe attacks, we randomly chose an attack year between 1 and 100 for each run. In high severity years 70% of stands containing pine >30 years old were attacked, killing 90% of trees in pure stands and 45% of overall trees in mixed-species stands. In lower severity attacks, 10% of stands containing pine >60 years old were attacked, killing 45% of trees in pure stands and 22% of overall trees in mixed-species stands. We randomly chose a year between 1 and 20 for the first low-severity mountain pine beetle attack, and then randomly selected subsequent low-severity attacks from between 16 and 24 years until 100 years was exceeded. This was repeated for all 10 patterns. For both Douglas-fir beetle and mountain pine beetle, attacks were simulated across the landscape and were grouped spatially as follows: 40% of attacks affected stands of less than 10 ha, 50% of attacks affected areas of 10–100 ha, and 10% of attacks covered 100–1000 ha.

We obtained historic fire data for our study area from records maintained by the BC Ministry of Forests and Range, Protection Branch, and recent updates to this database (see <http://cfs.nrcan.gc.ca/subsite/disturbance/sources>) to calculate the area burned annually from 1930 to 2007. From this distribution we randomly selected (with replacement) 10 possible patterns of the annual area burned for each of the 10 replicate simulations for each management-natural disturbance scenario. We assumed that the area burned during this time period reflected some level of fire suppression given the study area's close proximity to Kamloops. Based on these historic annual targets we applied both

Table 1

Effects of disturbances on landscape indicators. Values presented are relative to similarly aged undisturbed stands.

Disturbance	Stand volume	Crown closure	Large snags (>30 cm dbh)
Low severity Fires	↓ 30%	1–10 years: ↓ 30% 11–20 years: ↓ 15%	Existing: ↓ 50% New: ↑ 30% live trees
High severity Fires	↓ 100%	Refer to text for young stands.	Existing: ↓ 90% New: ↑ 90% live trees
Douglas-fir Beetle ^a	↓ 10%	1–10 years: ↓ 10%	Existing: no effect New: ↑ 10% live trees
Mountain pine beetle (severe) ^a	↓ 90%	1–10 years: ↓ 90% 11–20 years: ↓ 40%	Existing: no effect New: ↑ 90% live trees
Mountain pine beetle (low) ^a	↓ 45%	1–10 years: ↓ 45% 11–20 years: ↓ 20%	Existing: no effect New: ↑ 45% live trees
PC 45%	↓ 45%	1–10 years: ↓ 50% 11–20 years: ↓ 25%	Existing: ↓ 90% New: none
Fuel Management (1st Pass)	↓ 80%	1–10 years: ↓ 80% 11–20 years: ↓ 40%	Existing: ↓ 90% New: ↑ 15% live trees
Fuel Management (>1 Pass)	See text.	1–10 years: ↓ 80% 11–20 years: ↓ 40%	Existing: ↓ 90% New: ↑ 2.5 snags per ha
Salvage	↓ 100%	Refer to text for young stands.	Existing: ↓ 100% New: ↑ 5% live trees

^a For disturbances by beetles, values presented are for pure tree species stands; for the overall effect on mixed-species stands, divide values by half.**Table 2**

Summary of scenarios. Refer to text for description of scenarios.

Scenario	PC 45% harvest limit (m ³ /years)	Fuel Mgt. area (ha) ^a	Fuel Mgt. rate (ha/years)	No. of runs
Historic	—	0	0	10
PC_Only	95 000	0	0	10
Ext_FM	50 000	41 508	2670	10
WUI	90 000	5 363	290	10
WUI + PA	55 000	5 363	290	10

^a Includes small lakes and other non-fuel management classes.

low-severity and high-severity fires to the landscape each year, with low-severity fires killing 30% of trees in a stand and high-severity fires being stand-replacing. To reflect the complex topography in our study area, both fire types included “fire skips,” unburned patches within a burned area. We used information from DeLong and Tanner (1996) and a sample of fires in the vicinity of Kamloops for which fire skip estimates were calculated from aerial photographs (unpublished data) to establish fire skip rates of 20% for low-severity fires and 10% for stand-replacing events. Previous work in southern British Columbia (Arsenault and Klenner, 2005) indicated a mixed-severity fire regime is likely, and Wong (1999) found that low- and mixed-severity fires occurred on approximately 75% of the sites sampled in her study. We used these data to calibrate our fire disturbances such that non-stand replacing fires occurred on average three times more often than stand-replacing fires. Fuel management practices are designed to diminish fire behaviour and burn severity (Agee et al., 2000; Graham et al., 2004; Agee and Skinner, 2005), hence we only applied low-severity fires in our fuel management zones.

Historic fire data for our study area indicated relatively low levels of fire activity, with a median annual fire size of 13.2 ha and a maximum annual fire area of 496 ha. Since several fires greater than 5000 ha burned in adjacent similar forests over this time period, we introduced less frequent but more extensive fire events for some runs to capture this potential. We chose two potential large fire events: a 1 in 125-year fire that burned approximately 5000 ha, and a 1 in 250-year fire event that burned approximately 15 000 ha, representing fires that occur in extreme and very extreme fire weather conditions. For each of the 10 fire patterns we randomly determined if none, one or both fire events would occur based on their fire return interval probabilities. We incorporated

fire skips representing 10% of the area burned for the 1 in 125-year fires but reduced this to 5% for 1 in 250-year fires. In fuel managed areas we assumed low-severity and stand-replacing fires would be equally represented in the 1 in 125-year event, and would be completely stand-replacing in the rare but very extreme fire behaviour of 1 in 250-year fires.

Stands >80 years old affected by stand-replacing fires or severe mountain pine beetle attacks were salvage harvested. Salvage occurred up to 4 years after these disturbances and recovered 70% of the volume from the stand. Salvage was only possible on stands in the harvestable landbase. We set the annual volume limit for salvage harvesting at 200% of the normal harvest to account for limited mill capacity, and any volume salvaged in a year lowered the annual partial cutting limit by the same amount. For example, if we salvaged 75 000 m³ in a year and our annual partial cutting limit was set at 100 000 m³, only 25 000 m³ would be harvested in non-salvage areas that year.

Harvesting and natural disturbances affected the indicators used to track consequences of the different scenarios. Table 1 presents a summary of the effects of natural disturbances and management on these indicators.

5. Scenarios

We simulated five management scenarios over a 100-year projection (see Table 2). Each scenario was simulated 10 times using 10 randomly selected patterns of natural disturbances, and the same 10 disturbance patterns were repeated for each of the 5 scenarios. Annual partial cutting limits for each scenario were determined by iteratively testing different rates until we found the maximum harvest rate that maintained a consistent timber flow

over 100 years in the absence of natural disturbances. We applied natural disturbances to each scenario and assumed fire suppression would keep the total annual area burned within the range documented over the last 77 years. The two rare (1 in 125 or 250 year) fire events represent severe and very severe fire weather conditions during which fire suppression has limited efficacy beyond protecting local features. Five management scenarios were examined:

1. *Historic*: This scenario projects what could happen in the absence of any harvesting or fire suppression. Douglas-fir beetle and mountain pine beetle were present and we calibrated historic fire levels to be three times higher than levels recorded during our 77-year recorded history. Natural fire regimes in western conifer forests are highly variable but several studies have reported pre-European settlement fire frequencies of 2–4 times current levels (Brown et al., 1999; Bergeron et al., 2004; Klenner et al., 2008). The two large fire types burned at sizes of approximately 5000 ha and 15 000 ha for runs in which they appeared.
2. *Partial cut only (PC_Only)*: A widely applied harvest prescription for this area, with 45% of the standing volume removed across the diameter distribution in 0–40 ha blocks and re-entry prohibited for 50 years. Annual harvest rate was set at 95 000 m³ year⁻¹.
3. *Extensive fuel management (Ext_FM)*: Similar to PC_Only, except the entire very dry IDF (Fig. 1) was treated with heavy (80%) volume removal, managed to a density of approximately 150 stems ha⁻¹ and maintained in this state by re-entries every 15 years. Because partial cutting only occurs in the dry IDF in this scenario, we iteratively determined the maximum sustainable flow in the dry IDF as 50 000 m³ year⁻¹ in the absence of natural disturbances. Fuel management treatments were applied to 2670 ha annually.
4. *Wildland–urban interface (WUI)*: Similar to PC_Only, except fuel management was applied to 3 restricted areas representing 2 communities and a high investment forestry research area (Fig. 1). The fuel management zone adjacent to Kamloops Lake extends approximately 2 km beyond the boundaries of a fictitious city (Simulation City) and the zone north of this protects a small rural settlement. The maximum annual partial cutting limit was determined to be 90 000 m³ year⁻¹ in the absence of natural disturbances, and we fuel managed 290 ha year⁻¹ across the three wildland–urban interface areas (WUI), with re-entries scheduled every 15 years.
5. *Wildland–urban interface and protected areas (WUI + PA)*: Similar to the WUI scenario except 19 865 ha was removed from the harvestable landbase and placed into protected area designation. These protected areas were based on draft wildlife habitat area boundaries stemming from a land use planning process in the study area. We determined the maximum sustainable annual partial cutting limit to be 55 000 m³ year⁻¹ in the absence of natural disturbances.

6. Indicators

6.1. Harvest flow

We calculated merchantable volume for stands using volume curves based on stand age and species composition developed for this region (Ministry of Forests, 2001) and established an annual sustainable harvest rate for partial cutting in the absence of natural disturbances for each scenario. Salvage contributed to the annual harvest limit, but timber removed from harvestable stands during fuel management treatments was independent of the annual harvest limit. The first time a stand was fuel managed

80% of its volume was added to the total volume harvested for that year; in subsequent re-entries, fuel managed stands contributed 5 m³/ha after 15 years based on volume projections generated by the TASS model for densities of 150 stems ha⁻¹ in very dry IDF forest (unpublished data). At 20-year intervals we calculated the cumulative total volume harvested during the preceding 20 years.

6.2. Fuel treatment costs

Estimates of fuel treatment costs vary widely in relation to weather, risk to property, forest type, treatment area size, and how costs are calculated. Costs in the wildland–urban interface are particularly high because of risks to lives and property, smoke management and other factors. Wildland interface fuel management usually involves mechanical thinning and pruning, followed by piling and burning. Some costs may be offset by the harvest of merchantable stems, and future entries, once stand densities are reduced to 150 stems ha⁻¹, will be lower in cost. Cost of treating non-WUI areas is likely to be considerably less.

We based cost estimates on whether the treatment was inside or outside the WUI, on forest–non-forest designation, and on stem density and stand age as a rough surrogate for the risk and difficulty in applying a fuel removal treatment. Initial treatment costs in fully stocked very dry IDF forests ranged from \$3000 ha⁻¹ inside the WUI to \$1000 ha⁻¹ and \$1500 ha⁻¹ outside the WUI for stands in the harvestable and non-harvestable landbase, respectively. Subsequent entries incurred a reduced cost of \$1500 ha⁻¹ inside the WUI and \$500 ha⁻¹ outside the WUI. Costs were higher in the dry IDF because of higher tree densities and greater understory vegetation, ranging from \$4000 ha⁻¹ inside the WUI to \$1000 ha⁻¹ and \$2000 ha⁻¹ outside the WUI for stands in the harvestable and non-harvestable landbase, respectively. Subsequent re-entries incurred a reduced cost of \$2000 ha⁻¹ inside the WUI and \$1000 ha⁻¹ outside. Non-forested areas in fuel management zones cost \$500 ha⁻¹ inside the WUI and \$100 ha⁻¹ outside the WUI. Costs of fuel management inside the WUI were based on estimates from municipal and provincial agencies with experience in conducting fuel management in the area.

6.3. Crown fuel hazard

The likelihood of a crown–crown fire (a fire travelling from tree crown to tree crown) is affected by crown closure and weather conditions. Winds and temperature play a strong role in modifying the risk of a crown fire, complicating the relationship between canopy condition and crown fuel hazard (CFH). For this modelling exercise and for the limited range of forest types being evaluated we used crown closure to identify low (0–20% crown closure), moderate (21–50% crown closure) or high (51–100% crown closure) crown fuel hazard ratings. We used the relationship between stand density and crown closure developed in TASS simulations in similar dry IDF forests (unpublished data) to create our CFH categories in the context of recommendations in Agee et al. (2000) and Agee and Skinner (2005). Undisturbed stands older than 30 years at low productivity sites (site index = 14; i.e. height in m at age 50) have an average crown closure of 70% compared to 90% crown closure for undisturbed stands with higher growing potential (site index = 19 or 21). We assigned all stands less than 10 years old a crown closure value of 0%. For site index 14 stands, crown closure increased to 29% for 11–20-year-old stands and to 70% for 21–30-year-old stands. Crown closure in site index 19 and 21 stands increased to 33% and 44%, respectively, for 11–20-year old stands, and to 67% for 21–30-year-old stands. Table 1 presents a summary of the effects of natural disturbances and management activities on crown closure.

6.4. Forage production

Annual grass and forb production can be highly variable, making estimates at a landscape scale difficult. We used crown closure as a surrogate for forage production, based on relationships between crown closure and forage production (Pase, 1958; Halls and Schuster, 1965; Dodd et al., 1972; Mitchell and Bartling, 1991; Knowles et al., 1999). From these studies, the maximum forage production within a forested stand dropped to less than 50% of its potential after crown closure values exceeded 20%.

6.5. Snags

Large snags serve many important ecological roles (Hunter, 1990; Bunnell et al., 2002), providing foraging and nesting sites for cavity-nesting species such as woodpeckers, potential denning opportunities for marten and bear, and substrate for some lichen and moss species that grow preferentially on dead trees. We used tree plot data from two long-term research installations in our study area, the Opax Mountain Silvicultural Systems trial (159 plots, 0.04 ha each) and the Isobel Research Trial (169 plots, 0.03 ha each), to estimate the number of large (>30 cm dbh) live trees available ha^{-1} and the proportion of large snags in mature stands. Only pre-harvest plots from mature stands >120 old were used. Pooling all plots ($n = 276$) with more than one large stem present, we obtained an average of $136.2 (\pm 4.5 \text{ SE})$ large live stems ha^{-1} with an average of 3.7% large snags. Using these data as a guide, we generated a live stems ha^{-1} curve based on stand age, with large stems first appearing in a stand at 70 years and reaching an asymptote at 150 stems ha^{-1} at 150 years. We assumed a 5% presence of large snags and we used this to estimate the number of large snags present in relatively undisturbed stands based on stand age. Based on Landram et al. (2002), Conner and Saenz (2005) and Russell et al. (2006), we estimated snag fall rates. Trees killed by mountain pine beetles decayed at a relatively fast rate, with half the snags fallen within 10 years and all the snags fallen after 30 years. Trees killed by other disturbances decayed at a slower rate, maintaining half the snags after 20 years and dwindling to no standing snags after 60 years. Table 1 presents the specific effects of disturbances and management on the number of large snags ha^{-1} .

6.6. Forest undisturbed after 100 years

In dry forest ecosystems with complex structure arising from multiple non-stand-replacing disturbances, stand age is not a reliable indicator of old growth condition. As a measure of the proportion of forested stands on the landscape that were not affected by management or severe natural disturbances and which have had sufficient time to develop old forest structures (canopy gaps, large live trees, large snags and downed wood), we calculated the proportion of forested area for stands without a history of disturbances that reduced their live volumes by more than 33% at the end of the 100-year simulation. We excluded stands <50 years old at the start of the simulation because we focused on stands that could provide old forest structures such as large snags that only begin to develop at age 70 and which reach an asymptote at age 150 (see above).

7. Analyses

Prior to statistical testing, all variables were examined for assumptions of normality and variance. If transformations were ineffective, we used non-parametric statistics. Bonferroni-corrected Mann–Whitney U tests were used to make post hoc pairwise comparisons when Kruskal–Wallis tests found significant

differences among scenarios. For most analyses we chose the end of the simulation, 100 years from present, as the most relevant time period because it allowed enough time for the effects of different scenarios to be manifested.

8. Results

8.1. Harvest flow

Fig. 2 presents the cumulative volume harvested every 20 years for the four management scenarios with natural disturbances (Table 2). The scenario with extensive fuel management initially outperformed all other scenarios due to the high contribution of timber from heavily thinned stands in the initial fuel management treatment. After 20 years this advantage was lost. Not surprisingly given the relatively small amount of area covered by the three WUI zones, partial cut harvesting scenarios with and without WUIs were similar and provided the highest timber flows after 20 years. Removing almost 20 000 ha from the harvestable landbase and adding it to protected areas resulted in the lowest timber volumes, similar to the Ext_FM scenario after 20 years. We also tracked the total volume harvested over the 100-year simulation. Due to the pulse of volume from the initial fuel management entry, the Ext_FM scenario yielded similar volumes after 100 years as the PC_Only and WUI treatments (Table 3). The protected areas scenario produced significantly lower volume.

8.2. Fuel treatment costs

Applying fuel management treatments to the entire very dry IDF (Ext_FM) resulted in average cumulative costs of \$160 million after 100 years (Fig. 3), more than three times the cost of targeting the three wildland–urban interface areas (WUI scenario). Variation among runs within a scenario was slight, even with natural disturbances present, because the same target area was scheduled for fuel treatment annually. Although the Ext_FM scenario required treating almost 10 times the area as the WUI scenario

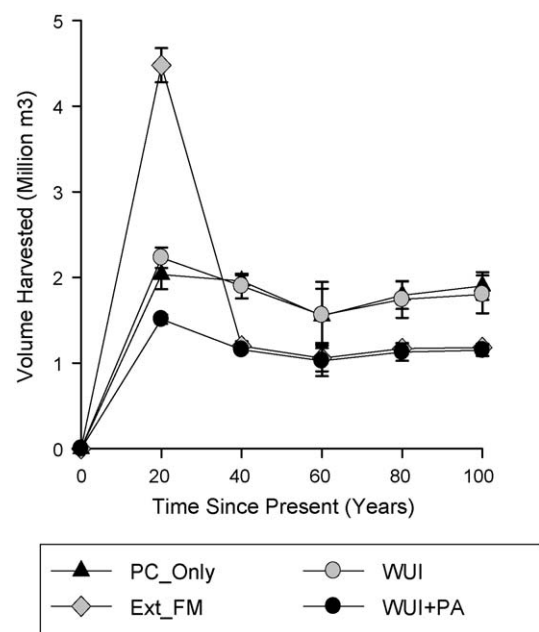


Fig. 2. Average (± 1 SD) cumulative volume harvested every 20 years for management scenarios with natural disturbances present. PC_Only: partial cutting only, Ext_FM: extensive fuel management, WUI: fuel management in wildland–urban interface areas only, WUI+PA: fuel management in wildland–urban interface areas only and no harvest in protected areas. Refer to text for a detailed description of scenarios.

Table 3

Average cumulative merchantable volume harvested after 100 years for scenarios with natural disturbances present.

Scenario	With natural disturbances million m ³ (mean $\pm$ 1 SD)
PC_Only	9.23 (0.25) ^a
Ext_FM	9.08 (0.40) ^a
WUI	9.23 (0.32) ^a
WUI + PA	5.97 (0.17) ^b

Lower case letters shared among scenarios indicate no significant difference ($p > 0.05$) in volume harvested (Bonferroni-corrected Mann–Whitney U tests). See text for description.

(2670 ha year⁻¹ vs. 290 ha year⁻¹), high treatment costs in the wildland–urban zone diminished the cumulative cost differences.

8.3. Crown fuel hazard

On average, the historic scenario maintained the forested landscape at a high crown fuel hazard rating for the duration of the simulation, with only 10–14% of the forest in lower hazard classes (Fig. 4). The Ext_FM scenario had the most dramatic effect, removing more than half of the forested landscape from high crown fuel hazard category. The PC_Only scenario, despite harvesting 45% of the timber volume at each entry, maintained a relatively high proportion of forest in the highest crown fuel hazard rating. Adding three small fuel management zones or creating more protected areas had little effect on crown fuel hazard risk at the overall landscape scale. There was a significant difference among scenarios in the proportion of the landscape found in the highest crown fuel hazard class after 100 years (Kruskal–Wallis test, $H = 30.9873$, $df = 4$, $p < 0.0001$). The historic scenario tended to have a higher (but non-significant) proportion of the area in the highest crown fuel hazard class, but only the Ext_FM scenario had a significantly lower proportion of the landscape in the highest crown fuel hazard class (Fig. 4; Bonferroni-corrected Mann–Whitney U tests, $p < 0.01$).

8.4. Forage production

The Ext_FM scenario had the greatest effect on forage production on the forested landscape, maintaining an average of over 30% of the landscape in high forage productivity (>50% of potential; Fig. 5). All other scenarios lowered forage productivity

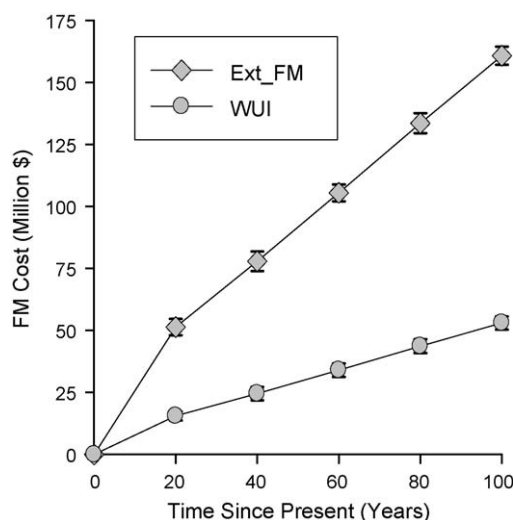


Fig. 3. Average (± 1 SD) cumulative cost of fuel management for the extensive fuel management scenario (Ext_FM) and the three smaller fuel management zones scenario (WUI) with natural disturbances present.

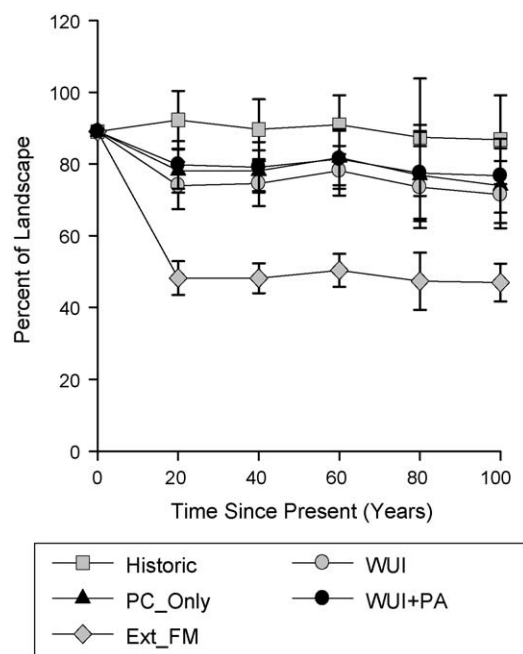


Fig. 4. Average (± 1 SD) percent of forested landscape in the high crown fuel hazard class (>50% crown closure) for scenarios with natural disturbances present.

from current conditions, although the WUI and WUI + PA scenarios had slightly higher forage productivity, reflecting the open canopy conditions that were created and maintained on the WUI areas which represent approximately 5% of the landscape. The Ext_FM scenario had significantly higher forage production than other treatments after 100 years (Fig. 5; Kruskal–Wallis test, $H = 26.8075$, $df = 4$, $p < 0.0001$; Bonferroni-corrected Mann–Whitney U tests, $p < 0.01$).

8.5. Large snags

The historic scenario produced the highest densities of snags over 100 years (Fig. 6c) and consistently maintained a density of 5–

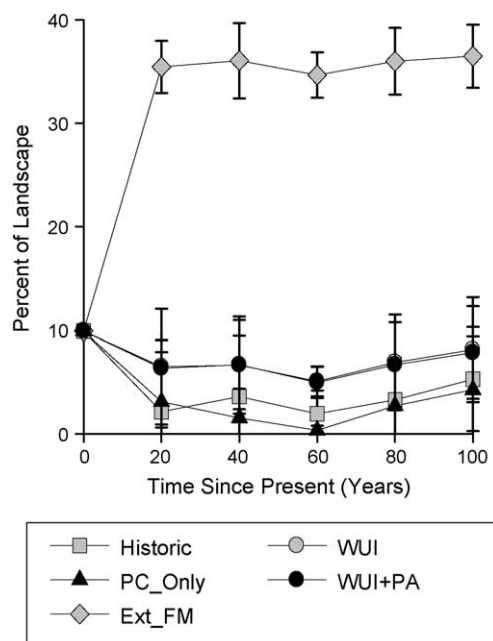


Fig. 5. Average (± 1 SD) percent of forested landscape with forage production potential greater than 50% for scenarios with natural disturbances present.

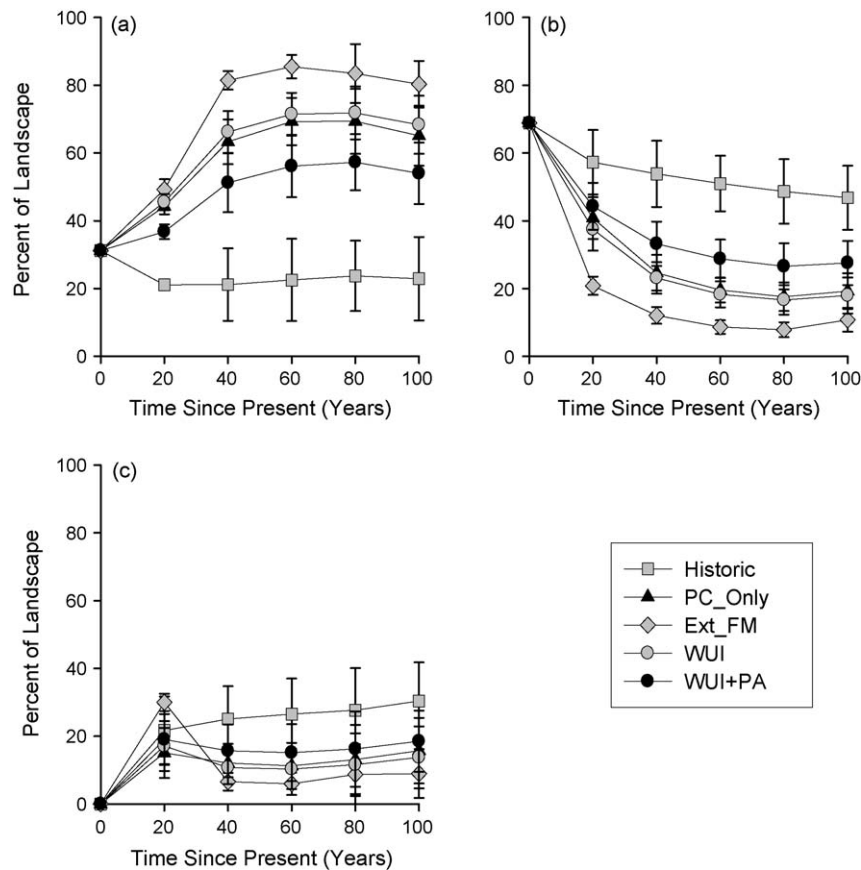


Fig. 6. Average (± 1 SD) percent of forested landscape in three large (>30 cm dbh) snag density classes: (a) <5 snags per ha; (b) 5–10 snags per ha; and (c) >10 snags per ha. Natural disturbances were present for all scenarios.

10 snags on half the landscape (Fig. 6b). Snag density was negatively correlated with management activity. The Ext_FM scenario dropped snag densities to less than five snags ha^{-1} on over 80% of the landscape, followed closely by partial cut management with and without small fuel management zones (Fig. 6a). Removing stands from the harvestable landbase and placing them in protected areas (WUI + PA) reduced management activity on the landscape and resulted in a higher proportion of stands with densities of 5–10 snags ha^{-1} (Fig. 6b).

Higher snag densities are beneficial for many species of wildlife. We compared the proportion of landscape after 100 years with fewer than five snags ha^{-1} and found significant differences among scenarios (Kruskal–Wallis test, $H = 38.0198$, $df = 4$, $p < 0.0001$). The Ext_FM scenario had the highest proportion of landscape in the lowest snag density class, differing significantly from all other scenarios (Fig. 6a; Bonferroni-corrected Mann–Whitney U tests, $p < 0.05$). Conversely, the historic scenario had the lowest proportion of the landscape with less than five snags ha^{-1} (Fig. 6a; Bonferroni-corrected Mann–Whitney U tests, $p < 0.01$). Differences between partial cutting scenarios (PC_Only, WUI) were not significant (Fig. 6a; Bonferroni-corrected Mann–Whitney U tests, $p > 0.05$). Although the WUI + PA scenario maintained a significantly smaller proportion of landscape in the lowest snag density class compared to the WUI scenario (Bonferroni-corrected Mann–Whitney U test, $p < 0.05$), it did not differ significantly from the PC_Only scenario (Bonferroni-corrected Mann–Whitney U test, $p > 0.05$).

8.6. Percent forest undisturbed after 100 years

We evaluated all forested stands projected to be at least 150 years old at the end of the 100-year simulation to identify how

Table 4

Percent of forested landscape^a without a major disturbance after 100 years for scenarios with natural disturbances present.

Scenario	Average % forest undisturbed (± 1 SD)
Historic	58.0 (6.7) ^a
PC_Only	12.3 (1.5) ^b
Ext_FM	4.4 (0.6) ^c
WUI	10.6 (1.5) ^b
WUI + PA	24.8 (2.3) ^d

Lower case letters shared among scenarios indicate no significant difference ($p > 0.05$) in percent undisturbed forest (Bonferroni-corrected Mann–Whitney U tests). See text for description.

^a For stands greater than 50 years old in current landscape (68 730 ha).

much of the landscape had not been affected by severe natural disturbances ($>33\%$ volume removal) or harvesting during the previous 100 years. Results for our five scenarios are presented in Table 4. We found significant differences among scenarios in the proportion of undisturbed forest remaining after 100 years (Kruskal–Wallis test, $H = 45.6696$, $df = 4$, $p < 0.0001$). Over half the forest remained undisturbed in the historic scenario. Placing part of the harvestable landbase into protected areas (WUI + PA) kept an average of 25% of the landscape in an undisturbed condition. The PC_Only scenario maintained only 12% of the forest in an undisturbed state, while the Ext_FM scenario reduced this to $<5\%$. Each scenario had significantly different proportions of undisturbed forest after 100 years, except for the PC_Only and WUI scenarios (Table 4).

9. Discussion

Our simulations clearly indicate that each of the management scenarios we examined had shortcomings with respect to the

Table 5

Summary of scenarios for individual indicators. Number of asterisks indicate, in descending order, the best to worst options for each indicator.

Scenario	Volume harvested	Standing volume	FM cost	CFH	Forage	Snags	No disturbance
Historic	—	***	—	*	*	****	****
PC_Only	***	**	—	*	*	**	**
Ext_FM	**	*	*	***	****	*	*
WUI	***	**	***	*	*	**	**
WUI + PA	*	*	**	*	*	***	***

wildlife habitat, fuel management, understory productivity, timber and cost indicators (Table 5). We incorporated natural disturbances into all scenarios since previous work has clearly identified the habitat (Klenner et al., 2000; Fall et al., 2004; Shifley et al., 2006) and timber (Peter and Nelson, 2005) concerns of not including natural disturbances in scenarios. Timber values were best maintained by partial cutting scenarios either with or without small fuel management zones, providing the highest level of harvest flows after 100 years. Applying extensive fuel management in our study area substantially diminished timber production, with a volume reduction of approximately 35 000 m³ year⁻¹ after the initial increase in timber harvest that resulted from the first pass of fuel management treatments creating open conditions on sites previously managed for timber. A similar reduction in timber flow was observed for the WUI + PA scenario that applied a combination of strategic wildland–urban interface treatments and an extensive protected area strategy. Altering the widely practised partial cutting approach to forest management had important consequences for timber values. Natural disturbances in our simulations were calibrated with wildfire and insect attack rates based on historic data, hence our results may represent conservative estimates of the role natural disturbances may play in the future. A warmer and drier future climate would likely lead to increased tree damage and mortality from insect attack (Bale et al., 2002), and wildfires may affect a greater area and potentially be more severe (Flannigan et al., 2005). These changes would likely diminish the volume of merchantable timber produced, and likely require more frequent or more intense fuel management treatments to manage fire behaviour within desired parameters.

Creating and maintaining fire-safe communities is an important dry forest management objective in British Columbia, brought into focus by extensive wildfires in 2003 (Filmon, 2004) that destroyed hundreds of homes and threatened several communities. Fuel management treatments can help reduce the likelihood of extreme fire behaviour under a broad range of conditions (Agee et al., 2000; Agee and Skinner, 2005), but the costs of treatments and long-term maintenance are seldom calculated. We explored the costs of implementing two different fuel management strategies, an extensive landscape level strategy and an approach concentrating fuel management around communities and areas of significance. Both strategies reduced crown fuel hazard in treated areas, although only the extensive fuel management scenario reduced crown fuel hazard on the overall landscape. Traditional partial cut harvesting, despite reducing timber volumes by 45% at each entry, had little effect on crown fuel hazard because partial cutting prescriptions are designed to maintain a well stocked and well distributed stand of trees for the next harvest entry. However, the extensive reduction in crown fuel hazard did not come without cost. Over the 100-year simulation, extensive fuel management costs averaged approximately \$1.6 million annually compared to \$0.5 million year⁻¹ for the three interface areas. Costs were proportionately higher in the wildland interface treatments, but localized fuel management zones around communities and areas of high investment potentially achieve effective and less costly fire protection overall. Narrow fuel management treatments (e.g. 100 m) are of questionable efficacy, but wider zones more than

300 m in width, or narrow zones combined with fire-proofing of individual homes (Cohen, 1999) appear to diminish both fire behaviour and burn severity (Agee et al., 2000; Graham et al., 2004; Agee and Skinner, 2005). Our wildland–urban interface fuel management areas were 2 km wide around communities and 1 km wide around high investment areas. These should be of adequate extent to provide fire management benefits. Furthermore, effective fire management in the interface zone near communities is enhanced through improved access and diminished response times by fire suppression crews (Agee et al., 2000; Keeley et al., 2004) should a fire enter the area.

Fuel management costs are notoriously variable, being sensitive to the physical characteristics of the area being treated, weather conditions, type of treatment, geographic location, treatment size and managerial factors (Rideout and Omi, 1995; González-Cabán, 1997; Cleaves et al., 2000; Hesseln, 2000). The cost estimates we used should be considered approximations but they represent a substantial annual expense and require a long-term commitment from resource managers. Calculating the absolute costs of fuel management treatments was beyond the scope of our study and would require a full accounting of the reduced timber quantity and quality from low density stands, the expected benefits of fewer stand-replacing fires (including lower fire suppression effort), post-fire restoration, timber losses and damage to personal property (Kline, 2004; Snider et al., 2006). Projecting long-term benefits into the future, however, makes many assumptions about the effectiveness of fuel management treatments in uncertain weather conditions and how treated stands will develop over time. Stand-replacing fires are still likely to occur under drought or extreme fire weather conditions, even in fuel managed areas (Fernandes and Botelho, 2003; Dellasala et al., 2004). Furthermore, without a commitment to maintenance treatments, the benefits of fuel management treatments will rapidly diminish (Fernandes and Botelho, 2003; Finney et al., 2005; Battaglia et al., 2008; Busse and Riegel, 2009).

Understory productivity is tightly coupled to canopy closure (Pase, 1958; Halls and Schuster, 1965; Dodd et al., 1972; Mitchell and Bartling, 1991; Knowles et al., 1999) and the open canopy conditions in fuel management areas benefit livestock and native ungulates. The extensive fuel management scenario easily outperformed all other scenarios, realizing over half the forage production potential on approximately 35% of the forested landscape and providing increased forage for some species of small mammals (Sullivan and Sullivan, 2001; Klenner and Sullivan, 2009) and ungulates (Peek et al., 2002). While stand-replacing fires and salvaged beetle kill stands provided localized grazing opportunities in all scenarios, only the extensive fuel management approach maintained a consistently high level of understory productivity. This level of forage production is much higher than was produced in our historic scenario, and may have several drawbacks. Native ungulates often rely on snow interception and thermal cover during the overwintering period (Armleder et al., 1994; Poole and Mowat, 2005) and open canopy conditions are unlikely to meet this need. However, by interspersing fuel management, partial cut and protected area or extended rotation prescriptions in a land use planning framework, a balance of

necessary habitats can be addressed. Open stand conditions may also increase the severity and behaviour of ground fires by creating a drier and windier microclimate in thinned stands and facilitating herbaceous growth (Agee and Skinner, 2005). Zimmerman and Neuenschwander (1984) reported reduced understory productivity on areas grazed by livestock, suggesting livestock management considerations should play an important role in addressing concerns about ground fuels. Suitable land use zonation and the use of fenced enclosures to focus livestock activity could be applied to strike a balance between diminishing understory fuels in critical areas and maintaining highly productive understories for wildlife.

Large snags have been widely reported as critical habitat features in stands that support a large number of vertebrate species (Hunter, 1990; Bunnell et al., 2002). In particular, cavity-nesting birds benefit from abundant snags arising from wildfires (Murphy and Lehnhausen, 1998; Smucker et al., 2005; Hutto, 2008) or extensive insect attack that provide important foraging, roosting and nesting opportunities for many wildlife species both as standing trees and later as downed wood. Because we did not know the disturbance histories associated with each stand at the beginning of our simulations, initial snag densities were based solely on inventory stand age, likely underestimating the proportion of the landscape in the lowest and highest snag classes (Fig. 6). For all scenarios the proportion of snags on the landscape declined with increasing management intensity, and extensive fuel management or partial cutting produced the lowest densities of snags on the landscape. Lower snag densities result from worker safety practices that discourage the retention of most snags during partial cutting operations, snag loss during operational salvage harvest, and because snags are often charred or consumed during wildfire or the application of prescribed fire. In this context, the historic (no management) scenario consistently provided high numbers of snags on the largest proportion of the landscape and the WUI + PA scenario represents a viable option in a managed landscape.

Old-growth forest is critical habitat for many species of vertebrates (Carey, 1989; Hunter, 1990) and is often considered a surrogate for the “naturalness” of a landscape, a measure of protection from anthropogenic disturbance. Because age is difficult to assign to multi-aged stands with complex structure that are found in many of the dry forest stands (Vyse et al., 1991) affected by ongoing, non-stand replacing disturbances, we instead chose to monitor the forested area that had experienced no major disturbances after 100 years. More management resulted in more disturbance on the landscape (Table 4), with the extensive fuel management scenario resulting in less than 5% of the landscape being undisturbed after 100 years. Placing land in protected areas was the best management strategy for this indicator, resulting in just under half the area from what would be expected from natural disturbances alone and doubling the undisturbed proportion realized through partial cut harvesting.

Forest landscape planning models have been used effectively for two decades to illustrate the effects of different management scenarios on landscape pattern and wildlife. Although the tendency has been to use the forest patterns created by natural disturbances as a reference condition, disturbances not only affect the size, shape and location of a forest patch but also the structural condition. The importance of large wildfires (Murphy and Lehnhausen, 1998; Hutto, 2008) and insect outbreaks (Martin et al., 2006) to many species of cavity-nesting birds clearly outlines the importance of forest structure to wildlife. In our simulations we evaluated changes in two stand-level forest structures known to be important to wildlife (snags and understory productivity) since measures of age and patch size are not particularly relevant to dry forest ecosystems in our study area. Our results indicate diverging structural conditions will be created by different forest

management practices, and are consistent with previous studies that have identified shifts in age structure (Klenner et al., 2000; Fall et al., 2004) or vegetation composition arising from forest management (Radeloff et al., 2006; Shifley et al., 2006). Although all aspects of historic conditions will likely never be achieved, a range of spatial and structural conditions that satisfy the habitat requirements of different species (Shifley et al., 2006; Zollner et al., 2008) can be met by ensuring diverse management practices are applied within and across multiple landscapes.

10. Management implications

Our analyses clearly indicate that of the scenarios we examined, no single strategy was best for all forest value indicators. Strategic forest landscape analyses of the nature undertaken here can only identify general trends and issues, but the scenarios we present provide a basic suite of tools that can be adjusted in an operational application. Forest landscape models have great potential in facilitating the incorporation of quantitative technical information into the development of landscape-scale, long-term forest management plans but several issues are critical and need to be addressed. (1) Choosing the best management system requires a clear understanding of the long-term goals for a landscape. This can be achieved most reliably through discussions with stakeholders who have access to quantitative information on the costs and benefits of activities. (2) Not all goals are likely to be met by one strategy, so a combination of management approaches is more likely to succeed. (3) Resource zonation is critical since one extensive treatment will not address all management objectives. (4) Location matters, and arbitrary zoning designations need to be replaced with spatially defined zones that reflect the greatest likelihood of achieving objectives. Forest landscape modelling can and should be used in an interactive manner with resource management professionals and stakeholders to assess the consequences of current or new management initiatives prior to their application to ensure expectations regarding costs and the achievement of objectives are realistic.

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References

- Agee, J.K., Skinner, C.N., 2005. Basic principles of forest fuel reduction treatments. *Forest Ecology and Management* 211, 83–96.
- Agee, J.K., Bahro, B., Finney, M.A., Omi, P.N., Sapsis, D.B., Skinner, C.N., van Wageningen, J.W., Weatherspoon, C.P., 2000. The use of shaded fuelbreaks in landscape fire management. *Forest Ecology and Management* 127, 55–66.
- Allen, C.D., Savage, M., Falk, D.A., Suckling, K.F., Swetnam, T.W., Schulke, T., Stacy, P.B., Morgan, P., Hoffman, M., Klingel, J.T., 2002. Ecological restoration of southwestern Ponderosa pine ecosystems: a broad perspective. *Ecological Applications* 12, 1418–1433.
- Armleder, H.M., Waterhouse, M.J., Keisker, D.G., Dawson, R.J., 1994. Winter habitat use by mule deer in the central interior of British Columbia. *Canadian Journal of Zoology* 72, 1721–1725.

- Arsenault, A., Klenner, W., 2005. Fire regime in dry-belt forests in British Columbia: perspectives on historic disturbances and implications for management. In: Taylor, L., Zelnik, J., Cadwallander, S., Hughes, B. (Eds.), *Mixed Severity fire regimes: Ecology and Management Symposium Proceedings*, November 17–19, Spokane, Washington. Association of Fire Ecology MISC03, Washington State University, Pullman, WA, pp. 105–121.
- Bale, J.S., Masters, G.J., Hodkinson, I.D., Awmack, C., Bezemer, T.M., Brown, V.K., Butterfield, J., Buse, A., Coulson, J.C., Farrar, J., Good, J.E.G., Harrington, R., Hartley, S., Jones, T.H., Lindroth, R.L., Press, M.C., Symrnioudis, I., Watt, A.D., Whittaker, J.B., 2002. Herbivory in global climate change research: direct effects of rising temperature on insect herbivores. *Global Change Biology* 8, 1–16.
- Baskett, E.Z., Jordan, G.A., 1991. Spatial wood supply simulation modelling. *Forestry Chronicle* 67, 610–621.
- Battaglia, M.A., Smith, F.W., Shepperd, W.D., 2008. Can prescribed fire be used to maintain fuel treatment effectiveness over time in Black Hills ponderosa pine forests? *Forest Ecology and Management* 256, 2029–2038.
- Bergeron, Y., Flannigan, M., Gauthier, S., Leduc, A., Lefort, P., 2004. Past, current and future fire frequency in the Canadian Boreal Forest: implications for sustainable forest management. *Ambio* 33, 356–360.
- Beukema, S.J., Kurz, W.A., Klenner, W., Merzenich, J., Arbaugh, M., 2003. Applying TELSA to assess alternative management scenarios. In: Arthaud, G.J., Barrett, T.M. (Eds.), *Systems Analysis in Forest Resources*. Kluwer Academic Publishers, pp. 145–154.
- Beyers, M., Omi, P.N., Hof, J., 2004. Random location of fuel treatments in wildland community interfaces: a percolation approach. *Canadian Journal of Forest Research* 34, 164–173.
- Brown, P.M., Kaufmann, M.R., Shepperd, W.D., 1999. Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. *Landscape Ecology* 14, 513–532.
- Bunnell, F.L., Houde, I., Johnston, B., Wind, E., 2002. How dead trees sustain live organisms in western forests. In: Laudenslayer Jr, W.F., Shea, P.J., Valentine, B.E., Weatherspoon, C.P., Lisle, T.E. (Eds.), *Proceedings of the Symposium on the Ecology and Management of Dead Wood in Western Forests*. November 2–4, 1999, Reno, Nevada. Pacific Southwest Research Station, USDA Forest Service General Technical Report. PSW-GTR-181, pp. 291–318.
- Busse, M.D., Riegel, G.M., 2009. Response of antelope bitterbrush to repeated prescribed burning in Central Oregon ponderosa pine forests. *Forest Ecology and Management* 257, 904–910.
- Carey, A.B., 1989. Wildlife associated with old-growth forests in the Pacific Northwest. *Natural Areas Journal* 9, 151–162.
- Cleaves, D.A., Martinez, J., Haines, T.K., 2000. Influences of Prescribed Burning Activity and Costs in the National Forest System. USDA Forest Service General Technical Report SRS-GTR-37, Asheville, NC.
- Cohen, J.D., 1999. Reducing the Wildland Fire Threat to Homes: Where and how much? PSW-GTR-173, USDA Forest Service General Technical Report PSW-GTR-203, Albany, CA.
- Conner, R.N., Saenz, D., 2005. The longevity of large pine snags in eastern Texas. *Wildlife Society Bulletin* 33, 700–705.
- Dellasala, D.A., Williams, J.E., Deacon Williams, C., Franklin, J.F., 2004. Beyond smoke and mirrors: a synthesis of fire policy and science. *Conservation Biology* 18, 976–986.
- DeLong, C., Tanner, D., 1996. Managing the pattern of forest harvest: lessons from wildfire. *Biodiversity and Conservation* 5, 1191–1205.
- Dodd, C.J.H., McLean, A., Brink, V.C., 1972. Grazing values as related to tree-crown covers. *Canadian Journal of Forest Research* 2, 185–189.
- ESSA Technologies, 2007. *Vegetation Dynamics Development Tool User Guide. Version 6.0*. ESSA Technologies, Vancouver, BC, 196 p. <http://www.essa.com/documents/vddt/VDDT-60-User-Guide.pdf>
- ESSA Technologies, 2008. *TELSA: Tool for Exploratory Landscape Scenario Analyses, Model Description. Version 3.6*. ESSA Technologies, Vancouver, BC, 64 p. <http://www.essa.com/documents/telsa/ModelDescription.pdf>
- Fall, A., Fortin, M.-J., Kneeshaw, D.D., Yamasaki, S.H., Messier, C., Bouthillier, L., Smyth, C., 2004. Consequences of various landscape-scale ecosystem management strategies and fire cycles on age-class structure and harvest in boreal forests. *Canadian Journal of Forest Research* 34, 310–322.
- Fernandes, P.M., Botelho, H.S., 2003. A review of prescribed burning effectiveness in fire hazard reduction. *International Journal of Wildland Fire* 12, 117–128.
- Filmon, G., 2004. *Firestorm 2003 Provincial Review*. Government of British Columbia, Victoria, BC, 100 p. www.2003firestorm.gov.bc.ca
- Finney, M.A., 2006. A computational method for optimizing fuel treatment locations. In: Andrews, P.L., Butler, B.V. (Eds.), *Fuels Management—How to Measure Success: Conference Proceedings*, 28–30, March 2006, Portland, OR. USDA Forest Service, Rocky Mountain Research Station Proceedings RMRS-P-41, Fort Collins, CO.
- Finney, M.A., McHugh, C.W., Grenfell, I.C., 2005. Stand- and landscape-level effects of prescribed burning on two Arizona wildfires. *Canadian Journal of Forest Research* 35, 1714–1722.
- Flannigan, M.D., Logan, K.A., Amiro, B.D., Skinner, W.R., Stocks, B.J., 2005. Future area burned in Canada. *Climate Change* 72, 1–16.
- González-Cabán, A., 1997. Managerial and institutional factors affect prescribed burning costs. *Forest Science* 43, 535–543.
- Graham, R.T., McCaffrey, S., Jain, T.B. (Eds.), 2004. *Science Basis for Changing Forest Structure to Modify Wildfire Behavior and Severity*. USDA Forest Service, Rocky Mountain Research Station, RMRS-GTR-120, Ft. Collins, CO.
- Gustafson, E.J., Crow, T.R., 1996. Simulating the effects of alternative management strategies on landscape structure. *Journal of Environmental Management* 46, 77–94.
- Halls, L.K., Schuster, J.L., 1965. Tree-herbage relations in pine-hardwood forests of Texas. *Journal of Forestry* 63, 282–283.
- Hansen, A.J., Garman, S., Marks, L.B., 1993. An approach for managing vertebrate diversity across multiple-use landscapes. *Ecological Applications* 3, 481–496.
- He, H.S., 2008. Forest landscape models: definitions, characterization and classification. *Forest Ecology and Management* 254, 484–498.
- He, H.S., Mladenoff, D.J., Gustafson, E.J., 2002. Study of landscape change under forest harvesting and climate warming-induced fire disturbance. *Forest Ecology and Management* 155, 257–270.
- Hesseln, H., 2000. The economics of prescribed burning: a research review. *Forest Science* 46, 332–334.
- Hunter, M.L., 1990. *Wildlife, Forests, and Forestry: Principles of Managing Forests for Biological Diversity*. Prentice-Hall, New Jersey.
- Hutto, R.L., 2008. The ecological importance of severe wildfires: some like it hot. *Ecological Applications* 18, 1827–1834.
- Keeley, J.E., Fotheringham, C.J., Moritz, M.A., 2004. Lessons from the October 2003 wildfires in Southern California. *Journal of Forestry* 102, 26–31.
- Klenner, W., Sullivan, T.P., 2009. Partial and clearcut harvesting of dry Douglas-fir forests: implications for small mammal communities. *Forest Ecology and Management* 257, 1078–1086.
- Klenner, W., Kurz, W., Beukema, S., 2000. Habitat patterns in forested landscapes: management practices and the uncertainty associated with natural disturbances. *Computers and Electronics in Agriculture* 27, 243–262.
- Klenner, W., Walton, R., Arsenault, A., Kremsater, L., 2008. Dry forests in the southern interior of British Columbia: historic disturbances and implications for restoration and management. *Forest Ecology and Management* 256, 1711–1722.
- Kline, J.D., 2004. *Issues in Evaluating the Costs and Benefits of Fuel Treatments to Reduce Wildfire in the Nation's Forests*. USDA Forest Service Research Note PNW-RN-542, Portland, OR.
- Knowles, R.L., Horvath, G.C., Carter, M.A., Hawke, M.F., 1999. Developing a canopy closure model to predict overstorey/understorey relationships in *Pinus radiata* silvopastoral systems. *Agroforestry Systems* 43, 109–119.
- Kurz, W.A., Beukema, S.J., Klenner, W., Greenough, J.A., Robinson, D.C.E., Sharpe, A.D., Webb, T.M., 2000. TELSA: the tool for exploratory landscape scenario analyses. *Computers and Electronics in Agriculture* 27, 227–242.
- Landram, F.M., Laudenslayer Jr, W.F., Atzet, T., 2002. Demography of snags in Eastside pine forests in California. In: Laudenslayer Jr, W.F., Shea, P.J., Valentine, B.E., Weatherspoon, C.P., Lisle, T.E. (Eds.), *Proceedings of the Symposium on the Ecology and Management of Dead Wood in Western Forests*. 1999 November 2–4, Reno, Nevada. Pacific Southwest Research Station, USDA Forest Service General Technical Report. PSW-GTR-181, pp. 605–620.
- Leitão, A.B., Ahern, J., 2002. Applying landscape ecological concepts and metrics in sustainable landscape planning. *Landscape and Urban Planning* 59, 65–93.
- Martin, K., Norris, A., Drever, M., 2006. Effects of mountain pine beetle outbreaks on avian biodiversity in the British Columbia interior: implications for critical habitat management. *BC Journal of Ecosystems and Management* 7, 10–24.
- Meidinger, D., Pojar, J., 1991. *Ecosystems of British Columbia*. Research Branch, Ministry of Forests, Victoria, BC.
- Ministry of Forests, 2001. *Kamloops Timber Supply Area Analysis Report*, July 2001. BC Ministry of Forests, Victoria, BC.
- Mitchell, K.J., 1975. Dynamics and simulated yield of Douglas-fir. *Forest Science Monograph* 17.
- Mitchell, J.E., Bartling, P.N.S., 1991. Comparison of linear and nonlinear overstorey-understorey models for ponderosa pine. *Forest Ecology and Management* 42, 195–204.
- Mladenoff, D.J., 2004. LANDIS and forest landscape models. *Ecological modelling* 180, 7–19.
- Mladenoff, D.J., White, M.A., Pastor, J., Crow, T.R., 1993. Comparing spatial pattern in unaltered old-growth and disturbed forest landscapes. *Ecological Applications* 3, 294–306.
- Murphy, E.C., Lehnhausen, W.A., 1998. Density and foraging ecology of woodpeckers following a stand-replacement fire. *Journal of Wildlife Management* 62, 1359–1372.
- Pase, C.P., 1958. Herbage production and composition under immature ponderosa pine stands in the Black Hills. *Journal of Range Management* 11, 238–243.
- Peek, J.M., Dennis, B., Hershey, T., 2002. Predicting population trends of mule deer. *The Journal of Wildlife Management* 66, 729–736.
- Peter, B., Nelson, J., 2005. Estimating harvest schedules and profitability under the risk of fire disturbance. *Canadian Journal of Forest Research* 35, 1378–1388.
- Pickett, S.T.A., White, P.S. (Eds.), 1985. *The Ecology of Natural Disturbance and Patch Dynamics*. Academic Press, Orlando, USA.
- Poole, K.G., Mowat, G., 2005. Winter habitat relationships of deer and elk in the temperate interior mountains of British Columbia. *Wildlife Society Bulletin* 33, 1288–1302.
- Radeloff, V.C., Mladenoff, D.J., Gustafson, E.J., Scheller, R.M., Zollner, P.A., He, H.S., Akcakaya, H.R., 2006. Modelling forest harvesting effects on landscape pattern in the Northwest Wisconsin Pine Barrens. *Forest Ecology and Management* 236, 113–126.
- Rideout, D.B., Omi, P.N., 1995. Estimating the cost of fuels treatment. *Forest Science* 41, 664–674.
- Romme, W.H., Clement, J., Hicke, J., Kulakowski, D., MacDonald, L.H., Schoennagel, T.L., Veblen, T.T., 2006. *Recent Forest Insect Outbreaks and Fire Risk in Colorado Forests: A Brief Synthesis of Relevant Research*. Colorado Forest Restoration Institute, Ft. Collins, CO.

- Russell, R.E., Saab, V.A., Dudley, J.G., Rotella, J.J., 2006. Snag longevity in relation to wildfire and postfire salvage logging. *Forest Ecology and Management* 232, 179–187.
- Schneider, R.R., Stelfox, J.B., Boutin, S., Wasel, S., 2003. Managing the cumulative impacts of land-uses in the Western Canadian Sedimentary Basin: a modeling approach. *Conservation Ecology* 7 (8) Available at <<http://www.ecologyand-society.org/vol7/iss1/art8/>>
- Sheppard, S.R.J., Harshaw, H.W. (Eds.), 2001. *Forests and Landscapes: Linking Ecology, Sustainability, and Aesthetics*. IUFRO Research Series No. 6. CABI, 294 p.
- Shifley, S.R., Thompson III, F.R., Dijak, W.D., Larson, M.A., Millspaugh, J.J., 2006. Simulated effects of forest management alternatives on landscape structure and habitat suitability in the Midwestern United States. *Forest Ecology and Management* 229, 361–377.
- Smucker, K.M., Hutto, R.L., Steele, B.M., 2005. Changes in bird abundance after wildfire: importance of fire severity and time since fire. *Ecological Applications* 15, 1535–1549.
- Snider, G., Daugherty, P.J., Wood, D., 2006. The irrationality of continued fire suppression: an avoided cost analysis of fire hazard reduction treatments versus no treatment. *Journal of Forestry* 104, 431–437.
- Stephens, S.L., Ruth, L.W., 2005. Federal forest-fire policy in the United States. *Ecological Applications* 15, 532–542.
- Sullivan, T.P., Sullivan, D.S., 2001. Influence of variable retention harvests on forest ecosystems. II. Diversity and population dynamics of small mammals. *Journal of Applied Ecology* 38, 1234–1252.
- Vyse, A., Smith, R.A., Bondar, B.G., 1991. Management of interior Douglas-fir stands in British Columbia: past, present and future. In: Baumgartner, D.M. (Ed.), *Interior Douglas-fir: The Species and its Management*. Symposium Proceedings. Washington State University, Department of Natural Resource Sciences, Pullman, WA.
- Wallin, D.O., Swanson, F.J., Marks, B., 1994. Landscape pattern response to changes in pattern generation rules: land use legacies in forestry. *Ecological Applications* 4, 569–580.
- Westfall, J., 2006. *Summary of Forest Health Conditions in British Columbia*. British Columbia Ministry of Forests and Range, Forest Practices Branch, Victoria, BC.
- Wong, C.M., 1999. *Memories of Natural Disturbances in Ponderosa pine–Douglas-fir age Structure, Southwestern British Columbia*. Master of Natural Resource Management Thesis. Simon Fraser University, Burnaby, British Columbia.
- Zimmerman, G.T., Neuenschwander, L.F., 1984. Livestock grazing influences on community structure, fire intensity, and fire frequency within the Douglas-fir/ninebark habitat type. *Journal of Range Management* 37, 104–110.
- Zollner, P.A., Roberts, L.J., Gustafson, E.J., He, H.S., Radeloff, V., 2008. Influence of forest planning alternatives on landscape pattern and ecosystem processes in northern Wisconsin, USA. *Forest Ecology and Management* 254, 429–444.