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Integrating climate change into forest management in South-Central British Columbia: An assessment of landscape vulnerability and development of a climate-smart framework

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ABSTRACT

The achievement of sustainable forest management requires the incorporation of risk and uncertainty into long-term planning. Climatic change will have significant impacts on natural disturbances, species and ecosystems, particularly on landscapes influenced by forest management. Understanding where vulnerabilities lie is important in managing the risks associated directly or indirectly with climatic change. The vulnerability of landscapes to natural disturbances, the resilience of ecosystems and distribution of species are all important components that need to be considered when undertaking forest planning, but climatic change is rarely factored into such planning. In this study, the vulnerability of fire potential, fire regimes, ecosystems and species to climatic change was modelled for a 145,000 ha landscape in the south-central interior of British Columbia, Canada. The results from these analyses were used to guide forest zoning, using the triad zoning framework, and for the development of a ''climate-smart'' management framework. The use of climate-smart management is advocated as a decision-making framework for managing forested landscapes based on an understanding of landscape vulnerability to future climatic change. From this understanding, the maintenance of ecosystem health and vitality could be achieved.

1. Introduction

Many conceptual frameworks have been developed that seek to either assess the impacts of climate change on systems (e.g., Ford and Smit, 2004) or to provide guidance on how to adapt management to the consequences of climate change (e.g., Ohlson et al., 2005; Tschakert and Olsson, 2005). Although many frameworks have been developed, most focus on single aspects (Lindner et al., 2002) or operate at a scale that is inconsistent with the needs of managers (Luers, 2005). Most existing frameworks therefore provide limited assistance to forestry decision-making, as they fail to identify vulnerability at the scale at which direct management actions are undertaken. Although the connection between consequences and decision-making have not been fully integrated by frameworks that provide a top-down approach, for example, frameworks that focus on adaptation and mitigation (i.e., Ohlson et al., 2005); they can be made effective by linking them to bottom-up approaches (Jones, 2001). Hinckley et al. (1998) states that top-down approaches have the appeal of parsimony although they are frequently restricted by uncertainty, while bottom-up approaches reduce this uncertainty through the examination of the driving principles and mechanisms that affect systems. To incorporate adaptation and mitigation strategies into long-term forest planning, an understanding of ecosystem and landscape vulnerabilities is therefore required so that the biophysical implications of management actions can be addressed (Duinker, 1990). This requires a framework that can evaluate the mechanisms which have a significant influence on our ability to manage landscapes in a sustainable manner. Manning et al. (2004) support this. They surmise that any assessment of climate change should seek to identify the determinants of uncertainty along with any conceptual or structural limits. Support is also provided by Turner et al. (2003) who state the need for developing conceptual frameworks with diverse and complex linkages that can account for the vulnerability of coupled human–environment systems.

In the context of sustainable forest management (SFM), which involves planning for long time horizons (100–200 years), the need to address the potential vulnerability over time and space is critical if current planning decisions and objectives are to be achievable (Turner et al., 2003). Spatial and temporal assessments of

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landscape vulnerability can be used to provide and understanding of potential ecosystem response to climatic change which in turn will remove some of the uncertainty on how to manage these systems. To achieve a reliable understanding we need to use meta-frameworks, essentially frameworks that operate from the bottom-up to provide the understanding of risk and uncertainty which will then allow for the effective use of top-down approaches. Turner et al. (2003) and Ohlson et al. (2005) all advocate the use of frameworks that integrate vulnerability assessments to identify the risks and reduce the uncertainty associated with climatic change.

von Gadow (2001) argued that identifying risks in forest management is difficult because both hazard probabilities and the consequences of a hazard are driven, either directly or indirectly, by allogenic or biogenic processes that are difficult to predict. The complexity of social and ecological systems also presents difficult forms of uncertainty and risk for land managers (Borchers, 2005). This will lead to any assessment of risk still containing a substantial degree of uncertainty, and the potential impacts of climate change increase that uncertainty. However, unlike other sources of uncertainty, that associated with climate change is non-stationary (White, 2004). Oppenheimer (2005) has argued that there is no unique system for categorising the risks posed by climate change, although risks can be separated as systematic or unsystematic (Figge, 2004). Risk is typically defined as the probability and severity of adverse effects (O'Laughlin, 2005). Haimes (1998) describes risk as having two components; one real (consequence) and one imagined (probability). In the context of climate change, risk has been defined as a function of hazard and vulnerability (risk = hazard $\times$ vulnerability) (Brooks et al., 2005). A hazard is defined as the cause of an adverse effect (O'Laughlin, 2005). Vulnerability is defined as the degree to which a system or system component is susceptible to sustaining damage from a hazard (Turner et al., 2003). This vulnerability-based definition of risk is supported by the Intergovernmental Panel on Climate Change (IPCC) (1998) as the best definition for assessing climate change as it allows for the assessment of system vulnerabilities rather than expected impacts. Vulnerability assessments need to be robust and consider the hazards (perturbations or stressors) that can affect the resilience of the system in question. Deciding what needs to be assessed in a framework is therefore an important step that requires the consideration of the hazards that will likely influence our ability to achieve SFM under climatic change. Identification of these hazards is required if risk reduction and spreading options (i.e., adaptation and mitigation strategies) are to be developed. The degree of risk that is identified by a vulnerability assessment will depend on the perceptions of the stakeholders (Weiss, 2001). For this reason, it is therefore important to identify the vulnerability of ecosystems to climatic change, so that stakeholders understand the potential hazards. Such awareness requires that both current problems and new threats are communicated in a transparent fashion (Blennow and Sallnäs, 2005). In addition to identifying vulnerabilities, it is also important to identify the climatic thresholds that may result in critical, irreversible impacts on a system, as these are required to inform adaptation strategies and actions that can be accomplished through the incorporation of vulnerability into planning (Jones, 2004).

The selection of vulnerability variables generally reflects the perceptions and values of the stakeholders, and the method used to evaluate the variables depends on the system being assessed. Consequently, it is not always possible to identify a method that is suitable for all cases, resulting in many assessment frameworks being generic rather than specific to a particular system or management area. Hollenstein (2001) developed a framework that integrates policy and risk analysis for use in forest planning that has the following structure: (1) define the system; (2) select risk variables; (3) select analysis technique; (4) calculate the risk for each variable; (5) aggregation of risk variables; (6) select management action. Step 5 can be excluded with management decisions (step 6) being based on step 4 (Hollenstein, 2001). The Unites States Environmental Protection Agency (US-EPA) has also developed guidelines for ecological risk assessment to evaluate the likelihood that adverse ecological effects may occur as a result of exposure to single or multiple stressors (US-EPA, 1998). The US-EPA (1998) defines an adverse ecological effect as an undesirable change to the structural or functional characteristics of an ecosystem or their components as the result of exposure to one or more stressors. A stressor is defined as any physical, chemical, or biological entity that can induce an adverse response (US-EPA, 1998). The US-EPA's ecological assessment framework has three main phases: (1) problem formulation, (2) analysis, and (3) risk characterisation. Problem formulation integrates available information to determine assessment endpoints and to identify the relationship between stressors and endpoints which then determines the analysis plan (US-EPA, 1998). The analysis phase investigates the two primary components of risk, exposure and effects, to determine or predict the ecological responses to stressors under exposure conditions (US-EPA, 1998). The risk characterisation phase is the final step where relationships between stressors, effects, and ecological entities are clarified and conclusions drawn about the occurrence of the exposure and adversity of existing or anticipated effects (US-EPA, 1998). An ecological entity is a component of an assessment endpoint that refers to a species, group of species, an ecosystem function or characteristic, or a specific function (US-EPA, 1998). An assessment endpoint is an explicit value that is to be protected or maintained (US-EPA, 1998). The frameworks developed by Hollenstein and the US-EPA are congruent with each other with many of Hollenstein's steps providing a means to meet the objectives require to conduct each phase of the US-EPA's ecological risk assessment.

In this study, the phases and steps of the US-EPA and Hollenstein frameworks were used to develop a framework that could be utilised to integrate climate change into SFM planning, through a assessment of stressors and ecological entities, to characterise landscape vulnerability in order to develop a decision-making process from which management decisions that can be made to achieve the assessment endpoint under the uncertainty of predicted climate change. The framework was then tested on a 145,000 ha landscape in South-Central British Columbia, Canada. The results of the vulnerability assessment were used to develop a decision-making framework for reducing the vulnerability and consequent risk of the tested landscape to climate change.

The phases and steps used in the vulnerability assessment are as follows:

- Phase 1. Problem formulation (methods)
  - Step 1. Definition of the system
  - Step 2. Selection of assessment variables
  - Step 3. Development of a conceptual model
  - Step 4. Selection of an analysis technique
- Phase 2. Vulnerability analysis (results)
  - Step 5. Calculating the vulnerability of stressor/response variables
  - Step 6. Development of management framework
- Phase 3. Characterisation of vulnerability (discussion and conclusion)

2. Problem formulation: methods

2.1. Step 1. Definition of the system

The system evaluated in this study was Tolko Industries Ltd.'s (Tolko) Tree Farm License (TFL) 49 near Kelowna, British Columbia, Canada (50°20′N, 119°55′W) (Fig. 1). The TFL 49 has an area of ca. 145,000 ha and is managed under an ecological stewardship plan (Riverside Forest Products Ltd., 2004). The implementation of this plan involves the allocation of the landscape through triad zoning, with the dual objectives of maintaining timber supply and conserving biodiversity. Three zones with fundamentally different management objectives were identified: a production (intensive) zone, an extensive (multiple use) zone and an ecological (reserves) zone (Seymour and Hunter, 1992). The objective of this triad approach is to produce a landscape design and management system that will conserve biodiversity and provide for other societal demands (Seymour and Hunter, 1999; Sample, 2005). The first step in the implementation of the triad system is an acceptance by forest managers that some of the landscape be set aside as reserves to ensure biodiversity conservation (Seymour and Hunter, 1999). This requires that representative samples of all ecosystems and areas containing key habitat elements and functions being delineated into reserves (Norton, 1998; Lindenmayer and Franklin, 2002). The second step is to assess the potential of the unreserved landscape for commodity production. Where timber demands are low, extensive/ecological forestry can be practiced; however, where timber demands are high, portions of the land should be managed under production/intensive silviculture to offset reserved land (Seymour and Hunter, 1999). Management in the extensive zone should be aimed at providing habitat elements at smaller spatial scales, buffering reserve zones and maintaining connectivity throughout the landscape (Franklin, 1993). Studies into use of the triad approach across North America have resulted in the proposed partitioning of a landscape into equal areas of reserve zone and intensive zone; however, the proposed amount of the landscape to be occupied by each zone varied between 10 and 25%, leaving 50–80% of the proposed landscapes to be placed into the extensive zone (Nitschke and Innes, 2005).

The landscape contains five forested ecosystems, classified as biogeoclimatic (BEC) zones (Meidinger and Pojar, 1991): the ponderosa pine (PP) zone, interior Douglas-fir (IDF) zone, Montane spruce (MS) zone, Engelmann spruce-Subalpine fir (ESSF) zone and the interior Cedar hemlock (ICH) zone. Analysis of Tolko's TFL 49 GIS database using ArcGIS 9.0 (Environmental Systems Research Institute (ESRI) 2004) identified that 13 tree species also occur in the landscape. Bunnell et al. (1999) suggested that 193 vertebrate species potentially occur in the landscape.

2.2. Step 2. Selection of assessment variables

Selecting variables for an assessment framework requires the incorporation of the values and risks of relevant stakeholders, while recognising that the perceptions of risk will most vary across social groups (Weiss, 2001). Choosing variables is thus a value-laden process. Consequently, the selection of variables usually reflects the values of the main stakeholders, in this case forest managers working in conjunction with community consultation and participation.

Forest fire, mountain pine beetle (*Dendroctonus ponderosae*) outbreaks, drought, loss of ecosystem resilience and loss of current biological diversity have all been identified by Tolko (Riverside Forest Products Ltd., 2004) as threats to achieving SFM on the TFL 49 landscape. This is consistent with the British Columbia Ministry of Forests and Range (MOFR) (2006), which has identified the loss of ecosystem resilience due to climate change as a major threat and that climate-induced changes in natural disturbance regimes, in particular fire and insect and pathogen epidemics, are likely to have major impacts on ecosystem resilience and biological conservation. The threats identified by the stakeholders in this study impact upon many criteria used to determine SFM but are directly relevant to the criterion identified by The Montréal Process (Montréal Process Technical Advisory Committee, 2003): the maintenance of forest ecosystem health and vitality. This criterion was selected as the assessment endpoint due to its relevance to SFM in relation and relationship with the threats identified by stakeholders. The metric(s) used to assess the endpoint may vary from region to region and over time. For example, Hickey and Nitschke (2007) identified 123 potential indicators being used for this criterion across the Pacific Rim region. Gough et al. (2008) found that public expectation of forest management is always evolving and as a result new indicators are constantly being proposed. In this study, the most relevant metrics to be used are: (1) a measure of the area and percent of forest land with diminished biological components indicative of changes in

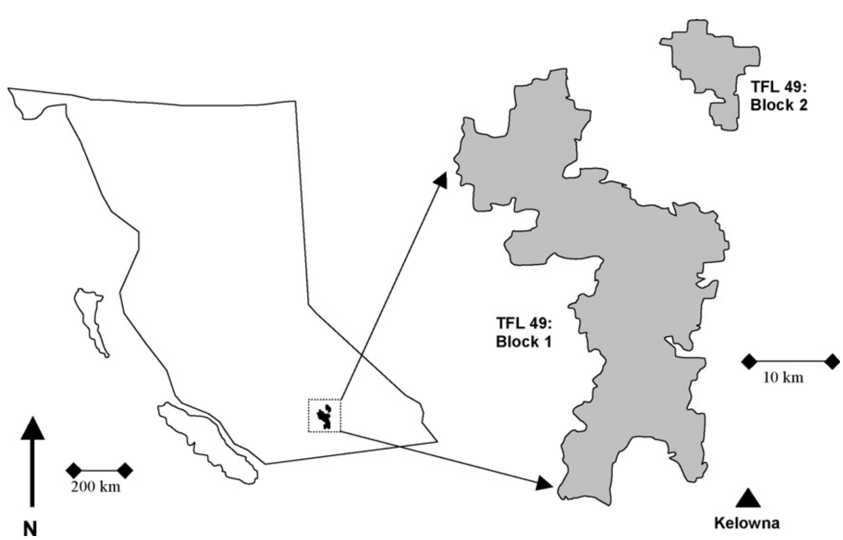

Fig. 1. Tree Farm License 49 landscape in the North Okanagan region of British Columbia, Canada.

fundamental ecological processes and/or ecological continuity; (2) area and percent of forest affected by processes or agents beyond the range of historic variation; (3) the success of forest regeneration following disturbance. These indicators provide measures of the status of fundamental ecological processes that underpin the maintenance of ecosystem health and vitality (Canadian Council of Forest Ministers, 2003; The State of Victoria, 2005).

2.3. Step 3. Development of a conceptual model

Having defined the system and identified the framework variables, we can create a conceptual model that describes the key relationships between stressors and the assessment endpoint (US-EPA, 1998). This model allows for the integration of multiple stressors to represent landscape vulnerability to climate change with the forest management objective of maintaining forest ecosystem health and vitality as an assessment endpoint (Fig. 2). The model provides a simple method for illustrating the relationship between the source, stressors, ecological entities, and endpoints into the context of decision-making for forest managers. The source in this model refers to the place where the stressors originate from (US-EPA, 1998). This makes it a simple and flexible decision-support framework, as recommended by Ohlson et al. (2005). The proposed framework combines vulnerability assessments of the following ecological entities: natural disturbances, floral and faunal species and species habitat. The framework does not contain an explicit temporal component, as vulnerability assessments require the evaluation of both current and future vulnerabilities in a system. The framework elements need to be examined in combination to provide decision support on the allocation of a landscape into management zones and how to manage within these zones. The goals and objectives of forest managers will vary from landscape to landscape, as will the stressors; however, conceptualising where the vulnerabilities and

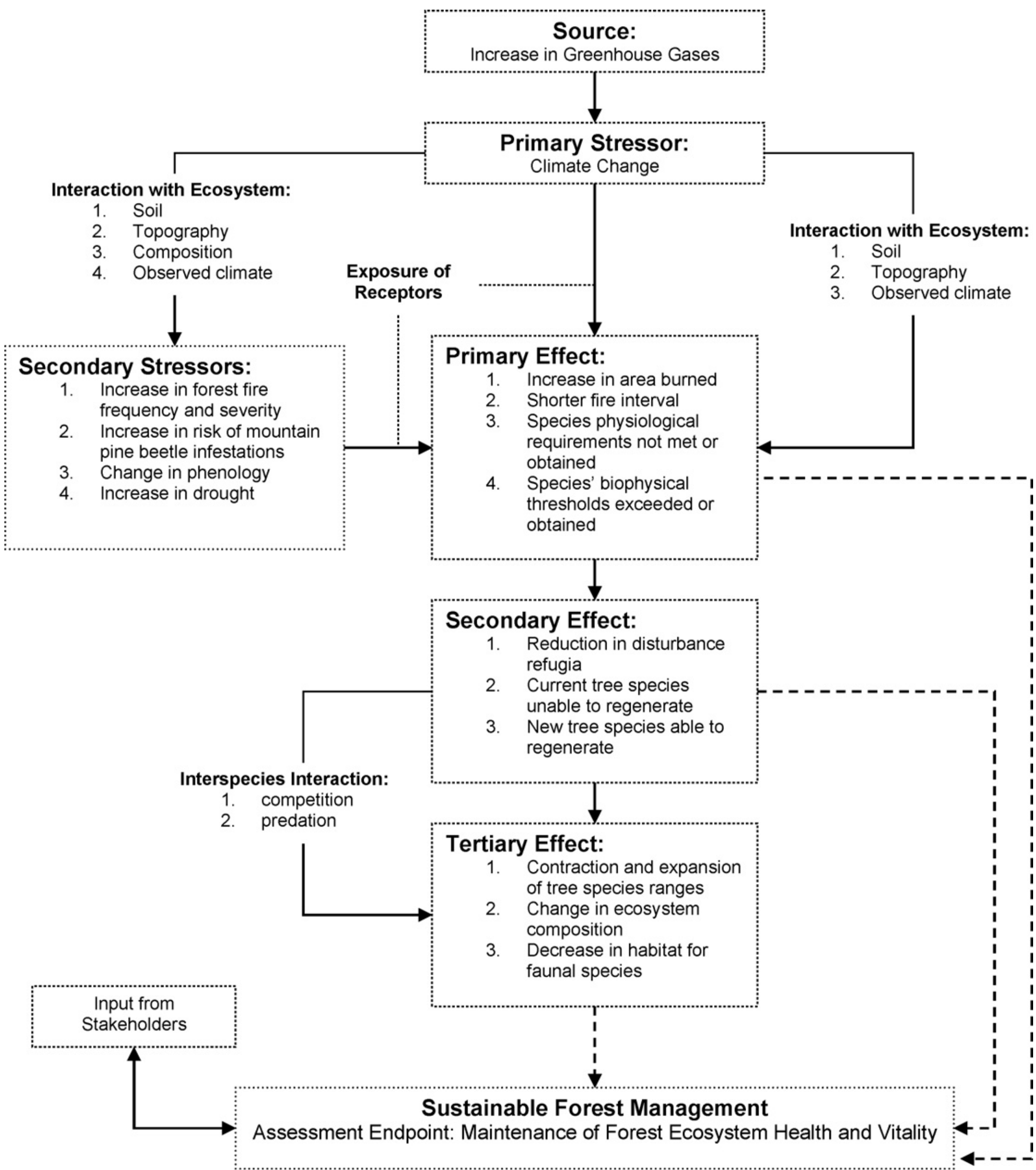

Fig. 2. Conceptual model for assessing landscape vulnerability to climate change in South-Central British Columbia. Dotted lines indicate pathways were receptors are exposed to stressors; dashed lines indicate where effects influence assessment endpoint. Arrow to stakeholder input illustrates the link between stakeholder perceptions of landscape vulnerability to predicted climate change and measures of ecosystem health and vitality.

management objectives meet can provide the starting point for selecting a technique for conducting an integrated assessment.

2.4. Step 4. Selection of an analysis technique

Planning over the time-scales relevant to SFM requires the use of decision support systems (DSS) that can examine the behaviour of a system under alternative scenarios and management decisions. Modelling is being increasingly used to do this (e.g., Messier et al., 2003). Modelling is one technique that can be used to conduct an ecological response analysis in order to assess the potential impact of stressors on an endpoint (US-EPA, 1998).

SFM planning requires a methodology that offers a trade-off between complexity and precision, such as the meta-model concept (Messier et al., 2003). Meta-modelling incorporates the strengths of many smaller models into a framework where the outputs of one model become the inputs of another (Luxmoore et al., 2002). By using the meta-model approach the issue of decreased precision and predictability commonly faced by complex, high-resolution models is avoided (Costanza and Voinov, 2004). The methodology permits the integration of climate change into forest planning (see Fig. 3) by allowing for an integrated assessment of multiple variables under multiple scenarios of climatic change. Each modelled component can be evaluated separately and then functionally coupled in a refined hierarchy with all components (Muzy et al., 2005). This approach enabled the integration of the multiple determinants driving the system.

The meta-model framework used in this study (Fig. 3) integrated multiple models to assess the impact of climatic change on the identified variables. The models used were:

1. ArcGIS 9.0 (ESRI, 2004): a geographic information system used to integrate the results of each modelling component/analysis into the same spatial information system. ArcGIS 9.0 was used to overlay raster coverages from the analysis of each risk variable, each raster coverage had a resolution of 50 × 50 m (0.25 ha).

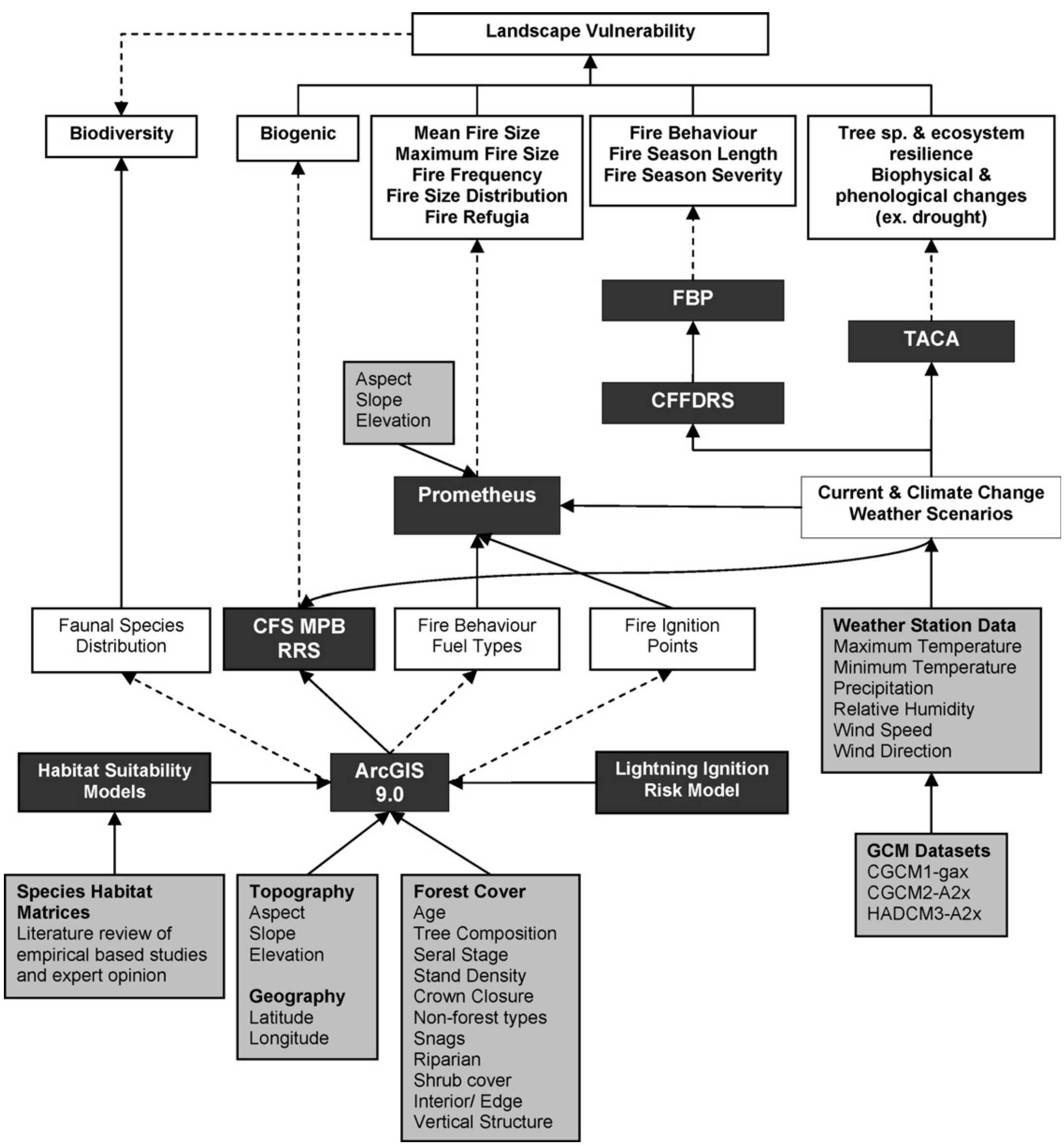

Fig. 3. Meta-model framework for analysing landscape vulnerability. Data inputs are presented in light grey boxes, models in dark grey boxes and data outputs in white boxes. Solid lines indicate data used within a model or analysis, while dashed lines illustrate data outputs (non-bolded) and results (bolded) from a particular model. Table 2 summarises the specific results of the landscape vulnerability analysis.

Table 1
Average predicted annual climate change for the North Okanagan, BC, Canada (ensemble predictions from three GCM models)

Climate period	Minimum temperature (°C)	Mean temperature (°C)	Maximum temperature (°C)	Precipitation (%)	Relative humidity (%)	Wind velocity (%)
2010–2039	1.1	1.1	1.2	2.9	−0.98	0.4
2040–2069	2.2	2.4	2.7	−1.8	−1.96	−1.4
2070–2100	3.8	4.0	4.3	1.7	−2.75	−3.6

2. Prometheus (CWFGM Steering Committee, 2003): a landscape-level fire growth model used to assess fire growth and spread in relation to fuel type, topography and weather.
3. Canadian Forest Service Mountain Pine Beetle Risk Rating System (CFS MPB RRS) (Shore and Safranyik, 1992).
4. Tree and Climate Assessment Model (TACA): an individual tree response model, developed for evaluating the vulnerability of tree species to climatic change in their fundamental regeneration niche (Nitschke and Innes, 2008a).
5. Eighty-one habitat suitability models developed from empirical studies and expert opinion (see Nitschke, 2006 for further detail).
6. Canadian Forest Fire Danger Rating System (CFFDRS) (Van Wagner, 1987).
7. Canadian Forest Fire Behaviour Prediction (FBP) System (Hirsch, 1996); the latter two models were used to evaluate fire season severity, length, and fire behaviour (Nitschke and Innes, 2008b).

The framework additionally incorporates weather station data and Global Climate Change Model (GCM) predictions. Nitschke and Innes (2008b) provide a detailed description on the generation of multiple climate scenarios utilised in this study to represent the variability in observed climate and predicted future climate. A brief

Table 2
Stressor/response profiles from the ecological response analysis of the TFL 49 landscape

Variables	Response and vulnerability to predicted climate change within the study area		
Fire season length	27% increase by 2085 in the Spring	2% increase by 2085 in the summer	1% increase by 2085 in the autumn
Fire season severity	40% increase by 2085 in the Spring**	95% increase by 2085 in the summer**	30% increase by 2085 in the autumn**
Fire behaviour	Increase in fire behaviour in all ecosystems by 2085 (PP, IDF, ESSF)+, (MS, ICH)*. Increase in intermittent to full crown fire behaviour		
Mean fire size	Increase in mean fire size by 2025*; by 2055 and 2085**. Overall, mean fire size increased from 641 to 1861 ha by 2085		
Maximum fire size	Average maximum fire size increased from 7961 to 19,076 ha by 2085. Largest fire size modelled increased was 35,390 ha in the 2055 climate period		
Fire size frequency distribution	Shift in distribution with fires <1000 ha in size occurring less frequently and fires >1000 ha occur more often by 2085. Increase in fires in the 5000 and 10,000 ha fire size classes and the presence of fires in the 20,000 and 50,000 ha size classes		
Fire frequency	A shift from the current 34% of the landscape experiencing frequent fires every 50 years or less to 93% of the landscape by 2085		
Fire refugia	Reduction in fire refugia area from 41 to 1.9% of the landscape by 2085		
Fire ignition	Proportion of landscape in each lightning fire ignition risk category: low (6%); medium (28%), high (48%); very high (18%)		
Tree species resilience	76.7% of the landscape area has at least one tree species at risk. Five species classified as at extreme risk (>90% contraction in range), five species at high risk (>30 to <70% contraction in range), three species classified as at low risk (net expansion but contraction of range at lower elevations). Ponderosa pine (*Pinus ponderosa*), Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) and western redcedar (*Thuja plicata*) had their regeneration niche expand to higher elevations but reduced at lower elevations. Lodgepole pine (*Pinus contorta* var. *latifolia*) was rated at high risk; its range was predicted to contract by 58.3%. Landscape became climatically suitable for grand fir (*Abies grandis*) and western white pine (*Pinus monticola*). Both species are major components in Northern Washington ecosystems		
Biophysical and phenological	Increases in heat stress, soil water deficits (drought stress), and growing season frosts. Reductions in bud burst date and the number of chilling weeks		
Ecosystem	38.7% of the area is occupied by highly vulnerable ecosystems		
Resilience	PP zone may become a Bunchgrass zone or savanna/woodland ecosystem dominated by sagebrush sp. (*Artemisia* sp.) and/or juniper sp. (*Juniperus* sp.) by 2085. IDF zone may become the new PP zone by 2085. ICH zone will likely resemble an integration of the current IDF and ICH zones; it is proposed that the ICH could become the Interior Cedar–Douglas-fir (ICDF) zone by 2085. 70% of species were classified at medium to high risk. Species requiring late-successional habitat in the ESSF and MS zones are at the highest risk. Species requiring mid to late successional habitat in the PP, IDF and ICH zones are at medium to high risk. These species are at risk due to the predicted increase in fire frequency and severity and loss of ecosystem resilience which will directly and indirectly reduce habitat and alter ecosystem function over time. The current MS zone could shift to a new stable state that may be comprised of multiple stable communities; it is proposed that this zone could become an Interior Mixed Conifer (IMC) zone, with ponderosa pine being the most abundant early to mid successional species. The ESSF zone, in the absence of fire, could develop into a mixed cedar/true fir forest dominated by western redcedar and grand fir. Pioneer stands could be dominated either by ponderosa pine, lodgepole pine, western white pine, Douglas-fir, western larch (*Larix occidentalis*) or trembling aspen (*Populus tremuloides*), or mixtures of all species. The current ESSF zone may become an Interior Cedar-Grand Fir (ICGF) zone by 2085 and beyond		
Biogenic	43.2% of the TFL 49 landscape is at some degree of risk from the current to mountain pine beetle (*Dendroctonus ponderosae*) epidemic. By 2085 vulnerability could increase to 51.5% if lodgepole pine is maintained		
Faunal species vulnerability	70% of species were classified at medium to high risk. Species requiring late-successional habitat in the ESSF and MS zones are at the highest risk. Species requiring mid to late successional habitat in the PP, IDF and ICH zones are at medium to high risk. These species are at risk due to the predicted increase in fire frequency and severity and loss of ecosystem resilience which will directly and indirectly reduce habitat and alter ecosystem function over time		

Results used to determine landscape vulnerability to climate change in order to evaluate assessment endpoint. Levels of significance: (+) 0.10, (*) 0.05, (**) 0.01.

description is provided; readers should however refer to Nitschke and Innes (2008b) for further explanation. The climate change scenarios were produced by the Canadian Global Circulation Model I (CGCM1-gax), Canadian Global Circulation Model II (CGCM2-A2x), and the Hadley Centre Global Circulation Model III (HadCM3-A2x) global circulation models. For a description of the Canadian models see Flato et al. (2000) and for the Hadley model see Johns et al. (2003). Climate change outputs were obtained from the Canadian Institute for Climate Studies, Canadian Climate Impacts and Scenarios Project (2005). For each GCM, the relevant grid point immediately above the study region was utilised for the output periods 2025 (2010–2039), 2055 (2040–2069), and 2085 (2070–2100). The average annual predicted changes for the region, based on an ensemble of the GCM models, are provided in Table 1.

The meta-model methodology provides direction and boundaries from which current and future management decisions can be evaluated. The use of spatially explicit models in the framework also provides a method for studying the ecological processes in the research area, from local to landscape scales, with a link to the global scale that climate change occurs at. The approach provides a methodology to assess ecosystem change in response to climatic change (Beissinger and Westphal, 1998). Multiple scenarios of climate change were incorporated to test the range of potential behaviour of natural disturbances and ecosystem response, enabling problems with increased uncertainty, interdependence and complexity to be addressed (Schoemaker, 1993). A multiple scenario approach, used in conjunction with the DSS, allowed the boundaries of landscape and ecosystem vulnerability under uncertain future conditions to be defined.

3. Vulnerability analysis: results

3.1. Step 5. Calculating the vulnerability of stressor/response variables

The detailed results of the ecological response analysis used to assess landscape vulnerability are summarised in Table 2. Overall, the results suggest that 93% of the landscape is predicted to be at risk from a longer, more severe, fire season and 51.5% from future mountain pine beetle epidemics. 39% of the landscape is occupied by ecosystems at high risk to climatic change with 77% of the landscape having at least one tree species at risk. An increase in drought stress in every ecosystem is also expected. In addition, 70% of the modelled vertebrate species can be considered to be at medium to high risk to climate change as a result of an increase in fire frequency and severity and a loss of key tree species within certain ecosystems. The results outline the diversity of vulnerabilities and risks that the TFL 49 landscape may face if climate change occurs within the exposure profile utilised in this study. Table 3 presents the climatic thresholds that resulted in a change in fire regime and a loss of ecological resilience. Ecological resilience refers to the magnitude of disturbance that can be absorbed before the system is restructured with different controlling variables and processes (Gunderson et al., 2002). These thresholds could be adopted in a monitoring program to enable the identification of climatic conditions that increase the risk of larger more severe fires and that will decrease the ability of tree species to regenerate within certain ecosystems and on sites with xeric edaphic conditions (Nitschke, 2006; Nitschke and Innes, 2008a).

3.2. Step 6. Development of management framework

The ecological response analysis allows for an understanding of landscape vulnerability to be developed. This understanding can be conceptualised into a decision-making framework where primary and secondary stressors can be linked to management actions that can be used to reduce system vulnerability. The stressor–response profile of this analysis highlighted the vulnerability of the landscape to potential changes in fire-driven natural disturbance events and to the contraction and expansion loss of key tree species that characterise the dominant ecosystems within the landscape and provide habitat for the majority of faunal species assessed. Based on this analysis two decision-making frameworks were developed. The degree of vulnerability to natural disturbance, in combination with the degree of ecosystem resilience, was used to determine a range of management actions that could be used reduce the risk of future disturbances and foster resilience; a "climate-smart" management framework is presented in Fig. 4.

The second framework was developed to determine the spatial delineation of the TFL 49 landscape into triad management zones based on the vulnerability assessment. The demarcation of the landscape was determined by the identification of areas that should be prioritized for biodiversity and that contain ecosystem elements regarded as essential for the conservation of species and the maintenance of ecosystem resilience under climatic change (Noss, 2001; Lindenmayer and Franklin, 2002). Areas identified as potential fire refugia under climate change are an example of an essential element. This should always the first step in achieving sustainable management (Poore, 2003). A spatial decision-making process based on knowledge, understanding and prediction was thereby created. The spatial application of the species analysis and the fire and ecosystem vulnerability analysis to climate change was used to allocate the TFL 49 landscape into triad zones. A decision tree used for selecting management zones is presented in Fig. 5. Using the information gained from the various analyses, 17% of the landscape was placed in a reserve zone (inclusive of the identified lake zone), 49% in the extensive zone (also referred to as the landscape matrix) and 34% in an intensive management zone (Fig. 6).

The final step in this assessment of landscape vulnerability involves the characterisation of vulnerability in order to clarify relationships between stressors, effects, and ecological entities and draw conclusions drawn about the occurrence of the climate change and how to deal with the existing or anticipated effects through forest management.

Table 3
Climatic thresholds for TFL 49 landscape's fire regime and ecosystems

Fire regime	Mean T[a] (°C)	Max T[a] (°C)	Min T[a] (°C)	Precip[a] (%)	RH (%)	WinSpd[a] (m/s)
2010–2039	+1.3	+1.5	+1.2	+2.1	−8	+0.6
Ecosystems						
2010–2039	1.3	1.2	1.4	+3	−6	Unknown
2040–2069	2.5	2.7	2.4	−2	−7	Unknown
2070–2100	4.2	4.3	4.0	+2	−9	Unknown

Exceeding these thresholds resulted in statistically significant change in fire behaviour and loss of one or more species from an ecosystem. Thresholds represent the degree of climatic change that will increase the vulnerability of a landscape and precipitate change in fire regimes and loss of ecological resilience.

[a] Min T, minimum temperature; mean T, mean temperature; Max T, maximum temperature; RH, relative humidity; precip, precipitation; WinSpd, wind speed.

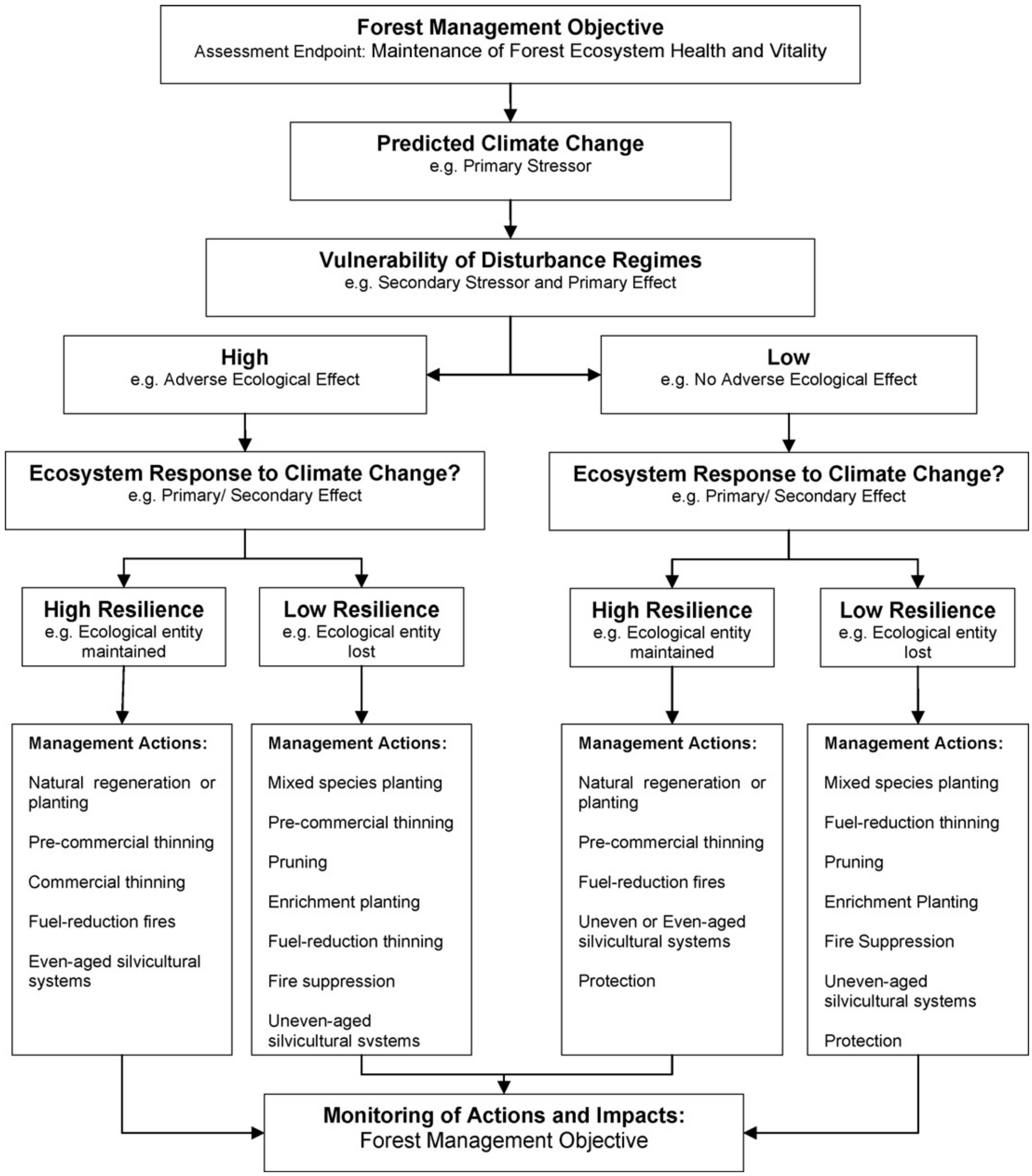

Fig. 4. Decision framework for climate-smart management. Example framework illustrates decision-making points and potential actions to combat changes in severity and frequency of wild fires and loss/maintenance of ecosystem resilience as a result of climate change at the stand- and landscape-level.

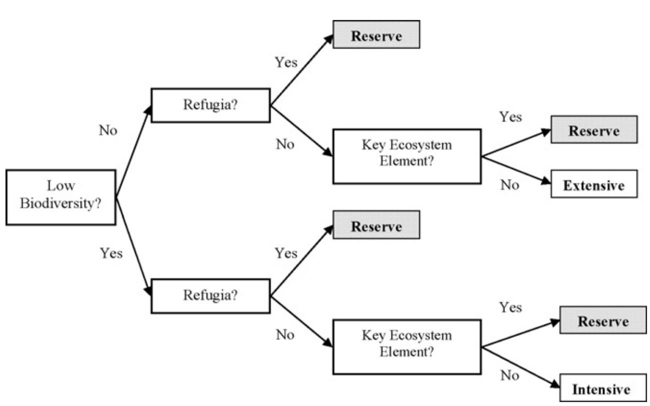

Fig. 5. Decision tree for selecting priority areas within a landscape for triad zone delineation. Under the context of climate change, "refugia" are areas identified to be at lowest risk to future forest fires while "key ecosystem elements" incorporate rare and vulnerable species/ecosystems.

4. Characterising vulnerability: discussion

A complex relationship exists between climate change, disturbance, forest management, ecosystem resilience and biodiversity conservation (Nitschke and Innes, 2006). This relationship is characterised by both positive and negative feedback loops that affect the ability of a system to recover from disturbance (Gunderson et al., 2002). The effects of climate-driven disturbances influence forest management actions and affect ecosystem resilience. Forest management activities also affect ecosystem resilience. The response of an ecosystem after disturbance will determine if an ecosystem will return to its former state or shift to a new stable state (Gunderson and Holling, 2001). The maintenance or loss of ecosystem resilience will shape the structure of ecosystems and the loss or change in ecosystem composition and/or key habitat structures will then influence faunal species. The loss of ecosystem resilience and structure will also affect the ability of the landscape to provide the same resources that society

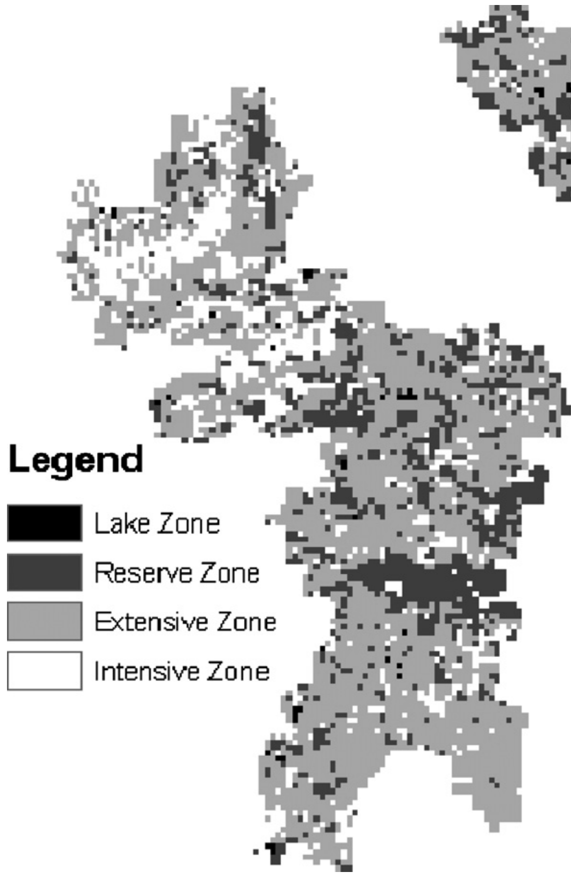

Fig. 6. Triad zone allocation of the TFL 49 landscape based on species analysis and climate change vulnerability assessment: 16% of the landscape was demarcated as a reserve zone, 1% as a lake zone, 49% as a extensive zone, and 34% as an intensive management zone. .

demands today. Understanding the broad pattern of causality can help identify the key pathways or feedback loops that need to be managed. In this assessment an increase in fire weather severity and drought conditions resulted in an increase in potential area burned and also pointed to a potential increase in fire frequency. The assessment also identified that current tree species will respond individually with some species having their future ranges contracting by greater than 90% while other species have their regeneration niche expand to higher elevation areas while suffering a range contraction in lower elevation areas. The combination of an increase in fire severity and frequency along with a decline in regeneration potential for many species increases the potential change in ecosystem composition and structure which in turn threatens faunal species that rely on these ecological entities for survival. Management actions that promote ecosystem resilience and reduce the risk of natural disturbances will therefore be important for the conservation of species and biodiversity and thus the achievement of sustainable forest management. The management of a landscape will require a myriad of different management actions that suit the degree of risk and the objectives of the management zone. For managers to reduce the risks associated with climate change, direct and definitive actions will need to be implemented that reduce risk and promote resilience. The climate-smart management framework presented in Fig. 4 is proposed here as a decision-making tool that can be used to conduct management actions that can maintain ecosystem health and vitality under climate change and thus aid in the achievement of sustainable management.

4.1. Climate-smart management framework

The climate-smart management framework is rooted in the use of existing forest management practices to reduce the risk from the impacts of natural and anthropogenic disturbances and to promote or maintain ecological resilience under climate change. The proposed framework is founded on the understanding and assessment of landscape vulnerability to climatic change in this study. The framework requires the identification of potential changes to disturbance regimes and ecosystem resilience/resistance to climatic change.

The initial step of climate-smart management is the identification of management objectives. In this study the maintenance of ecosystem health and vitality was selected as the key objective to assess landscape vulnerability. The division of a landscape into different management zones typically leads to the development of different management objectives for separate areas of a landscape. For example, the triad zoning system specifies three different types of management, each having its own range of possible management actions (Nitschke and Innes, 2005). The distribution of multiple stressors and the variation of ecosystem resilience measured across the landscape in this study means that a blend of management strategies will be required across all management zones, necessitating a spatial and temporal understanding of system vulnerability in order to incorporate landscape vulnerability into forest management. The vulnerability analysis conducted in this study was used in the demarcation of triad management zones; however, on landscapes where management zones are already demarcated the analysis could be used to assist managers in changing the objectives of each established management zone which will then allow for the framework to be utilised.

The second step in the climate-smart decision-making process is the identification of the vulnerability of disturbance regimes to climate change. In this study, we found that 93% of the landscape is at risk to fires every 50 years or less, an increase from 34% at the present. This suggests that nearly 60% of the landscape is vulnerable to a climate change-driven shift in fire regime. This understanding of potential fire risk can be used to classify the areas within a landscape based on future disturbance risk. In the developed framework, high and low risks were used, but any number of categories can be used based on the system and management objectives. After the classification of disturbance risk, the third step involves the demarcation of management actions based on the degree of species and ecosystem resilience to climate change. In this study, ponderosa pine was found to be vulnerable to climate change in the ponderosa pine ecosystem but resilient across the remainder of the landscape (see Table 2). Ponderosa pine, and the ecosystem it currently dominates, could therefore be considered to have low resilience to predicted climate change while at higher elevations this species exhibited high resilience. In locations where resilience is low, management actions will likely need to be more intensive and complex than where resilience is high. The management actions that influence resilience also need to be tied back to disturbance since disturbance processes act as a secondary stressor on ecological resilience (Nitschke and Innes, 2006). In the developed framework, high and low risks were used but any number of categories can be used based on the complexity of species and ecosystem response.

4.2. Forest management and climate change

The partitioning of a landscape into different management zones can be an important first step towards SFM, but the allocation process alone will not guarantee sustainability. The overwhelming degree of vulnerability measured for the TFL 49

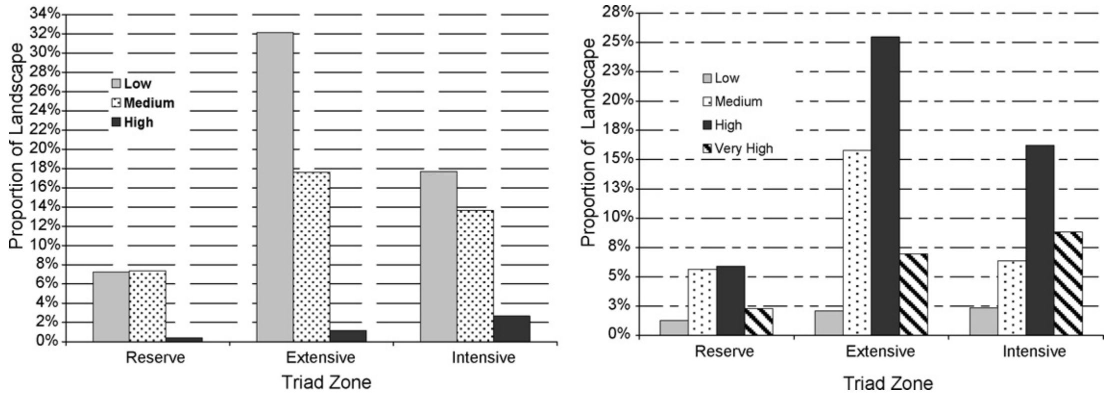

Fig. 7. Distribution of mountain pine beetle and lightning ignition risk across the three triad zones [mountain pine beetle risk (left), lightning ignition risk (right)].

landscape illustrates this point. Due to the high degree of landscape vulnerability, a trade-off exists between avoiding risk and maximising a value, this is demonstrated by the distribution of mountain pine beetle and lightning ignition risk across the three triad zones used in this study (Fig. 7). The reserve and extensive zones contain areas of high biodiversity but also high risk to these two elements that will require active management. The intensive zone will, by definition, be actively managed. However, management actions that solely reduce the risk over the rotation period without improving stand yield may be required. The risk of removing ecosystem resilience will likely be higher where intensive short-rotation forest management is practised, as will the ability to adapt the species composition in these areas to fit future climatic conditions. These results demonstrate the overwhelming degree of risk faced by the TFL 49 forest managers to climatic change, and they also outline the need for flexible and proactive forest management actions that can help reduce this risk. Managers who choose not to incorporate this understanding of system vulnerability will be making decisions over time that are increasingly weighted with risk and thus more likely to be negatively impacted by climatic change. Incorporating this understanding into management is therefore a logical step in reducing the risks associated with climatic change.

Flexible management actions that reduce the risk to ecosystems and species are necessary if sustainable forest management is to be achieved. If the initial allocation process is to conserve the current distribution of biodiversity and resource supply, adaptive management must be implemented at the same time as the allocation process. Under the uncertainty associated with climate change, the pro posed management framework provides a flexible system that incorporates an understanding of system vulnerabilities, in turn allowing risk to be reduced. This creates a feedback loop between proactive risk management and sustainable forest management that gradually adapts the landscape and management system to climatic change as our understanding and knowledge of predicted impacts increase over time. The predicted vulnerabilities observed in this study provide an initial assessment of landscape vulnerability; this understanding might be used to inform management decisions and actions. Management actions based on this understanding can be designed to be both robust to uncertainties yet reversible (Carpenter et al., 2001).

4.3. Climate-smart management actions

The appropriate management actions will depend on the stage of stand development, the potential disturbance frequency and severity and the degree of ecological resilience. The timing of management actions will also depend on the climatic conditions. For example, fire weather severity will influence when fuel-reduction burning can take place. In this study, the fire season length was predicted to increase by 27% in the spring but remain relatively unchanged in both summer and autumn. Fire severity was also found to increase in all seasons but less so in autumn. This could shift fuel-reduction burning from spring to autumn. The timing of budburst also needs to be considered when applying treatments such as thinning or prescribed burning. Trees are highly susceptible to disturbances in the week immediately prior to and following budburst and the week preceding leaf-fall or dormancy (Kramer, 1996). The increase in fire season length and severity along with early budburst dates could result in a smaller window in the early spring or late autumn for conducting these treatments. The use of a particular management action does not pertain to a single risk factor, as multiple factors can be achieved through the same management action. The following section will discuss the role of select management actions in the climate-smart framework.

4.3.1. Regeneration: natural or artificial; single or mixed species?

Following a disturbance (natural or anthropogenic), the option exists to allow natural regeneration or to plant seedlings. The use of artificial planting often involves the establishment of single-species stands that are then augmented by natural regeneration from adjacent stands. In highly resilient systems, the use of either natural regeneration or planting will enable the ecosystem to follow a successional pathway toward one of many possible states (Gunderson et al., 2002). In ecosystems with low resilience to climatic change, the use of natural regeneration alone may not guarantee that the ecosystem will develop along a traditional pathway, as a new stable state may develop. In this study, 10 of 13 species were found to be highly vulnerable to climate change within their regeneration niche. This vulnerability places many of the current ecosystems at risk and highlights the risk associated with relying solely on natural regeneration for maintaining ecosystem health and vitality. The utilisation of mixed-species planting will diversify the risk ecosystems face and will lessen the risks posed by disturbance and biodiversity loss (Gitay et al., 2002; Whitehead et al., 2004). Conversely, the planting of mono-specific stands will increase the risk to biodiversity and resource values in systems with low resilience (Noss, 2001). Artificial regeneration can also be used to facilitate the persistence of species and ecosystems through "human-assisted migration". This may become an important management strategy to deal with increased drought risk (Hogg and Bernier, 2005). All trees assessed in the study were predicted to contract from lower elevations as a result of increased drought. Particular care is needed to ensure that short-term incentives to achieve prescriptive regeneration objectives do not result in future forests with low resilience. For example, the continual planting of pure lodgepole pine stands across the majority of the studied landscape will facilitate larger- more severe fires and future mountain pine beetle infestations within forest types that are rated as highly vulnerable to climate change (58.3% contraction in area available for regeneration).

4.3.2. Thinning and pruning

Thinning can lessen the risk posed by multiple disturbances. Three types of thinning have been recommended for climate-smart management, namely pre-commercial, commercial and fuel-reduction thinning. Pre-commercial thinning is generally utilised in young stands to reduce stand density in order to optimise the growth of the remaining trees (Smith et al., 1996). Commercial thinning, also referred to as thinning from below, involves removing less vigorous trees and some dominant and co-dominant individuals to reduce competition (and thereby water stress) and to accelerate the growth of the residual trees. It is undertaken during the stem exclusion and early understorey re-initiation phases (Smith et al., 1996). Fuel-reduction thinning involves the removal of surface and ladder fuels from forest stands to reduce the risk of fire ignition and spread (Raymond and Peterson, 2005).

Fuel-reduction thinning is a part of fire-smart management for reducing forest fuel loads (Hirsh et al., 2001), and the removal of canopy fuels reduces the potential for crown fires (Raymond and Peterson, 2005). Thinning is also recommended in late-successional stands to decrease fire risk by reducing ladder and surface fuels (Spies et al., 2006). Pruning can also reduce ladder fuels within stands, if required. The reduction of both ladder and surface fuels is required for fire risk and mortality to be reduced (Raymond and Peterson, 2005). The predicted increase in fire severity, size and frequency under current fuel conditions highlight the need to undertake fuel-reduction activities that both fire-smart and climate-smart the landscape to protect vulnerable species and ecosystems in order to maintain ecosystem health and vitality.

Pre-commercial thinning and pruning have also been found to be effective in reducing the susceptibility of stands to Dothistroma needle blight (*Dothistroma septosporum*) outbreaks (van der Pas et al., 1984). Dothistroma outbreaks in British Columbia have been linked to changes in climate and can cause extensive mortality in young lodgepole pine (*Pinus contorta* var. *latifolia*) stands (<20 years old) (Woods et al., 2005). Dothistroma affects over 60 species of pines (*Pinus* sp.) worldwide, including lodgepole pine, ponderosa pine (*Pinus ponderosa*) and whitebark pine (*Pinus albicaulis*) (Bradshaw, 2004), all of which are found in the TFL 49 study area.

Climatic change has been identified as a major driving factor in past and current mountain pine beetle epidemics (Allen and Breshears, 1998; Carroll et al., 2004). Commercial thinning to reduce stand density can reduce mortality in pine stands during such epidemics and can also prevent incipient infestations through the alteration of microclimatic conditions (Whitehead et al., 2004). Thinning from below reduces competition for site resources, increasing the growth and vigour of the remaining trees. This reduces the susceptibility of the residual pine to successful pine beetle attack by stimulating trees' resin defence systems (Sartwell and Dolph, 1976; Mitchell et al., 1983; Speight and Wainhouse, 1989; Whitehead et al., 2004). Reduced competition also lowers the vulnerability of trees, particularly understorey seedlings, to drought (Man and Lieffers, 1999). Droughts can lead to the loss of forest cover and alter forest succession and can also cause both direct mortality and an increase in mortality caused by bark beetles and fire (Allen and Breshears, 1998; Hogg and Bernier, 2005). An increase in drought stress was modelled in this assessment.

4.3.3. Fuel-reduction fires and fire suppression

Fuel-reduction fires (prescribed burning) can be used in conjunction with thinning to decrease the potential of crown fires and to reduce surface fire intensity (Raymond and Peterson, 2005). If under-burning is not utilised after fuel-reduction thinning, then both surface and ladder fuels will need to be mechanically removed to reduce the fine-fuel loads generated by the thinning. Mortality is greatest in stands that are thinned but not under-burned, with stands that are thinned and then burned incurring significantly less mortality after wildfire (Raymond and Peterson, 2005). Fuel-reduction fires are also a recommended action in the fire-smart management paradigm (Hirsch et al., 2001).

In ecosystems with high resilience to climate change, fuel-reduction fires can be applied after thinning to reduce the potential spread and severity of wildfires. In ecosystems with low resilience, burning is only suitable for stands dominated by fire-tolerant species (for example, the Douglas-fir and ponderosa pine dominated ecosystems). In stands dominated by fire-intolerant species, fire is best avoided. In areas of low resilience and fire intolerance, mechanical removal of fuels followed by fire suppression may be the most suitable option (for example, in the spruce dominated ecosystems). Thinning without fuel removal could be counter-productive, increasing the risk of fire-mortality due to a likely increase in fire intensity and severity. Fuel reduction burns should be used as frequently as required, with a reduction on forest fuels helping fire suppression. Increased fire severity and size were predicted to occur in the study area as a consequence of climatic change. An increase in fire severity and size will result in an increase in the costs and reduce our ability to suppress fires unless fuel loads are reduced (Arno and Fiedler, 2005). The continual reduction in forest fuels will reduce the risk of large, uncontrollable fires and increase the ability of forest managers to suppress fires in ecosystems of low resilience. An investment in fuel reduction will benefit both biodiversity and forest resources and will be a key requirement in maintaining ecosystem resilience under climatic change. For fuel reduction activities to benefit all levels of biodiversity, key habitat elements, such as snags and coarse woody debris, will need to be maintained at the stand- and landscape-levels in quantities that will conserve species dependent on these structures. Seventy percent of the species assessed in this study were determined to be at medium to high risk with species requiring key habitat elements associated with late-successional spruce-fir forests to have the highest vulnerability. The use of fuel reduction burns may also be required to maintain species that require early seral habitat.

4.3.4. Enrichment planting

Enrichment planting involves the planting of seedlings within a forest where natural regeneration is poor or non-existent (d'Oliveir, 2000). It has been used successfully to establish desired timber and non-timber species in combination with shelterwood, nurse-tree, and selective harvesting systems (Ashton et al., 1997; Lozada et al., 2003). Enrichment planting has also been successfully used after single and group-tree selection harvesting in the temperate rimu (*Dacrydium cupressinum*) forests of New Zealand (Jamesa and Norton, 2002). It may become a requirement in forest reserves and after fuel-reduction treatments or selective harvesting in vulnerable ecosystems where natural regeneration continually fails under climate change. In this study, the spruce-dominated ecosystems were all assessed to be extremely vulnerable in their regeneration niche to climate change. In these ecosystems, enrichment planting of seedlings may be required to maintain a younger cohort of trees that can gradually replace the loss of mature individuals.

Silvicultural systems that promote large openings will have a significantly different microclimate than systems that maintain an intact canopy (Noss, 2001), with an increasing size of canopy opening proportionally increasing the light levels and temperatures and decreasing humidity (Mason, 1996). Forests that provide higher humidity, cooler temperatures and wetter edaphic conditions are important for maintaining intolerant species on sites that are exposed and have lower humidity, higher temperatures and drier edaphic conditions (Aide and Rivera, 1998). Enrichment planting could be used to establish late-successional species that are vulnerable to climatic change in the understorey of established forests where temperatures are lower, humidity is higher and wetter edaphic conditions may persist over the summer season. Enrichment planting could aid the establishment of species in areas where they have low resilience to climate change within their current regeneration niche. Likewise, it could also be used following artificial or natural regeneration planting to fill in gaps resulting from disturbance or climate-induced mortality. Enrichment planting is most suitable when natural regeneration is poor or non-existent in areas that require the maintenance of ecosystem resilience for biological conservation. It could also be used to facilitate the persistence of species and ecosystems through "human-assisted migration", and to establish new species that are better adapted to the future climates. The latter may enable a gradual and controlled transition to a more climatically adapted ecosystem. Enrichment planting is best considered as a "stop gap" to help mediate the transition of ecosystems and to provide habitat in current and future areas for species that are vulnerable to climate change. In the long-term, enrichment planting alone will not sustain or conserve an ecosystem or any species requiring it.

4.3.5. Silvicultural systems

A silvicultural system is defined as a process by which forests are tended, harvested and regenerated to produce a stand and eventually a forest with a distinct form (Toumey, 1928; Matthews, 1989). Even-aged silvicultural systems include clear-cut, patch-cut, coppice, seed tree and shelterwood systems while uneven-aged systems include single-tree selection, group-tree selection, and irregular shelterwood systems (Smith et al., 1996). Variable retention can also be considered an uneven-aged system if it promotes the retention of multiple age class structures that can provide habitat elements associated with older forests in younger stands (O'Hara and Nagel, 2006). The objectives of each system will determine the extent and timing of canopy removal over the length of the rotation. The use of even-aged versus uneven-aged systems will affect microclimatic conditions with mid-summer water stress usually being reduced in multi-aged stands versus single-aged stands (O'Hara and Nagel, 2006). The greater the percentage of the canopy removed at one time, the greater the change in microclimate (Nitschke, 2005). The decision to use uneven- or even-aged silvicultural systems usually depends on the ecology of the tree species being managed for, the prevalent disturbance regime and prevailing social pressures. For management to be climate-smart, the degree of ecosystem resilience must also be considered. In systems with low resilience to climatic change, areas with cooler microclimates will be important for sustaining species that cannot persist on exposed sites (Aide and Rivera, 1998).

Uneven-aged systems or forest reservations should be used in ecosystems with low disturbance risk and low resilience. This reflects the longer disturbance intervals associated with lower disturbance risk and will maintain suitable microclimatic conditions. In areas of high disturbance risk (areas with frequent stand-replacing disturbances) and low resilience, uneven-aged systems could be used to represent the loss of biomass to frequent disturbances while maintaining microclimatic conditions that facilitate resilience. In areas of high ecosystem resilience, uneven- and even-aged systems can be used, although systems that maintain key habitat elements are generally preferred. Variable retention systems can be used in preference to clear-cut systems in order to maintain ecologically important features, as these will provide habitat structures and help ameliorate microclimate conditions in larger harvest openings.

4.4. Pros and cons of using climate-smart management

The proposed climate-smart management actions would require an increase in investment into forest management by forest stakeholders. Any decision to implement climate-smart management will need to be viewed in the form of risk management; the cost of no action (i.e., having to undertake remedial treatments and increased costs associated with fires and other disturbances) may be considerably greater than the costs associated with reducing risk. Cost–benefit analyses that include market and non-market forest values are required to determine the intensity of the risk reduction actions undertaken. In the case of fire-smart landscapes, Mason et al. (2006) identified that substantial benefits are possible over the long-term by investing in fuel reduction to reduce fire risk. Another important determinant in undertaking climate-smart management is consensus on the perception of risk by forest stakeholders, the perception of risk will vary across social groups and stakeholders (Weiss, 2001). Despite this potential challenge, it is up to stakeholders to decide which measures of risk to use and how to judge the significance of any vulnerability assessment (Oppenheimer, 2005). The perceptions of forest managers will be the critical determinants in the incorporation of climate change into forest management, making geographically specific knowledge of ecosystem vulnerability a necessity (Ogden and Innes, 2007). Assessments of system vulnerabilities can provide an understanding, within a spatial context, of the potential risks forest managers will face as climate changes. This knowledge will be essential in any decision-making process adopted to determine the management policy, objectives and actions required to achieve sustainable management.

Climate-smart management is derived from an assessment of landscape vulnerability to climatic change, and implementation of this paradigm is necessary if we want to manage in a sustainable manner the range of forest values and ecosystem services that are present today. Engaging in climate-smart management will allow forest managers to foster ecological resilience and mitigate the impacts of small- and large-scale disturbances associated with climate change on both current and future ecosystem compositions. Climate-smart management will aid in the gradual transition of ecosystems to new forest values and ecological services. This form of management is not about managing the status quo; it is about adapting ecosystems to help mitigate the environmental impacts of climate-change in a proactive rather than reactive manner. It will allow forest managers to foster resilience and adapt ecosystems so that they can continually provide the social and economic needs of societies while providing societies time to adapt their social and economic systems to climate-driven changes in ecosystems

5. Conclusion

Understanding how complex systems react to change is a prerequisite for sustainable management (Gunderson and Holling, 2001). Integrating climate change into forest management therefore requires an understanding of ecological response and vulnerability to this stressor. Our objective was to gain knowledge and understanding of system vulnerabilities in order to integrate the uncertainties and vulnerabilities of climatic change into forest management, with these system components being the "pieces of the forest management jigsaw puzzle" (Kimmins et al., 2005) needed to practise sustainable management. The first step involved using existing knowledge to gain an understanding of each piece of the puzzle through prediction. The amalgamation of each analysis was then used to create a picture from the puzzle pieces, with that picture having two parts: the spatial demarcation of the landscape into defined management zones, and management actions required to maintain the health and vitality of that picture in an uncertain future. Understanding system vulnerabilities is key to achieving this objective (Turner et al., 2003).

Although the exact nature of future climatic change is uncertain, it has always occurred and will continue to do so. Current predictions for western Canada suggest that the climate will be warmer and wetter in the winter and hotter and drier in the summer under the majority of modelled scenarios (Lemmen and Warren, 2004). Based on the climate change scenarios derived from three global circulation models, significant vulnerability exists to climate change on the TFL 49 landscape. Climate change will increasingly impact fire weather behaviour, fire regimes, ecosystem resilience, biogenic disturbance, and biodiversity over the next 100 years. The results of these analyses identified that a cascading relationship exists between climate change, natural disturbances and ecosystem resilience. The relationship suggests that managing forest landscapes under climate change will require multiple actions to achieve sustainability. Active management of landscape vulnerability is the most pressing need. The allocation of a landscape into management zones is a first step that will ensure that current features required for conserving biological diversity are represented but alone will not ensure sustainability. Climate-smart management is required to ensure that the allocation process is successful. Integrating an understanding of system vulnerabilities is essential for developing management plans and actions that incorporate risk and uncertainty in a manner that fosters sustainability.

Sustainable management requires long planning horizons fraught with uncertainty, making an adaptive management approach desirable. Despite this need, two conflicting responses exist amongst forest practitioners in relation to the integration of climate change into management; an active and a passive approach (Ogden and Innes, 2007). Climate-smart management is an active approach that is based on an understanding of ecosystem vulnerability. Passive management is a reactionary approach that will be constrained by the vulnerability of a system and result in an increase in both systemic and unsystemic risks. Management will need to be flexible and proactive to deal with the multiple aspects of risk faced by forest stakeholders.

Sustainability relies on the maintenance and exploitation of ecological resilience. Where resilience is low we will need to tread lightly but be proactive; where resilience is high we can be less cautious yet still proactive. Due to the uncertainty in the exact degree and direction of change in future water supply it is important that we refine management actions as we gain a greater understanding of climate change impacts on the water cycle. In this study, proactive management actions are directed at reducing disturbance risk and fostering ecological resilience under predicted warmer/drier summer conditions. If these predictions are wrong and warmer and wetter summer conditions occur and/or an increase in water-use efficiency occurs, then management actions will need to be refined because risk could be perceived differently. Climate-smart management would still be important because a warmer/wetter environment will still impact disturbance risk and an increase in water-use efficiency will still impact ecological resilience. Consequently, the management actions may change but the objectives will not.

As we gain new knowledge about systems we can re-evaluate vulnerability, and the results of this analysis can be used to refine proactive management through an adaptive management framework. Understanding future system vulnerability will always be challenged by uncertainty but by using our current knowledge and understanding we can redefine much of this uncertainty as risk. The conservation of biodiversity and resource values relies on management objectives that seek out ways to reduce risk, and this will only be achievable through an understanding of landscape vulnerability.

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References

Aide, T.M., Rivera, E., 1998. Geographic patterns of genetic diversity in *Poulsenia armata* (Moraceae): implications for the theory of Pleistocene refugia and the importance of riparian forest. J. Biogeogr. 25, 695–705.

Allen, C.D., Breshears, D.D., 1998. Drought-induced shift of a forest-woodland landscape in response to climate variation. Proc. Natl. Acad. Sci. U.S.A. 95, 14839–14842.

Arno, S.E., Fiedler, C.E., 2005. Mimicking Nature's Fire: Restoring Fire-Prone Forests in the West. Island Press, Washington, USA.

Ashton, M.S., Gamage, S., Gunatilleke, I.A.U.N., Gunatilleke, C.V.S., 1997. Restoration of a Sri Lankan rainforest: using Caribbean pine *Pinus caribaea* as a nurse for establishing late-successional species. J. Appl. Ecol. 34 (4), 915–925.

Beissinger, S.R., Westphal, M.I., 1998. On the use of demographic models of population viability in endangered species management. J. Wildl. Manage. 62, 821–841.

Blennow, K., Sallnäs, O., 2005. Decision support for active risk management in sustainable forestry. J. Sustain. For. 21 (2–3), 199–210.

Borchers, J.G., 2005. Accepting uncertainty, assessing risk: decision quality in managing wildfire, forest resource values, and new technology. For. Ecol. Manage. 211, 26–46.

Bradshaw, R.E., 2004. Dothistroma (red-banded) needle blight of pines and the dothistromin toxin: a review. For. Pathol. 34, 163–185.

Brooks, N., Adger, W.N., Kelly, P.M., 2005. The determinants of vulnerability and adaptive capacity at the national level and the implications for adaptation. Global Environ. Change 15, 151–163.

Bunnell, F.L., Huggard, D.J., Lisgo, K.A., 1999. Vertebrates and Stand Structure in TFL 49. Draft Report. Centre for Applied Conservation Biology, University of British Columbia. Vancouver, British Columbia.

Canadian Council of Forest Ministers, 2003. Defining Sustainable Forest Management in Canada: Criteria and Indicators 2003. Canadian Council of Forest Ministers. Available from http://ccfm.org/ci/CI_Booklet_e.pdf (accessed April 10, 2008).

Canadian Institute for Climate Studies, 2005. Canadian Climate Impacts and Scenarios. Available from http://www.cics.uvic.ca/scenrios (accessed January 10, 2005).

Canadian Wildland Fire Growth Model Steering Committee, 2003. Prometheus User Manual Version 2.2.5. Available from http://www.firegrowthmodel.com/public_download.cfm (accessed February 17, 2004).

Carpenter, S., Walker, B., Anderies, J.M., Abel, N., 2001. From metaphor to measurement: resilience of what to what? Ecosystems 4, 765–781.

Carroll, A.L., Taylor, S.W., Régnière, J., Safranyik, L., 2004. Effects of climate change on range expansion by the mountain pine beetle in British Columbia. In: Shore, T.L., Brooks, J.E., Stone, J.E. (Eds.), Mountain Pine Beetle Symposium: Challenges and Solutions. Natural Resources Canada Information Report BC-X-399, pp. 223–232.

Costanza, R., Voinov, A., 2004. Introduction: spatially explicit landscape simulation models. In: In Landscape simulation modelling: a spatially explicit, dynamic approach, Springer-Verlag, New York, 3–20.

d'Oliveir, M.V.N., 2000. Artificial regeneration in gaps and skidding trails after mechanised forest exploitation in Acre Brazil. For. Ecol. Manage. 127 (1–3), 67–76.

Duinker, P.N., 1990. Climate change and forest management, policy and land use. Land Use Policy 7, 124–137.

Environmental Systems Research Institute, 2004. ESRI ArcGIS 9.0. Environmental Systems Research Institute Ltd., USA.

Figge, F., 2004. Bio-folio: applying portfolio theory to biodiversity. Biodiv. Conserv. 13, 827–849.

Flato, G.M., Boer, G.J., Lee, W.G., McFarlane, N.A., Ramsden, D., Reader, M.C., Reader, A.J., 2000. The Canadian centre for climate modelling and analysis global coupled model and its climate. Clim. Dyn. 16, 451–467.

Franklin, J.F., 1993. Preserving biodiversity: species, ecosystems or landscapes? Ecol. Appl. 3 (2), 202–205.

Ford, J.D., Smit, B., 2004. A framework for assessing the vulnerability of communities in the Canadian Arctic to risks associated with climate change. Arctic 57 (4), 389–400.

Gitay, H., Suárez, A., Wilson, R., 2002.Climate Change and Biodiversity. Technical Paper V. Intergovernmental Panel on Climate Change, Geneva.

Gough, A.D., Innes, J.L., Allen, S.D., 2008. Development of common indicators of sustainable forest management. Ecol. Indic. 8 (5), 425–780.

Gunderson, L.H., Holling, C.S., 2001. Panarchy: understanding transformations in humans and natural systems. Island Press, Washington, DC.

Gunderson, L.H., Holling, C.S., Pritchard Jr., L., Peterson, G.D., 2002. Resilience of large-scale resource systems. In: SCOPE 60: Resilience and Behaviour of Large-Scale Systems, Island Press, Washington, USA, pp. 3–20.

Haimes, Y.Y., 1998. Risk Modelling, Assessment and Management. Wiley, New York, NY.

Hinckley, T.M., Sprugel, D.G., Brooks, J.R., Brown, K.J., Martin, T.A., Roberts, D.A., Schaap, W., Wang, D., 1998. Scaling and integration. In: Ecological Scale: Theory and Application, Columbia University Press, New York, pp. 309–338.

Hickey, G.M., Nitschke, C.R., 2007. Monitoring sustainable forest management in the Pacific Rim. J. Sustain. For. 24 (2–3), 245–278.

Hirsch, K.G., 1996. Canadian Forest Fire Behavior Prediction (FBP) System: Users's Guide. Natural Resources Canada, Canadian Forest Service, Northwest Region, Northern Forestry Centre, Edmonton, Alberta. Special Report 7.

Hirsch, K., Kafka, V., Tymstra, C., McAlpine, R., Hawkes, B., Stegehuis, H., Quintilio, S., Gauthier, S., Peck, K., 2001. Fire-smart forest management: a pragmatic approach to sustainable forest management in fire-dominated ecosystems. For. Chron. 77 (2), 357–363.

Hogg, E.H., Bernier, P.Y., 2005. Climate change impacts on drought-prone forests in western Canada. For. Chron. 81 (5), 675–682.

Hollenstein, K., 2001. Risk analysis as a tool for policy evaluation: a multi-level comparison example. In: Risk Analysis in Forest Management, Kluwer Academic Press, The Netherlands, pp. 75–112.

Intergovernmental Panel on Climate Change, 1998. The Regional Impacts of Climate Change: An Assessment of Vulnerability. Cambridge University Press, New York.

Jamesa, I.L., Norton, D.A., 2002. Helicopter-based natural forest management for New Zealand's rimu (*Dacrydium cupressinum*, Podocarpaceae) forests. For. Ecol. Manage. 155 (1–3), 337–346.

Johns, T.C., Gregory, J.M., Ingram, W.J., Johnson, C.E., Jones, A., Lowe, J.A., Mitchell, J.F.B., Roberts, D.L., Sexton, D.M.H., Stevenson, D.S., Tett, S.F.B., Woodage, M.J., 2003. Anthropogenic climate change for 1860 to 2100 simulated with the HadCM3 model under updated emission scenarios. Clim. Dyn. 20, 583–612.

Jones, R.N., 2001. An environmental risk assessment/management framework for climate change impact assessments. Natural Hazards 23, 197–230.

Jones, R.N., 2004. Incorporating agency into climate change risk assessments. Clim. Change 67, 13–36.

Kimmins, J.P., Welham, C., Seely, B., Meitner, M., Rempel, R., Sullivan, T., 2005. Science in forestry: why does it sometimes disappoint or even fail us? For. Chron. 81 (5), 723–734.

Kramer, K., 1996. Phenology and Growth of European Trees in Relation to Climate Change. PhD Thesis. Landbouar Universikit Wageningen.

Lemmen, D.S., Warren, F.J., 2004. Climate Change Impacts and Adaptation: A Canadian Perspective. Natural Resources Canada, Government of Canada.

Lindenmayer, D.B., Franklin, J.F., 2002. Conserving Forest Biodiversity: A Comprehensive Multiscaled Approach. Island Press, Washington, D.C.

Lindner, M., Sohngen, B., Joyce, L.A., Price, D.T., Bernier, P.Y., Karjalainen, T., 2002. Integrated forestry assessments for climate change impacts. For. Ecol. Manage. 162, 117–136.

Lozada, J.R., Moreno, J., Suescun, R., 2003. Plantations in enrichment strips. Experiences in 4 units of forest management at the Venezuelan Guiana. Interciencia 28 (10), 568.

Luers, A.L., 2005. The surface of vulnerability: an analytical framework for examining environmental change. Global Environ. Change 15, 214–223.

Luxmoore, R.J., Hargrove, W.W., Tharp, M.L., MacPost, W., Berry, M.W., Minser, K.S., Cropper Jr., W.P., Johnson, D.W., Zeide, B., Amateis, R.L., Bukhart, H.E., Baldwin Jr., V.C., Peterson, K.D., 2002. Addressing multi-use issues in sustainable forest management with signal-transfer modelling. For. Ecol. Manage. 165, 295–304.

Man, R., Lieffers, V.L., 1999. Effects of shelterwood and site preparation on microclimate establishment of white spruce seedlings in a boreal mixedwood forest. For. Chron. 75, 837–844.

Manning, M., Petit, M., Easterling, D., Murphy, J., Patwardhan, A., Rogner, H.-H., Swart, R., Yohe, G., 2004. Describing Scientific Uncertainties in Climate Change to Support Analysis of Risk and of Options. IPCC Workshop Report. National University of Ireland, Maynooth, Co. Kildare, Ireland, May 11–13, 2004.

Mason, C.L., Lippke, B.R., Zobrist, K.W., Bloxton, T.D., Ceder, K.R., Comnick, J.M., McCarter, J.B., Rogers, H.K., 2006. Investments in fuel removals to avoid forest fires result in substantial benefits. J. For. 104 (1), 27–31.

Mason, D., 1996. Responses of Venezuelan understorey birds to selective logging, enrichment strips, and vine cutting. Biotropica 28 (3), 296–309.

Matthews, J.D., 1989. Silvicultural Systems. Clarendon Press, Oxford.

Meidinger, D., Pojar, J., 1991. Ecosystems of British Columbia. B.C. Ministry of Forests, Victoria, BC. Special Report SRS06.

Messier, C., Fortin, M.J., Schmiegelow, F., Doyon, F., Cumming, S.G., Kimmins, J.P., Seely, B., Welham, C., Nelson, J., 2003. Modelling tools to assess the sustainability of forest management scenarios. In: Towards Sustainable Forest Management of the Boreal Forest, NRC Research Press, Ottawa, Ontario, Canada, pp. 531–580.

Ministry of Forests and Range, 2006. The Future Forest Ecosystems of BC: Exploring the Opportunities. Symposium and Workshop Report from the Ministry of Forests and Range Chief Foresters Office, February, 2006. Government of British Columbia. Available from http://www.for.gov.bc.ca/hts/Future_Forests (accessed March 1, 2006).

Mitchell, R.G., Waring, R.H., Pitman, G.B., 1983. Thinning lodgepole pine increases tree vigor and resistance to mountain pine beetle. For. Sci. 29 (1), 204–211.

Montréal Process Technical Advisory Committee, 2003. Montréal Process: First Forest Overview Report 2003. The Montréal Process. Available from http://www.rinya.maff.go.jp/mpci/rep-pub/2003/overview/index_e.html (acessed January 10, 2008).

Muzy, A., Innocenti, E., Aïello, A., Santucci, J.-F., Santoni, P.-A., Hill, D.R.C., 2005. Modelling and simulation of ecological propagation processes: application to fire spread. Environ. Modell. Software 20, 827–842.

Nitschke, C.R., 2005. Does forest harvesting emulate fire disturbance? A comparison of effects on selected attributes in coniferous-dominated headwater systems. For. Ecol. Manage. 214, 305–319.

Nitschke, C.R., 2006. Integrating Climate Change into Forest Planning: A Spatial and Temporal Analysis of Landscape Vulnerability. PhD Dissertation. University of British Columbia. Vancouver, Canada.

Nitschke, C.R., Innes, J.L., 2005. The application of forest zoning as an alternative to multiple use forestry. In: Forestry and Environmental Change: Socioeconomic and Political Dimensions, CAB International, Oxford, pp. 97–124.

Nitschke, C.R., Innes, J.L., 2006. Interactions between fire, climate change and forest biodiversity. Perspect. Agric. Vet. Sci. Nutr. Nat. Resour. 1 (6), 1–9.

Nitschke, C.R., Innes, J.L., 2008a. A tree and climate assessment tool for modelling ecosystem response to climate change. Ecol. Modell. 210 (3), 263–277.

Nitschke, C.R., Innes, J.L., 2008b. Climatic change and fire potential in South-Central British Columbia Canada. Global Change Biol. 14 (4), 841–855.

Norton, D.A., 1998. Indigenous biodiversity conservation and plantation forestry: options for the future. New Zeal. For. 43 (2), 34–39.

Noss, R.F., 2001. Beyond Kyoto: forest management in a time of rapid climate change. Conserv. Biol. 15 (3), 578–590.

Ogden, A.E., Innes, J.L., 2007. Perspectives of forest practitioners on climate adaptation in the Yukon and Northwest Territories of Canada. For. Chron. 83 (4), 557–569.

O'Hara, K.L., Nagel, L.M., 2006. A functional comparison of productivity in even-aged and multi-aged stands: a synthesis for *Pinus ponderosa*. For. Sci. 52 (3), 290–303.

Ohlson, D.W., McKinnon, G.A., Hirsch, K.G., 2005. A structured decision-making approach to climate change adaptation in the forest sector. For. Chron. 81 (1), 97–103.

O'Laughlin, J., 2005. Conceptual model for comparative ecological risk assessment of wildfire effects on fish, with and without hazardous fuel treatment. For. Ecol. Manage. 211, 59–72.

Oppenheimer, M., 2005. Defining dangerous anthropogenic interference: the role of science, the limits of science. Risk Anal. 25 (6), 1399–1407.

Poore, D., 2003. Changing Landscapes. International Tropical Timber Organisation and Earthscan Publications Ltd, London, UK.

Raymond, C.L., Peterson, D.L., 2005. Fuel-treatments alter the effects of wildfire in a mixed evergreen forest, Oregon, USA. Can. J. For. Res. 35 (12), 2981–2995.

Riverside Forest Products Ltd., 2004. TFL 49 Ecological Stewardship Plan. Riverside Forest Products Ltd.

Sample, V.A., 2005. Sustainable forestry and biodiversity conservation: toward a new consensus. J. Sustain. For. 21 (4), 137–150.

Sartwell, C., Dolph, R.E., 1976. Silvicultural and Direct Control of Mountain Pine Beetle in Second Growth Ponderosa Pine. USDA Pacific Northwest Forest and Range Experiment Station, Research Note 258.

Schoemaker, P.J.H., 1993. Multiple scenario development: its conceptual and behavioural foundation. Strateg. Manage. J. 14, 193–213.

Seymour, R.S., Hunter Jr., M.L., 1999. Principles of ecological forestry. In: Maintaining Biodiversity in Forest Ecosystems, Cambridge University Press, Cambridge, pp. 22–61.

Seymour, R.S., Hunter Jr., M.L., 1992. New forestry in eastern spruce-fir forests: principles and applications in Maine. Maine Agriculture and Forestry Experiment Station Miscellaneous Publication 716.

Shore, T.L., Safranyik, L., 1992. Susceptibility and Risk Rating Systems for the Mountain Pine Beetle in Lodgepole Pine Stands. Canadian Forest Service. Pacific and Yukon Region. Information Report BC-X-336.

Smith, D.M., Larson, B.C., Kelty, M.J., Ashton, P.M.S., 1996. The Practice of Silviculture: Applied Forest Ecology, 9th ed. John Wiley & Sons, New York.

Speight, M.R., Wainhouse, D., 1989. Ecology and Management of Forest Insects. Clarendon Press, New York.

Spies, T.A., Hemstrom, M.A., Youngblood, A., Hummel, S., 2006. Conserving old-growth forest diversity in disturbance-prone landscapes. Conserv. Biol. 20 (2), 351–362.

The State of Victoria, Department of Sustainability and Environment, 2005. Victoria's State of the Forests Report. Sustainable Forest Management Group, Parks

and Forests Division, Department of Sustainability and Environment, Victoria, Australia.
Toumey, J.W., 1928. Foundations of Silviculture Upon An Ecological Basis, vol. 1. John Wiley & Sons, New York.
Tschakert, P., Olsson, L., 2005. Post-2012 climate action in the broad framework of sustainable development policies: the role of the EU. Clim. Policy 5 (3), 329–348.
Turner, II.B.L., Kasperson, R.E., Matson, P.A., McCarthy, J.J., Corell, R.W., Christensen, L., Eckley, N., Kasperson, J.X., Luers, A., Martello, M.L., Polsky, C., Pulsipher, A., Schiller, A., 2003. A framework for vulnerability analysis in sustainable science. Proc. Natl. Acad. Sci. 100 (14), 8074–8079.
Unites States Environmental Protection Agency, 1998. Guidelines for Ecological Risk Assessment. EPA/630/R-95/002F. Federal Register 63(93): 26846–26924.
van der Pas, J.B., Bulman, L., Horgan, G.P., 1984. Disease control by aerial spraying of *Dothistroma pini* in tended stands of *Pinus radiata* in New Zealand. New Zeal. J. For. Sci. 14, 23–40.
Van Wagner, C.E., 1987. Development and Structure of the Canadian Forest Fire Weather Index System. Canadian Forest Service Forestry Technical Report 35.
von Gadow, K., 2001. Risk Analysis in Forest Management. Kluwer Academic Publishers, The Netherlands.
Weiss, G., 2001. Administrating risk—a social science perspective on natural jazard prevention based on an Austrian case study. In: Risk Analysis in Forest Management, Kluwer Academic Press, The Netherlands, pp. 171–200.
White, R.R., 2004. Managing and interpreting uncertainty for climate change risk. Build. Res. Inform. 32 (5), 438–448.
Whitehead, R.J., Safranyik, L., Russo, G.L., Shore, T.L., Carroll, A.L., 2004. Silviculture to reduce landscape and stand susceptibility to the mountain pine beetle. In: Shore, T.L., Brooks, J.E., Stone, J.E. (Eds.), Mountain Pine Beetle Symposium: Challenges and Solutions. Natural Resources Canada Information Report BC-X-399, pp. 233–244.
Woods, A., Coates, D., Hamann, A., 2005. Is an unprecedented Dothistroma needle blight epidemic related to climate change? Bioscience 55 (9), 761–769.