USE OF PRESCRIBED FIRE IN ECOLOGICAL RESTORATION: LESSONS FROM CHITTENDEN MEADOW, SKAGIT VALLEY, BRITISH COLUMBIA

by

Darren Witt
B.Sc., Simon Fraser University, 1994

RESEARCH PROJECT SUBMITTED IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE OF

MASTER OF RESOURCE MANAGEMENT

In the
School
of
Resource and Environmental Management
Report No. 402

SIMON FRASER UNIVERSITY

Summer 2006

APPROVAL

Name:	**Darren Witt**
Degree:	**Master of Resource Management**
Title of Research Project:	**Use of Prescribed Fire in Ecological Restoration: Lessons from Chittenden Meadow, Skagit Valley, British Columbia**
Report No.	**402**

Supervisory Committee:

Senior Supervisor:	**Dr. Kenneth P. Lertzman** Senior Supervisor Associate Professor
Supervisor:	**Dr. Alton Harestad** Supervisor Professor of Biological Sciences

Date approved: Jun. 6/06.

ABSTRACT

Fire suppression results in dramatic structural and compositional changes in many vegetation communities. The use of prescribed fire to restore a community to historical conditions may be unsuccessful if the trade-off between adequate fire severity and mitigation of fire risk is too conservative. Managers need detailed information on fire behaviour and vegetation response to effectively make decisions about the trade-off between risks and effects. In this report I describe the immediate effects of a prescribed fire in a meadow that has experienced tree and shrub encroachment due to fire suppression and climatic factors. Less than three months after the fire, the meadow community as a whole showed little response to the treatment. Of the life form groups and culturally important plants that I examined, only herbs and grasses showed significant burn effects. Species richness was unaffected by the burn.

Keywords: prescribed fire, restoration ecology, fire severity, fire suppression, management strategies

ACKNOWLEDGEMENTS

I would like to thank my senior supervisor K. Lertzman for his support, comments, and sagely advice on all topics. I would also like to thank A. Harestad for his guidance and pragmatic contributions. He also gets "credit" for helping me decide to enter grad school in the first place. T. Blackbird took the initiative to get this project started, and continues to push the project forward. I would like to thank him for inviting me to be a part of this fun, exhausting endeavour.

T. Semeniuk, O. Love, J. Passmore, V. Michelfelder, and J. Smith (and others I am forgetting) have helped push this document to completion. T. Witt deserves a Medal of Tolerance for her editing and continued presence. I would also like to thank all the Remmers (and my brother) who came out to the meadow to do grunt labour, help with the burn, or just hold up a shovel. Benjamin provided some wonderfully effective distractions.

An NSERC Postgraduate Scholarship provided support for my studies, supplemented by my wife and family.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

1 INTRODUCTION

1.1 Impact of Fire Suppression

Protection of the forest resource through fire suppression has been the dominant paradigm in North America for decades (Langston 1995, Pyne et al. 1996). The absence of fire from the landscape has had significant consequences which may be both unforeseen and undesirable (Rinne 1996, Perry and Amaranthus 1997, Covington 2003). Some of the effects of fire suppression are: encroachment of fire intolerant species, establishment of exotics, altered community structure, and increased fuel loading (Covington 1994, Whelan 1995, Covington et al. 1997, Zimmerman 2003); structural and compositional changes in vegetation communities (Stephenson 1999, Anderson et al. 2000, Fule et al. 2002, Lepofsky et al. 2003, Gildar et al. 2004); more severe and difficult to control wildfires (Covington 1994, Langston 1995, USDA Forest Service 1995, Covington 2000, USDA Forest Service 2001); and escalating costs of suppression activities (Pyne et al. 1996, British Columbia Ministry of Forests and Range Protection Branch 2005e).

Fire suppression is widespread in large tracts of natural and semi-natural lands (Agee 1993, Fule and Covington 1998, Anderson et al. 2000, Copeland et al. 2002, Kuuluvainen et al. 2002, Heuberger and Putz 2003, Wrobleski and Kauffman 2003). Since its formation in 1912, the British Columbia Forest Service has emphasized fire suppression and prevention as a means to protect timber resources, property, and people, and its publications continue to espouse this policy (British Columbia Ministry of Forests

1995, British Columbia Ministry of Forests and Range Protection Branch 2005a, d). Over the last 10 years, British Columbia has spent an average of $90 million per year fighting fires. In extreme years the costs rose to over $265 million, as the interplay of climate, weather, and fuels caused the number of fires to increase 25%, and the area burned to increase tenfold (British Columbia Ministry of Forests and Range Protection Branch 2005e). The escalating costs of fire suppression efforts over the last decade and increased concern from the public (Government of British Columbia 2003, British Columbia Ministry of Forests and Range Protection Branch 2005e) are putting pressure on management agencies to reconsider the policy towards fire and fire suppression (Government of British Columbia 2003, Canadian Forest Service 2004). In light of the ecological, economic, and social consequences of the fire suppression and prevention paradigm, it will be necessary to incorporate additional management tools to build an alternative paradigm for ecological and social settings where it is desirable. Central to this alternative paradigm is the use of prescribed fire (and various mechanical treatments) to manage fuels and control stand structure, with the intent to at least partially restore fire to its historically "natural" role in the environment.

1.2 Fire and Restoration Ecology

If the exclusion of fire from the landscape has significant effects the corollary is that, historically, fire played important roles in shaping the environment. Fire affects organisms directly by burning, and indirectly by changing their environment (Connell 1978, Agee 1990, 1993, Whelan 1995, Pyne et al. 1996, USDA Forest Service 2000, Wrobleski and Kauffman 2003). Direct effects include tissue damage, mortality, and altered plant productivity, phenology, and competitive abilities. Fire indirectly affects an organism by changing species composition and richness, altering hydrological regimes and water quality, altering soil chemistry, nutrient content, insolation, temperature and moisture, and removing and altering the structure of biomass and cover. These factors interact in the post-fire environment in ways that are complex and poorly understood, leading to net effects that can be difficult to predict (Pyne et al. 1996, Boyd and Bidwell 2002, Slocum et al. 2003) and even contradictory (Pyne et al. 1996). The overall effects of a fire depend on the ecology of the species (or community or ecosystem) in question, the specifics of the fire (e.g., timing and intensity), and the response being measured (Whelan 1995, Pyne et al. 1996).

When the consequences of fire suppression on an ecosystem, community, or species have reached levels that are unacceptable to resource managers (or other stakeholder groups), ecological restoration is an option that has been adopted increasingly over recent decades (Hobbs and Norton 1996, Young 1999, Pfadenhauer 2001, Friederici 2003, Davis and Slobodkin 2004). Ecological restoration is the act of recreating an historic ecosystem, or, more generally, the act of restoring one or more processes or attributes to a stand or landscape (Hobbs and Norton 1996). In this context, fire might be introduced into the environment in an attempt to simulate the altered disturbance regime and restore some aspect of the community that has been altered by fire exclusion (Christensen et al. 1989, Davis et al. 2000, Gildar et al. 2004). In an effort to expedite restoration, and in recognition that many ecosystems have followed developmental trajectories that would make the direct application of fire problematic, invasive or encroaching vegetation is often removed mechanically prior to burning (Moore et al. 1999, Radeloff et al. 2000, Meyer et al. 2001, Bailey and Covington 2002, Waltz et al.

2003). The use of mechanical removal of vegetation can be an attempt to speed up recovery of the system (Provencher et al. 2000, Hutchinson et al. 2005), or a recognition that, short of a dangerously intense fire, fire alone may not remove well-established unwanted species or individuals (Provencher et al. 2000, Fule et al. 2002). For example, 16 years of fire exclusion in a Southern Illinois pine barrens caused a successional shift towards a closed forest understory (Anderson et al. 2000). Three successive years of prescribed fire partially reversed the effects of exclusion: prairie species increased, but tree density and basal area were unaffected. Anderson et al. (2000) concluded that the reintroduction of fire alone was insufficient and that mechanical removal of trees would be necessary to restore historic stand structure, and to further promote open woodland and prairie understory vegetation.

In British Columbia, prescribed fire has been used to improve habitat for wildlife and domestic stock (by stimulating forage growth), maintain or enhance habitat diversity, decrease the risk of wildfire or intensity of natural fires (by reducing fuel loads), control fire (through fire breaks), return an essential process to ecosystems, and improve tree growth and overall forest health (Pyne et al. 1996, British Columbia Ministry of Forests and Range Protection Branch 2005d, c, b).

Just as the effects of fire suppression may be unforeseen and undesirable (Pyne et al. 1996, Boyd and Bidwell 2002, Slocum et al. 2003), the results of a prescribed fire may be difficult to predict and contrary to management objectives (Tiedemann et al. 2000). Despite predictive difficulties, a careful consideration of the historic fire regime and the ecological effects of fire on relevant biota and will improve the chances of success (Fule and Covington 1998, Boyd and Bidwell 2002, Copeland et al. 2002, Mulligan and

Kirkman 2002, Slocum et al. 2003). The timing of a prescribed fire, for example, can influence the success of a restoration project. Idaho fescue (*Festuca idahoensis*) prairies in western Washington (USA) benefit from fall fires which promote native species, but the adjacent Garry oak (*Quercus garryana*) woodlands get mixed results from a fall fire (Tveten and Fonda 1999). While fall burning reduces shrub and woody encroachment in the woodlands, it also favours invasive species over native ones. Furthermore, in both ecosystems, fire intervals that are too short or too long (compared to the historic interval) cause loss of prairie and woodland species respectively. This example illustrates the complexity of community response to management treatments, and that incorrect information (or assumptions) concerning the historic fire regime and species responses to fire can affect results in unpredictable and potentially undesirable ways (Whelan 1995).

The use of fire is particularly complicated where suppression (and other management practices) has caused a build-up of fuels which can alter fire behaviour significantly (Stephenson 1999, Anderson et al. 2000, Holmes et al. 2000, Provencher et al. 2000, Fule et al. 2002, Heuberger and Putz 2003, Lepine et al. 2003). Elevated fuel levels can increase the chance that a fire will become extremely intense and difficult to control (Harrington and Sackett 1992, Pyne et al. 1996), providing a strong incentive to burn in cooler and wetter weather to minimize the likelihood of an escaped fire and help preserve valuable community resources (Swezy and Agee 1991, Ruthven and Synatzske 2002). Unfortunately, in many ecosystems cool and wet weather does not correspond with the historic fire season which often occurs when the weather is hot and dry and when a natural ignition source, such as lightning, is present (Agee 1993, Whelan 1995, Pyne et al. 1996). Burning outside of the historic fire season can result in a more patchy, less severe burn (Sparks et al. 2002, Slocum et al. 2003) that favours undesired or invasive species (Boyd and Bidwell 2002, Mulligan and Kirkman 2002). Some research cautions that the use of fire can induce changes in the understory that promotes the spread of invasives, particularly after severe fires (Griffis et al. 2001, Honnay et al. 2002, Lesica and Martin 2003, Sieg et al. 2003).

Finally, failures of fire prescriptions to meet objectives are unlikely to be well documented compared to the successes (Pyne et al. 1996), so managers are less able to learn from and avoid the mistakes of others, such as issues around poor ignition and burning. Although escaped fires are difficult to conceal, where possible, small escapes are likely to be underreported (i.e., classified as wildfires) (Pyne et al. 1996).

The use of prescribed fire is most common in fire-suppressed ecosystems that have historically experienced high frequency, low-severity (stand-maintaining) fire regimes. These include prairies, parklands, barrens, and a few forest ecosystems (e.g., Anderson et al. 2000, Bailey and Covington 2002, Copeland et al. 2002, Kuuluvainen et al. 2002, Heuberger and Putz 2003). In these systems, fire suppression is a major contributor to increased occurrence and increased severity of fires (Covington and Moore 1994, Allen et al. 2002). These systems are rapidly and significantly affected by fire suppression efforts because more fire cycles are interrupted in a given period of suppression activity, causing more divergence from natural ecosystem states (Turner et al. 2003). In British Columbia, the Bunchgrass, Ponderosa Pine, and portions of the Interior Douglas-fir biogeoclimatic zones are most susceptible to significant alteration from suppression activities, with historic fire return intervals ranging from 4 to 50 years (Parminter 1990, British Columbia Ministry of Forests 1995). In systems that experience fire only rarely – for example every one or two centuries – the effects of fire suppression efforts are likely to be very small or non-existent over the span of decades (Turner et al. 2003).

1.3 Case Study: Chittenden Meadow

My study looked at the application of prescribed fire in restoration ecology using Chittenden Meadow as a case study. Lepofsky et al. (2003) conducted a multidisciplinary study of the cultural and ecological history of Chittenden Meadow to determine how it has changed over time and to identify the key factors responsible for those changes. For an undetermined time prior to 1880, Chittenden Meadow was open ponderosa pine parkland probably maintained by frequent surface fires (a keystone process in most ponderosa pine systems; Agee 1998). In this period, fire was frequent enough to kill encroaching Douglas-fir (*Pseudotsuga douglasii*) and grand fir (*Abies grandis*). Approximately 100 years ago, all but the nine remaining mature ponderosa pines were killed (likely by fire, but possibly felled by homesteaders), creating an open meadow with scattered pines similar to what is there today. These conditions persisted until climatic changes (resulting in high spring temperatures and low spring snow packs in the 1970s) permitted the successful establishment of Douglas-fir and grand fir seedlings, while fire exclusion prevented their natural removal from the meadow. The result was a significant decrease in the extent of the meadow and a shift in the composition and structure of the forest community (Figure 1a).

The stewardship and restoration of Chittenden Meadow is important to the Stó:Lô First Nation for historic and cultural reasons and they have been actively involved in research in the area (e.g., Lepofsky et al. 2003). There is substantial evidence of aboriginal activity in the area surrounding Chittenden Meadow over the past 8400 years. This includes the presence of hearths, berry-drying trenches, and various utilitarian artifacts (Franck 2000, Lepofsky et al. 2003). The Nlaka'pamux, Nooksack, Stó:lô, and Upper Skagit used the valley for trade, travel, hunting and gathering food, and gathering non-food products.

Although the historic forest structure and composition in and around Chittenden Meadow was reconstructed by examining tree cores and charcoal deposits (Lepofsky et al. 2003), there has been no similar reconstruction of the meadow vegetation itself. However, the same forces that facilitated tree encroachment have likely also affected the herb, grass, and shrub communities, causing a shift towards a more fire-intolerant, shrub-dominated community that inhibits the establishment of shade-intolerant ponderosa pine (Agee 1993). The same forces that altered the forest community have created a meadow that is substantially divergent not only from the community that existed prior to 1880, but also from that which was protected by BC Parks in 1973 (as a provincial Recreation Area) and again in 1997 (when Skagit Valley Provincial Park was established).

Based partially on the assessment by Lepofsky et al. (2003), BC Parks decided to undertake some restoration to return the meadow to the ecological and cultural state for which it was originally protected. The goals of the Chittenden Meadow Restoration Project were 1) to remove evidence of industrial activities from the meadow, 2) to open up the meadow to its historic extent, 3) to re-establish the presumed historic vegetation community, 4) to return the meadow to an open ponderosa pine parkland, and 5) to increase the cultural value of the meadow vegetation. The first three objectives would be met through the levelling of berms created by industrial activities, the felling and removal of encroaching trees and shrubs, and the reintroduction of fire into the meadow. The remaining two objectives were intended to be met indirectly through completion of the first three. As it was initially conceived, the project would involve years of incremental restoration work and repeated applications of fire in the meadow. The progress of the project would be documented through long-term, continued monitoring that would begin prior to any restoration activities. I was approached by BC Parks to design an experiment around the initial application of prescribed fire, to outline the protocols for long-term monitoring, and to describe the immediate vegetation response to the treatment.

In its initial phase, the Chittenden Meadow Restoration Project had two components: the felling and removal of encroaching Douglas-fir, grand fir, and large shrubs; and the reintroduction of fire to the meadow. These activities were planned and executed by BC Parks with my input when activities affected, or were limited by, experimental design. Whenever necessary, I also provided and organized volunteer labour.

My objectives in this paper are 1) to describe baseline pre-treatment conditions, 2) to describe the change in meadow community in response to the restoration treatment, 3) to establish protocols for monitoring the long-term dynamics of the vegetation change in and around the meadow in response to the initial (and any subsequent) restoration activities, and 4) to use this case study as an opportunity to discuss some of the pitfalls and opportunities of prescribed fire use in hopes of developing better management options for future prescriptions. I have designed the application of the initial restoration treatments as an experiment (within the context of the long-term monitoring program) to describe the immediate vegetation response to prescribed fire in the meadow. I will analyze and discuss how the restoration treatment affected the meadow in terms of: understory species composition; herb, shrub, grass, and total vegetation abundances (cover); abundances (cover) of several culturally important plants for First Nations (Nootka rose [*Rosa nutkana*], Saskatoon serviceberry [*Amelanchier alnifolia*], and tall Oregon-grape [*Mahonia aquifolium*]; (Turner 1995, 1998); and species richness. Finally, I will discuss the correlation between fire severity and vegetation response.

2 METHODS

2.1 Study Site

Chittenden Meadow is in Skagit Valley Provincial Park, British Columbia, Canada, just north of the Canada-United States border (elevation 500 m, latitude 49°01' N, longitude 121°04' W). The meadow is in the warm wet subzone of the Interior Douglas-fir zone (IDFww; BC biogeoclimatic classification system; Meidinger and Pojar 1991). The meadow is in an area where mesic coastal forests meet dry interior forests (Agee and Kertis 1987). The forest surrounding the study area is in the Coastal Western Hemlock zone to the north and west (submaritime and dry submaritime subzones: CWHms1 and CWHds1), and the Englemann Spruce-Subalpine Fir zone to the east (moist warm subzone: ESSFmw). Ponderosa pine – common in the dry interior forests and generally uncommon west of the Cascade crest – occurs in the meadow and in some nearby forests. A nearby ponderosa pine meadow (10 km south) historically experienced fire an average of every 50 years (Agee et al. 1990).

Chittenden Meadow lies on the floodplain of the Skagit River, in the rain shadow of the Pickett Range. The mean annual precipitation is 790 mm (International Joint Commission 1971), most of which falls as winter snow. Snowpack generally persists into April or May. The C horizon underlying the meadow (and surrounding forest) is glacio-fluvial outwash and till, indicating that it is probably an extinct bar of the Skagit River. The meadow soil has a shallow O horizon and a well-defined A horizon. Immediately south of the meadow is the north tip of the Ross Reservoir, which was logged in the

1940s-1950s (Pitzer 1978), and reached its current extent in 1952. When full (a seasonal occurrence), it comes to within a few meters of the south end of the meadow.

Tree species in and around Chittenden Meadow included ponderosa pine (*Pinus ponderosa*), grand fir (*Abies grandis*), Douglas-fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), and poplar (*Populus* sp.). Common herb and shrub species in the meadow included yarrow (*Achillea millefolium*), field chickweed (*Cerastium arvense*), wild strawberry (*Fragaria virginiana*), silky lupine (*Lupinus sericeus*), tall Oregon-grape (*Mahonia aquifolium*), timothy (*Phleum pratense*), Nootka rose (*Rosa nutkana*), Menzies' campion (*Silene menziesii*), common snowberry (*Symphoricarpos albus*), and common dandelion (*Taraxacum officinale*). See Table 1 for a complete list of the species that I identified in the meadow.

Immediately prior to treatments, the meadow had extensive areas of tree and shrub encroachment along the perimeter of the meadow, and several areas dominated by shrubs alone ("shrub fields"; Figure 2). The encroachment stratum varied greatly in depth from a maximum of 33 m on the south-facing side of the meadow, to complete absence over some areas of the north-facing side of the meadow.

2.2 Sampling Design

2.2.1 Stratification of the Meadow

I identified three spatial strata at the study site based on the pre-treatment dominant vegetation there: meadow, encroaching tree and shrub growth ("encroachment"), and mature forest (Figure 3). The meadow was easy to define: it is characterized by the absence of tree or shrub canopy and the dominance of grasses and herbaceous vegetation, so the meadow-encroachment edge is simply where the vertical projection of the tree or shrub canopy begins. Large tracts of shrubs (shrub fields) that were less than 2 m in height were classified as meadow; if over 2 m, they were considered encroachment. In general, when shrub fields were over 2 m, the dominant species was Nootka rose (*Rosa nutkana*), and when shrub fields were under 2 m, they contained primarily Nootka rose, tall Oregon-grape (*Mahonia aquifolium*), or Saskatoon serviceberry (*Amelanchier alnifolia*). The encroachment-forest transition was more difficult to define. In the absence of both dendrochronological data and an obvious border, I defined the end of the encroachment (and the beginning of the mature forest) as the point at which a line (such as a transect) running from the meadow and perpendicular to the meadow-encroachment edge had passed within 5 m of at least 2 trees, each with a diameter at breast height (DBH) of 30 cm or greater. This appears consistent with the edge of the meadow prior to the encroachment of the 1970s as it appears in Figure 1a.

2.2.2 Transect and Quadrat Layout

Before transects were be laid out, I established a reference line down the centre of the meadow. At approximately 25-m intervals along the short axis of the meadow (roughly northwest to southeast), I measured the width of the meadow along a line that ran as perpendicular to the meadow edges as possible, and located the centre point. These centre points formed the reference line from which transects began at random intervals.

I used a clustered sampling design: I collected data from 1 m^2 quadrats along transects that ran from the centre of the meadow, through all 3 strata (Figure 3). Transects ran approximately northwest or southeast from the reference line, and were laid out on a bearing such that they were perpendicular to the edge of the meadow (Figure 3). I rejected transects if their randomly selected starting points were less than 13 m from an adjacent transect. This separation ensured that quadrats on adjacent transects were at least 7 m apart even when they were displaced laterally to increase sampling effort in the encroachment stratum (i.e., when the stratum was too narrow for quadrats to be placed along the transect; see below). Where a transect ran parallel to the meadow-encroachment boundary, it needed to be at least 5 m from that boundary.

After measuring the length of each stratum, I distributed quadrats systematically (evenly spaced) within each stratum. Because transects (and strata) varied in length, the distance separating quadrats also varied. By avoiding a uniformly systematic sampling regime, I have minimized the effect that patterns or clumping in vegetation distribution will have on my estimates (Whysong and Miller 1987). Along a transect, each stratum had at most 4 quadrats, and at least 2, unless the stratum was absent on that transect (the encroachment stratum was absent from 9 transects, and the mature forest from 4). At least 2 m separated any quadrat from a stratum boundary and at least 4 m separated any 2 quadrats on a transect.

The layout of quadrats in the encroachment depended on the length of the encroachment, the minimum separation requirements for adjacent quadrats, and the minimum separation requirements for quadrats adjacent to stratum boundaries. If the length of transect in the encroachment stratum was <20 m long, quadrats were displaced 2 m from either side of the transect to increase sampling effort within the encroachment when minimum separation requirements would only allow 2 or 3 quadrats along the transect. Where the encroachment was less than 9 m in length, there was room for only two laterally displaced quadrats. With less than 5 m of encroachment, there was no room for any quadrats. Because the forest continued beyond the perceived influence of our restoration treatments, I ended all forest transects at 18 m into the forest (the length which 4 minimally spaced quadrats required).

I decided on a 1 m^2 quadrat based on a simple pilot study from which I plotted a species-area curve. Above 1 m^2, the species-area curve flattened, meaning that any increases in quadrat size resulted in only small increases in number of species detected. The practicalities of data collection also affected this decision. To estimate percentage cover, I had to view the entire plot from directly above without disturbing the vegetation. Above 1 m^2, it became difficult to view the entire quadrat from directly above.

2.2.3 Monitoring and Restoration Treatment Activities

To describe the pre-treatment conditions, I sampled the meadow, encroachment, and mature forest in July 2003 – the summer prior to the burn. Initially, I had randomly selected 4 control zones within the meadow; however, a limited budget and concerns about controlling the fire resulted in the treatment (burn) areas being clustered (and therefore non-randomly distributed) in two zones, as depicted in Figure 3.

In late September through October 2003, a cutting crew felled encroaching trees in the meadow and along its periphery. The extent of encroaching tree removal along the edge of the meadow varied from 0 to 13 m in depth. Trees felled were approximately 25 cm in diameter or smaller, although not all trees meeting this size limit were felled. Tree removal was originally scheduled to begin in early August, but the hot and dry weather delayed it until when chainsaws were not likely to spark a wildfire. Due to this delay and the large volume of trees present, the extent of cutting fell short of our goal to remove, from intended burn areas, all trees that had established since the 1970s (i.e., all designated treatment areas in the encroachment stratum). Two days before the burn, a work crew cleared the felled trees and other coarse woody debris from the edge of the forest to avoid the fire burning from this slash into standing trees. Other than at the periphery of the meadow near intact forest, most of the felled trees were left to burn where they had fallen. Many trees that were cleared from their original location in the north end of the meadow were piled in a single large slash pile.

The prescribed fire began at 9:30 AM and ended at 4:30 PM on April 22, 2004. The weather was cool and the vegetation was damp, especially in the morning, when the north end of the meadow was burned. The resulting burn was quite low in severity. Figure 3 shows those portions of the meadow that were burned during the prescribed fire. On May 1, an unplanned burn of unknown origin burned a third region of the meadow, which provided additional treatment data (Figure 3). The result is four time by treatment groups: Control 2003 (transects that were not burned in 2004), Pre-Burn 2003 (transects that were burned in 2004), Control 2004 (Control 2003 transects that had not been burned), and Burned 2004 (transects that have had half or more of their quadrats burned). The comparison of Control 2003 vs. Control 2004 describes the inter-annual variation, while comparing Pre-Burn 2003 vs. Burned 2004 describes the effect of the burn *plus* inter-annual variation (the treatment effect). The net effect of the prescribed fire can be estimated by the difference between the treatment effect and inter-annual variation.

Post-burn vegetation monitoring took place in July 2004. Each transect and quadrat established in 2003 was resampled 2.5 months after the burn. A few quadrats were inaccessible due to high water levels in Ross Reservoir or because debris prevented access. In the pre-burn season, 28 transects and 276 quadrats were sampled. Fifteen transects contained one or more quadrats that were burned the following spring, and 11 transects ran through encroachment zones from which trees had been removed. In the post-burn season of 2004, 33 transect and 280 quadrats were sampled. I placed five additional transects in burned areas of the meadow, and only sampled the meadow stratum of the new transects. The purpose of these extra transects was to increase the sampling effort in burned areas.

To aid in the long-term monitoring of vegetation change in the meadow, I collected baseline, pre-treatment data beyond the requirements of the short-term analysis. Specifically, transects extended into the mature forest to facilitate monitoring the effects of the new meadow edge on forest floor vegetation.

2.2.4 Data Collected

2.2.4.1 Community response to the treatment(s)

Within each quadrat, I estimated cover – the vertical projection of the perimeter of an organism to the ground (McCune et al. 2002) – for individual species, life form groups, and all vegetation (total cover) and averaged the results across quadrats to the stratum level within transects. Only plants under approximately 2 m in height, which originated within the quadrat, and that cast projections within the area of the quadrat were recorded. This was an arbitrary threshold that was my imperfect attempt to trade-off two sometimes-conflicting goals: the need to look directly down on vegetation to estimate cover accurately, and a desire not to exclude tall individuals of species that I considered important to sample, particularly Nootka Rose. Only that part of the vertical projection that fell within the quadrat counted towards estimates of cover. When summed for a quadrat, total cover can exceed 100% because of the presence of multiple layers of plants.

Visual cover estimates are fast and non-destructive, but subjective. To minimize this last source of error, I used a standardized cover estimation key (British Columbia Ministry of Sustainable Resource Management (Terrestrial Information Branch) 2002) and collected two independent estimates (from two individuals) for each datum. If the two estimates disagreed, the measurement was taken again. If the two measurements still disagreed, I averaged them. Cover was scored in classes using a modified Braun-Blanquet (Braun-Blanquet 1965) system with increased resolution at low levels of cover for more sensitivity to, and better estimation of, less abundant species (Mueller-Dombois and Ellenberg 1974, Jensen 1978). Classes have been shown to be effective detectors of community changes over time (Mitchell et al. 1988), are quick to use in the field (United States Forest Service and Fire Sciences Lab 2004), and do not pretend to achieve more accuracy than is realistic given information on human estimation errors (Hatton et al. *in* (Scheller and Mladenoff 2002). Grouping estimates into cover classes has the benefit of preventing the analysis from overemphasizing the dominant species at the expense of those species with medium to low cover values (McCune et al. 2002).

I counted the total number of species in each quadrat excluding grasses, the majority of which were indistinguishable in the field due to the absence of fruiting bodies on most plants and the similarity of the species present. For herbs, trees, and shrubs, I identified each species present, or collected samples for later identification.

2.2.4.2 Fire severity

In the pre-burn field season, on approximately every second quadrat, I placed metal pins into the soil to measure the severity of the burn. Two inches out from each quadrat corner, I sunk a large nail so that its head lay flush with the top of the fermentation layer of the LFH horizon (the partially decomposed organic material on the ground that lies beneath the freshly fallen and individually identifiable needles, twigs and leaves and which rests on the mineral soil). In the post-burn field season, I estimated the amount of fine fuel consumed by the fire by the distance from the top of the nail to the newly exposed surface of the fermentation layer of the LFH horizon layer (or soil, if this layer is entirely consumed).

2.3 Statistical Analyses

2.3.1 Limitations of the Data / Data Set Construction

This preliminary analysis on the short-term effects of treatment on the vegetation community is limited to the meadow stratum. The extent of cutting fell short of our goal to remove all trees that had established since the 1970s. As a result, tree removal affected very few transects and I obtained insufficient data from areas of tree removal for analysis. Furthermore, from an anecdotal point of view, recolonization in areas of tree removal could only have barely begun when post-burn sampling began. At 2.5 months after the burn most of the area where trees had been removed were devoid of vegetation. Change should be slower and less dramatic in the forest and encroachment because the fire treatment was not applied there. However, I expect changes to grasses, herbs, and shrubs will occur because of alterations to the moisture and light regimes. These changes should be gradual and occur through recolonization at the new edge, and result in a resorting of community structure and composition. I expect that as the monitoring program continues, and more data are collected in the encroachment and forest strata, analyses in these areas will reveal changes to the community that have not yet been expressed over the short period of my project. In particular, the removal of the encroachment canopy will result in dramatic changes to the vegetation community of this stratum. The trees (and shrubs) that were removed from the encroachment acted as foundation species, defining the community through their influence on microclimate, nutrient cycling, hydrology, and insolation, and their removal will result in a restructuring of the vegetation community around the altered environmental conditions (Ellison et al. 2005). Finally, I have excluded the trees, moss, and lichen life form groups from all analyses due to the small amount of data that were collected in these categories.

2.3.2 Analysis of Community Response to the Treatment(s)

2.3.2.1 Univariate analyses

To describe the response in abundance of life form groups (shrubs, herbs, grasses), bare ground, and three culturally significant plant species to the prescribed fire, I performed Kruskal-Wallis tests (the non-parametric analog of the ANOVA). I analyzed each relevant pair of treatment groups: Control 2003 vs. Control 2004, and Pre-Burn 2003 vs. Burned 2004. The culturally significant plant species selected by BC Parks for monitoring are: Nootka rose, Saskatoon, and Tall Oregon grape (Turner 1995, 1998). I analyzed species richness in the same manner.

2.3.2.2 Multivariate analyses

To describe changes in the patterns of community structure (abundance and composition) in response to the treatment, I applied non-metric multidimensional numeric scaling ordination (NMDS or NMS; Kruskal 1964 JRS). NMS is well-suited for ecological data because it avoids the assumption of linear relationships among variables (e.g., percent plant cover), makes no distributional assumptions concerning input variables, and is robust to the large number of zeroes common to community datasets

(Fasham 1977, Clarke 1993, Pitkanen 1997, McCune and Mefford 1999, Peterson and McCune 2001). NMS takes the arrangement of plots in the original, n-dimensional data set and reduces the relationship to a small number of dominant, synthetic axes. The intent of this dimensional reduction is to preserve the relationship between plots in n-dimensions by representing as much of the variation in the original data as possible with the smallest number of axes possible to facilitate interpretation. The distance between points in the ordination space is proportional to the distance (dissimilarity) in the original data set.

I performed all multivariate analyses using the PC-ORD 4.25 software package (McCune and Mefford 1999) set on "slow and thorough" autopilot mode: 40 runs with real data, a stability criterion of 0.00001 over 15 iterations, and random starting coordinates. I used a rank-transformed Sorensen's distance measure in the analyses (McCune et al. 2002). To ensure that the NMS is extracting stronger solutions/axes than expected by chance (probability of finding equal or lower stress solutions by chance <0.05), stress in the real data is compared to stress on randomized data (50 runs, Monte Carlo test).

To test explicitly for statistical significance of these graphical patterns, further multivariate tools are necessary. The multi-response permutation procedure (MRPP) is a nonparametric test similar to a multivariate ANOVA that does not require the data to meet the assumption of multivariate normality that is required of parametric tests. The MRPP explicitly tests if the treatment groups contain different communities of plants (i.e., are in different regions of species space in the ordination plots). MRPP describes effect size with the A statistic: the chance-correlated within-group agreement. When all sample units are identical, A=1; when A=0, heterogeneity within groups is equal to that expected by chance. In community ecology, A>0.3 is quite high and significant differences can be found with an effect size less than 0.1 (McCune et al. 2002).

3 RESULTS

3.1 Qualitative Description of the Burn

The north end of the meadow was burned in the morning, when the weather was cool and the vegetation was damp. Early in the day, ignition was difficult, propagation of the fire was poor, and the resulting burn was patchy and very low in severity (Figure 4a, b). Figure 4a shows the ignition problems in the morning: the vegetation burned under the direct heat of the torch, but went out and merely smoked when the torch was removed. In some areas, dry fuels were placed on hot spots in an effort to encourage the fire. As the day wore on, the temperature rose, the wind picked up, and the vegetation dried out, resulting in better ignition and propagation, and a less patchy and more severe burn.

Even later in the day, when the meadow ignited and burned readily, burn severity varied substantially. In some areas the fire would burn as a surface fire under the Nootka rose, leaving new green growth above the fire unscorched (Figure 4c and 4d), while in other areas it would jump into the crown of the shrub canopy and have no effect on the surface vegetation below. Where surface fuels were abundant, the front moved more rapidly and the burn was more intense and homogeneous (Figure 4e and 4f) than in those areas where fuels were less abundant, for example, in areas of low grass (Figure 4g and 4h).

3.2 Univariate Analyses: Life Form Groups and Culturally Important Species

In the meadow stratum, I identified 64 species of plants (I did not distinguish grass species) and three life form groups: grasses, shrubs, and herbs. In all treatment groups (Control 2003, Control 2004, Pre-Burn 2003, and Burned 2004), shrubs were the most dominant life form in terms of median cover, followed by herbs and grasses (Figure 5). The median cover of bare ground – the complement of total cover – was greater than all life form groups, across all treatment groups. Notwithstanding statistical significance, the net effect of the fire (treatment effect minus year effect) was positive for herbs, grasses, and bare ground and negative for shrubs (Table 2). Due to the effects of multiple layers of vegetation, the sum of all cover values for a treatment group in Figure 5 and Table 2 may exceed 100%.

Table 2 shows summary statistics and the results of significance tests between the treatment groups. Using a non-parametric (Kruskal-Wallis) test to compare Control 2003 vs. Control 2004 and Pre-Burn 2003 vs. Burned 2004 with data grouped into life form categories (shrub, herb, and grass) and bare ground, I found that three comparisons of within-group response were statistically significant. Two of the three significant results concerned herbs. For herbs, there was both a significant treatment effect (burn plus inter-annual variation; asymptotic significance = 0.003), and a significant effect of year (asymptotic significance = 0.032). Average herb cover increased 6.21% in the control group, and 8.22% in the burn treatment group, so the net effect of the fire was a 2.01% increase in herb cover (Table 2). The third significant result was in grasses, which showed a significant treatment effect (burn plus inter-annual variation) that was not confounded by inter-annual variation (asymptotic significance = 0.012). Average grass cover increased 3.87% in response to the fire. Net average shrub cover decreased and bare ground increased non-significantly in both treatment and control comparisons implying, perhaps, the beginnings of trends.

None of the three culturally important species (Nootka Rose, Saskatoon, and Tall Oregon grape) or species richness overall showed any significant effects of treatment (burn plus inter-annual variation) or time (Figures 6 and 7, Table 2). Nootka rose, tall Oregon-grape, and species richness exhibited non-significant net decreases in response to the fire, with tall Oregon-grape showing the largest change, making it a likely candidate for the beginning of a trend: -2.97% (Table 2). In contrast, average Saskatoon cover showed a non-significant net increase in response to the fire: 1.28% (Table 2). The magnitude of the non-significant change in tall Oregon-grape may indicate the beginning of a trend. However, based on their ability to survive light severity fires, I would expect Nootka Rose, Saskatoon, and Tall Oregon grape to be unaffected by the prescribed fire (Haeussler et al. 1990, Walkup 1991, Howard 1997).

Although I was primarily interested in the three culturally important species and community-level changes, I examined the data of individual species to look for interesting trends. I found none. Of 64 species, none had a net change in response to the burn (absolute value of the treatment effect minus year effect) greater than 4%. Fifty-nine species had a net change that was less than 1%. The 5 remaining species included tall Oregon-grape and Saskatoon, on which I have reported above. Silky lupine (*Lupinus sericeus)* showed a net increase of 1.25% in response to the burn, and thimbleberry (*Rubus parviflorus*) a net decrease of 1.19%. Birch-leaved spirea (*Spirea betulifolia*) exhibited the greatest net response to the treatment: a 3.19% increase. None of these changes were statistically significant.

3.3 Establishing the Multivariate Statistical Argument

3.3.1 Comparing Control 2003 and Pre-Burn 2003 Transects

Figure 8 shows the results of the nonmetric multidimensional numeric scaling (NMS) reduction of the 61 dimensions of the Control 2003 and Pre-Burn 2003 treatment group data (percent cover of all species present in at least one transect, and bare ground). The NMS suggests that a 3 dimensional solution best represents the data (85.7% of variation explained). The results of the multi-response permutation procedure (MRPP) indicate that the two groups – Control 2003 and Pre-Burn 2003 – are statistically distinct ($A = 0.089$, $P = 0.001$). This prevents the two groups of data from being pooled in further analyses. I therefore ran two parallel analyses, comparing Control 2003 with Control 2004, and Pre-Burn 2003 with Burned 2004. The difference between Control 2003 and Control 2004 reflects the inter-annual variation. Comparing Pre-Burn 2003 and Burned 2004 will give the magnitude of the treatment effect combined with the effect of one year passing (inter-annual variation). The difference between the effect sizes of these two comparisons – Control 2003 vs. Control 2004 and Pre-Burn 2003 vs. Burned 2004 – is an estimate of the effect size of the burn treatment.

3.3.2 Effect of Time: Comparing Control 2003 Transects and Control 2004 Transects

Figure 9 shows the results of the NMS reduction of the 63 dimensions of the Control 2003 and Control 2004 treatment group data (percent cover of all species present in at least one transect, and bare ground). The NMS again suggests that a 3 dimensional solution best represents the data (83.2% of variation explained). The results of the MRPP indicate that the two groups – Control 2003 and Control 2004 – are statistically indistinguishable: there is no statistically significant effect of year in the meadow ($A = 0.022$, $P = 0.134$). This means that I can assume that any effect between groups in the next analysis is due to the treatment that was applied and not the passage of one year.

3.3.3 Effect of Burning (and Time): Comparing Pre-Burn 2003 Transects and Burn 2004 Transects

Figure 10 shows the results of the NMS reduction of the 65 dimensions of the Pre-Burn 2003 and Burn 2004 treatment group data (percent cover of all species present in at least one transect, and bare ground). The NMS suggests that a 3 dimensional solution best represents the data (84.6% of variation explained). The results of the MRPP indicate that the two groups are statistically indistinguishable: at the time of sampling, there is no statistically significant effect of the restoration treatment on the meadow community ($A = 0.044$, $P = 0.090$).

3.3.4 Non-Significant Trends in the Meadow Community Response

Although there was no measurable community response to the prescribed fire, there is a potentially interesting non-significant trend in the results: the effect size of the treatment (burn plus inter-annual variation) was twice the size of inter-annual variation. The effect size of inter-annual variation (or time; represented by the chance-corrected within-group agreement statistic, A; McCune et al. 2002) is 0.022 (Figure 9), whereas the effect size of the treatment is 0.044 (Figure 10). This difference in effect sizes, though statistically insignificant, may indicate the beginnings of a response to the prescribed fire that could develop over time. This difference is evident in the 3-dimensional ordination plots of the burn effect and the effect of time: the separation among groups in Figure 10 is somewhat greater than in Figure 9.

3.4 Fine Fuel Consumption

The severity of the fire was low enough that it only burned organic matter above the fermentation layer of the LFH horizon at most locations in the meadow. Where the fire appeared to burn down into the fermentation layer, it was to such a small degree that I was not able to detect a change in fermentation layer depth on any of the 48 fuel consumption pins.

Although the sampling data describe a fire of uniformly low severity (i.e., undetectable), this was not strictly the case. During the burn, the fire burned much more severely at the bases of large trees than in the rest of the meadow due to the presence of large amounts of dry fuel (twigs and needles). The number of large trees in the burned areas (5 ponderosa pines) means that these high severity burn areas are not substantial contributors to the overall severity of the burn and are not representative of the prescribed fire. I did not avoid sampling these areas, but the random placement of transects did not pick them up.

4 DISCUSSION

4.1 Summary of the Immediate Response to the Treatment

One of the goals of the Chittenden Meadow restoration project was to use prescribed fire to reduce tree and shrub cover and re-establish the historic, fire-dependent, vegetation community. Unfortunately, in the short-term, the meadow community as a whole showed no statistically significant response to the prescribed fire. With the exception of herbs and grasses, breaking down the community dataset into life form groups and culturally important species (and species richness) did not yield any statistically significant results. Although the herb and grass results were statistically significant, it is questionable whether the results are biologically significant: herbs exhibited a net 2.01% and grasses a 3.87% increase in average cover in response to the treatment.

4.2 An Explanation of the Response

It is neither unprecedented nor surprising for a prescribed fire to produce no substantial changes in a vegetation community (Haeussler et al. 1990, Pyne et al. 1996, Tiedemann et al. 2000, Fule et al. 2002, Rideout et al. 2003, Carter and Foster 2004). The prescribed fire in Chittenden Meadow failed to produce significant, detectable effects either because the fire was of inadequate severity, we were unable to detect changes (due to the timing of post-fire measurements, weather variations, or an unrepresentative control group), or due to some combination of these factors.

I suspect that the primary reason for the lack of significant changes in Chittenden Meadow is that the severity of the burn was too low – so low that there was no measurable consumption of the fermentation layer. The fire burned through only the loosely packed litter layer (above the fermentation layer), which produced a dramatic yet superficial burn with little to no effect on vegetation cover. A higher severity burn would likely have yielded significant results in the short-term (i.e., prior to post-treatment observation) by killing more plants or by consuming more aboveground biomass. Although I lack data to directly describe biomass consumption, the cover data (and a subjective assessment) imply that few plants were consumed by the fire, or that any loss of living tissue was offset by regrowth prior to post-burn observation.

Simply having a higher severity fire in Chittenden Meadow would not guarantee a dramatic vegetation response. Even prescribed fires with a statistically detectable effect on fuel loads or other species may not be of adequate severity to affect the understory vegetation. For example, despite prescribed fires that reduced forest floor depth 41-78% in a ponderosa pine forest, cover and species richness of the understory plant community were not affected (Fule et al. 2002). Similarly, in a Texas "pineywoods" ecosystem (mixed pine/deciduous forest), prescribed fire significantly affected sapling mortality and litter depth, but herbaceous species abundance and species richness were not affected (Rideout et al. 2003). These results indicate that managers are more likely to meet restoration objectives if they have detailed information on plant responses to fires of varying severity, and a clear sense of what level of severity is needed to achieve a desired response.

After severity, the most likely explanation for the lack of significant changes in Chittenden Meadow is the timing of post-fire measurement. My analysis only describes the immediate vegetation response to the burn: the short-term results of what should be a long-term restoration and monitoring project. Post-burn vegetation sampling occurred 2.5 months after the prescribed fire, so it is not surprising that there is no significant response to the burn (and mechanical treatments). A response documented immediately after treatment may not reflect trends over subsequent years (Haeussler et al. 1990, Fule et al. 2002); it is possible (although I believe unlikely) that even with our low severity burn, further changes in Chittenden Meadow may become evident over time. For example, the fertilization effect of a fire (an immediate increase in soil nitrogen and other nutrients in the soil) may stimulate the growth of understory vegetation in the short-term (i.e., 1-2 years post-burn), while erosion and leaching in subsequent years may cause a decrease in nutrients below pre-fire levels, resulting in further, more dramatic changes to the understory vegetation (Pyne et al. 1996, Tiedemann et al. 2000, Carter and Foster 2004). Further monitoring in the meadow is therefore warranted.

Two additional details of the experiment may have affected my ability to detect changes in meadow vegetation. First, the weather differed dramatically between my two field seasons. Pre-treatment data was collected in July 2003, which was a very hot and dry month. In July 2004, when the post-burn data was collected, it was comparably cool and damp. Although I attempted to address inter-annual variation in my analyses, this variation in weather may still obscure the effects of the burn. This pattern of weather variation could exaggerate the effect of the burn on fire tolerant species. Held to low abundances by the hot dry pre-burn weather, fire-tolerant species might appear to benefit greatly from the burn, when some of the increase was attributable to more favourable growing conditions in the post-burn months. Conversely, the boost in growth from favourable post-burn weather may mask some of the deleterious effects of the burn on fire intolerant species. It is almost certain that the variation in weather between field seasons adds noise to the analysis, and impairs my ability to detect an effect of the treatment.

Second, and more problematic, is that the MRPP results indicate that Control 2003 and Pre-Burn 2003 were significantly different prior to burning, a fundamental shortcoming for a control group. This is unsurprising, given that transects were not randomly assigned to treatment groups and the fact that a brief walk through the pre-treatment meadow shows that regions of the meadow only a few meters apart are often quite strikingly dissimilar. On the day of the burn, this was compounded by changes made to the distribution of the actual area burned, which confounded the original, more formal, layout of control and treatment groups. In the assignment of transects to treatment groups, logistics necessarily dictated experimental design – grouping together the areas to be burned made the fire much easier to control, as did selecting burn areas for the absence of ladder fuels – but this meant that we did not achieve true interdigitation of replicates (Hurlbert 1984). The dissimilarity between Control 2003 and Pre-Burn 2003 requires a leap of faith that is statistically indefensible. To be able to use the control group to net-out the inter-annual variation, I have to assume that my distinctly different pre-treatment group nonetheless responded similarly to my controls.

4.3 Management Lessons from Chittenden Meadow

4.3.1 Burn Severity: Trading Off Effect Size with Controllability and Over-burning

The results from Chittenden Meadow illustrate a key trade-off associated with prescribed fire that needs to be made explicit. With increasing burn severity, both the likelihood of achieving meaningful results in the target community and the risk of losing control of a fire increase (Whelan 1995, Pyne et al. 1996; Table 3). Furthermore, variation of severity within the burn (Figure 11) means that with increasing severity, a larger proportion of the plant community experiences a burn that may produce deleterious effects on the plant community (Agee 1993, Whelan 1995, Pyne et al. 1996, Fule and Covington 1998, Slocum et al. 2003, Clarke et al. 2005). Careful consideration of these trade-offs is an important part of preparing a burn prescription, but the process was not transparent in the Chittenden Meadow prescribed fire.

A low severity burn, such as the top curve in Figure 11, will maximize the controllability of the fire (and thus minimize the risk of an escaped fire) but it will also minimize the proportion of the community that responds measurably to the treatment. In this scenario, only a very small proportion of the community experiences a burn severity beyond the threshold necessary to produce ecological meaningful (and statistically significant) effects. At intermediate (medium) burn severities (center scenario in Figure 11), the majority of the community will respond significantly to the fire while the risk of escapement is increased, but still acceptable. As severity increases further (the bottom curve in Figure 11), the probability of having an uncontrollable fire increases to unacceptable levels, although more of the community may fall between the minimum and deleterious effect thresholds.

The net effect of a prescribed fire depends not only on the overall severity, but also on the distribution of severity within the community (Agee 1993, Whelan 1995, Pyne et al. 1996, Fule and Covington 1998, Slocum et al. 2003, Clarke et al. 2005). Figure 11 shows the range of impacts (burn severity) for low, medium, and high severity burns. With a low severity fire (the top scenario of Figure 11), very little of the community experiences a high severity burn (over-burning) and the negative consequences associated with it (e.g., removal of desirable species, removal of all vegetation, exposure of bare soil, promotion of invasive species, or burning off of the organic layers in the soil). With increasing severity, the proportion of the community experiencing a burn severity beyond the deleterious effect threshold will increase. Because it is unlikely that managers will know the exact shape of their particular curve prior to the burn, they will not know what proportion of the community will experience a severe burn with deleterious effects and the net effect on the community (and project) will be impossible to predict. However, it is clear that at some point the negative consequences in over-burned areas will outweigh the positive effects elsewhere. In Chittenden Meadow, we noted that the areas under large trees burned with a greater severity than the rest of the meadow. The high severity burn in these areas should be a concern to anyone burning in meadows with a larger proportion of area under large trees because they may present control issues or result in larger areas of high severity burn than expected.

The trade-off between the effect size and controllability of a burn has particular relevance to the results of the prescribed fire in Chittenden Meadow. It was our intent to produce a fire in which most of the meadow would respond measurably to the treatment and that the risk of escape would be acceptably low, similar to the middle scenario of Figure 11. However, we were so risk-averse concerning controllability and burning the meadow too severely (and producing deleterious effects) that we unintentionally "chose" an extremely low severity burn that is better described by the top scenario of Figure 11. We burned in the cool, wet spring, whereas fires in this community are more likely to occur naturally at the end of summer or in early fall when fuel is drier and a fire is likely to burn more severely (Van Wilgen et al. 1990, Agee 1993). Furthermore, the burn day was selected because weather ensured that indices of fire risk were low. The result was an extremely low severity burn that compromised our ability to get meaningful effects in the meadow, at least as it stood 2.5 months after the prescribed fire. Had we more explicitly considered the trade-offs associated with the Chittenden Meadow prescription we may have altered our approach.

On the other hand, there are benefits to our risk-averse strategy that may not be obvious initially. It is not trivial to have reintroduced fire into the meadow and avoided catastrophe! Burning in the spring may have meant compromising our ecological objectives (at least in the short-term), but it helped us produce a fire that was easy to control and that did not escape and burn down the forest outside of our planned burn perimeter. A low severity burn also meant that we did no (or very little) harm to the meadow community: there were no large-scale negative consequences of the burn to the vegetation such as those described in Table 3. Starting off gradually allows us to increase burn severity in future prescriptions to produce the desired effects in the community. It would have been difficult to impossible to recreate the community had we over-burned large portions in a high severity fire. Furthermore, the low severity burn allowed us to gain experience with planning, executing, and controlling a burn in circumstances that were safe and low-risk. We gained knowledge about how a low severity fire behaves in Chittenden Meadow, and established a baseline estimate of vegetation response to low-severity prescribed fires. Both will help us calibrate future prescriptions within the meadow.

Finally, it is worth noting that for any prescription (i.e., any intended severity) there is a chance of obtaining a burn of unintended severity. Even with thorough planning and preparation, a fire can burn more or less severely than intended and result in a discrepancy between intended and realized effects that can be dramatic and potentially catastrophic (Pyne et al. 1996). In terms of Figure 11, this represents a jump from the intended scenario to one that is more severe.

4.3.2 Managing Risk and Burn Severity

There are various strategies managers can employ to manage the risk and burn severity concerns that I have discussed above. The primary means for exerting control over a prescribed burn are through fuel reduction, burn timing (season and weather), fire fighting resources, and experienced and knowledgeable personnel (Pyne et al. 1996).

Fuel load reduction is a strategy for managing the risk of escaped and potentially catastrophic fires and for reducing burn severity where fuel loads are high. Fire is often prescribed in systems that have experienced prolonged fire suppression where elevated quantities of surface and ladder fuels can alter fire behaviour, increasing the risk of a catastrophic runaway fire (Agee 1993, Whelan 1995, Pyne et al. 1996, Agee 1997, USDA Forest Service 2000). Fuel loads can be reduced or removed prior to burning by mechanically thinning stands or removing surface fuels (Moore et al. 1999b, Radeloff et al. 2000b, Meyer et al. 2001b, Bailey and Covington 2002b, Waltz et al. 2003b, Zimmerman 2003). Mechanical removal of encroaching vegetation in Chittenden Meadow served not only to open up the meadow, but also to remove ladder fuels from the forest edge.

As we have seen from the results in Chittenden Meadow, burning in the season (and on days) when weather conditions inhibit fire can be an effective strategy to minimize the risk of an escaped fire (Snyder et al. 1999, Rideout et al. 2003, Zimmerman 2003) and minimize over-burning (Pyne et al. 1996). However, managers must take care because a significant departure from the historic season (or frequency or extent of burn) may cause deleterious or counterproductive effects to biota, making the system deviate further from historic conditions (Whelan 1995, Copeland et al. 2002, Sparks et al. 2002, Slocum et al. 2003). For example, it can take several years for plant structures and seed banks to recover from an early fire that has disrupted growth and seed production (Whelan 1995). Season of burn can also influence the severity of a burn and therefore its effects on vegetation. The higher severity of a late summer/early fall burn is more successful at reducing the numbers of encroaching or invasive species in a number of systems (including Chittenden Meadow), as compared to off-season burns that are generally lower in severity (Tveten and Fonda 1999, Boyd and Bidwell 2002, Copeland et al. 2002, Sparks et al. 2002). To better identify the practical tradeoffs associated with burn timing, many researchers have investigated the ecological effects of burn season for their specific systems (Boyd and Bidwell 2002, Copeland et al. 2002, Mulligan and Kirkman 2002, Ruthven and Synatzske 2002, Sparks et al. 2002, Slocum et al. 2003, Wrobleski and Kauffman 2003). This information would be particularly valuable when planning a prescribed fire.

Methods for controlling a fire through burn timing and fuel reduction are not mutually exclusive. Elevated fuel loads can be reduced gradually over a series of off-season prescribed burns, using a strategy that addresses both risk and burn season explicitly (Agee 1993, Pyne et al. 1996, Rideout et al. 2003). In this approach, accumulated fuels are consumed over successive burns (over several years) that begin in the off-season when the risk of an escaped fire is low. As fuel loads diminish over successive burns, burning is shifted towards the natural fire season. Once the reduction in surface and ladder fuels has adequately reduced risks of escape and negative effects on target species, fire can be prescribed in warmer, drier months when fires are historically more common. This phasing-in of natural-season fires will require substantial knowledge on fire behaviour, fuel loads, and ecology of target species. Although it initially has the same caveats as a simple strategy of off-season burns, the implicit presumption is that, over successive burns (each closer to the natural fire season), the potential negative ecological effects of off-season burning will disappear as burning is phased-in to the natural fire season.

Beyond fuel load reduction and burn timing, managers can exert control over a fire directly through the presence of appropriate levels of on-site and emergency firefighting resources and indirectly through the use of skilled personnel. Appropriate levels of firefighting resources are necessary for any burn. Experienced and knowledgeable personnel planning a prescription will be able to address fuel loads and burn timing issues in a manner that does not compromise the goals of a project and does not create unnecessary risk. During the execution of the burn, they will be able to deal with situations as they arise and, more importantly, prevent them from arising in the first place.

The Chittenden Meadow Restoration Project suffered somewhat from the lack of integration between burn planning and monitoring personnel. With better integration (and more explicit and frequent communication) we could have ensured that the monitoring design better complemented the logistics of the fire prescription and that the prescription did not needlessly compromise the design of the monitoring program. In general, more decisions should have been made with consultation of other members of the team. Greater expertise in both the planning and monitoring areas of the project may have helped avoid pitfalls that now seem fairly obvious. For example, I should have been able to foresee that grouped burn zones were more pragmatic than dispersed zones and that deliberately placing burn zones in a manner that suited the specific characteristics of the site was more appropriate given the size of our team on the day of the burn.

4.4 The Future of Chittenden Meadow

The future of Chittenden Meadow will depend largely on how restoration activities proceed. The restoration of Chittenden Meadow was conceived as a long-term, multi-year project which began with an attempt to open up the meadow to its historic extent (through removal of encroaching vegetation) and to shift the meadow vegetation closer to the presumed historic community (through the reintroduction of fire). Neither of these two goals was satisfied completely. Due to logistical problems (poor weather and insufficient labour), the extent of cutting fell well short of the goal. As I have found, the single prescribed fire did not (in the short-term) shift the meadow community to its presumed historic structure and composition.

Management-induced changes in community composition and structure are often temporary, even when change is dramatic, unlike the results in Chittenden Meadow (Antos et al. 1983, Pendergrass et al. 1999, Boyd and Bidwell 2002b). Even with repeated applications of fire over years, a return to management without fire can reverse beneficial effects within a few years (Anderson et al. 2000). Therefore, it is logical to assume that with no further interventions (and, presumably, with continued fire suppression) the most likely future for Chittenden Meadow is a return to its pre-restoration state: an increasingly encroached, fire-intolerant, shrub-dominated community. However, there is a small possibility that the response to the prescribed fire was delayed beyond 2.5 months, and it is possible, although unlikely, that these changes could increase over time.

I have recently been informed by the project manager at BC Parks that additional cutting and disposal of encroaching vegetation has progressed significantly, and that all but one of the industrial berms have been levelled. BC Parks plans to flatten the remaining berm and remove an additional 4 hectares of encroachment over the next two years, which will result in (approximately) a 40 % increase in the size of the meadow over its current extent. In addition to these mechanical treatments, BC Parks plans to burn the entire meadow in or before June 2006, and burn a third time in the following two years.

The recent (and imminent) restoration activities will move the Chittenden Meadow Restoration Project closer to its goals, however, additional intervention will still be necessary to complete and maintain the project. We need to assess whether the felling (and disposal) of encroaching vegetation has restored the historic extent of the meadow, and do additional work if necessary. The levelled berms should be monitored closely, and action taken if necessary, to ensure that invasive species do not dominate the exposed soil. A second burn over the entire meadow is good progress, but is not and end point of restoration. With the application of periodic and more severe burns (consistent with the historic fire regime), we can affect substantial and lasting change in the meadow community and re-establish the presumed historic community. To ensure that restoration proceeds appropriately and to guide the frequency, timing, and severity of future activities, I recommend regular monitoring of the plant community.

Details of this regular monitoring program are not critical to my recommendation and I will leave them to whoever designs the program. However, I will suggest the framework of a basic program to monitor changes in the plant community. To reduce the labour involved (compared to my methods), 5 to 10 permanent sites could be established in the meadow. Every 2 to 3 years, the following data should be collected:

- photographs at each direction perpendicular to the forest edge and parallel to the meadow's long axis;
- a photograph directly down at the meadow vegetation (capturing the entire sampling unit or quadrat);
- vegetation cover and;
- an aerial photograph of the meadow.

For the cover data, it would be sufficient to sample life form groups (i.e., trees, shrubs, herbs, and grasses) during the late spring or early summer after green-up of the meadow. If this is done before the summer heat dries the vegetation, the life forms and percent cover can be identified more easily and accurately. If sampling is warranted or desired for a limited number of focal or indicator species, it would not add greatly to the cost of the monitoring program. Some potential indicator species are marked in Table 2 with an asterisk and are consistent with indicators in Meidinger and Pojar (1991) and USDA Forest Service (2006). If the results of this monitoring program warrant it, a more detailed survey can be conducted (similar to the my methods) on an infrequent basis.

5 CONCLUSION

The Chittenden Meadow restoration burn failed to produce significant changes in the short term because minimizing the risk of a catastrophic fire dominated over considerations of the ecological impacts of an off-season burn. However, in the context of an ongoing restoration project, this approach meant that we avoided catastrophe and will be able to calibrate future prescriptions based on the experience gained.

Our experiences in the first phase of the Chittenden Meadow Restoration Project can help guide managers who are considering similar restoration projects. Some of the important lessons include:

- Managers need to design a prescription that addresses the potential risks and logistics without sacrificing their ability to produce results. The prescription should recognize the expected format of the project (single or multiple interventions);
- Prescriptions should incorporate the best available information on predicted fire behaviour and plant responses under natural and elevated fuel conditions and in various weather conditions;
- Managers need to understand that, despite all efforts, they may not get the burn (and results) that they had intended and that an adaptive management perspective is appropriate; and
- The prescription needs to ensure appropriate on-site, ground-based firefighting resources as well as availability of air support in the event of an escaped fire.

Figure 1 (a) Aerial photographs of Chittenden Meadow in 1946 and 1998 showing the recent encroachment of trees and shrubs. (b) Photographs showing encroaching vegetation from the ground.
Photographs in (a) are reprinted with permission from Dana Lepofsky, © Lepofsky et al. 2003.

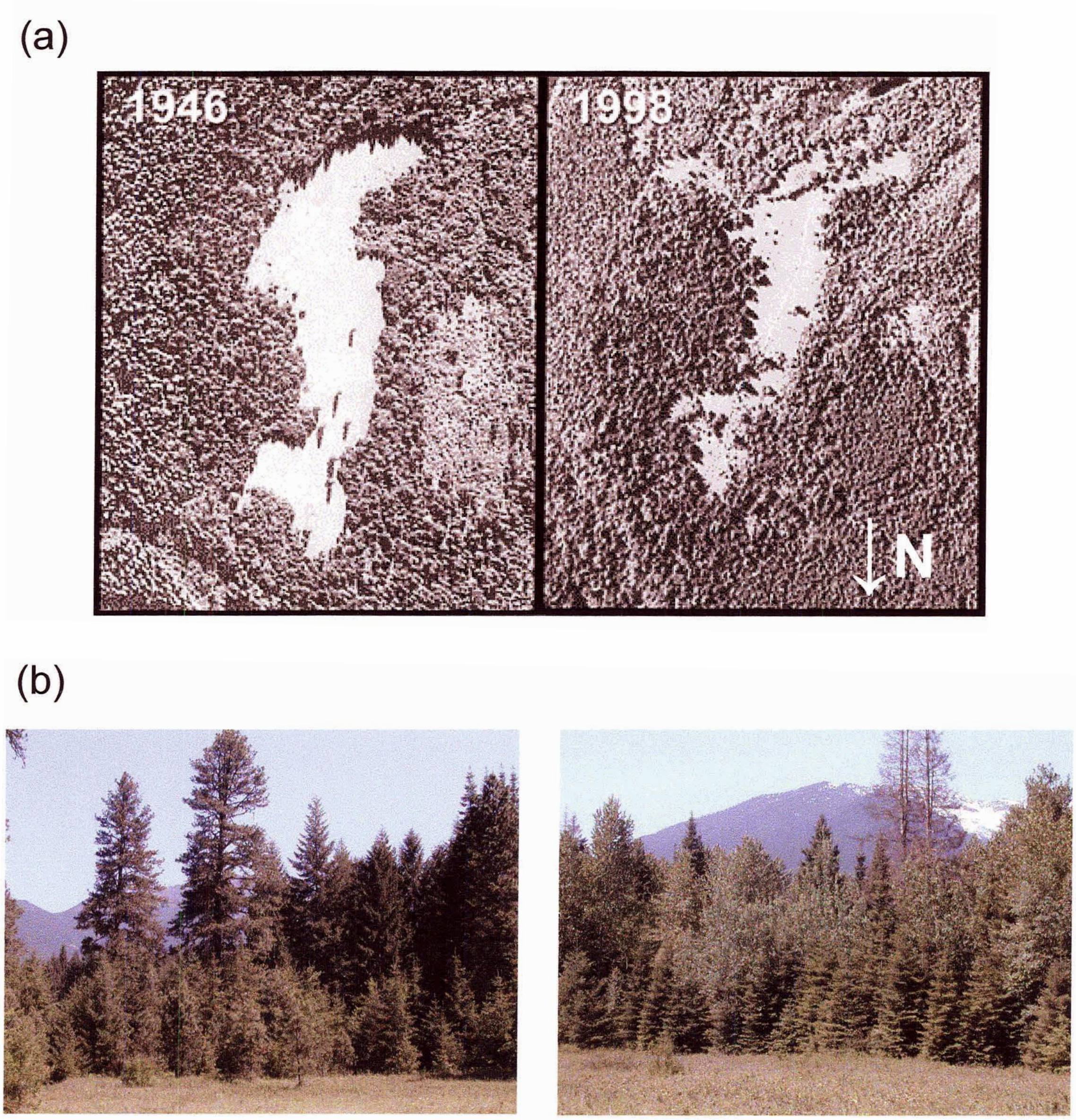


Figure 2 Map of Chittenden Meadow showing the approximate extent of pretreatment tree and shrub encroachment and the locations of shrub fields.

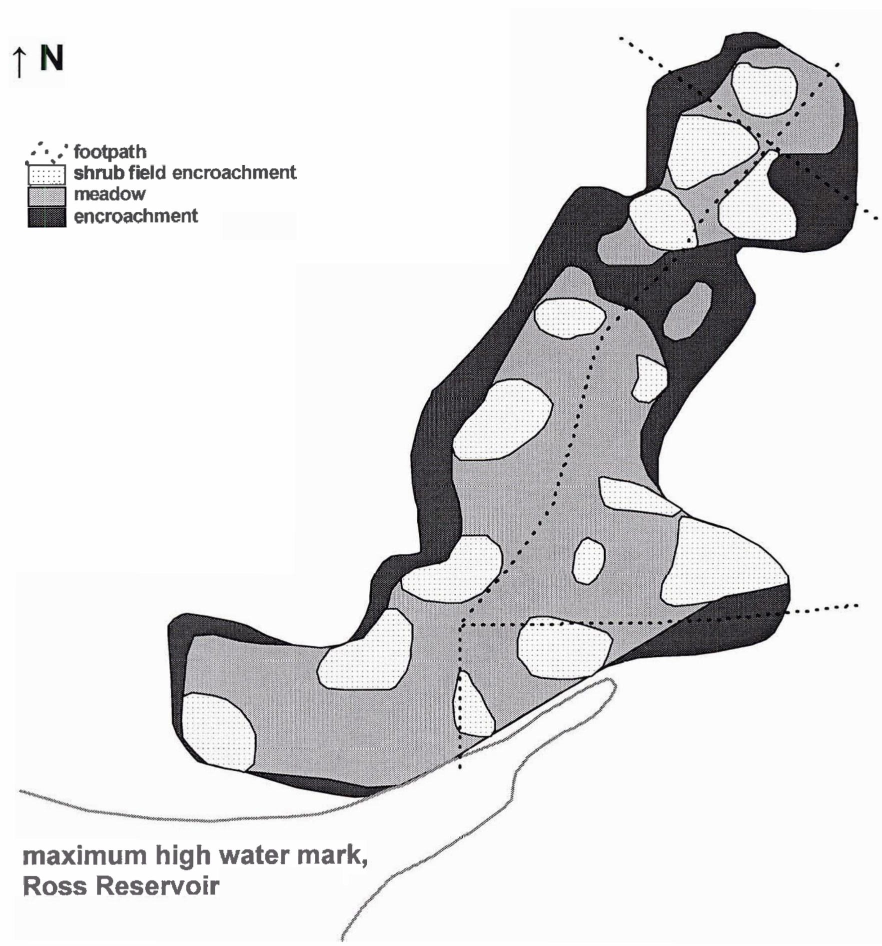


Figure 3 Map of Chittenden Meadow showing strata (meadow, encroachment, and mature forest), transects, and the areas of the meadow that were burned.
In most cases, transects begin in the meadow, run through the encroachment, and end in the forest (the white beyond the meadow). The dotted regions indicate the two areas burned during the prescribed fire on 22 April, 2004. Diagonal lines indicate another area that was burned during a wildfire of unknown origin 9 days later.

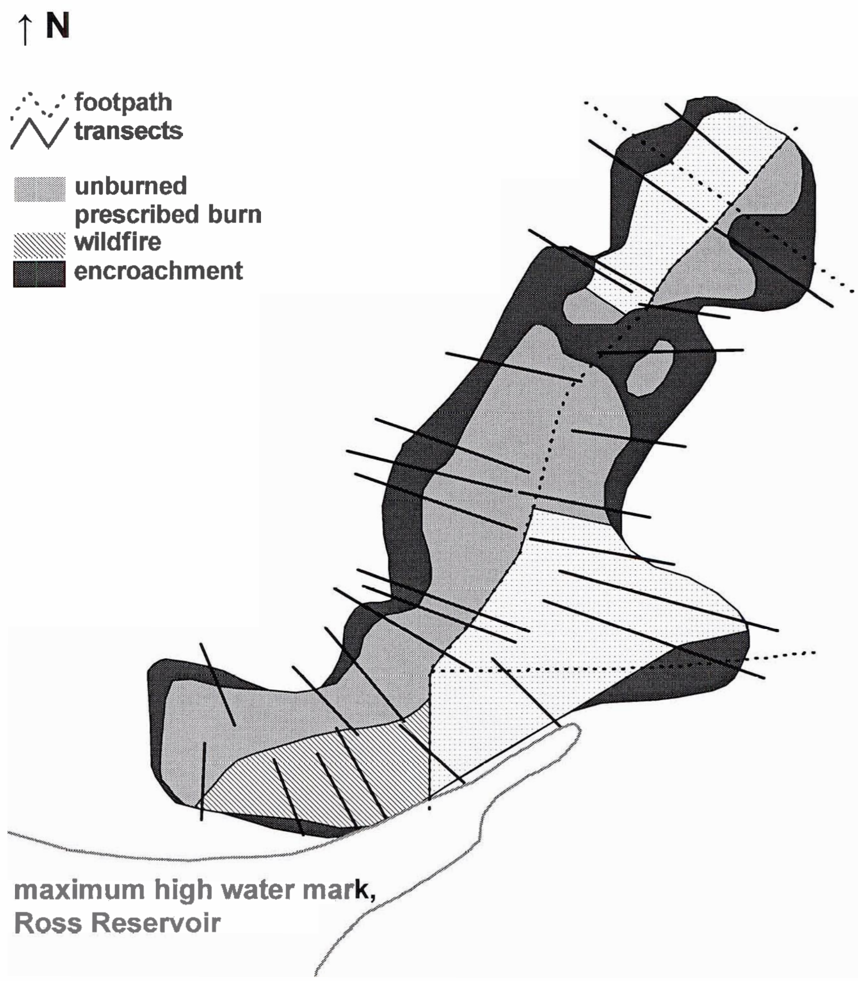


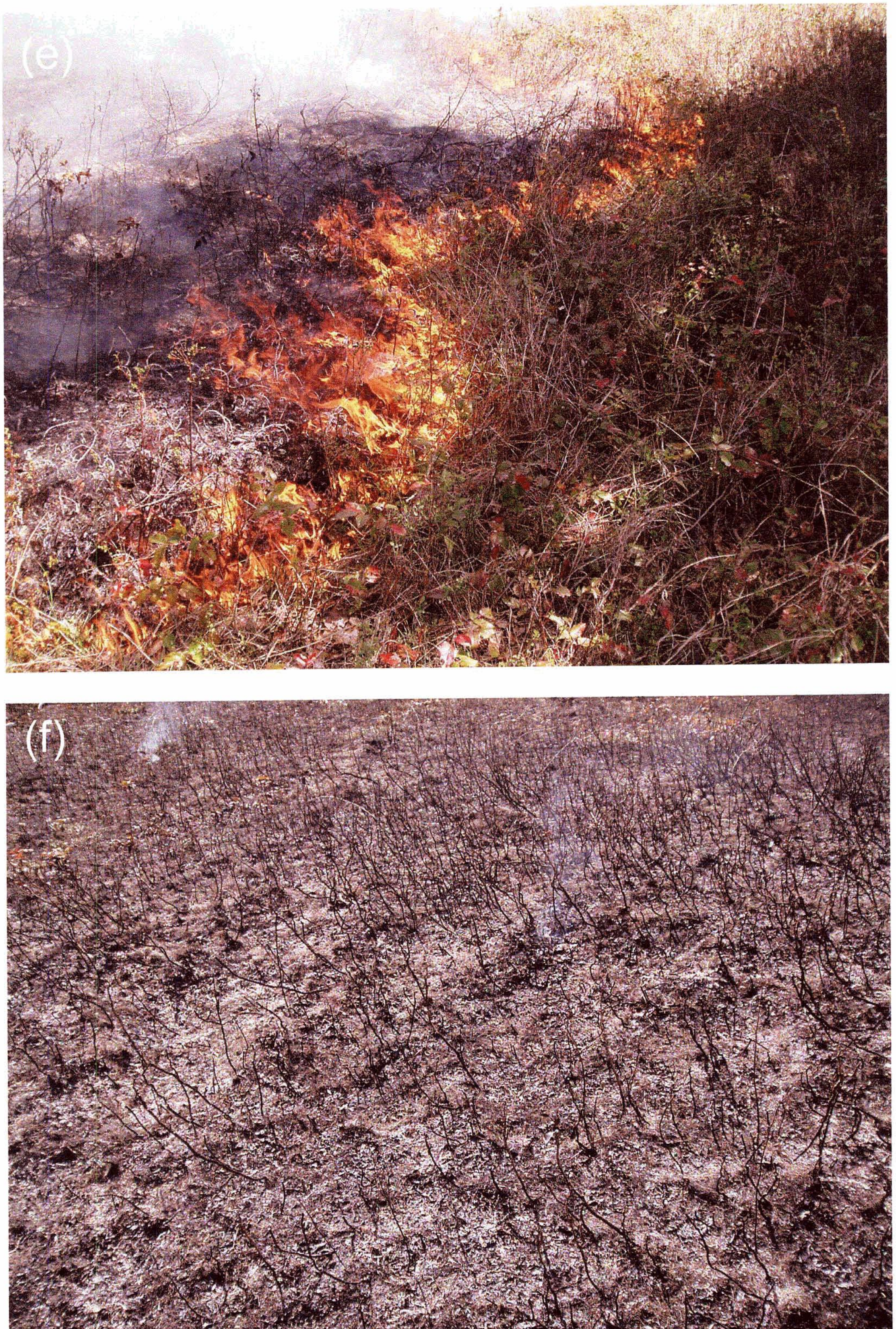
Figure 4 **Photographs of the prescribed burn (April 22, 2004) showing the variation in severity. (a) Early in the day (when it was cool and damp), the vegetation burns under the influence of the torch but does not continue to burn when it is removed, and (b) the burn was extremely patchy. (c) and (d) The fire burns under a stand of Nootka rose (*Rosa nutkana*) without killing the roses and leaving new growth unscorched. (e) and (f) Where fuels were abundant, the burn was more intense and homogeneous. (g) and (h) Where fuels were less abundant, the burn was less intense and more heterogeneous.**

Photos are found on the following four pages.


(a)
(b)


(c)


(d)


(e)
(f)


(g)


(h)

Figure 5 **Box plots of the original cover data for selected life form categories, within each of the 4 treatment groups: Control 2003, Pre-Burn 2003, Control 2004, and Burned 2004. The thick black bar indicates the median, the ends of the boxes mark the upper and lower quartiles and the whiskers mark the upper and lower extremes of the data. Circles are outliers. Significant differences are marked with paired uppercase letters.**

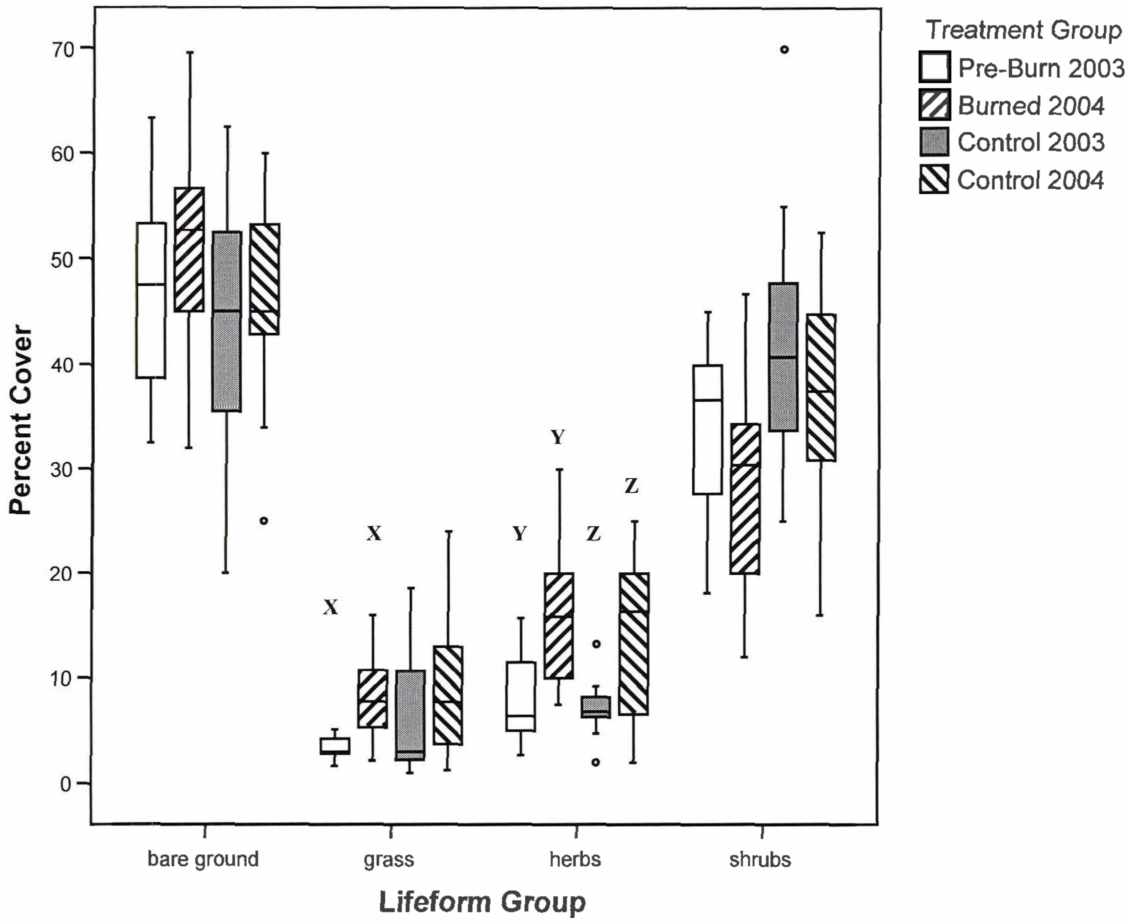


Figure 6 Box plots of species richness (of all herb, shrub, and tree species) for transects in each of the 4 treatment groups: Control 2003, Pre-Burn 2003, Control 2004, and Burned 2004.
The thick black bar indicates the median, the ends of the boxes mark the upper and lower quartiles and the whiskers mark the upper and lower extremes of the data. Circles are outliers. There are no significant differences.

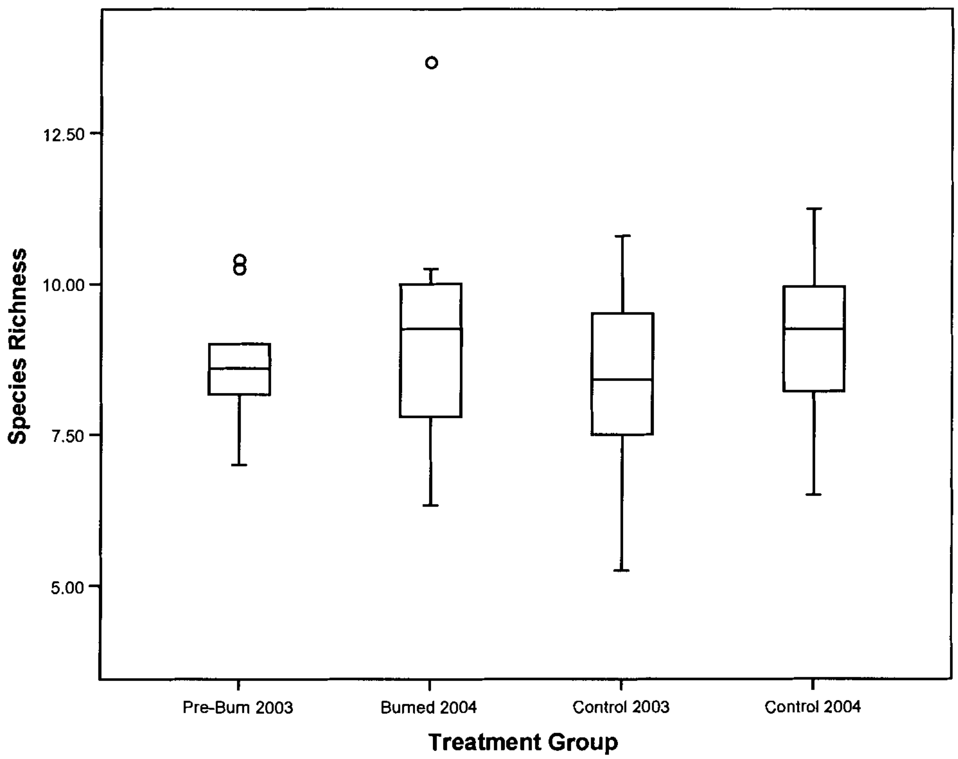


Figure 7 **Box plots of original cover data for selected culturally important species, in each of the 4 treatment groups: Control 2003, Pre-Burn 2003, Control 2004, and Burned 2004. The thick black bar indicates the median, the ends of the boxes mark the upper and lower quartiles and the whiskers mark the upper and lower extremes of the data. Circles are outliers. There are no significant differences.**

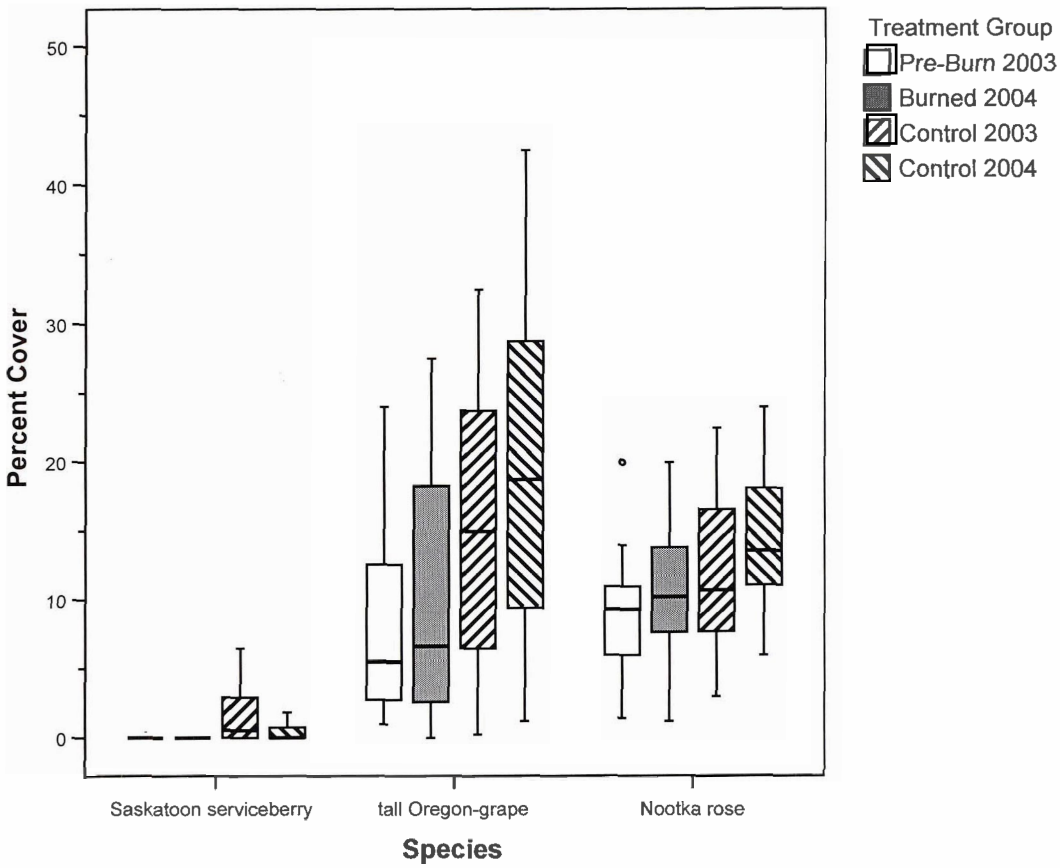


Figure 8 NMS ordination results comparing Control 2003 and Pre-Burn 2003 transects. The ordination was run using data for all species present in at least one transect in either group, plus bare ground.
The variation in the original data is best represented by a 3 dimensional solution (final stress = 12.63). Squares represent transects from the Pre-Burn 2003 group and triangles the Control 2003 group. The results of the MRPP indicate that the two groups are statistically distinct (A = 0.089, P = 0.001).

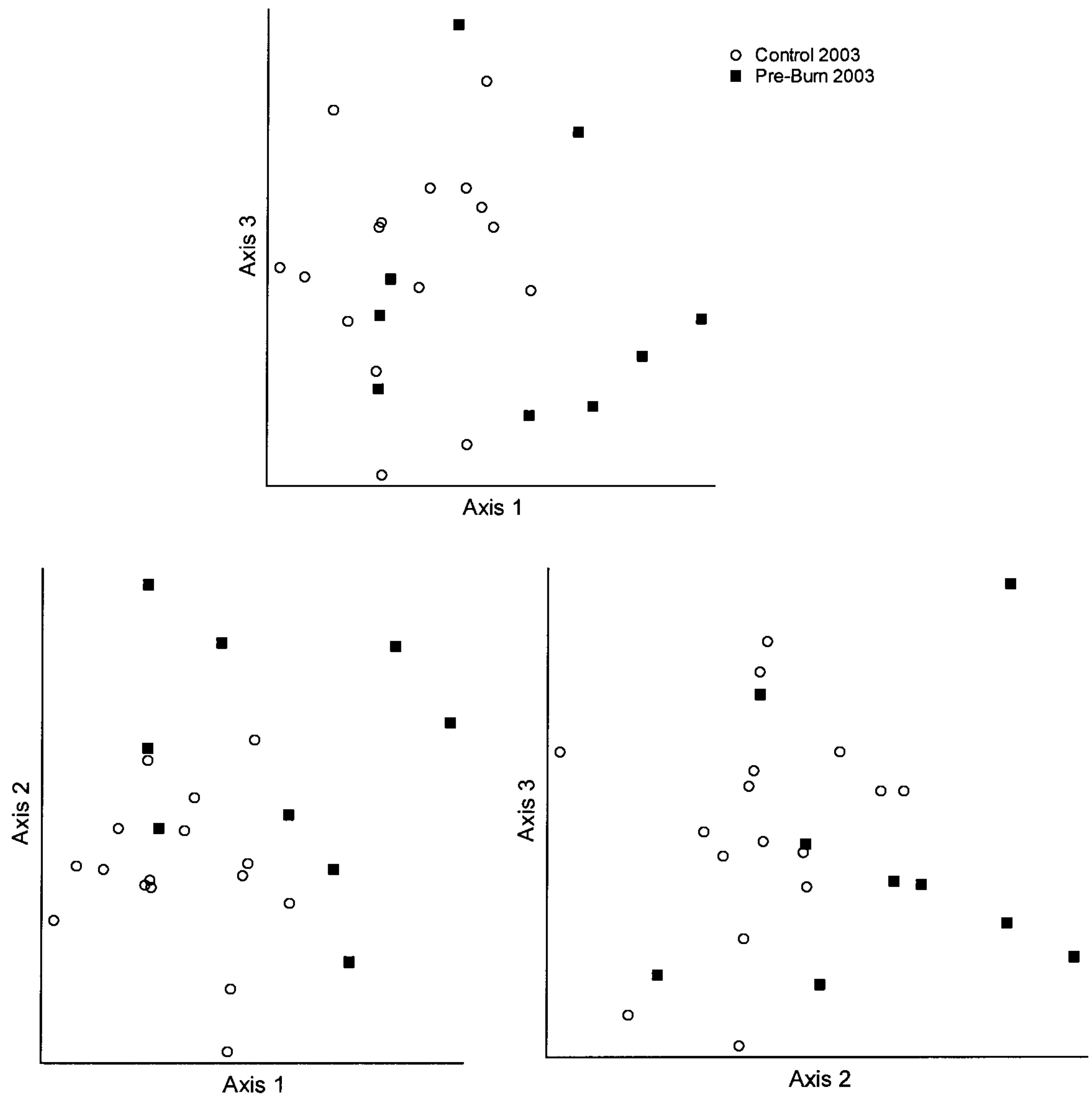


Figure 9 NMS ordination results comparing Control 2003 and Control 2004 transects. The ordination was run using data for all species present in at least one transect in either group, plus bare ground.
The variation in the original data is best represented by a 3 dimensional solution (final stress = 13.76). Squares represent transects from the Pre-Burn 2003 group and triangles the Control 2003 group. The results of the MRPP indicate that the two groups are statistically indistinguishable (A = 0.022, P = 0.134).

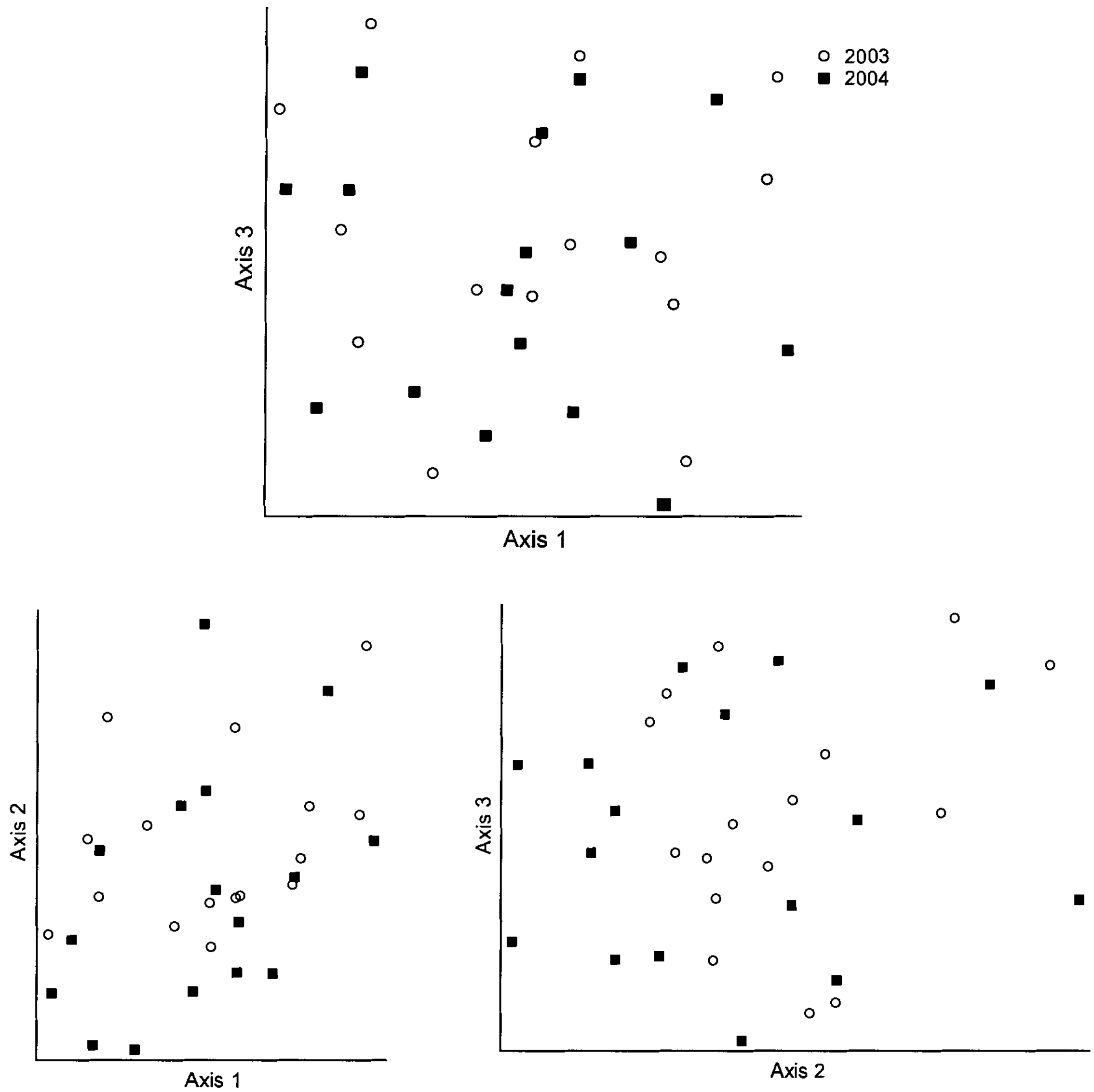


Figure 10 **NMS ordination results comparing Pre-Burn 2003 and Burned 2004 transects. The ordination was run using data for all species present in at least one transect in either group, plus bare ground.**
The variation in the original data is best represented by a 3 dimensional solution (final stress = 11.32). Squares represent transects from the Pre-Burn 2003 group and triangles the Control 2003 group. The results of the MRPP indicate that the two groups are statistically indistinguishable (A = 0.044, P = 0.090).

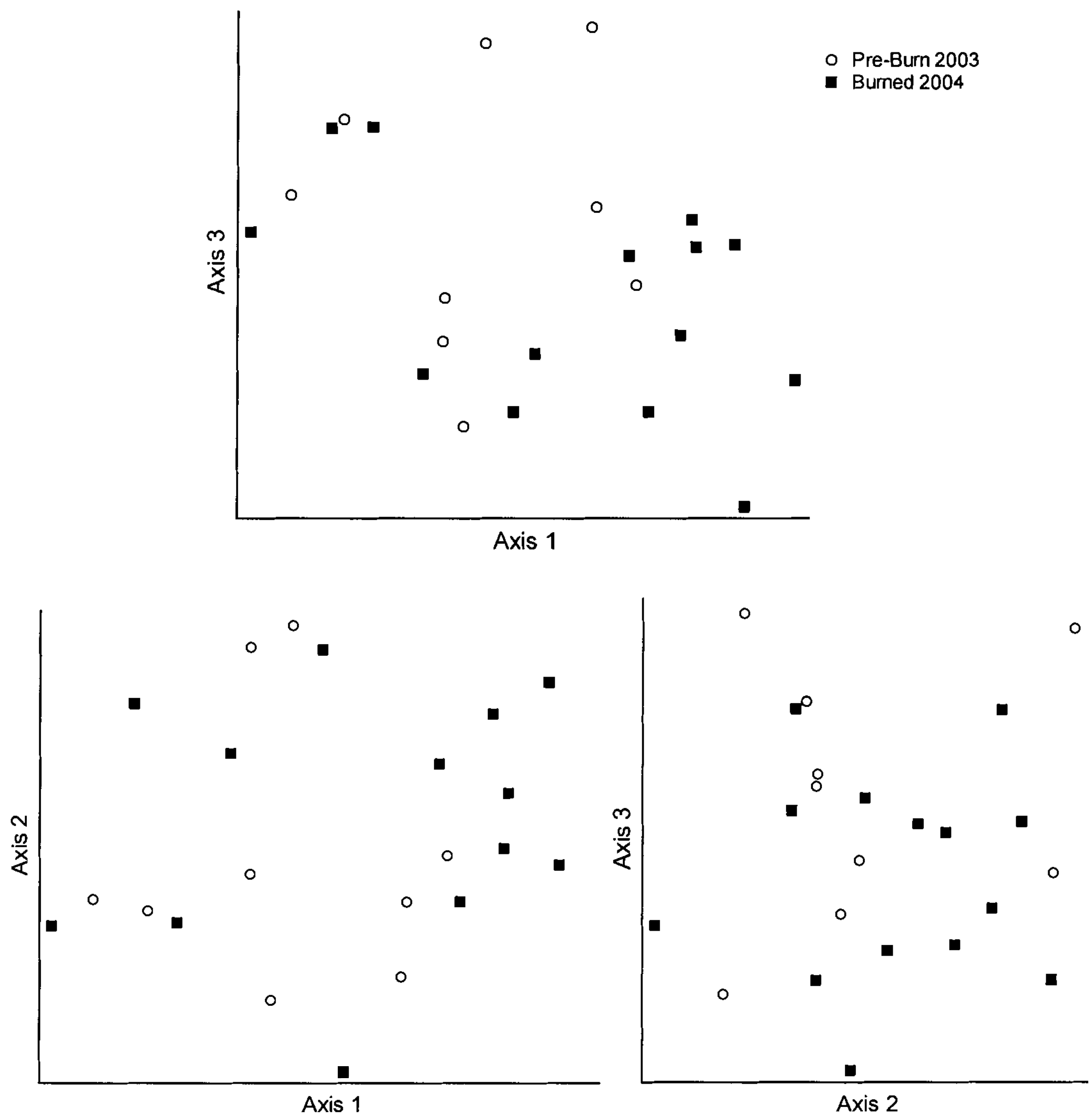


Figure 11 **Curves showing the hypothetical distribution of burn severity under 3 scenarios: low, medium, and high severity fires.**
The minimum effect threshold (a) represents the minimum burn severity required to produce a statistically significant, measurable response in the community. Past the deleterious effect threshold (b), the target community is adversely affected by the burn. In an idealized burn such as that represented by the middle curve, most of the area burned will experience a burn severity between these two thresholds. The top curve approximates the burn that we had in Chittenden Meadow.

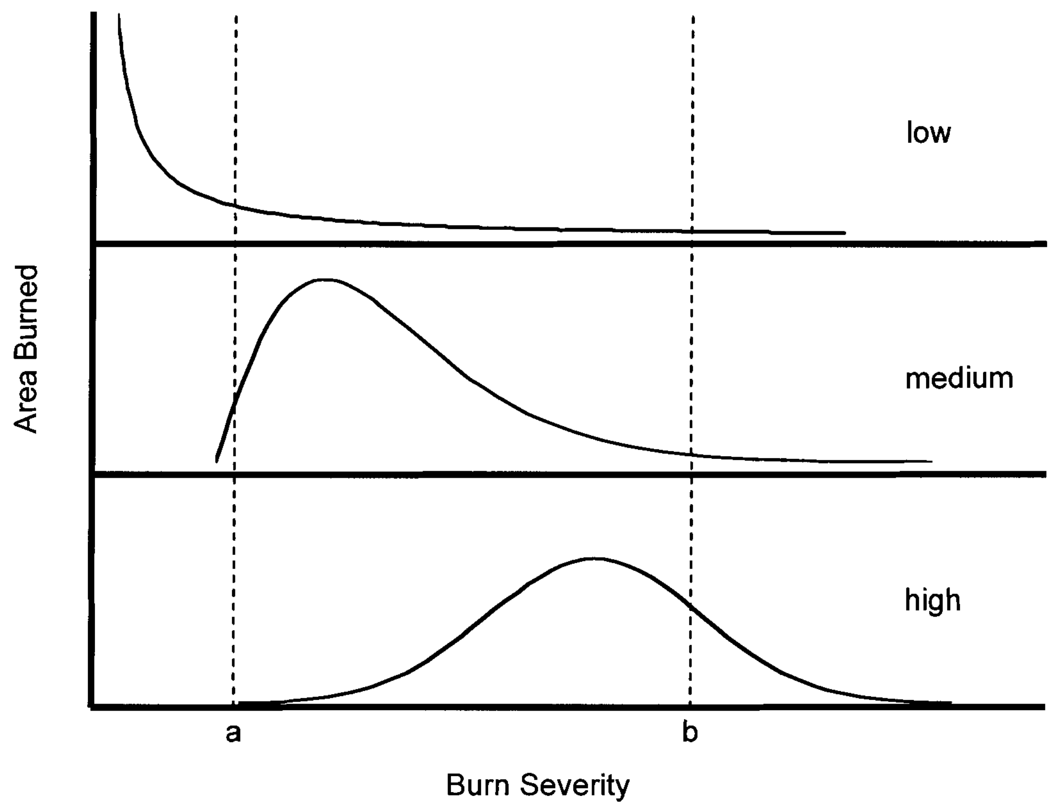


Table 1 **A list of all species identified in Chittenden Meadow. Potential indicator species are marked with an asterisk (*).**

grand fir	***Abies grandis***
vine maple	***Acer circinatum***
• yarrow	***Achillea millefolium***
pathfinder	***Adenocaulon bicolor***
orange agoseris	***Agoseris aurantiaca***
• saskatoon	***Amelanchier alnifolia***
pearly everlasting	***Anaphalis margaritacea***
• rosy pussytoes	***Antennaria microphylla***
field pussytoes	***Antennaria neglecta***
spreading dogbane	***Apocynum androsaemifolium***
kinnikinnick	***Arctostaphylos uva-ursi***
• showy aster	***Aster conspicuous***
common red paintbrush	***Castilleja miniata***
field chickweed	***Cerastium arvense***
Menzies' pipissewa	***Chimaphila menziesii***
unknown thistle	***Cirsium sp.***
narrow-leaved collomia	***Collomia linearis***
bunchberry	***Cornus canadensis***
red-osier dogwood	***Cornus stolonifera***
black hawthorn	***Crataegus douglasii***
hooker's fairybells	***Disporum hookeri***
fireweed	***Epilobium angustifolium***
common horsetail	***Equisetum arvense***
scouring-rush	***Equisetum hyemale***
showy daisy	***Erigeron speciosus***
wood strawberry	***Fragaria vesca***
wild strawberry	***Fragaria virginiana***
sweet-scented bedstraw	***Galium triflorum***
large-leaved avens	***Geum macrophyllum***
ocean spray	***Holodiscus discolor***
common St. John's-wort	***Hypericum perforatum***
wall lettuce	***Lactuca muralis***
purple peavine	***Lathyrus nevadensis***
oxeye daisy	***Leucanthemum vulgare***
tiger lily	***Lilium columbianum***
unknown lily	***Lilium sp.***
orange honeysuckle	***Lonicera ciliosa***
black twinberry	***Lonicera involucrata***
* silky lupine	***Luoinus sericeus***
tall Oregon-grape	***Mahonia aquifolium***
dull Oregon-grape	***Mahonia nervosa***
pink twink	***Microsteris gracilis***
blunt-leaved sandwort	***Moehringia lateriflora***
mountain sweet-cicely	***Osmorhiza chilensis***
falsebox	***Pachistima mvrsinites***
timothy	***Phleum pratense***
ponderosa pine	***Pinus ponderosa***
unknown poplar	***Populus sp.***
sticky cinquefoil	***Potentilla glandulosa***
graceful cinquefoil	***Potentilla gracilis***
unknown cherry	***Prunus sp.***
Douglas-fir	***Pseudotsuga menziesii***
bracken	***Pteridium aquilinum***
pink wintergreen	***Pyrola asarifolia***
Nootka rose	***Rosa nutkana***
unknown raspberry	***Rubus***
thimbleberry	***Rubus parviflorus***
fragile sour weed	***Rumex acetosella***
red elderberry	***Sambucus racemosa***
unknown senecio	***Senecio sp.***
• Menzies's campion	***Silene menziesii***
unknown Solomon's-seal	***Smilacina sp.***
star-flowered false Solomon's-seal	***Smilacina stellata***
Canada goldenrod	***Soladego canadensis***
lady's tresses	***Spiranthes romanzoffiana***
• birch-leaved spirea	***Spirea betulifolia***
common snowberry	***Symphoricarpos albus***
common dandelion	***Taraxacum officinale***
yellow salsify	***Tragopogon dubius***
broad-leaved starflower	***Trientalis latifolia***
unknown clover	***Trifolium sp.***
western hemlock	***Tsuga heterophylla***
unknown vetch	***Vicia sp.***
unknown violet	***Viola sp.***

Table 2 **Summary statistics and univariate tests of significance between treatment groups. This table summarizes the results of the (non-parametric) Kruskal-Wallace tests for each paired comparison of treatment groups discussed in the text – Control 2003 vs. Control 2004, and Pre-Burn 2003 vs. Burned 2004 – for selected life form groups and culturally significant species (average cover; raw and untransformed) and average species richness. The asymptotic significance value is analogous to the parametric P-value. A statistically significant difference is present if the asymptotic significance value is ≤ 0.05; these are marked by an asterisk (*). For grasses, the estimate of the net effect of the fire does not consider the net change in the control because the treatment effect was not confounded by inter-annual variation.**

	treatment group compared	n	mean	net change in control or treatment group	estimated net effect of the fire (treatment minus control)	Kruskal-Wallis test results	
						Chi-square	asymptotic significance
Shrubs	Control 2003 vs.	16	41.16			0.754	0.385
	Control 2004	16	37.26	-3.9			
	Pre-Burn 2003 vs.	9	33.51		-0.5	0.955	0.328
	Burned 2004	14	29.06	-4.45			
Herbs	Control 2003 vs.	16	7.71			4.625	* 0.032
	Control 2004	16	13.92	6.21			
	Pre-Burn 2003 vs.	9	7.80		2.01	8.962	* 0.003
	Burned 2004	14	16.02	8.22			
Grasses	Control 2003 vs.	16	6.59			1.598	0.206
	Control 2004	16	9.29	2.7			
	Pre-Burn 2003 vs.	9	4.49		3.87	6.368	* 0.012
	Burned 2004	14	8.36	3.87			
Bare ground	Control 2003 vs.	16	42.81			0.796	0.372
	Control 2004	16	46.48	3.67			
	Pre-Burn 2003 vs.	9	47.09		0.15	0.290	0.590
	Burned 2004	14	50.91	3.82			
Nootka Rose	Control 2003 vs.	16	11.94			1.077	0.229
	Control 2004	16	14.03	2.09			
	Pre-Burn 2003 vs.	9	9.45		-0.52	0.255	0.614
	Burned 2004	14	11.02	1.57			
Saskatoon	Control 2003 vs.	16	2.31			2.350	0.125
	Control 2004	16	0.51	-1.8			
	Pre-Burn 2003 vs.	9	0.94		1.28	0.017	0.896
	Burned 2004	14	0.42	-0.52			
Tall Oregon-grape	Control 2003 vs.	16	14.99			0.821	0.365
	Control 2004	16	19.12	4.13			
	Pre-Burn 2003 vs.	9	8.92		-2.97	0.009	0.925
	Burned 2004	14	10.08	1.16			
Richness	Control 2003 vs.	16	8.45			1.601	0.206
	Control 2004	16	9.12	0.67			
	Pre-Burn 2003 vs.	9	8.69		-0.29	0.361	0.548
	Burned 2004	14	9.07	0.38			

Table 3 **Some characteristics of burns of varying severity.**
With increasing severity, the effect of the burn on the target community will increase, the proportion of the community experiencing a burn severity high enough to produce deleterious effects will increase, and the burn will become more difficult to control. The goal of a manager is to find the optimal burn severity that meets the project goals, minimizes areas of excessive burn, and maximizes controllability of the fire.

	low severity	**medium severity**	**high severity**
effect on target community	none / small	desired / meaningful	deleterious
area severely burned (i.e., experiences deleterious effects)	none / small	small	large
controllability	high	moderate / acceptable	low / unacceptable

6 REFERENCE LIST

Agee, J. K. 1990. The historical role of fire in Pacific Northwest forests. Pages 25-38 *in* J. Walstad, S. R. Radsevich, and D. V. Sandberg, editors. Natural and Prescribed Fire in Pacific Northwest Forests. Oregon State University Press, Corvallis.

Agee, J. K. 1993. Fire Ecology of Pacific Northwest Forests. Island Press, Washington, DC, USA.

Agee, J. K. 1997. Fire management for the 21st century. Pages 191-201 *in* K. A. Kohm and J. F. Franklin, editors. Creating a forestry for the 21st century: the science of ecosystem management. Island Press, Washington, D.C.

Agee, J. K. 1998. Fire and Pine Ecosystems. *in* D. M. Richardson, editor. Ecology and biogeography of Pinus. Cambridge University Press, New York.

Agee, J. K., M. Finney, and R. de Gouvenain. 1990. Forest fire history of Desolation Peak, Washington. Canadian Journal of Forest Research **20**:350–356.

Agee, J. K., and J. Kertis. 1987. Forest types of the North Cascades National Park Complex. Canadian Journal of Botany **65**:1520-1530.

Allen, C. D., M. Savage, D. A. Falk, K. F. Suckling, T. W. Swetnam, T. Schulke, P. B. Stacey, P. Morgan, M. Hoffman, and J. T. Klingel. 2002. Ecological restoration of Southwestern ponderosa pine ecosystems: A broad perspective. Ecological Applications **12**:1418-1433.

Anderson, R. C., J. E. Schwegman, and M. R. Anderson. 2000. Micro-scale restoration: A 25-year history of a southern Illinois barrens. Restoration Ecology **8**:296-306.

Antos, J. A., B. McCune, and C. Bara. 1983. The effect of fire on an ungrazed Western Montana grassland. American Midland Naturalist **110**:354-364.

Bailey, J. D., and W. W. Covington. 2002. Evaluating ponderosa pine regeneration rates following ecological restoration treatments in northern Arizona, USA. Forest Ecology and Management **155**:271-278.

Boyd, C. S., and T. G. Bidwell. 2002. Effects of prescribed fire on shinnery oak (*Quercus havardii*) plant communities in western Oklahoma. Restoration Ecology **10**:324-333.

Braun-Blanquet, J. 1965. Plant Sociology: the study of plant communities. Hafner, London, England.

British Columbia Ministry of Forests. 1995. Forest Practices Code of British Columbia: Biodiversity Guidebook. British Columbia Ministry of Forests, Communications Branch, Victoria, B.C.

British Columbia Ministry of Forests and Range Protection Branch. 2005a. www.for.gov.bc.ca/protect/.

British Columbia Ministry of Forests and Range Protection Branch. 2005b. Ministry Policy Manual. www.for.gov.bc.ca/tasb/manuals/policy/resmngmt/rm9-5.htm.

British Columbia Ministry of Forests and Range Protection Branch. 2005c. Prescribed Fire Planning. www.for.gov.bc.ca/protect/burning/burnplanhelp.htm#c_sum.

British Columbia Ministry of Forests and Range Protection Branch. 2005d. Prescribed Fire. www.for.gov.bc.ca/protect/burning/prescribedfire.htm.

British Columbia Ministry of Forests and Range Protection Branch. 2005e. Report on Average hectares, fires and dollars. www.for.gov.bc.ca/protect/reports/HistoricalAverages.htm.

British Columbia Ministry of Sustainable Resource Management (Terrestrial Information Branch). 2002. Vegetation Resources Inventory: Sample Selection Procedures for Ground Sampling Version 3.3. Resources Information Standards Committee.

Canadian Forest Service. 2004. Rethinking Canada's Approach to Forest Fires. www.for.gov.bc.ca/protect/reports/HistoricalAverages.htm.

Carter, M. C., and C. D. Foster. 2004. Prescribed burning and productivity in southern pine forests: A review. Forest Ecology and Management **191**:93-109.

Christensen, N. L., J. K. Agee, P. F. Brussard, J. Hughes, D. H. Knight, G. W. Minshall, J. M. Peek, S. J. Pyne, F. J. Swanson, J. W. Thomas, S. Wells, S. E. Williams, and H. A. Wright. 1989. Interpreting the Yellowstone fires of 1988: ecosystem responses and management implications. BioScience **39**:678-685.

Clarke, K. R. 1993. Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology **18**:117-143.

Clarke, P. J., K. J. E. Knox, K. E. Wills, and M. Campbell. 2005. Landscape patterns of woody plant response to crown fire: disturbance and productivity influence sprouting ability. Journal of Ecology **93**:544-555.

Connell, J. H. 1978. Diversity in tropical rainforests and coral reefs. Science:1302-1310.

Copeland, T. E., W. Sluis, and H. F. Howe. 2002. Fire season and dominance in an Illinois tallgrass prairie restoration. Restoration Ecology **10**:315-323.

Covington, W. W. 1994. Southwestern ponderosa pine forest structure: Changes since Euro-American settlement. Journal of Forestry **92**:39-47.

Covington, W. W. 2000. Helping western forests heal: The prognosis is poor for U.S. forest ecosystems. Nature:135-136.

Covington, W. W. 2003. The Evolutionary and Historical Context. Pp. 26-47 *in* P. Friederici, editor. Ecological Restoration of Southwestern Ponderosa Pine Forests. Island Press, Washington.

Covington, W. W., P. Z. Fule, M. M. Moore, S. C. Hart, T. E. Kolb, J. N. Mast, S. S. Sackett, and M. R. Wagner. 1997. Restoring ecosystem health in the ponderosa pine forests of the southwest. Journal of Forestry **95**:23-29.

Covington, W. W., and M. M. Moore. 1994. Southwestern Ponderosa Forest Structure - Changes since Euro-American Settlement. Journal of Forestry **92**:39-47.

Davis, M. A., D. W. Peterson, P. B. Reich, M. Crozier, T. Query, E. Mitchell, J. Huntington, and P. Bazakas. 2000. Restoring savanna using fire: Impact on the breeding bird community. Restoration Ecology **8**:30-40.

Davis, M. A., and L. B. Slobodkin. 2004. The Science and Values of Restoration Ecology. Restoration Ecology **12**:1-3.

Ellison, A. M. , M. S. Bank, B. D. Clinton, E. A. Colburn, K. Elliott, C. R. Ford, D. R. Foster, B. D. Kloeppel, J. D. Knoepp, G. M. Lovett, J. Mohan, D. A. Orwig, N. L. Rodenhouse, W. V. Sobczak, K. A. Stinson, J. K. Stone, C. M. Swan, J. Thompson, B. Von Holle, and J. R. Webster. 2005. Loss of foundation species: consequences for the structure and dynamics of forested ecosystems. Frontiers in Ecology and the Environment **3**:479-486.

Fasham, M. J. R. 1977. A comparison of nonmetric multidimensional scaling, principal components and reciprocal averaging for the ordination of simulated coenoclines and coenoplanes. Ecology **58**:551-561.

Franck, I. C. 2000. An archaeological survey of the Galene Lakes in the Skagit Range of the North Cascade Mountains, Skagit Valley Park, B.C. Thesis (MA). Simon Fraser University, Vancouver, B.C., Canada.

Friederici, P. 2003. The "Flagstaff Model". Pages 7-25 *in* P. Friederici, editor. Ecological Restoration of Southwestern Ponderosa Pine Forests. Island Press, Washington.

Fule, P. Z., and W. W. Covington. 1998. Spatial patterns of Mexican pine-oak forests under different recent fire regimes. Plant Ecology **134**:197-209.

Fule, P. Z., W. W. Covington, H. B. Smith, J. D. Springer, T. A. Heinlein, K. D. Huisinga, and M. M. Moore. 2002. Comparing ecological restoration alternatives: Grand Canyon, Arizona. Forest Ecology and Management **170**:19-41.

Gildar, C. N., P. Z. Fule, and W. W. Covington. 2004. Plant community variability in ponderosa pine forest has implications for reference conditions. Natural Areas Journal **24**:101-111.

Government of British Columbia. 2003. Firestorm 2003 Provincial Review. www.2003firestorm.gov.bc.ca/default.htm, Victoria, BC.

Griffis, K. L., J. A. Crawford, M. R. Wagner, and W. H. Moir. 2001. Understory response to management treatments in northern Arizona ponderosa pine forests. Forest Ecology and Management:239-245.

Haeussler, S., D. Coates, and J. Mather. 1990. Autecology of Common Plants in British Columbia: A Literature Review. FRDA 158, B.C. Ministry of Forests.

Harrington, M. G., and S. S. Sackett. 1992. Past and present fire influences on southwestern ponderosa pine old growth. Pages 44-50 *in* M. R. Kaufmann, W. H. Moir, R. L. Bassett, editors. Proceedings of old-growth forest in the Southwest and Rocky Mountain Regions. U.S. Department of Agriculture, Fort Collins, CO.

Heuberger, K. A., and F. E. Putz. 2003. Fire in the suburbs: Ecological impacts of prescribed fire in small remnants of longleaf pine (*Pinus palustris*) sandhill. Restoration Ecology **11**:72-81.

Hobbs, R. J., and D. A. Norton. 1996. Towards a conceptual framework for restoration ecology. Restoration Ecology **4**:93-110.

Holmes, P. M., D. M. Richardson, B. W. Van Wilgen, and C. Gelderblom. 2000. Recovery of South African fynbos vegetation following alien woody plant clearing and fire: implications for restoration. Austral Ecology **25**:631-639.

Honnay, L., K. Verheyen, and M. Hermy. 2002. Permeability of ancient forest edges for weedy plant invasion. Forest Ecology and Management:109-122.

Howard, J. L. 1997. Fire Effects Information System: Amelanchier alnifolia. *in* http://www.fs.fed.us/database/feis/. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory.

Hurlbert, S. H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs **54**:187-211.

Hutchinson, T. F., E. K. Sutherland, and D. A. Yaussy. 2005. Effects of repeated prescribed fires on the structure, composition, and regeneration of mixed-oak forests in Ohio. Forest Ecology and Management **218**:210-228.

International Joint Commission. 1971. Environmental and ecological consequences in Canada of raising Ross Lake in the Skagit Valley to elevation 1725. Appendix F. International Joint Commission, Canada and United States. North Cascades National Park, Sedro Woolley, Washington, USA.

Jensen, S. 1978. Influences of transformation of cover values on classification and ordination of lake vegetation. Vegetatio **48**:47-59.

Kuuluvainen, T., K. Aapala, P. Ahlroth, M. Kuusinen, T. Lindholm, T. Sallantaus, J. Siitonen, and H. Tukia. 2002. Principles of ecological restoration of boreal forested ecosystems: Finland as an example. Silva Fennica **36**:409-422.

Langston, N. 1995. Forest Dreams, Forest Nightmares, The Paradox of Old Growth in the Inland West. University of Washington Press.

Lepine, F., C. Opio, and D. Ayers. 2003. An analysis of escaped fires from broadcast burning in the Prince George Forest Region of British Columbia. BC Journal of Ecosystems and Management **3**:1-9.

Lepofsky, D., E. K. Heyerdahl, K. Lertzman, D. Schaepe, and B. Mierendorf. 2003. Historical Meadow Dynamics in Southwest BC: a Multidisciplinary Analysis. Conservation Ecology **7**:article 5.

Lesica, P., and B. Martin. 2003. Effects of prescribed fire and season of burn on recruitment of the invasive exotic plant, Potentilla recta, in a semiarid grassland. Restoration Ecology **11**:516-523.

McCune, B., J. B. Grace, and D. L. Urban. 2002. Analysis of Ecological Communities. MjM Software Design, Gleneden Beach, Oregon, USA.

McCune, B., and M. J. Mefford. 1999. PC-ORD. Multivariate analysis of ecological data. MjM Software Design, Gleneden Beach, Oregon, USA.

Meidinger, D., and J. Pojar. 1991. Ecosystems of British Columbia. British Columbia Ministry of Forests.

Meyer, C. L., T. D. Sisk, and W. W. Covington. 2001. Microclimatic changes induced by ecological restoration of ponderosa pine forests in northern Arizona. Restoration Ecology **9**:443-452.

Mitchell, J. E., P. N. S. Bartling, and R. O'Brien. 1988. Comparing cover-class macroplot data with direct estimates from small plots. American Midland Naturalist **120**:70-78.

Moore, M. M., W. W. Covington, and P. Z. Fule. 1999. Reference conditions and ecological restoration: A southwestern ponderosa pine perspective. Ecological Applications **9**:1266-1277.

Mueller-Dombois, D., and H. Ellenberg. 1974. Aims and Methods of Vegetation Ecology. John Wiley & Sons, New York, USA.

Mulligan, M. K., and L. K. Kirkman. 2002. Burning influences on wiregrass (Aristida beyrichiana) restoration plantings: Natural seedling recruitment and survival. Restoration Ecology **10**:334-339.

Parminter, J. 1990. Typical historic patterns of wildfire disturbance by biogeoclimatic zone. *in* Old-growth forests: problem analysis. British Columbia Ministry of Forests, Research Branch, Victoria, BC.

Pendergrass, K. L., P. M. Miller, J. B. Kauffman, and T. N. Kaye. 1999. The role of prescribed burning in maintenance of an endangered plant species, Lomatium bradshawii. Ecological Applications **9**:1420-1429.

Perry, D. A., and M. P. Amaranthus. 1997. Disturbance, recovery, and stability. Pages 31-56 *in* K. A. Kohm and J. F. Franklin, editors. Creating a forestry for the 21st century: the science of ecosystem management. Island Press, Washington, D.C.

Peterson, E. B., and B. McCune. 2001. Diversity and succession of epiphytic macrolichen communities in low-elevation managed conifer forests in Western Oregon. Journal of Vegetation Science **12**:511-524.

Pfadenhauer, J. 2001. Some remarks on the socio-cultural background of restoration ecology. Restoration Ecology **9**:220-229.

Pitkanen, S. 1997. Correlation between stand structure and ground vegetation: an analytical approach. Plant Ecology **131**:109-126.

Pitzer, P. C. 1978. Building the Skagit. Galley Press, Portland, Oregon, USA.

Provencher, L., B. J. Herring, D. R. Gordon, H. L. Rodgers, G. W. Tanner, L. A. Brennan, and J. L. Hardesty. 2000. Restoration of northwest Florida sandhills through harvest of invasive Pinus clausa. Restoration Ecology **8**:175-185.

Pyne, S. J., P. L. Andrews, and R. D. Laven. 1996. Introduction to Wildland Fire, Second edition. John Wiley & Sons, Inc., New York.

Radeloff, V. C., D. J. Mladenoff, and M. S. Boyce. 2000. A historical perspective and future outlook on landscape scale restoration in the northwest Wisconsin pine barrens. Restoration Ecology **8**:119-126.

Rideout, S., B. P. Oswald, and M. H. Legg. 2003. Ecological, political and social challenges of prescribed fire restoration in east Texas pineywoods ecosystems: a case study. Forestry **76**:261-269.

Rinne, J. N. 1996. Short-term effects of wildfire and aquatic macroinvertebrates in the southwestern United States. North American Journal of Fisheries Management **16**:653-658.

Ruthven, D. C., and D. R. Synatzske. 2002. Response of herbaceous vegetation to summer fire in the, western south Texas plains. Texas Journal of Science **54**:195-210.

Scheller, R. M., and D. J. Mladenoff. 2002. Understory species patterns and diversity in old-growth and managed northern hardwood forests. Ecological Applications **12**:1329-1343.

Sieg, C. H., B. G. Phillips, and L. P. Moser. 2003. Exotic Invasive Plants. Pages 251-267 *in* P. Friederici, editor. Ecological Restoration of Southwestern Ponderosa Pine Forests. Island Press, Washington.

Slocum, M. G., W. J. Platt, and H. C. Cooley. 2003. Effects of differences in prescribed fire regimes on patchiness and intensity of fires in subtropical savannas of Everglades National Park, Florida. Restoration Ecology **11**:91-102.

Snyder, S. A., L. E. Tyrrell, and R. G. Haight. 1999. An optimization approach to selecting research natural areas in national forests. Forest Science **45**:458-469.

Sparks, J. C., R. E. Masters, D. M. Engle, and G. A. Bukenhofer. 2002. Season of burn influences fire behavior and fuel consumption in restored shortleaf pine-grassland communities. Restoration Ecology **10**:714-722.

Stephenson, N. L. 1999. Reference conditions for giant sequoia forest restoration: Structure, process, and precision. Ecological Applications **9**:1253-1265.

Swezy, D. M., and J. K. Agee. 1991. Prescribed-Fire Effects on Fine-Root and Tree Mortality in Old-Growth Ponderosa Pine. Canadian Journal of Forest Research **21**:626-634.

Tiedemann, A. R., J. O. Klemmedson, and E. L. Bull. 2000. Solution of forest health problems with prescribed fire: are forest productivity and wildlife at risk? Forest Ecology and Management **127**:1-18.

Turner, M. G., W. H. Romme, and D. B. Tinker. 2003. Surprises and lessons from the 1988 Yellowstone fires. Frontiers in Ecology and the Environment **1**:351-358.

Turner, N. J. 1995. Food plants of coastal First Peoples. UBC Press, Vancouver, BC, Canada.

Turner, N. J. 1998. Plant technology of First Peoples in British Columbia. UBC Press, Vancouver, BC, Canada.

Tveten, R. K., and R. W. Fonda. 1999. Fire effects on prairies and oak woodlands on Fort Lewis, Washington. Northwest Science **73**:145-158.

United States Forest Service, and Fire Sciences Lab. 2004. Fire Effects Monitoring and Inventory Protocol. http://fire.org/index.php?option=content&task=section&id=5&Itemid=42.

USDA Forest Service. 2006. Fire effects information system. http://www.fs.fed.us/database/feis/.

USDA Forest Service. 1995. Federal Wildland Fire Management Policy and Program Review. http://www.fs.fed.us/fire/management/index.html.

USDA Forest Service. 2000. Wildland Fire in Ecosystems: Effects of Fire on Flora. U.S. Government Printing Office, Washington, D.C.

USDA Forest Service. 2001. Policy Implications of Large Fire Management: A Strategic Assessment of Factors Influencing Costs. http://www.fs.fed.us/fire/management/Large_Fire_Mgt.pdf.

Van Wilgen, B. W., C. S. Everson, and W. S. W. Trollope. 1990. Fire management in southern Africa: some examples of current objectives, practices and problems. Pages 179-215 *in* J. G. Goldammer, editor. Fire in the Tropical Biota: Ecosystem Processes and Global Challenges. Springer-Verlag, Berlin.

Walkup, C. J. 1991. Fire Effects Information System: Mahonia repens. *in* http://www.fs.fed.us/database/feis/. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory.

Waltz, A. E. M., P. Z. Fule, W. W. Covington, and M. M. Moore. 2003. Diversity in ponderosa pine forest structure following ecological restoration treatments. Forest Science **49**:885-900.

Whelan, R. J. 1995. The Ecology of Fire. Cambridge University Press, Cambridge.

Whysong, G. L., and W. H. Miller. 1987. An evaluation of random and systematic plot placement for estimating frequency. Journal of Range Management **40**:475-479.

Wrobleski, D. W., and J. B. Kauffman. 2003. Initial effects of prescribed fire on morphology, abundance, and phenology of forbs in big sagebrush communities in southeastern Oregon. Restoration Ecology **11**:82-90.

Young, T. P. 1999. Restoration Ecology and Conservation Biology. Biological Conservation:73-83.

Zimmerman, G. T. 2003. Fuels and fire behavior. Pages 126-143 *in* P. Friederici, editor. Ecological Restoration of Southwestern Ponderosa Pine Forests. Island Press, Washington.