Plant Community Response to Post-wildfire Management Activities in Interior Douglas-Fir Forests of Southern BC

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Abstract

Post-wildfire disturbances such as salvage logging and seeding of agronomic species occur over large parts of the forested land base in British Columbia. However, there is surprisingly little research on the effects of these management practices on plant community composition and species diversity. The future of plant and animal biodiversity will depend increasingly on regional floras surviving in highly managed and disturbed environments. I examined vascular and nonvascular plant community responses four years after wildfire and post-wildfire management practices in interior Douglas-fir (*Pseudotsuga menziesii var. glauca*) forests following two separate 2003 wildfires near Kamloops, BC, Canada. Wildfire sites with all combinations of seeding and salvage logging disturbance were selected in similar post-wildfire environments. Analysis at the plot ($400m^2$, n=104) and stand ($400\text{-}1200m^2$, n=42) scales suggested that post-wildfire disturbance had a significant negative effect on native species richness and reduced the frequency of some shrub and shade tolerance species including *Ceanothus sanguineus* and *Prosartes hookeri*. As well, multivariate analysis showed evidence for altered post-wildfire community composition and structure mainly due to increased dominance of exotic and graminoid species. These negative effects were most apparent in wildfire sites that were both seeded and salvaged-logged. Sustainable forest management requires a thorough understanding of the cumulative impacts of post-wildfire management practices on understory vegetation and ecosystem processes. The results of this study can aid resource managers by helping them incorporate the effects of natural and anthropogenic disturbance into future post-wildfire management activities.

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Chapter 1 Introduction

Plant Community Response to Post-wildfire Management Activities in Interior Douglas-Fir Forests of Southern BC

Wildfire is the most prominent large-scale, global disturbance process in terrestrial ecosystems, and is second in magnitude only to human caused landscape modifications (Bond and Van Wilgen 1996, Wright and Bailey 1982). Fire has features in common with other types of mortality-causing disturbances such as logging, flood, windstorms, and pathogen or insect outbreaks. Knowledge of the interactions of disturbance and the rates at which populations and communities recover from different types of disturbance is of great importance in ecology. Although there are potentially extreme ecological consequences involved, disturbance is thought to be essential for the proper functioning and maintenance of the inherent biodiversity of many ecosystems (Petraitis et al. 1989). When the frequency and intensity of disturbances are known, the effects on resource levels, plant community composition, and competitive exclusion by dominant species can be highly predictable. Such information provides the foundation for many ecological concepts including plant succession (Clements 1916, Cowles 1899), the intermediate disturbance hypothesis (Connell 1978) the resource ratio hypothesis (Tilman 1982) and ecosystem resilience (Elmqvist et al. 2003, Holling 1973).

1.1 Review of Concepts and History of Succession and Disturbance

Early studies of changes in plant communities revolved around the dynamics of succession as either a directional process (Clements 1916) or a coincidental grouping based on environment and individual life-history traits (Gleason 1926). One of the first researchers to account for the role of disturbance in succession was (Watt 1947), who saw the existence of plant communities as a complex of fluctuating patches that are in phase with their environment. Egler (1954) introduced the concept of 'initial floristic composition', a view whereby secondary succession develops from the species present at any one site after disturbance and those that arrive early on. Thirty years later, after continuing debate between the Clementsian and Gleasonian camps, Connell and Slatyer (1977) synthesized what was known about succession into a theoretical model with three pathways: facilitation, tolerance and inhibition. In this way disturbance could be factored into the establishment of multiple trajectories for plant community assembly but did not necessarily culminate in one predetermined climax grouping. Differences in disturbance and successional history could result in multiple stable states for the same community type (Sutherland 1974).

Further incorporation of disturbance, specifically for fire prone ecosystems, came with (Noble and Slatyer 1980) “vital attributes model” which allowed for effects of recurrent disturbances and defined plant communities relative to the method of arrival, establishment and maturation of species following fire. Rowe (1983) condensed Noble and Slatyer’s 19 patterns of persistence after fire into two basic propagule categories and five survival methods. The first propagule category includes plants that have a seed based strategy of survival by 1) *invading* after the fire,

2) *evading* the fire by storing seed in the canopy or soil or, 3) *avoiding* fire prone areas all together. The second of Rowe's propagule categories, vegetative-based fire survival, includes 4) *resisters* that can tolerate low severity fire and 5) *endurers* that have the ability to resprout from buried rhizomes. These five modes of persistence are not mutually exclusive however, and it is foreseeable that the same species could be an invader, an evader and an endurer.

Many studies have also focused on competition for limited resources as a major driver of succession (Smith and Huston 1989, Tilman 1985, 1982). These are often mechanistic models that are difficult to apply to whole plant communities but can help elucidate the process of succession on a plant by plant or group by group replacement basis. Competition models rarely take into account the nuances of recurring disturbance but are helpful when one or more resources (such as nutrients, light, water or microsites) are limited. Tilman (1982) termed the concept of succession being determined by changes in the ratios of limiting resources the "resource ratio hypothesis". In this view, the successional pattern of a community is driven by the resource interactions between competing plants at the population level and nutrient processes at the ecosystem level. Resource interactions may be crucial in the early colonization stage after fire. The resource approach to community structure relies on basic mechanisms being able to infer larger, more complex processes. Predictions of the future state of communities however, are difficult and idiosyncratic events can wreak havoc on successional models incorporating species biology and use of resources. McCune and Allen (1985) found that the random or cyclic nature of seed production, drought and disease could create large variation in community composition for hundreds of years, even between very similar sample sites. As well, McCune and Allen (1985) note that the rate of competition is often "much too slow (relative to disturbance frequencies in the context of individual longevity) to play an overriding role in structuring these forest communities”.

In the search for generalities in the patterns and processes of succession it seems that the number of pathways for succession is only surpassed by the myriad plant communities that succession can lead to. Cook (1996) summarized some of the underlying components of community response to disturbance, mainly that disturbance is integral to vegetation dynamics and that there are many processes preventing eventual establishment of a “climax” community. Disturbance can accelerate, stall or reverse movement towards a late successional stage and random events along with species life history attributes must be taken into account when describing succession. As well, while disturbance is a key component of succession, some researchers have pointed out that spatial and temporal gradients apart from disturbance can also affect community change (Grubb 1988, Whittaker 1975). This brings into question the definition of disturbance. van der Maarel (1993) suggests the most precise definition was offered by (Grime 1979): “disturbance is considered to consist of the mechanisms which limit plant biomass by causing its total or partial destruction”. The description of disturbance can involve many metrics for describing disturbance events including (Agee 1993, Pickett and White 1985):

1) Type of disturbance (e.g. fire, logging, insects, biotic invasion),

2) Frequency or level of recurrence of disturbance events,

3) Severity or the amount of biomass removed by disturbance,

4) Intensity or physical strength of the disturbance event (e.g. flame size),

5) Distribution in space and time of disturbance events (e.g. seasonality),

6) Extent or size of disturbed patches,

7) Synergism of different disturbance agents or events involved (e.g. a severe surface burn may damage roots and predisposes trees to windthrow).

van der Maarel (1993) describes the three *main* dimensions of disturbance as spatial extent, duration and magnitude. These three dimensions combine to have variable effects on the stability of three ecological processes: (1) patch dynamics, (2) community dynamics and (3) regeneration succession. Therefore a more complete definition of disturbance may be "any relatively discrete event in time that disrupts ecosystem, community or population structure and changes resources, substrate availability or the physical environment" (Pickett and White 1985, see Laska 2001, or White and Jentsch 2001 for more on the definition of disturbance).

The traditional concept that underlying physical factors such as topography, slope, aspect, climate, and soil type dictate vegetation type is incomplete if the historic disturbance regime is not accounted for. Ecosystems are often adapted to a long term disturbance regime and changes in disturbance dynamics that alter the physical environment, microclimate, species richness/composition, patch size and seed bank in new ways may result in drastically different plant communities (Vitousek and Walker 1989).

The study of variation in disturbance has led to one of the most well known hypotheses in ecology today, namely the intermediate disturbance hypothesis (IDH) (Connell 1978). The IDH predicts that maximum species diversity occurs at intermediate levels of disturbance; while at high levels of disturbance long-lived species cannot persist and, at low levels species dependant on disturbance may disappear from the community. Although the IDH may not apply to every disturbance scenario (Svensson et al. 2009, Mackey and Currie 2001), it is, nonetheless a widely accepted ecological concept (Shea et al. 2004). The IDH provides a mechanism that promotes species coexistence and therefore plays a critical role in the maintenance of species diversity, as well as ecosystem function and viability (Grime 1998).

Multiple forms of disturbance are increasingly common on the land base as anthropogenic disturbances are added to existing natural disturbances. For example, many forests in North America affected by wildfire and insect attack are subjected to further large scale secondary disturbances such as salvage logging (clear-cutting after fire), grass seeding, silvicultural site preparation and cattle grazing. However, while the IDH can apply to the cumulative effects of multiple disturbances it does not readily account for variable interactions or positive and negative effects of sequential disturbances.

Disturbance not only directly affects survivorship but can also have an indirect effect by changing community structure and the long term environmental conditions that species have become adapted to. Any combination of disturbance, from a change in plant-plant interaction to alteration of the physical environment can occur. The spatial scale and frequency of disturbance also affect vegetation response and can alter habitat heterogeneity. Roberts (2004) summarizes eight of the main mechanisms by which disturbance affects composition of the herbaceous layer in forests including: 1) competition with higher strata, 2) competition within the herb layer, 3) microclimate, 4) coarse woody debris substrate, 5) pits and mounds or microtopography, 6) mineral soil substrates, 7) damage to pre-existing plants and, 8) propagule availability. Multiple disturbances can negatively affect species richness as predicted by the IDH; however, the effects will vary depending on the nature of the disturbances.

Another set of organizing ideas in disturbance ecology is concerned with ecosystem resilience. Ecosystem resilience is defined as the level of disturbance a system can handle while still retaining its basic functional and successional characteristics (Elmqvist et al. 2003, Holling 1973, Walker et al. 1999). The concept of resilience suggests that there is a threshold at which the natural range of plant community variation may shift irreversibly and lose the capacity to renew itself. One runs into a dilemma here because there is a wide range of ecosystems that can legitimately be considered “natural” as discussed in Sprugel (1991). Any community composed of native species in a state anywhere along the successional continuum may be considered “natural” and therefore assessing the impact of multiple disturbances is complicated. Often plant communities are in a natural state of flux as is the case, for example, in the transition between woodland and grassland. However, shifts in plant community composition may be especially disruptive when replacement by introduced, non-native species is involved.

1.2 Disturbance and Invasive Species

Because disturbance opens up new growing sites there is often an increase in ruderal and non-native species, another factor that can affect the process of regeneration succession. It has been suggested that *any* change in the historic disturbance regime will result in a reduction in native species and/or an increase in non-native species (Hobbs and Huenneke 1992). Exotic plant "invasions" have become one of the most persistent changes in the natural landscape in recent decades (Mooney and Hobbs 2000). There is little evidence that all introduced plants are harmful to native plant communities, however many of the non-native plant species in BC are considered invasive (widespread and dominant with potentially harmful effects to local plant communities) in habit and therefore should be considered a potential risk. Unfortunately, there is a lack of information on the current invasive status (i.e. the extent and degree of dominance) of many non-native species in BC.

Invasive species can alter ecosystem processes, community dynamics and disturbance regimes beyond any historical terms of reference for a particular ecosystem. Effects on ecosystem processes involve the alteration of resource availability and soil stability, acceleration of erosion, and retention of litter, salts, or other soil resources (Vitousek 1990, Gordon 1998). Invasive species can alter community dynamics by changing stand structure, recruitment rates, competing for resources and replacing functional groups (D'antonio and Vitousek 1992, Gordon 1998). Brooks et al. (2004) present a model of the relationship between plant invaders and fire disturbance regimes and explain how invasive plants can affect fire behavior and alter fire frequency, intensity, extent, type, and seasonality. These changes are largely due to the change in fuel properties afforded by introducing new plant species with life history and structural traits novel to a particular environment.

Biotic change by invasives has the capacity to alter many ecosystems beyond a reasonable means of restoring them back to their pre-invasive state. Large scale biotic change can be compared with global climate change, and it has been suggested that reducing greenhouse gases is easier than eradicating invasive species and undoing the transformations they have caused (Mooney and Hobbs 2000).

1.3 Natural History of Interior Douglas-fir Forests

Interior Douglas-fir (*Pseudotsuga menziesii var. glauca*) occurs along the western slopes of the Rocky Mountains and the eastern slopes of the Cascade, Coastal, and Sierra Nevada ranges from northern British Columbia (BC) to the southwestern United States and northern Mexico. In BC, the interior Douglas-fir forests generally occupy an elevational band between ~300-1400m (but up to 2400m) and are characterized by a continental climate with warm, dry summers, a fairly long growing season, cool winters, and often a substantial moisture deficit (Lloyd et al. 1990). Annual precipitation averages approximately 379 mm over the study area with more than 45% falling from May–September (Lloyd et al. 1990). The term dry Douglas-fir forests includes forests dominated by Douglas-fir in most of the Interior Douglas-fir (IDF) biogeoclimatic subzones, plus moister sites in the Ponderosa Pine (PP) zone and drier sites in the Montane Spruce (MS) and Sub-boreal Spruce (SBS) zones of BC (Vyse et al.1998, Lloyd et al. 1990).

Douglas-fir forests support a rich diversity of plant and wildlife communities reaching from the lower grasslands to the upper montane forests. Due to moderate climate and easy access, Douglas-fir forests also support many anthropogenic uses such as resource extraction, cattle grazing, recreation activities and urban development. In BC the IDF biogeoclimatic zone *sensu lato*, contains 107 plants and 44 animals that are endangered or threatened (BC Conservation Data Center 2009).

Douglas-fir is considered a late successional, moderately shade tolerant species capable of regenerating under its own canopy. Young Douglas-fir trees have a somewhat smooth bark with resinous blisters that in older trees ($\geq$30 yrs) thickens to form an insulative cork cambium layer, protecting mature trees from low-intensity fires. Interior Douglas-fir (hereafter Douglas-fir) can be distinguished from coastal Douglas-fir (FDC, *Pseudotsuga menziesii var. menziesii*) by its spreading instead of straight bracts, smaller cones (4-7 cm - interior, 6-10 cm - coastal) and its blue green/grey green leaves as opposed to the yellow green leaves of FDC.

At lower elevations Douglas-fir may be associated with ponderosa pine (*Pinus ponderosa*) and is less common on more xeric sites. At mid and upper elevations Douglas-fir and lodgepole pine (*Pinus contorta*) form extensive stands, while in cooler areas Douglas-fir may be found with white spruce (*Picea glauca*) and trembling aspen (*Populus tremuloides*) and on rocky, mountainous crests with limber pine (*Pinus flexilis*) and whitebark pine (*Pinus albicaulis*). In south central BC common Douglas-fir forest understory plant associations on mesic and dryer sites include shrubs such as falsebox (*Paxistima myrsinites*), soopolallie (*Shepherdia* *canadensis*), rose (*Rosa acicularis*) and maple (*Acer glabrum*) with abundant pinegrass (*Calamagrostis rubescens*) in the herb layer or Oregon-grape (*Mahonia aquafolium*), snowberry (*Symphoricarpos albus*), juniper (*Juniperus communis*), saskatoon berry (*Amelanchier alnifolia*) and spirea (*Spiraea betulifolia*) with bluebunch wheatgrass (*Pseudoroegneria spicata*) and rough fescue (*Festuca campestris*) as the dominant herbs (Lloyd et al. 1990). Grass or shrubs species often dominate the understory of open woodland areas while a moderate to extensive coverage of moss and lichen species may occur under denser canopies and on rock outcrops and woody debris. Extensive undisturbed 'climax' Douglas-fir forests are uncommon due to the relatively high frequency of fire and other disturbances in this ecosystem.

1.4 Disturbance History and Post-fire Communities of Dry Douglas-fir Forests

The idea that wildfire destroys forests and other natural ecosystems and should be suppressed at all costs dominated the fire management paradigm for more than half a century (Kozlowski and Ahlgren 1974). Contrary to common belief, major disturbances such as wildfire have benefits to ecosystems including the generation of heterogeneity and structural complexity (Inions et al. 1989), recycling of nutrients and the creation of early successional habitat for wildlife (Noss et al. 2006). Fire suppression results in an accumulation of woody plant fuels and may contribute to an increased occurrence of large-scale, high-severity fires (Agee and Skinner 2005, Jain et al. 2007). A recent management doctrine that dry Douglas-fir forests experienced mostly frequent, low severity fires is now being questioned in favor of a more highly varied fire management plan for this dynamic ecosystem. As well, while BC forest regulations do recommend the maintenance of biodiversity across the landscape, there is no mention of this type of conservation in salvaged post-fire or beetle attacked ecosystems. In fact BC is the only highly forested province not to have any policy specific to post-fire salvage (Schmeigelow 2006)

Klenner et al. (2008) give evidence for a consistent mixed-severity fire regime (a combination of high, moderate and low severity fires) in dry Douglas-fir forests, resulting in a mosaic of disturbance levels across the landscape (see also Arsenault and Klenner 2004, Hessburg et al. 2005). As well, they point out that bark beetles and defoliators have played a role in determining the historic and present forest structure in the Douglas-fir forests. The current landscape mosaic of the study area is composed of large patches of forest and regenerating clearcut areas, small patches of grassland, agricultural fields and lakes and some areas of natural early successional forest in parks and ecological reserves. Exactly how disturbance, climate and shifts in plant communities interact to create this mosaic is a very complex topic currently undergoing a great deal of research.

Post-wildfire understory plant communities are generally dominated initially by shade intolerant early successional species that have long distance dispersal abilities (Rowe 1983). In Douglas-fir forests these include plants such as fireweed (*Chamerion angustifolium*), many species in the *Asteraceae* family, and other weedy, annual or ruderal species. However, many species can survive fire as underground rhizomes (*C. angustifolium* included) or buried propagules in the soil seedbank (Stark et al. 2006). Some rhizomatous species may be more shade tolerant and therefore rely on the partially shaded microclimate offered by the many snags and patches of surviving trees present after fire (Carleton and MacLellan 1994, Eberhart and Woodard 1987, Geiger 1975, Harrington and Kelsey 1979). Little is known about the effects of post-fire standing dead and residual trees on microclimate; however, if salvage logging is similar to conventional logging greater extremes in surface temperatures and reduced water retention can be expected (Keenan et al. 1993).

The major impetus for salvage logging is the economic value of the timber removed. Other suggested reasons include the control of insect pests and the reduction of fuels, chance of reburn and erosion (McIver and Starr 2001). It has been suggested that salvage logging "adds little if any additional impact and serves to reduce the long term cumulative watershed impacts already imposed on the watershed by wildfire" (Barker 1989). However, recent research into the effects of salvage logging strongly suggest that many important ecological processes are altered and that the negative ecological impacts may out weight potential economic benefits (Lindenmayer et al. 2008, Lindenmayer and Noss 2006, Peterson et al. 2009).

During the summer of 2003, British Columbia experienced fifteen major wildfires. These fires consumed upwards of 100 000 hectares of forest and parkland, and caused serious damage to a number of residential and commercial areas. This study focuses on the McLure and McGillivray fires of 2003. Immediate rehabilitation efforts included aerially seeding slopes with agronomic grasses and legumes (hereafter referred to as seeded sites), as well as horticulturally bred native grass species in an attempt to provide forage, prevent erosion and avoid promoting invasive species. Post-wildfire reforestation was always preceded by salvage logging. The normal BC forest practice codes are relaxed in post-wildfire areas and as such the decisions regarding post-wildfire management vary widely.

1.5 Similar Studies

Previous studies comparing vegetation communities in natural post-fire environments to salvage logged areas are surprisingly limited because studies of forest disturbances often focus mainly on tree regeneration or changes in structure and age class distributions (Bock et al. 1978, Donato et al. 2006, Miller et al. 1993, Shatford et al. 2007). Studies of forest recovery after natural disturbance often indicate that tree seeding and planting following disturbances may not be necessary and may actually reduce forest recovery (Donato et al. 2006, Turner et al. 2003).

A critical effect of post-fire harvesting is a change in the patterns of biodiversity. Current research in the boreal forests and in northern California and Oregon suggests that salvage logging has negative ecological consequences for plant communities. Some of these consequences include negative effects on bryophyte communities (Bradbury 2006), increased edge influence on surrounding forests (Hanson and Stuart 2005), reductions in species richness and diversity (Sexton 1998, Purdon et al. 2004) and significant changes in vegetation structure, composition, and abundance (Stuart et al. 1993). In addition, logging after wildfire may result in loss of foraging and nesting habitat for wildlife, reduction of seed sources for regeneration, a decline in naturally seeded conifers and creation of less-favorable conditions for understory vegetation growth (Nappi et al. 2004, Hutto 2006, Donato et al. 2006). For example, Heinrich

(2007) found that ungulates in the study area depend on both mature forest and natural post-wildfire areas for winter forage and that deer use of the salvage logged and clear-cut areas was “non-existent”.

There is also growing evidence that artificial seeding after natural disturbance does not achieve the desired goals of slope stabilization or noxious weed control (Beyers 2004, Keeley et al. 2006, Robichaud et al. 2006). Nevertheless, if naturally burnt areas are not seeded for erosion control and slope failure does occur resource managers will be seen as not taking the necessary precautions needed to “preserve” the landscape. Livestock grazing and range economics also have an influential role in determining post-fire management and political views. This creates a conflict between the ecological role of natural recovery processes and prudent management perspectives.

This is the first large scale study to examine the effects of salvage logging and grass seeding on understory plant communities in Douglas-fir forests. The Douglas-fir forest system is a common forest type with a long history of intensive management applied consistently over a vast region, yet little is known about the understory plant community dynamics in this system. Given the prominence of salvage logging and grass seeding in BC, it is important that any changes in plant community dynamics be quantified in order to better manage and maintain the historic range of variability and biological diversity found in BC ecosystems.

In this study, I assessed the influence of post-wildfire salvage logging and grass seeding on community structure and species richness. The main research questions of this study are:

1) What effects do artificial seeding and salvage logging have on species richness in post-wildfire habitats?

2) How do community composition dynamics differ between natural post-wildfire communities and those managed with artificial seeding and/or salvage logging?

3) What are the combined biotic and abiotic effects of seeding and salvage logging?

Chapter 2 Methods

2.1 Study Area

The impacts of post-wildfire management on understory plant community dynamics were examined in two large fires areas near Kamloops, BC. The McLure and McGillvray wildfires burned approximately 26 420 and 11 400 ha respectively in the summer of 2003 (Figure 1). The McLure fire started due to human causes on July 30th and reached maximum area on September 4th. The McGillivray fire was started by lightning on August 15th and was contained and under control by firefighters by September 8th. Of the 22 000 ha of Douglas-fir forest affected roughly 34% was salvage logged and 29% was seeded. The biogeoclimatic classification for the study area covers primarily IDFmw2 (Thompson moist warm) and upper IDFxh2 (Thompson very dry hot) variants (Lloyd et al. 1990). The soils consist mostly of medium to coarse grained Dystric or Eutric Brunisols with a Mor or Moder humus form (Lloyd et al. 1990). There was no noticeable historic fire occurrence overlapping with any of the study area plots since 1950 (when formal fire records start for the study area).

2.2 Study Design

A stratified random approach was used to sample Douglas-fir-dominated plant communities that covered the range of fire and salvage disturbance severities while minimizing ecological differences prior to disturbance. Salvage logging and seeded “treatments” were not manipulated on the landscape in an *a priori* fashion and sampling occurred in habitat that was left after post-wildfire management. As a result, sites where not randomly assigned to be perturbed but nevertheless, given the large size of the area affected, sites with very similar abiotic and biotic attributes were available. In wildfire areas, unsalvaged and salvaged forest polygons were delineated. From these areas, nested polygons greater that 1 ha with a warm aspect (135˚ - 285˚), gentle to moderate slope (15 – 55%) and an elevation between approximately 600m and 1000m were identified using the GIS software ArcMap version 9.2., ESRI Inc. Redlands CA.

From the outlined polygons, a random selection of plot centers located at least 30m from the polygon edge and 50m from another plot center was created. Sites were then field-identified as having a mesic to subxeric soil moisture regime (classes 2–4 in Lloyd et al. 1990) and dominated by mature or near-mature Douglas-fir prior to disturbance. In wildfire plots, no evidence of anthropogenic disturbance was permitted. In clearcut plots, no evidence of prelogging human impact was permitted (except one prefire selectively logged area). Deciduous-dominated patches, old roads, and heavy use cattle areas were also avoided.

Circular sample plots (one to three per stand polygon) had an area of $400m^2$ (11.28 m radius) and contained 12 randomly placed $1m^2$ circular vegetation sampling quadrats (Figure 2). Relative frequency per plot was based on the total number of quadrats in which a species was present. The sample plot was also systematically searched for species that did not occur in the individual quadrats. Any additional species found were added to the list for that plot and used for species richness and diversity calculations. Any cow pies encountered were also tallied. Presence of all vascular and nonvascular species was recorded and voucher specimens of all unknown species were collected for later identification. Categorization of species as native, exotic or invasive was based on BC Ministry of Forests and Range data, Perzoff (2009) and the USDA Plants Database (USDA 2009). The percent cover of surficial substrates including exposed mineral soil, DWM (downed woody material), rocks, bedrock and organic matter >1cm thick was also recorded. Standardized approaches and definitions for collection of ecological data were adopted from the BC Ministry of Forests and Range manual “Describing ecosystems in the field” (Luttmerding et al. 1990).

To determine tree seedling regeneration a 3.99 meter radius silviculture plot (50 m^2 or 0.005 ha) was placed randomly within the larger 400 m^2 plot. The total number and health status of individual seedlings as well as the number of well spaced seedlings (approximately 2 m apart) was documented. Planted trees were distinguished by the presence of a root plug after brushing the soil from the seedling base. These greenhouse grown seedlings were not included in this analysis because they only occurred in the salvage logged areas. *Prunus pensylvanica*, *Prunus virginiana* and *Salix scouleriana* were treated as tall shrubs in this analysis.

Six 15cm deep soil samples were randomly collected within the 400 m^2 plot using a small auger and observations on soil disturbance in the 1m^2 area around the soil sample were noted. Soil horizon attributes (to a depth of 1m) were documented for a sub sample of 16 plots in each of the unmanaged (wildfire only) and managed (salvaged and/or seeded) stands. Tree species, diameter at breast height (DBH) and stump diameter/height measurements were obtained for all trees (dead or alive) in a 400 m^2 plot. Basal areas measured at stump height were later adjusted to breast height using allometric equations developed for similar forest types by the BC Ministry of

Forests and Range. As well, information on average scorch height, wood burn intensity class (Stepnisky 2003), crown mortality and ground scorch (Robichaud 2000) were recorded at each tree/stump. These metrics were then averaged for each plot, ranked evenly into three categories and used to designate the fire severity index (1 = low disturbance; 2 = moderate disturbance; 3 = high disturbance).

Data for equivalent, mature, non-burnt IDF forests (FOR) were selected from the Ministry of Forests and Range Biogeoclimatic Ecosystem Classification (BEC) database. These data consists of percent cover of vegetation in 400 m^2 plots. The same site selection criteria (aspect, elevation, slope, etc.) were used to select data from the BEC database. Only presence/absence information was used when analyzing FOR data that were collected between 1979 and 2002. Historical climate data for mean temperature at the Kamloops airport, BC were obtained from the National Climate Data and Information Archive (CDCD 2009).

2.3 Data Analysis

The six combined soil samples for each plot were air dried and sieved thru a 2 mm screen. A subsample of the composite soil for each plot was analyzed at the BC Ministry of Forests and Range analytical chemistry laboratory, Victoria, BC. Total C and N data were collected via combustion elemental analysis and other soil elements were obtained for the Mehlich III extraction using an ICP spectrometer. Exploratory regression analyses indicated that soil chemistry data were highly non-normal and therefore nonparametric Bonferroni-corrected pair wise comparisons were performed using Wilcoxon / Kruskal-Wallis tests in JMP 7.0, SAS Institute, Cary, NC. There was little detectable effect of salvage logging or seeding on mean soil chemistry concentrations, nor was there any significant correlation between soil chemistry and species richness, as a result, no further statistical analyses were conducted on these data

Species richness depends on the number of individuals sampled (Gotelli and Colwell 2001); therefore species accumulation curves were generated for each site type (treatment) in order to compare species richness. Total species richness for each site type was approximated using a first-order jackknife estimator as implemented in PCord version 5.10 (McCune and Mefford 2006).

All data sets were evaluated to determine whether they significantly deviated from normality or homogeneity of variance, and transformations were performed when necessary. When appropriate, one-way ANOVA was used to test for differences between species richness in wildfire (WF), seeded wildfire (WFSE), salvaged wildfire (WFSA) and seeded and salvaged wildfire (WFSESA) site types at the plot and stand scales. Two-way ANOVA was also performed to test the interaction between site type*fire severity, site type*fire area (McLure and McGillivray) and site type*disturbance season for total species richness. The season of salvage logging disturbance was acquired from the Ministry of Forests and Range RESULTS database. When ANOVA tests showed significant differences, Tukey's HSD post hoc tests were undertaken to determine specific treatment differences with pair-wise multiple comparisons.

For every site type the Whittaker index of beta diversity (Bw) (Whittaker 1960, 1972) was calculated. This was done by dividing the total number of species in one site type by the average number of species in the 400 m^2 plots of that site type minus one. Coefficient of variation for plots and total among-plot variance, both using raw species data tables, were calculated for each site type using PCord ver. 5.10. Measures of diversity including evenness, Simpson's index and Shannon’s diversity were also calculated using PCord ver. 5.10 (McCune and Mefford 2006).

2.4 Community Composition Analysis

MRPP (multi-response permutation procedure) was used to test the hypothesis of no difference in community composition between site types. This randomized test for significance suited the non-normal species matrix data collected (McCune et al. 2002) and is particularly suitable for studies where vegetation is unevenly distributed due to high levels of disturbance (Biondini et al. 1988).

When MRPP identified significant groupings, indicator species analysis (ISA) was performed (Dufrene and Legendre, 1997). This test provides an indication of how well the presence of a species indicates a site type. A threshold level of 20% for the index was chosen as a slightly more lenient cut off for identifying significant indicator species than the 25% value recommended by Dufrene and Legendre (1997).

Nonmetric multidimensional scaling (NMS) ordination was chosen to provide an ecologically meaningful visualization of the compositional differences among forest types. For NMS analysis the main matrix was also used as a secondary matrix in order to produce vectors in a joint plot graph that showed which species were driving the variation of the axes. In order to determine the best solution (dimensionality with lowest final stress) for NMS, the ‘‘slow and thorough’’ autopilot procedure was utilized (McCune et al. 2002, McCune and Mefford 2006). Only species or species groups with an r^2 greater than 0.3 are shown in the ordinations. Kendall and Pearson correlations with ordination axes were used to aid in interpretation of the ordination trends.

MRPP, ISA and NMS calculations were computed using the Bray/Curtis (Sorenson) distance measures in PCord version 5.10 (McCune and Mefford 2006) and no transformations where made to the data.

Uncommon species were defined as those that occurred in less than 10% of the plots (this included additional species that were not found in quadrats), occasional species occurred in 10–33% of the plots, and common species in more than 33% (Dodson et al. 2007). Richness was then calculated for uncommon, occasional and common native species and ANOVA was used to quantify differences. All ANOVA and related post hoc tests were calculated using JMP 7.0, SAS Institute, Cary, NC, USA.

Native to exotic ratio (N:E) was calculate as native species richness divided by exotic species richness in individual plots. Linear ANCOVA regression models of native versus exotic (including seeded agronomics) species richness were also performed in JMP 7.0, SAS Institute, Cary, NC, USA. In ANCOVA, regression lines have the same slope but different y-intercepts and separation between these lines in this case relates to significant differences between the least squares means for the four site types. Plot and stand scale data satisfied the assumptions of ANCOVA (Gotelli and Ellison 2004), however the quadrat scale data was highly non-normal, subject to pseudoreplication and did not have equal slopes. A covariance model with different slopes was applied to the quadrat data to illustrate the relationship between native and exotic richness and allow for unequal slopes. Generalized linear model (GLM) analysis using a poisson distribution and log link function was also used to confirm the relationship between natives and exotics at the quadrat scale.

Chapter 3 Results

Climate data analysis revealed a mean temperature increase of 1.2 degrees for the years data were available (Figure 3). Mean spring and summer precipitation and mean temperature three years before the study fires showed no significant deviation from means of the previous 20 years ($p = 0.51$, precipitation and $p = 0.95$ temperature).

3.1 Patterns of Species Composition

In total, 193 species from 42 families were observed in the two wildfire areas:

- 148 herbs/forbs,
- 25 shrubs,
- 15 nonvascular and,
- 5 tree species.

A survey of 104 plots (Table 1) found more than twice as many species in post-wildfire Douglas-fir forest sites than previous smaller scale studies in similar habitat (Stark et al. 2006). The wildfire and salvage logged wildfire sites (not separating grass seeded sites) had 163 and 133 species respectively, with the wildfire sites having 60 unique species and the salvaged logged areas having 30 unique species (Figure 4). Of the four site types sampled, the non-seeded, non-salvaged wildfire (WF) sites had the highest number of species (130) and the most unique species (25) (Figure 4). The seeded wildfire (WFSE) had 121 species, followed by the salvaged and seeded wildfire sites (WFSESA) with 103 species and the salvage wildfire sites (WFSA) with 90 species, the latter being ~30% lower than the WF only sites. All of the species proposed to have been aerially seeded on the fire sites in various mixtures (Table 2) were found.

The most frequently occurring species (plot frequency >70% in all four site types) included *Ceratodon purpureus*, *Calamagrostis rubescens*, *Spiraea betulifolia*, *Chamerion angustifolium*, *Bryum caespiticium*, *Mahonia aquifolium*, *Lactuca serriola*, *Epilobium brachycarpum* and *Taraxacum officinale* (Table 3). The resprouting, nitrogen fixing shrub *Ceanothus sanguineus* showed a 46% frequency reduction in WFSA sites compared to WF sites, the largest difference of all shrubs encountered. *Arnica cordifolia* was the most common shade-tolerant herb, while *Calamagrostis rubescens* and *Poa pratensis* had the highest relative abundance of native and exotic graminoid species, respectively (Table 3).

While the species accumulation curves (Figure 5) did not reach horizontal asymptotes, the slopes of the tails ranged from 0.013 for the WFSA sites to 0.107 for the WF sites suggesting that an adequate prediction of regional diversity had been obtained. The smaller sample size of the WFSA did not seem to impair the species accumulation curve from nearing a plateau; thus, it seems reasonable to compare the four site types even though they are not of equal sample size. A comparison of total species richness (including additional species found in $400m^2$ plots outside of the quadrats) and jackknife estimates in each of the site types indicates that between 83% (for the WFSA sites) and 97% (for the WFSE sites) of the species predicted to occur in each site type were captured by the sampling plan (Figure 5). In addition, although the two seeded sites had differing sample sizes they had very similar accumulation curves and jackknife estimates.

Ecological data obtained from the BC Ministry of Forests and Range for equivalent mature forested ecosystems (FOR) totaled 219 taxa, 105 of which also occurred in the sampled wildfire sites. Of the 114 taxa found only in the mature forest sites, 56 (49%) were nonvascular species (not including epiphytes) and 10 (9%) were shrubs.

One red-listed species, *Poa fendleriana* (Steud.) Vasey ssp. *fendleriana*, whose population is considered to be at risk in BC, was identified in a WF site. Furthermore, in all sites sampled 69% of species were considered uncommon (occurred in <10% of plots) and 28% of the species occurred in only one plot (Table 4). Thirty-six of the species surveyed are considered introduced (exotic) in BC (USDA 2009), with approximately twice as many of these found in the WFSA and/or WFSESA sites as in the WF or WFSE sites (Table 4). Sixteen of these introduced species are considered "invasive, noxious or problem plants" in BC (Perzoff 2009).

MRPP revealed significant differences in species composition among site types ($p = <0.001$, Table 5). The only pair wise comparison not to show a significant difference was between WF and WFSE ($p = 0.128$, Table 5). There was no significant difference in the species composition among three levels of estimated fire severity between site types ($p = 0.382$). The largest effects were found between all site types together ($A = 0.067$) and between the WF and WFSESA sites

(A = 0.082, p = 0.001). An MRPP effect size of 0.1 is common for significantly different observations in community ecology (McCune and Grace 2002).

Indicator species analysis suggested that most of the species causing differences among the site types are in the WF and WFSESA types (Table 6). Seven native species were significantly associated with the WF sites, including the common Douglas-fir understory herbs *Arnica cordifolia* and *Aster conspicuous*, as well as the grass, *Calamagrostis rubescens*, and the shrub *Rosa acicularis*. Eight introduced species were indicative of the WFSESA sites including five agronomic species and three invasive species. The most common species in the WFSE sites were two agronomic species one of which (*Poa pratensis*) was not listed in the proposed seeding mix (Table 2). The two non-salvaged site types (WF, WFSE) also counted native moss species among their indicators. Two native ruderal species (*Conyza canadensis* and *Chamerion angustifolium*) and one invasive (*Logfia arvensis syn. Filago arvensis*) were the only significant, highly-indicative plants in the WFSA sites.

A 3-D solution NMS (final stress = 10.2) illustrated the separation of the four site types and the species groups associated with each type at the plot scale (Figure 6). Axis 1 explained the most variation with an R^2 value of 0.466 and largely reflects a strong separation between lifeforms such as conifer and broadleaf trees (r = -0.344 and -0.402), shrubs (r = -0.444) and nonvasculars (r = -0.57) versus exotic (r = 0.866) and graminoid (r = 0.552) species. Generally, the WFSE and WFSA sites are separated from the along axis 2 (R^2 = 0.344), with graminoid species (r = -0.894) at one end and conifer and broadleaf trees (r = 0.421 and 0.638) at the other. The NMS ordination showed a high degree of separation between the WF and the WFSESA site types suggesting species composition of the understory plant community was altered the most in WFSESA sites. There was considerable overlap between the WF sites and both the WFSA and WFSE sites.

At the stand (polygon) scale a 3-D solution NMS (final stress 14.0) using an individual species matrix yielded very similar trends to those found at the plot scale (Figure 7). In this case the variation explained by axes 1, 2 and 3 is more evenly distributed (0.329, 0.227 and 0.277 respectively). One of the greatest distinctions between the Pearson and Kendall correlations with axis 1 was for the seeded species *Festuca brevipila* (-0.672, FESTRA) and the moss *Bryum caespiticium* (0.477, BRYUCAE). For Axis 2, the invasive weeds *Lactuca serriola* (-0.824) and *Sisymbrium loeselii* (-0.629) were negatively correlated with the shrubs *Rosa acicularis* (0.67) and *Paxistima myrsinites* (0.608). Axis three is difficult to portray in the 2 dimensional Figure 7 but species with the lowest axis 3 scores included *Calamagrostis rubescens*, *Spiraea betulifolia* (low shrub) and *Aster conspicuus*. Species with the highest axis 3 scores included *Arenaria serpyllifolia* (exotic) and *Cerastium arvense* (native). In general, the relationships among site types displayed in the NMS ordinations concur with the results of the MRPP comparisons and the indicator species analysis.

3.2 Patterns of Species Richness

The patterns of species richness and Shannon's diversity index (H') were very similar (linear model; $r^2 = 0.9$, p = <0.001) therefore the simplest metric (richness) is presented here (except in the regression with downed woody material where H' better satisfied the assumptions of normality). One-way ANOVA results at the plot scale revealed site type as a significant factor of species richness within the species groups examined (Figure 8). When analyzing all understory vegetation combined, only species richness at the WFSA site type was significantly lower than the WF type (22% decline in mean richness, Tukey's HSD $p<0.05$: ANOVA, $F = 3.46$, $p = 0.010$). However, when agronomic seeded species were excluded from the WFSE and WFSESA site types the WFSESA was also significantly lower (Tukey's HSD $p<0.05$; ANOVA, $F = 17.68$, $p = <0.001$) in species richness than WF. Table 7 shows a summary of mean species richness by site type. No significant interactions were detected between site type and fire severity index ($p=0.4964$) or disturbance season ($p=0.0782$) in these ANOVA analyses. Disturbance season did have a nearly significant interaction term, however, and a separate one-way ANOVA indicated that two stands (two plots each) in the McGillivray fire and one stand in the McLure fire (three plots) that were salvaged in the winter had a significantly lower species richness than could be attributed to chance alone (Tukey's HSD $p<0.05$; ANOVA, $F = 5.92$, $p = 0.001$). These three stands also had between 50 and 80% cover of downed woody material (DWM).

For the plot scale, mean exotic richness at the WFSESA sites was significantly higher (Tukey's HSD $p<0.05$; ANOVA, $F = 35.8$, $p = <0.001$) than the other site types and this relationship held for all but the WFSA site type when agronomics were removed (Figure 8). For example, *Lactuca serriola* was nearly two times more frequent in WFSESA than WF sites (0.34 in WF vs. 0.60 in WFSESA). Conversely, the undisturbed forest (FOR) had significantly lower exotic richness than any other site type even with agronomics removed (Tukey's HSD $p<0.05$).

Native species richness among site types at the plot scale had the same pattern as that of all vegetation combined. The WFSA site type had a significantly lower native richness than the WF site type (Tukey's HSD $p<0.05$; ANOVA, $F = 4.93$, $p = 0.0010$).

Shrub species richness in the FOR, WFSE and WFSESA sites was significantly lower than in the WF site type (Tukey's HSD $p<0.05$; ANOVA, $F = 10.51$, $p = <0.001$), suggesting an effect of seeding on shrub regeneration at the plot scale. There was a strong decline in nonvascular taxa in the post-fire environment (Tukey's HSD $p<0.05$; ANOVA, $F = 40.97$, $p = <0.001$; Figure 8).

The WF site type had the greatest beta diversity (100.2) followed by the WFSE (92.7), WFSESA (75.1) and WFSA (66.5) site types. Conversely, the WFSESA sites had the highest coefficient of variation and total among plot variance in relative abundance, indicating a spatial increase in variability at these sites (Table 7).

Mean species richness at the stand scale resembled the trends found at the plot scale with three important differences (Figure 9). First, excluding agronomic species for all vegetation combined did not show a significant decrease in species richness in WFSESA compared to WF. Second, there was no significant difference in stand scale shrub richness found between any of the site types (ANOVA, F = 2.29, p = 0.0938). Thirdly, richness of nonvascular species showed a moderately significant reduction in WFSESA compared to WF sites.

The ratio of native to exotics species richness (N:E) further elucidated relationships between site types at the plot scale (Figure 10). One-way ANOVA results indicated that WFSE, WFSESA and WFSA sites had significantly lower N:E ratios compared to WF sites (Tukey's HSD $p<0.05$; ANOVA, F = 56.4, p = <0.001). Furthermore, WFSESA had a significantly lower N:E than WFSE and WFSA sites (Figure 10). These trends were repeated at the stand scale except the WFSESA was not significantly lower than the WFSE and WFSA sites (Figure 11).

Common native species richness differed among the site types at the plot scale, with the WFSA and WFSESA sites showing a significant reduction (F = 7.2; p = 0.0002), but there was no detectable difference at the stand scale (Figure 12 & 13). Richness of occasional native species in the WFSE and WFSESA sites was reduced from that of WF at the plot scale (p = 0.001; F = 11.2), but again there was no difference among site types at the stand scale. Uncommon species richness, however, differed among site types at both the plot and stand scales with the WF sites having a significantly higher richness than all but the stand scale WFSE sites (Figure 12 & 13).

Analysis of covariance (ANCOVA) regression demonstrated a positive relationship between native and exotic species richness at the plot scale (r^2 = 0.598, p = <0.001; Figure 14). However, the intercept of the fit line for each site type increased from the least disturbed site types (FOR, 1.3 and WF, 4.6) to the most disturbed site type (WFSESA, 10.8) indicating a greater presence of exotic species with the addition of secondary disturbances. The relationship between native and exotic richness shows locally positive correlations despite broad scale increases in exotic richness. All individual regression lines are significantly different ($p = <0.05$) except the WFSE (6.59) and WFSA (6.55) lines which overlap. Most importantly the WFSESA sites were significantly different from all other sites types. Stand scale ANCOVA results were nearly identical to plot scale results and are not presented here. There was negative relationship between native and exotic species richness at the quadrat scale, however the statistical robustness of this relationship is weak due to its non-normal nature (Shapiro-Wilk's test $W = 0.9814$, p-value = <0.001) and therefore interpretation of ANCOVA regression results should be done with caution. Nevertheless, nonparametric generalized linear model (GLM) analysis suggested a significant ($p = <0.001$) negative relationship between native and exotic species richness. ANCOVA results at the quadrat scale suggest that the intercepts of WF (1.4) and WFSA (1.2) sites are not different while WFSE (2.2) and WFSESA (3.6) sites are different from the WF and WFSA sites and each other (Figure 15). At the $1m^2$ scale there is generally a negative relationship between native and exotic species richness (Figure 15).

3.3 Environmental Characteristics

The mean thickness of the Ah horizon (incorporating humus, organics or charred organics) in WF and WFSA sites was 2.2 cm and 0.56 cm respectively, resulting in a significant difference in Ah horizon thickness ($F=13.1$, $p=0.002$, $n=16$). Twelve of the sixteen soil descriptions for the WFSA site had no Ah horizon at all. At the plot and stand scales soil chemistry was similar among site types. There were slight separations of soil chemistry variables such as aluminum, phosphorus, calcium, magnesium and carbon to nitrogen ratio among site types; however these variables failed to meet assumptions of normality and equal variances. See Appendix B for a list of the soil data collected.

Nonparametric Wilcoxon / Kruskal-Wallis tests suggested a somewhat significantly higher mean presence of cattle in WFSESA sites at the stand scale as measured by cow pie counts ($p = <0.008$; Bonferroni corrected $\alpha=0.008$). As well, these tests indicated a significantly higher maximum fire severity index on the WF and WFSE sites ($p=0.006$), indicating that WF areas may have been left unharvested due to low quality timber. However, high fire severity may simply be more difficult to estimate in the salvage logged sites. As well, while the plots may have been perceived as uniform in fire severity this may not have been the case. This highlights the inherent subjectivity of the fire severity index. The mean cover of DWM was significantly greater in the two salvage logged stands ($p = <0.001$). DWM also showed a negative relationship with mean Shannon's diversity at the stand scale and, as shown in Figure 16, the relationship is driven by influence of the two salvage-logged site types ($r^2 = 0.54$, $p = <0.001$).

The mean canopy cover given by standing dead and residual trees in the WF and WFSE sites was 15% and ranged between 1% and 77%. At the plot scale there was no significant difference in Douglas-fir basal area ($p=0.12$) or tree stem density ($p=0.59$).

3.4 Tree Seedling Regeneration

Natural regeneration of tree seedlings in the post-wildfire environment was variable with 53% of plots showing no natural seedling regeneration and 58% of plots showing no natural conifer regeneration. Tree seedlings encountered included *Pseudotsuga menziesii, Pinus contorta*, *Populus tremuloides*, *Betula papyrifera*, and *Populus balsamifera*. Differential responses to site type treatments were found at the plot scale but very little difference was found at the stand scale.

Comparison of 95% confidence intervals (Figure 17) suggest a reduction in naturally regenerated tree seedlings at the plot scale in the WFSE and WFSESA sites as indicated by a lack of overlap in 95% C.I.'s. This trend is confirmed when comparing seeded and non-seeded plots (Figure 18). Furthermore, figure 19 and 20 indicate reductions in seedlings at seeded sites for *P. menziesii* and *P. contorta* seedlings, but not for broadleaf species.

At the stand scale there was little difference in natural tree seedling regeneration (Figures 21, 22, 18), except for a marginally significant reduction in number of seedlings ha^{-1} at seeded sites (Figure 23).

Chapter 4 Discussion

Early post-wildfire plant communities were significantly altered by multiple disturbances four years after wildfire in interior Douglas-fir forests. The cumulative effects of wildfire, post-wildfire grass seeding and post-wildfire salvage logging had negative impacts on community composition, relative frequency (regeneration success) of some plant groups, species richness and the ratio of native to non-native species. These effects diminish the quality of habitat for species that use early successional forests and may hinder the reestablishment of Douglas-fir forest structure and species composition and reduce general biodiversity.

Current forest management practices recognize the ecological and economic importance of maintaining biodiversity in natural and managed ecosystems (Grime 1998, Edwards and Abivardi 1998, Hansen et al. 1991). Douglas-fir forests and associated ecosystems shelter the second highest number of red listed plant species (107) of any ecosystem type in BC (BC CDC 2008; Coastal Western Hemlock zone = 118 red listed spp.). These dry forests represent a unique component of North American and Canadian flora. In this study Douglas-fir forests burned by wildfire had the greatest number of species and also contained many species not found in other sites examined. Because wildfire can occur on such a large scale, the cumulative negative effects of multiple post-wildfire disturbances may have implications for the conservation of biodiversity within Douglas-fir forests. This is concerning because Douglas-fir forests are predicted to expand in range with changes in climate, adding more challenges for the long term maintenance of biodiversity in this system.

4.1 Project Limitations

In this project, environmental variation was minimized at the landscape scale to focus on ecosystems with broadly similar climatic, geological and biotic influences. There were many unmeasured environmental factors such as microclimate, soil compaction and historic disturbance effects that could be at play, however any variation in these factors are likely consistently expressed in the patterns of richness and community composition between site types. It was the goal of this study to encompass the full range of wildfire and post-wildfire disturbance intensities thereby capturing any variation evenly across the landscape.

In analyzing the effects of natural disturbance there are methodological limitations due to the fact that technically the main level of independent replication is the disturbance event itself. Hurlbert (1984) suggests that many ecological studies are guilty of clustering sample plots in such a way that their responses are not independent. In this study, I have tried to follow Hurlbert's mantra of replication, randomization and interspersion whenever possible. My landscape scale questions were best answered by lumping the two temporally and ecologically similar fire events together. However, I have made every attempt to address the potential problem of pseudoreplication and to provide robust empirical data. The vast area of post-fire Douglas-fir forests and management treatments available for sampling allowed a random selection of plot sites from countless potential sites. All plots were separated by 50 m or more to foster independence in species composition at scales representative of typical understory vegetation communities (Palmer 1995). As well, variability in local site conditions was minimised as much as possible in order to better distinguish the effect of a “treatment” from some other systematically varying or unmeasured factor.

There may be confounding factors due to the influence of varied post-salvage silvicultural practices in the McLure and McGillivray fire areas. In particular, some salvaged logged sites were “stumped”; a process whereby the salvaged stumps are mechanically pried out of the ground and flipped upside down in the same spot. This practice provides mineral planting sites for tree seedlings, is thought to be a treatment for *Armillaria* spp. (root rot fungi) and may also dissuade cattle from using stumped areas (Dana Manhard, BC Ministry of Forests and Range, pers. com. 2007). Stumping was considered part of the normal post-harvest site preparation and reforestation process on the landscape and was not analyzed separately. As well, the salvage harvesting and related disturbance started in late August 2003 and for some sampling areas did not end until December 2005. It is likely that the direct damage of salvage logging on vegetation would be reduced if all harvesting were done in the winter when the soil is frozen and/or covered with snow. However, salvaged sites disturbed in the winter season were found to have the lowest species richness, suggesting these sites were not harvested during sufficiently cold temperatures.

Climatic factors were implicitly controlled for as sample sites were in reasonably close proximity, had comparable topographic features and shared similar weather patterns on a yearly basis. As with most natural experiments there was no true “control” and the site types in this study do not represent classical “treatments”. While this study did make use of data collected in similar pre-fire ecosystems, it is questionable whether species richness in early successional and late successional forests is comparable. In mature forests many species are dormant in the seedbank and therefore vegetation surveys may only detect late successional species unless the site contains recent disturbance patches of some sort. For example, Stark et al. (Stark et al. 2006) found that 40% of mature Douglas-fir forest seedbank species were not represented in the above ground vegetation. As well, because the number of individuals of a particular species and the total species richness can vary significantly between different communities, comparable samples will not necessarily be achieved using a balanced sampling design (Colwell and Coddington 1994, Loya and Jules 2008).

The results of this short term study of shifts in species compositions and richness may be difficult to apply to long term trends. Nevertheless, short term natural experiments avoid account of larger climatic and physical changes over time and can incorporate information from a larger spatial scale than is realistic for classic field or laboratory experiments. Collecting environmental data 3-4 years post-disturbance may not adequately represent initial conditions that influenced vegetation regeneration. However, there is evidence that many post-fire plant communities stabilize after 3-4 yrs for upwards of 20 yrs suggesting that community patterns observed in this study may have long term implications (D'Antonio et al. 2000, Grace and Keeley 2006, Keeley et al. 2005, Pyke and Archer 1991, Stuart et al. 1993).

4.2 Effects of Post-wildfire Grass Seeding Disturbance on Douglas-fir Plant Communities

The high persistence and frequency of artificially seeded agronomic species four years after wildfire suggests the effects of these species on native plant communities may be long lasting. This result confers with other research suggesting agronomic species can dominate native plant communities and may persist for 12 yrs or more (D'Antonio et al. 2000, Kruse et al. 2004, Keeley 2004). Early successional wildfire (WF) plant communities were characterized by native herbs, grasses, shrubs and trees, while seeded wildfire (WFSE) and seeded and salvage logged wildfire (WFSESA) sites showed increasing levels of non-native and ruderal species. Aerial seeding sources have been shown to often have a high level of contamination and are responsible for the introduction of many invasive species (Keeley 2006). The presence of *Bromus inermis ssp. inermis, Sisymbrium loeselii, Circium spp.* and *Poa pratensis* in the WFSE and WFSESA sites (and its absence in the WF sites) suggests contamination of the seeding mix may be to blame for at least some of the increase in non-native species in these site types. *B. inermis* is often associated with agricultural fields where much of the grass seed is grown and was 80% more likely to occur on seeded sites than non-seeded sites in this study (Table 3). Pre-2003 data indicated some areas of the McLure and McGillivray fire sites had exotic agronomic species such as *Poa pratensis, Medicago sativa* and *Agropyron cristatum*, reflecting the historical influence of livestock grazing in the area. It is unknown whether historic seeding events have taken place, or whether species such as *P. pratensis, M. sativa* and *A. cristatum* have invaded from nearby areas, however it is assumed that most or all of the agronomic species surveyed in this project originated from the aerial seeding campaign shortly after the forest fire events. However, *P. pratensis* is considered by some botanist to be naturalized and it has been argued that some populations in western North America may be native (Gleason and Cronquist 1963).

The native species *Pascopyrum smithii* was not found on any of the non-seeded sites and therefore it is assumed that all occurrence of *P. smithii* is due to the presence of the horticulturally produced variant of this species in the seed mix. Interestingly, this species was to be one of most represented species in the erosion, weed control and wildlife forage seed mixes (between 30 and 35% by weight, Table 2) and yet this species had the lowest occurrence rate of any of the seeded species.

Grass seeded areas are known to attract cattle due to the increased forage quality offered compared to areas with native vegetation. There is some evidence that carefully managed grazing can help control noxious weeds (Olson 1999), however there is also research documenting reduced biodiversity and an increase in exotic species and noxious weeds due to grazing (Freilich et al. 2003, Kauffman and Pyke 2001). The effects of grazing are complex as grazers can potentially limit dominance of exotics by reducing competitive interactions and also act a vector for the spread of invasives. Site selection methods for this study avoided sites with signs of livestock grazing; however, pre-fire and post-fire grazing may still have affected species composition.

While grass seeding may not typically be thought of as a disturbance that causes physical damage to ecosystem, community or population structure, it can disrupt many processes and functions in forest systems. Previous studies have indicated that post-fire grass seeding resulted in lower native species richness and diversity in various ecosystems (Keeley 2004, Kruse et al. 2004, Amaranthus et al. 1993). Other effects of grass seeding observed in this study were a reduction in shrub and tree richness and relative abundance. These effects are primarily attributable to resource competition and altered post-fire regeneration conditions.

Immediately after fire, competition effects may be evident between the sprouting survivors/seedbank and the sown agronomic species. The reestablishment of native species may be suppressed by aggressive grass species competing for light, water and other resources. Exotic grasses are better adapted to high nutrient and moisture conditions and often have higher maximum growth rates than native species that are adapted to low nutrient environments (Brooks 2003, Maron and Jefferies 1999). The shallow, fibrous root systems (Gordon and Rice 1993) and high wet season transpiration rates of most agronomic grasses effectively allow them to deplete rooting zone soil water (Davis and Mooney 1985, Eissenstat and Caldwell 1988) giving them a competitive advantage over other life forms. These grasses may die back during the dry season, producing a thick persistent grass litter or mulch that may promote grass productivity (Heady 1956) and inhibit the growth of non-grass survivors and newly arrived propagules. This change in soil surface substrate may affect arrival patterns, establishment of propagules and the ability of nearby residual or newly regenerated species to colonizing after intense disturbances. It is interesting that the WFSE sites did have a relatively high occurrence of the important native grass *Festuca idahoensis*. Native grasses such as *F. idahoensis* are important browse for native ungulates in Southern BC. Recovery of native grass species may be hampered by a high cover of exotic grasses (Brooks 2003, Humphrey and Schupp 2004), but some species may recover overtime from the change in environmental conditions (Corbin and D'Antonio 2004). Native grass species common to mature Douglas-fir forests such as *Hesperostipa comata* ssp. *comata* and *Achnatherum occidentale* were not present in any of the study plots sampled. It may be important to note that the seeded species *Festuca brevipila* (syn. *F. trachyphylla*) is highly variable and it is unknown whether this species hybridizes with *F. idahoensis*, a species capable of hybridization with other *Fescue* species (Jones et al. 2008). Introduction of agronomic species capable of hybridizing with native species may cause genetic erosion of native populations, therefore the interactions between *F. brevipila* and *F. idahoensis* warrant further investigation.

The degree to which exotic grasses out compete native shrubs such as *Artemesia spp.* has been shown to increase with grass density (Eliason and Allen 1997). Newly sprouting shrubs grow slowly and may have low carbohydrate reserves due to fire damage (Bond and Midgley 2003, Bond and Van Wilgen 1996). It may be difficult for native shrubs to establish in areas where thick fibrous root mats form due to heavy grass seeding. The same may also be true in areas with large accumulations of grass litter (aka. mulch, thatch), however as observed in the field, native plants such as *Chamerion angustifolium* grow vigorously after fire and may also produce enough litter to suppress other species from growing underneath them. Competition effects such as these are obviously density dependant. Although the number of individuals for each species was not measured directly, relative frequency may serve as a rough surrogate for density. Opportunistic post-wildfire species like *C. angustifolium* already dominate patches of the landscape after fire and, with the addition of agronomic species, it seems shrubs may be displaced by further patch dominance and high relative frequency of seeded species in the WFSE and WFSESA site types.

Grass seeding may have also had a strong effect on wind dispersed tree species such as *P. menziesii* which showed the lowest recruitment numbers in seeded areas. Tree seeds and seedlings may suffer the same grass competition effects as mentioned above, however grass seeding may also be associated with other sources of mortality. It has been shown that small mammals are more frequent in grass seeded areas (Sullivan and Sullivan 1984) potentially leading to increased levels of predation on native seeds and seedlings as well. Indeed, it has been suggested that in some areas rodents are the primary predators of *P. menziesii* seeds (Caccia and Ballaré 1998). As well, Mills (1983) presented evidence that small-mammal herbivory influences the first-year establishment of shrub seedlings including *Ceanothus* species.

Yet another indirect effect of grass seeding on tree and shrub seedlings is reduced mycorrhyza formation (Amaranthus et al. 1993). Grass species do not support ectomycorrhizal fungi that are important for the establishment of trees and shrubs on sites where regeneration is difficult (Amaranthus and Perry 1987). Grass seeding, therefore, may cause changes in the below ground community that do not favour tree and shrub establishment leading to their reduced richness and frequency on seeded sites.

4.3 Effects of Post-wildfire Salvage Logging Disturbance on Douglas-fir Plant Communities

Previous investigation of the effects of salvage logging on species richness in other systems such as boreal forests have shown an increase (Kurulok and Macdonald 2007) or a decrease (Purdon et al. 2004) in richness due to logging after natural disturbance. Much of the increase in richness was due to a higher proportion of non-native, ruderal and shade intolerant species. Similar research in north-western USA forests has indicated little difference in species richness (Stuart et al. 1993) or a reduction in species richness (Sexton 1998), but both studies did note increased cover of native herbs and shrubs on non-salvaged sites and pronounced effects of salvage logging on species composition. These compositional effects included a reduction in native herbs and nitrogen fixing shrubs as well as an increase in exotic species and graminoids. In the present study, salvage logging in the McLure and McGillivray fire areas resulted in a reduction in native and overall species richness, decreased occurrence of uncommon species and an increase in the proportion of exotic species. These changes in post-fire successional vegetation dynamics and structure due to salvage logging are primarily attributable to physical damage to regenerating species and alteration of post-fire microenvironment and substrates.

The process of removing the charred and uncharred trees involves the use of a “feller-buncher”, a large machine that harvests, delimbs and piles the trees. The shoots and roots of regenerating vegetation lying in the track paths and under piled logs are subject to mechanical damage, rutting and soil compaction (Tan et al. 2009). Germination of the seedbank takes place shortly after wildfire and before or during post-fire harvesting. Damage to pre-existing and newly sprouting vegetation before adequate establishment has occurred severely hinders post-fire regeneration and can alter the successional trajectory of post-fire plant communities. The timing of salvage logging effectively stunts seedbank regeneration and reduces input from other seed sources (Greene et al. 2006), however the severity of mechanical damage may depend greatly on seasonality. Wet and fine textured soils have been shown to be more susceptible to compaction and rutting (Moehring and Rawls 1970, Brais and Camire 1998) therefore it is predicted that direct damage from salvage logging may be greatest April through June, moderate in the summer and least in the winter when soils are frozen. However, as indicated in the results, winter disturbance was associated with the lowest levels of species richness in WFSA sites, possibly indicating that soils in the study area do not reach adequate sub zero temperatures or that another unknown effect of winter salvage logging is at play.

In addition to physically damaging regenerating plants and seed sources, salvage logging also greatly modifies the microenvironment of post-wildfire ecosystems. Depending on the intensity of the fire, the trees left standing after wildfire may have been scorched some distance up the stem or, if fire reached the crown, the whole tree may be scorched or partially consumed. In the study area, after salvage logging was complete the cut areas were left completely devoid of standing trees with a varying coverage of downed wood material (DWM) left behind. Removal of standing dead trees (snags) can greatly impact the microclimate (light, air and soil temperature, and soil moisture) in early successional, post-wildfire ecosystems. Forest microclimate influences many ecological processes and plays an important role in landscape scale ecosystem function and structure (Chen et al. 1999).

Although a hillside of standing dead trees may seem a bleak landscape there is a significant amount of shade, averaging 15% coverage in sites sampled (a value that would increase at low sun angles). Many shade tolerant species are dependent on low light regeneration niches and may decline or go extinct if there is a drastic change in light levels (Henry and Aarssen 1997, Grubb 1977). For example, large leaved forest species such as the lily, *Prosartes hookerii* were very uncommon or absent from WFSA and WFSESA sites. The lack of cool, partially shaded microsites with adequate moisture in areas where the canopy has been removed can greatly affect the germination success of shade tolerant species (Swank and Vose 1994, Purdon et al. 2004, Sexton 1998). Shaded microsites have improved early seedling survival on sites with high insolation (Childs and Flint 1987) and may be important in dry Douglas-fir forests where high summer temperatures can cause seedling mortality. These microsites are limited but may occur in the shade of tree boles or in soil pockets where roots are burnt out after fire and are very uncommon in salvaged areas (personal observation).

The high light environment of salvaged sites may actually improve productivity if there are enough of other resources to support the use of the extra light. Even after fire the boreal forest can have moderately thick organic layers (Kurulok and Macdonald 2007, Greene et al. 2006) and therefore may be able to take advantage of the increased light by using stored nutrients and water. The relatively thin and dry Douglas-fir forest floor (Isaac and Hopkins 1937) may be more susceptible to disturbance, especially after fire. In fact, soil horizons incorporating organics averaged only ~2cm in depth for WF sites and ~0.5 cm in salvaged areas indicating that surficial soil horizons are significantly transformed and that the Douglas-fir may indeed be nutrient limited.

Snags in unsalvaged stands may also affect the microenvironment in burnt areas by decreasing wind speed and in turn decreasing plant desiccation (Chen et al. 1992, 1995). Sexton (1998) found a nearly 10% reduction in soil moisture in salvaged areas two years after fire and suggested that these areas may also be subject to greater extremes in surface temperatures similar to results obtained from clear-cut forest studies. However, there is very little research comparing surface temperatures in wildfire and salvage logged areas. There is also very little research on the ecological role of moss cover after fire. Salvage logging disturbs surficial soil layers and in turn disrupts the ubiquitous post-fire moss cover. Recent research in high latitude environments suggests that moss removal results in higher soil temperatures and decomposition rates (Gornall et al. 2009).

The fact that tree seedling occurrence was not significantly affected by salvage suggests that any mechanical damage done to this species group may have been offset by seed rain during the salvage process or dispersal from nearby seed sources. However, approximately 10% of the tree seedlings in salvaged areas appeared chlorotic (yellow in color) or damaged and only future study will reveal their long term survival. Salvage logging and associated soil disturbance may affect many individual regenerating plants and change post-fire communities by favouring species that are adapted to frequent disturbance or can recolonize via long distance seed dispersal. Shade tolerant species often have slow rates of dispersal and may drop out of the community if suitable microsites do not exist and the native seedbank is exhausted. Exotic species, however, can move into disturbed areas along the network of roads needed for post-fire logging and invade when natural succession in perturbed.

Leaving occasional standing trees after timber harvest is a commonly used and recommended strategy for biodiversity management (Szaro 1995). It is likely that as the number of standing dead trees increases, there would be reduced damage to soil and regenerating vegetation, and greater amounts of shading. Both of these factors would promote survival of shade tolerant species and foster increased understory plant diversity (Halpern and Spies 1995). Unfortunately, while current forest practice regulations in BC do call for retention of wildlife tree patches after green tree clear-cutting, these regulations are rarely applied to clear-cutting after fire.

Along with standing dead trees, post-wildfire salvage logging removes future large diameter DWM which is an important resource for post-fire regeneration, wildlife and ecosystem heterogeneity (Bunnell et al. 2002, Grove and Meggs 2003, Stevens 1997). Harvested post-wildfire stands tend to have a higher cover of small diameter (< 7.5cm) DWM compared to natural post-wildfire stands (Song 2002, Brown 1980). Salvaged wildfire sites sampled had significantly higher percent cover of DWM which was observed to consist mostly of small diameter branches and tree tops, a feature that may inhibit seedling emergence by acting as a mechanical barrier as well as increase fire hazard ratings and reburn potential compared to non-salvaged stands (Brown et al. 2003, Donato et al. 2006, Thompson et al. 2007).

Small diameter DWM decays faster than larger DWM and also fails to provide the range and variation of substrates and microsites needed to support long term establishment of bryophytes, lichens and vascular plants that larger DWM does (Harmon et al. 1986, Song 1997). Post-fire Douglas-fir snags have an average half life of approximately 15–16 years in unlogged burnt dry forests and therefore add DWM, heterogeneity and complexity to the forest for many decades after fire (Russell et al. 2006). Salvage logging is a cumulative post-fire disturbance that not only reduces species richness in early successional Douglas-fir forests but may also reduces the potential for future plant diversity due to long lasting changes in microenvironment and surficial and standing woody substrates.

Finally, salvage logging may also affect the long-term productivity of Douglas-fir forests. Similar to this study, Stuart (1993) notes that salvaged sites had a lower frequency of shrubs, particularly the nitrogen-fixing *Ceanothus* species, which may result in decreased long-term fertility. The reduction of the nitrogen fixing shrub *Ceanothus sanguineus* in WFSA sites compared to WF sites indicates that the ability of resprouters to the regenerate in the WFSA environment may be hampered. Productivity losses of up to 14% have been associated with landscape scale mechanical harvesting disturbance in Douglas-fir forests (Grigal 2000 and refs. there in).

4.4 Combined Effects of Seeding and Salvage Logging

The combined effects of aerially seeding agronomic species and clear-cut harvesting post-wildfire areas resulted in substantial alteration of the natural post-wildfire environment. Large scale changes to soil characteristics, DWM, microenvironment and the magnitude and duration of disturbance along with the introduction of several rootzone altering, competitive species changed the species composition and likely the long term successional trajectory of the post-wildfire landscape.

Wildfire and post-fire management are part of a progression of potentially interacting disturbance pressures. Drought, wildfire, seeding, post-wildfire salvage logging, grazing, and climate change are particularly important for regeneration succession in dry Douglas-fir forests. Lindenmayer et al. (2008) point out in their salvage logging review that drought (1) and high temperatures are common pre-wildfire disturbances that negatively affect many plant species. Wildfire (2) has variable disturbance intensity and while ~40% of species were missing from WF sites compared to mature forest sites, overall species richness was comparable (Figure 8). In this case a released seedbank and influx from remnant or highly dispersive post-fire seed sources provides a large species pool. Wildfire also offers a high degree of environmental heterogeneity in the form of snags, DWM, microenvironment and surficial substrates that promote a wide range of life forms and general biodiversity. Mass seeding (3) of agronomic species introduces a new level of competition into early post-fire successional communities. The successful addition of highly competitive species to post-wildfire plant communities can shift community composition due to the fact that species with the highest fitness in a given environment will come to dominate that environment. Salvage logging (4) removes and/or alters many of the features that support heterogeneity and biological diversity after wildfire. Post-fire tree harvesting also adds another disturbance to the landscape and effectively increases the magnitude and duration of disturbance which tests the resilience of post-fire plant communities and reduces the overall species pool. In addition, grazing (5) may add another complex suite of disturbance effects that can have positive and/or negative effects on species richness and community composition (Harrison et al. 2003, Noy-Meir 1995, Keeley et al. 2003). Although grazing was not the part of the focus of this study it may be an important factor in order understand the long-term effects of the previous four disturbances. Finally, in the face of climate change (6) maintenance of the regional species pool is important for sustaining diversity over the long term due to the predicted expansion of the IDF forest type throughout British Columbia (Hamann and Wang 2006). Warmer annual temperatures, drier summer conditions and increased fire occurrence may affect ecosystem stability and the persistence of species groups that rely on the presence of natural early successional habitats.

Hypotheses on the effects of multiple disturbances propose that sequential disturbances are likely to have compounding ecological consequences and therefore go beyond the disturbance levels any single natural perturbation may incur (Paine et al. 1998, Peterson et al. 2009, Roberts 2004). If cumulative effects are large enough, dramatic changes in forest structure and community composition may result. The long term ability of the forest system to recover may then be reduced and result in major changes in ecosystem state. The multiple disturbance events in this study had clear impacts on floristic composition. Seeding and salvage logging events had individually discernable impacts but the combination of the two resulted in the largest change in floristic composition. The most notable change was in the richness and proportion of exotic species.

ANCOVA analysis (Figures 14 and 15) shows a general increase in exotic species richness for the WFSASE sites. This implies that multiple disturbances ‘raise the bar’ and promote greater diversity of exotics. There was a negative relationship found between native and exotic species richness at the $1m^2$ scale (Figure 15) while there was a positive relationship at the plot scale

(Figure 14). These trends are thought to arise because large landscapes with favourable conditions for native species also have favourable conditions for exotic species and cover greater spatial environmental heterogeneity (Davies et al. 2005). On the other hand, the effects of competition are more prevalent at smaller scales (Huston 1999) and therefore negative relationships are often found. It would appear that greater heterogeneity and species richness in WF sites protects these communities against high levels of invasion. The effects of salvage logging and grass seeding (with seed mix potentially contaminated by exotic species) however reduce the cover and diversity of native plant communities and favour the establishment of exotics. Natives can out-compete exotics due to their long-term presence in the seed bank but multiple disturbances can reduce this advantage.

Even though native species richness was not significantly lower in WFSESA sites the ratio of natives to exotic was greatly reduced (Figure 10 and 11), reflecting a decline in total diversity and a change in dominance of species groups as shown by the NMS lifeform ordination (Figure 6). Thus, the combined effects of seeding and salvage logging may be pushing the plant communities towards an alternate successional trajectory, shifting it from a woodland-forest matrix to a system dominated by weedy and heliophyllic species. This shift has also been documented in Colorado (USA) windthrow areas where salvaged logged sites exhibited a shift towards graminoid dominance (Rumbaitis del Rio 2006). Increased disturbance frequency after fire has also been shown to shift dominant vegetation from evading to enduring shrubs (Pausas 1999) or to exotic grasses and forbs (Keeley and Fotheringham 2000, Pausas 1999). Further experimental investigation into the competitive relationship between native, exotic and seeded species is required in order to fully understand the processes behind the net effects shown in the present study.

Historically, mature Douglas-fir forests had a low proportion of exotic species (see Figure 10) with the highest concentration of exotics likely clustered around sites of anthropogenic disturbance such as roads and agricultural fields. One way wildfire, seeding and salvage logging increase the proportion of exotic species is by reducing the distance and barriers between patches of exotics, resulting in an opportunity for exotic species to spread across the landscape if favourable sites exist. This scenario is described theoretically as the Percolation Theory (Gardner et al. 1987, With and Crist 1995) which states that at a certain threshold clusters of individuals will combine and create a connected network, making it possible for a species or genotype to move across the landscape using only sites favourable for its growth.

Long-term effects of multiple disturbances and changes in successional trajectory may have varied outcomes. Disturbance influences understory plant response by affecting the vegetation that is present at each stage of post-disturbance succession. If a shift in understory community lasts until the thinning or stem-exclusion stage of stand development (van der Maarel 1996, White and Jentsch 2001) there may be little residual understory vegetation available to recolonize following thinning or future disturbances. Mature forest growth stages can have a rich understory herbaceous layer but altered disturbance regimes and successional pathways may permanently effect vegetation states and weaken future forest resilience and biodiversity. The question of whether this a short term reversible effect or a permanent one can only be answered with further long term study. Theory on succession, however, would suggest that plant communities that are significantly altered from their historical range of variability will affect the composition and resilience of future successional stages. Whatever the long term effects, early successional stands should be managed for biodiversity and habitat in their own right, a failure to do so would negate the role these habitats play in providing heterogeneity and diversity across the landscape.

Chapter 5 Conclusions

The importance of non-salvaged, early successional, post-wildfire plant communities and related snag attributes as habitat for invertebrates, birds, ungulates and plants has been recognized (Drapeau et al. 2002, Ralph Heinrich 2007, Nappi et al. 2004, Cobb et al. 2007). With so little of the post-wildfire land base allowed to regenerate naturally, this habitat may be of greater conservation value than other successional stages including old growth (Lindenmayer et al. 2008). As Franklin (1988) states "conserving the biodiversity of temperate forests requires the maintenance of all forest successional stages". One of the most over looked of which in recent times is natural, early successional, post-wildfire habitat.

The most apparent and publicized effects of wildfires are the destruction of vegetation and property during fires and the increase in erosion, sedimentation, and flooding that can follow severe fires in steep terrain. However, understory plant communities are not destroyed by fire as often described (Stadt 2001) but instead renewed and the long term effects of community change and loss of biodiversity due to unmitigated post-wildfire management may be staggering.

This study highlights the importance of considering the effects of post-fire management practices on native plant communities and reveals the need for long-term monitoring of post-fire recovery. Salvage logging and seeding after fire have many implications for dry Douglas-fir forest management. The increase in exotics and reduced frequency of species such as *C. sanguineus* in salvaged sites may influence the seasonal browse quality for ungulates and long term forest productivity. A reduction in total species richness and a shift towards plant communities dominated by exotics and graminoids may alter the successional pathway of study area forests and change the species composition of future successional stages.

While there are many reasons to harvest forests following disturbances, there are equally numerous arguments for conserving post-wildfire environments with natural levels of snags and DWM. Where economics and human safety are deemed most important harvesting should be conducted in a careful manner minimizing disruption of ecosystem function and the introduction of exotics. However, there is little evidence suggesting that natural disturbances cause long term negative ecological consequences when they occur away from urban areas (Foster and Orwig 2006). Therefore, post-wildfire management regimes aiming to increase long-term forest health and productivity need to review whether the practice of seeding and salvage logging after fire is worth the potential ecosystem damage involved.

Tables

Table 1. Sampling summary for McLure and McGillivray fire areas, 2007.

Site Type	Quadrats	# Plots	# Stands
Wildfire	324	27	11
Seeded Wildfire	384	32	12
Salvage Logged Wildfire	264	22	9
Salvaged and Seeded Wildfire	276	23	10
Total	1248	104	42

Table 2. Species mixtures for seeding application in McClure and McGillivray post-fire areas showing proposed density and proportion of species for each seed mix.

Seed Mix (Density)	Common_Name	Latin Name	% By Weight
Erosion Control (10kg/ha),Weed Control (5kg/ha)	Italian Ryegrass	*Lolium perenne* L. ssp. *multiflorum* (Lam.) Husnot	32%
	Creeping Red Fescue	*Festuca rubra* L. ssp. *rubra*	5%
	Canada Bluegrass	*Poa compressa* L.	4%
	Timothy	*Phleum pratense* L.	4%
	Western Wheatgrass	*Pascopyrum smithii* (Rydb.) A. Löve	35%
	Rambler Alfalfa	*Medicago sativa* L. ssp. *sativa*	20%
Forage Management (3kg/ha)	Italian Ryegrass	*Lolium perenne* L. ssp. *multiflorum* (Lam.) Husnot	25%
	Hard Fescue	*Festuca brevipila* Tracey	5%
	Slender Wheatgrass	*Elymus trachycaulus* (Link) Gould ex Shinners ssp. *trachycaulus*	45%
	Orchardgrass	*Dactylis glomerata* L.	15%
	White Clover	*Trifolium repens* L.	10%
Wildlife Forage/Weed Control (5kg/ha)	Italian Ryegrass	*Lolium perenne* L. ssp. *multiflorum* (Lam.) Husnot	24%
	June Grass	*Koeleria macrantha* (Ledeb.) Schult.	2%
	Western Wheatgrass	*Pascopyrum smithii* (Rydb.) A. Löve	30%
	Bluebunch Wheatgrass	*Pseudoroegneria spicata* (Pursh) A. Löve ssp. *spicata*	20%
	Slender Wheatgrass	*Elymus trachycaulus* (Link) Gould ex Shinners ssp. *trachycaulus*	24%

Table 3. Proportion of plots occupied by individual species in each site type. Bold values indicate top 20 most common species for that site type (N = native, E = exotic, Agro = seeded agronomic species)

Species		Proportion by Site Type			
		Wildfire (n=27)	Seeded Wildfire (n=32)	Salvaged Wildfire (n=22)	Seeded Salvaged Wildfire (n=23)
Ceratodon purpureus	N	**1.00**	**1.00**	**1.00**	**1.00**
Chamerion angustifolium	N	**1.00**	**0.97**	**1.00**	**0.91**
Bryum caespiticium	N	**0.96**	**0.94**	**0.95**	**0.83**
Calamagrostis rubescens	N	**0.96**	**1.00**	**0.82**	**0.83**
Spiraea betulifolia	N	**0.93**	**1.00**	**0.86**	**0.91**
Taraxacum officinale	E	**0.89**	**0.72**	**0.95**	**0.83**
Mahonia aquifolium	N	**0.89**	**0.91**	**0.68**	**0.87**
Epilobium brachycarpum	N	**0.85**	**0.81**	**0.86**	**0.87**
Rosa acicularis	N	**0.81**	**0.59**	**0.82**	**0.43**
Carex rossii	N	**0.81**	**0.81**	**0.77**	**0.52**
Lactuca serriola	E	**0.78**	**0.88**	**0.86**	**0.91**
Aster conspicuus	N	**0.74**	**0.69**	**0.55**	**0.30**
Symphoricarpos albus	N	**0.70**	**0.88**	**0.50**	**0.70**
Arnica cordifolia	N	**0.70**	**0.63**	0.41	0.48
Conyza canadensis	N	**0.63**	0.41	**0.77**	**0.52**
Pseudotsuga menziesii	N	**0.59**	0.22	0.50	0.17
Ceanothus sanguineus	N	**0.59**	0.38	0.27	0.35
Salix scouleriana	N	**0.56**	0.31	**0.55**	**0.30**
Cirsium vulgare	E	0.52	0.34	0.50	0.52
Achillea millefolium	N	**0.52**	0.41	0.18	0.35
Amelanchier alnifolia	N	**0.52**	0.38	0.18	0.30
Polytrichum juniperinum	N	0.48	0.47	**0.50**	**0.26**
Paxistima myrsinites	N	0.48	0.13	0.50	0.00
Filago arvensis	E	0.44	0.34	**0.82**	**0.65**
Vicia americana	N	0.44	**0.59**	0.23	0.74
Betula papyrifera	N	0.41	0.13	0.32	0.00
Populus tremuloides	N	0.37	0.38	**0.50**	**0.22**
Shepherdia canadensis	N	0.37	0.03	0.32	0.04
Rubus parviflorus	N	0.37	0.06	0.18	0.04
Prosartes hookeri	N	0.37	0.13	0.05	0.00
Rubus idaeus	N	0.33	0.19	0.27	0.09
Hieracium sp.	N	0.33	0.13	0.23	0.09
Pohlia nutans	N	0.30	**0.56**	0.23	0.17
Poa pratensis	E	0.30	**0.59**	0.09	0.57
Collinsia parviflora	N	0.30	0.28	0.09	0.26
Astragalus miser	N	0.30	0.22	0.09	0.17
Erigeron speciosus	N	0.30	0.03	0.05	0.04
Tragopogon dubius	E	0.26	0.22	**0.59**	**0.22**
Fragaria virginiana	N	0.26	0.06	0.14	0.09
Funaria hygrometrica	N	0.26	0.09	0.00	0.00

Table 3. Proportion of plots occupied by individual species in each site type. Bold values indicate top 20 most common species for that site type (N = native, E = exotic, Agro = seeded agronomic species)

Species		Proportion by Site Type			
		Wildfire (n=27)	Seeded Wildfire (n=32)	Salvaged Wildfire (n=22)	Seeded Salvaged Wildfire (n=23)
Crepis tectorum	E	0.22	0.00	0.14	0.04
Arabis holboellii	N	0.22	0.13	0.09	0.17
Arenaria serpyllifolia	E	0.22	0.06	0.09	0.39
Pinus contorta	N	0.19	0.06	**0.68**	**0.13**
Anaphalis margaritacea	N	0.19	0.09	0.27	0.09
Carex concinnoides	N	0.19	0.16	0.23	0.22
Erythronium grandiflorum	N	0.19	0.13	0.18	0.00
Fritillaria affinis var. affinis	N	0.19	0.25	0.14	0.00
Calochortus macrocarpus	N	0.19	0.28	0.00	0.04
Crepis atribarba	N	0.19	0.06	0.00	0.00
Balsamorhiza sagittata	N	0.19	0.25	0.09	0.13
Bromus tectorum	E	0.19	0.03	0.09	0.22
Lathyrus ochroleucus	N	0.15	0.09	0.14	0.13
Campanula rotundifolia	N	0.15	0.16	0.00	0.17
Pseudoroegneria spicata	N	0.15	0.09	0.00	0.04
Aster occidentalis	N	0.15	0.03	0.00	0.09
Elymus glaucus	N	0.15	0.03	0.00	0.04
Acer glabrum	N	0.15	0.09	0.09	0.04
Fragaria vesca	N	0.15	0.09	0.09	0.04
Lilium columbianum	N	0.15	0.06	0.09	0.00
Allium cernuum	N	0.11	0.09	0.09	0.04
Bromus japonicus	E	0.11	0.09	0.00	0.13
Marchantia polymorpha	N	0.11	0.06	0.00	0.04
Apocynum androsaemifolium	N	0.11	0.03	0.00	0.00
Unknown herb sp.	N	0.11	0.00	0.00	0.00
Hieracium scouleri var. albertinum	N	0.11	0.03	0.09	0.00
Lomatium dissectum	N	0.11	0.13	0.05	0.00
Zigadenus venenosus	N	0.11	0.13	0.05	0.00
Antennaria neglecta	N	0.11	0.00	0.05	0.09
Maianthemum racemosum	N	0.11	0.00	0.05	0.00
Poa compressa (Agro)	E	0.07	0.34	0.18	0.57
Cirsium sp.	N	0.07	0.25	0.14	0.30
Collomia linearis	N	0.07	0.03	0.09	0.04
Festuca idahoensis	N	0.07	0.44	0.00	0.17
Polygonum douglasii ssp. douglasii	N	0.07	0.16	0.00	0.13
Lithospermum ruderale	N	0.07	0.03	0.00	0.00
Rosa gymnocarpa	N	0.07	0.03	0.00	0.00
Arabis sp.	N	0.07	0.00	0.00	0.04
Castilleja hispida	N	0.07	0.00	0.00	0.00
Festuca campestris	N	0.07	0.00	0.00	0.00

Table 3. Proportion of plots occupied by individual species in each site type. Bold values indicate top 20 most common species for that site type (N = native, E = exotic, Agro = seeded agronomic species)

Species		Proportion by Site Type			
		Wildfire (n=27)	Seeded Wildfire (n=32)	Salvaged Wildfire (n=22)	Seeded Salvaged Wildfire (n=23)
Iliamna rivularis var. rivularis	N	0.07	0.00	0.00	0.00
Linnaea borealis	N	0.07	0.00	0.00	0.00
Oxytropis sericea	N	0.07	0.00	0.00	0.00
Tortula ruralis	N	0.07	0.00	0.00	0.00
Bromus inermis ssp. inermis	E	0.07	0.38	0.05	0.30
Peltigera malacea	N	0.07	0.03	0.05	0.00
Hieracium umbellatum	N	0.07	0.00	0.05	0.00
Ribes viscosissimum	N	0.07	0.00	0.05	0.00
Aralia nudicaulis	N	0.04	0.00	0.14	0.00
Verbascum thapsus	E	0.04	0.00	0.14	0.35
Festuca brevipila (Agro)	E	0.04	**0.59**	0.09	0.78
Koeleria macrantha	N	0.04	0.06	0.09	0.09
Sisymbrium loeselii	E	0.04	0.06	0.09	0.57
Hieracium albiflorum	N	0.04	0.00	0.09	0.00
Dactylis glomerata (Agro)	E	0.04	0.25	0.00	0.30
Festuca occidentalis	N	0.04	0.13	0.00	0.00
Epilobium ciliatum	N	0.04	0.09	0.00	0.04
Polytrichum piliferum	N	0.04	0.09	0.00	0.00
Carex concinna	N	0.04	0.06	0.00	0.00
Penstemon fruticosus	N	0.04	0.06	0.00	0.00
Arctostaphylos uva-ursi	N	0.04	0.03	0.00	0.00
Crepis occidentalis	N	0.04	0.03	0.00	0.00
Deschampsia cespitosa	N	0.04	0.03	0.00	0.00
Elymus x albicans	N	0.04	0.03	0.00	0.09
Galium boreale	N	0.04	0.03	0.00	0.04
Lithophragma parviflorum	N	0.04	0.03	0.00	0.00
Potentilla gracilis var. fastigiata	N	0.04	0.03	0.00	0.00
Adenocaulon bicolor	N	0.04	0.00	0.00	0.00
Antennaria racemosa	N	0.04	0.00	0.00	0.00
Aster occidentalis var. intermedius	N	0.04	0.00	0.00	0.00
Chimaphila umbellata	N	0.04	0.00	0.00	0.00
Erigeron divergens	N	0.04	0.00	0.00	0.00
Galium trifidum	N	0.04	0.00	0.00	0.00
Leucanthemum vulgare	E	0.04	0.00	0.00	0.00
Lonicera utahensis	N	0.04	0.00	0.00	0.00
Osmorhiza berteroi	N	0.04	0.00	0.00	0.04
Peltigera membranacea	N	0.04	0.00	0.00	0.00
Peltigera ponojensis	N	0.04	0.00	0.00	0.00
Peltigera rufescens	N	0.04	0.00	0.00	0.04
Phlox longifolia	N	0.04	0.00	0.00	0.00

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Species		Proportion by Site Type			
		Wildfire (n=27)	Seeded Wildfire (n=32)	Salvaged Wildfire (n=22)	Seeded Salvaged Wildfire (n=23)
Piperia unalascensis	N	0.04	0.00	0.00	0.00
Poa fendleriana	N	0.04	0.00	0.00	0.00
Poa palustris	N	0.04	0.00	0.00	0.04
Prunus pensylvanica	N	0.04	0.00	0.00	0.00
Ranunculus uncinatus	N	0.04	0.00	0.00	0.00
Ribes hudsonianum	N	0.04	0.00	0.00	0.00
Silene antirrhina	N	0.04	0.00	0.00	0.00
Sisymbrium altissimum	E	0.04	0.00	0.00	0.17
Soil Crust	N	0.04	0.00	0.00	0.00
Unknown grass		0.04	0.00	0.00	0.00
Vaccinium membranaceum	N	0.04	0.00	0.00	0.00
Elymus glaucus ssp. glaucus	N	0.04	0.03	0.05	0.00
Ribes lacustre	N	0.04	0.00	0.05	0.00
Calamagrostis canadensis	N	0.00	0.13	0.14	0.00
Pseudotsuga menzisii (Planted)		0.00	0.00	0.14	0.17
Sonchus asper	E	0.00	0.00	0.14	0.00
Phleum pratense (Agro)	E	0.00	**0.69**	0.09	0.61
Elymus trachycaulus (Agro)	E	0.00	0.31	0.09	0.70
Aster sp.	N	0.00	0.00	0.09	0.04
Descurainia sp.	N	0.00	0.00	0.09	0.00
Hieracium scouleri	N	0.00	0.03	0.05	0.00
Agoseris glauca	N	0.00	0.00	0.05	0.00
Deschampsia elongata	N	0.00	0.00	0.05	0.04
Erigeron corymbosus	N	0.00	0.00	0.05	0.00
Hordeum jubatum	N	0.00	0.00	0.05	0.13
Medicago lupulina	E	0.00	0.00	0.05	0.04
Phlox gracilis	N	0.00	0.00	0.05	0.00
Phlox gracilis ssp. gracilis	N	0.00	0.00	0.05	0.00
Pinus contorta (Planted)		0.00	0.00	0.05	0.00
Populus balsamifera	N	0.00	0.00	0.05	0.00
Rubus leucodermis	N	0.00	0.00	0.05	0.13
Sonchus arvensis	E	0.00	0.00	0.05	0.00
Medicago sativa (Agro)	E	0.00	**0.59**	0.00	0.65
Lolium multiflorum (Agro)	E	0.00	0.16	0.00	0.43
Pascopyrum smithii	N	0.00	0.16	0.00	0.30
Antennaria microphylla	N	0.00	0.13	0.00	0.04
Peltigera didactyla	N	0.00	0.13	0.00	0.00
Astragalus collinus	N	0.00	0.06	0.00	0.00
Festuca rubra (Agro)	E	0.00	0.06	0.00	0.26
Agropyron cristatum	E	0.00	0.03	0.00	0.09

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Species		Proportion by Site Type			
		Wildfire (n=27)	Seeded Wildfire (n=32)	Salvaged Wildfire (n=22)	Seeded Salvaged Wildfire (n=23)
Anaphalis sp.	N	0.00	0.03	0.00	0.00
Antennaria sp.	N	0.00	0.03	0.00	0.00
Cerastium nutans	N	0.00	0.03	0.00	0.00
Delphinium nuttallianum	N	0.00	0.03	0.00	0.00
Descurainia pinnata	N	0.00	0.03	0.00	0.00
Elymus repens	E	0.00	0.03	0.00	0.00
Elymus trachycaulus ssp. subsecundus	N	0.00	0.03	0.00	0.00
Epilobium sp.	N	0.00	0.03	0.00	0.17
Ericameria nauseosus	N	0.00	0.03	0.00	0.00
Erigeron pumilus	N	0.00	0.03	0.00	0.00
Lathyrus nevadensis	N	0.00	0.03	0.00	0.00
Lathyrus sp.	N	0.00	0.03	0.00	0.00
Lilium sp.	N	0.00	0.03	0.00	0.00
Melilotus officinalis	E	0.00	0.03	0.00	0.00
Phalaris arundinacea	N	0.00	0.03	0.00	0.00
Prosartes trachycarpa	N	0.00	0.03	0.00	0.00
Prunus virginiana	N	0.00	0.03	0.00	0.00
Ribes sp.	N	0.00	0.03	0.00	0.00
Aster ciliolatus	N	0.00	0.00	0.00	0.04
Berteroa incana	E	0.00	0.00	0.00	0.09
Centaurea biebersteinii	E	0.00	0.00	0.00	0.04
Cerastium arvense	N	0.00	0.00	0.00	0.04
Deschampsia cespitosa ssp. cespitosa	N	0.00	0.00	0.00	0.04
Descurainia sophia	E	0.00	0.00	0.00	0.04
Festuca saximontana	N	0.00	0.00	0.00	0.04
Matricaria discoidea	E	0.00	0.00	0.00	0.09
Myosotis stricta	E	0.00	0.00	0.00	0.04
Peltigera sp.	N	0.00	0.00	0.00	0.04
Plantago sp.	N	0.00	0.00	0.00	0.04
Potentilla recta	E	0.00	0.00	0.00	0.13
Trifolium pratense	E	0.00	0.00	0.00	0.04
Veronica sp.	N	0.00	0.00	0.00	0.04

Table 4. Summary of understory species attributes and trait groups (1 = coniferous tree, 2 = broad-leaved tree, 3 = evergreen shrub, 4 = deciduous shrub, 6 = graminoid, 7 = forb, 9 = moss, 10 = hepatic, 11 = lichen, 12 = dwarf woody plant, A = annual, B = biennial, P = perennial, I = introduced, Iv = invasive, Av = avoider - fire sensitive species, In = invader - highly dispersive, ruderal species, Ev = evader - long lived species with propagules stored in the soil or canopy and germinate post-fire, En = endurer - sprouts from above or below ground structures post-fire, C = common >33%, O = occasional 10-33%, U = uncommon <10% of plots).

Family	Species	Life Form	Life History	Weed Type	Native/ Exotic	Agronomic	Fire Response	Occurrence (104 plots)
Aceraceae	*Acer glabrum*	4	P		N		En	U (10)
Apiaceae	*Lomatium dissectum*	7	P		N		Av/En	U (8)
	Osmorhiza berteroi	7	P		N		Ev/En	U (2)
Apocynaceae	*Apocynum androsaemifolium*	7	P		N		En	U (4)
Araliaceae	*Aralia nudicaulis*	7	P		N		Av	U (4)
Asteraceae	*Achillea millefolium*	7	P		N		In	C (39)
	Adenocaulon bicolor	7	P		N		En	U (1)
	Agoseris glauca	7	P		N		Ev	U (1)
	Anaphalis margaritacea	7	P		N		In/En	O (16)
	Anaphalis sp.	7	P		N		In/En	U (1)
	Antennaria microphylla	7	P		N		Av	U (5)
	Antennaria neglecta	7	P		N		Av	U (6)
	Antennaria racemosa	7	P		N		Av	U (1)
	Antennaria sp.	7	P		N		Av	U (1)
	Arnica cordifolia	7	P		N		Av/Ev	C (59)
	Aster ciliolatus	7	P		N		Av	U (1)
	Aster conspicuus	7	P		N		En	C (61)
	Aster occidentalis	7	P		N		Av	U (7)
	Aster occidentalis var. intermedius	7	P		N		Av	U (1)
	Aster sp.	7	P		N		Av	U (3)
	Balsamorhiza sagittata	7	P		N		En	O (18)
	Centaurea biebersteinii	7	B/P	Iv	E		In	U (1)
	Cirsium sp.	7	A		N		In	O (20)
	Cirsium vulgare	7	A	Iv	E		In/En	C (48)
	Conyza canadensis	7	A		N		In	C (59)
	Crepis atribarba	7	P		N		In	U (7)
	Crepis occidentalis	7	A/P		N		In	U (2)
	Crepis tectorum	7	A	waif	E		In	U (10)
	Ericameria nauseosus	4	P		N		Av	U (1)

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Family	Species	Life Form	Life History	Weed Type	Native/ Exotic	Agronomic	Fire Response	Occurrence (104 plots)
	Erigeron corymbosus	7	P		N		Av	U (1)
	Erigeron divergens	7	P		N		Av	U (1)
	Erigeron pumilus	7	P		N		Av	U (1)
	Erigeron speciosus	7	P		N		Av	U (11)
	Filago arvensis	7	A	I	E		In	C (56)
	Hieracium albiflorum	7	P		N		Ev	U (3)
	Hieracium scouleri	7	P		N		Ev	U (2)
	Hieracium scouleri var. albertinum	7	P		N		Ev	U (6)
	Hieracium sp.	7	P		N		Ev	O (20)
	Hieracium umbellatum	7	P		N		Ev	U (3)
	Lactuca serriola	7	A	I	E		In	C (89)
	Leucanthemum vulgare	7	P	Iv	E		Av	U (1)
	Matricaria discoidea	7	A	Iv	E		Av	U (2)
	Sonchus arvensis	7	P	Iv	E		In	U (1)
	Sonchus asper	7	P	Iv	E		In	U (3)
	Taraxacum officinale	7	A	I/N	E		In	C (87)
	Tragopogon dubius	7	A/B	Iv	E		In	O (32)
Berberidaceae	*Mahonia aquifolium*	3	P		N		En	C (88)
Betulaceae	*Betula papyrifera*	2	P		N		Av	O (22)
Boraginaceae	*Lithospermum ruderale*	7	P		N		Av	U (3)
	Myosotis stricta	7	A	I	E		Av	U (1)
Brassicaceae	*Arabis holboellii*	7	B/P		N		Av	O (16)
	Arabis sp.	7	B/P		N		Av	U (3)
	Berteroa incana	7	A/B/P	Iv	E		Av	U (2)
	Descurainia pinnata	7	A/B/P		N		In	U (1)
	Descurainia sophia	7	A/B	Iv	E		In	U (1)
	Descurainia sp.	7	A/B/P		N		In	U (2)
	Sisymbrium altissimum	7	A/B	I	E		In	U (5)
	Sisymbrium loeselii	7	A/B	I	E		In	O (18)

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Family	Species	Life Form	Life History	Weed Type	Native/ Exotic	Agronomic	Fire Response	Occurrence (104 plots)
Bryaceae	*Bryum caespiticium*	9	P		N		In	C (96)
	Pohlia nutans	9	P		N		In	C (35)
Campanulaceae	*Campanula rotundifolia*	7	P		N		Ev/En	U (13)
Caprifoliaceae	*Linnaea borealis*	12	P		N		Ev/En	U (2)
	Lonicera utahensis	4	P		N		Av	U (1)
	Symphoricarpos albus	4	P		N		En	C (74)
Carophyllaceae	*Arenaria serpyllifolia*	7	A	I	E		Av	O (19)
	Cerastium arvense	7	P		N		In	U (1)
	Cerastium nutans	7	A/P		N		Av	U (1)
	Silene antirrhina	7	A		N		Av	U (1)
Celastraceae	*Paxistima myrsinites*	3	P		N		Av	O (28)
Cyperaceae	*Carex concinna*	6	P		N		Ev	U (3)
	Carex concinnoides	6	P		N		Ev	O (20)
	Carex rossii	6	P		N		En	C (77)
Ditrichaceae	*Ceratodon purpureus*	9	P		N		In	C (104)
Elaeagnaceae	*Shepherdia canadensis*	4	P		N		En	O (19)
Ericaceae	*Arctostaphylos uva-ursi*	12	P		N		In/En	U (2)
	Vaccinium membranaceum	4	P		N		Av	U (1)
Fabaceae	*Astragalus collinus*	7	P		N		Av	U (2)
	Astragalus miser	7	P		N		Av	O (21)
	Lathyrus nevadensis	7	P		N		In/Ev	U (1)
	Lathyrus ochroleucus	7	P		N		In/Ev	U (13)
	Lathyrus sp.	7	P		N		In/Ev	U (1)
	Medicago lupulina	7	A/P	I	E		Av	U (2)
	Medicago sativa	7	A/P	I	E	Yes	Av	O (34)
	Melilotus officinalis	7	A/B/P	I	E		Av	U (1)
	Oxytropis sericea	7	P		N		Av	U (2)
	Trifolium pratense	7	B/P	I	E		Av	U (1)
	Vicia americana	7	P		N		In/Ev	C (53)

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Family	Species	Life Form	Life History	Weed Type	Native/ Exotic	Agronomic	Fire Response	Occurrence (104 plots)
Funariaceae	*Funaria hygrometrica*	9	P		N		In	U (10)
Grossulariaceae	*Ribes hudsonianum*	4	P		N		Ev/En	U (1)
	Ribes lacustre	4	P		N		Ev/En	U (2)
	Ribes sp.	4	P		N		Av	U (1)
	Ribes viscosissimum	4	P		N		Ev/En	U (3)
Liliaceae	*Allium cernuum*	7	P		N		En	U (9)
	Calochortus macrocarpus	7	P		N		En	O (15)
	Erythronium grandiflorum	7	P		N		En	U (13)
	Fritillaria affinis var. affinis	7	P		N		En	O (16)
	Lilium columbianum	7	P		N		En	U (8)
	Lilium sp.	7	P		N		En	U (1)
	Maianthemum racemosum	7	P		N		In/En	U (4)
	Prosartes hookeri	7	P		N		En	O (15)
	Prosartes trachycarpa	7	P		N		En	U (1)
	Zigadenus venenosus	7	P		N		En	U (8)
Malvaceae	*Iliamna rivularis var. rivularis*	7	P		N		Av	U (2)
Marchantiaceae	*Marchantia polymorpha*	10	P		N		In	U (6)
Onagraceae	*Chamerion angustifolium*	7	P		N		In/En	C (101)
	Epilobium brachycarpum	7	A		N		In	C (88)
	Epilobium ciliatum	7	P		N		In	U (5)
	Epilobium sp.	7	P		N		In	U (5)
Orchidaceae	*Piperia unalascensis*	7	P		N		En	U (1)
Peltigeraceae	*Peltigera didactyla*	11	P		N		Av	U (4)
	Peltigera malacea	11	P		N		Av	U (4)
	Peltigera membranacea	11	P		N		Av	U (1)
	Peltigera ponojensis	11	P		N		Av	U (1)
	Peltigera rufescens	11	P		N		Av	U (2)
	Peltigera sp.	11	P		N		Av	U (1)
Pinaceae	*Pinus contorta*	1	P		N		In/En	O (25)

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Family	Species	Life Form	Life History	Weed Type	Native/ Exotic	Agronomic	Fire Response	Occurrence (104 plots)
	Pinus contorta (Planted)		P				N/A	U (1)
	Pseudotsuga menziesii	1	P		N		In/En	O (34)
	Pseudotsuga menzisii (Planted)		P				N/A	U (7)
Plantaginaceae	*Plantago sp.*	7	P		N		In	U (1)
Poaceae	*Agropyron cristatum*	6	P	I	E	yes	Av	U (3)
	Bromus inermis spp. inermis	6	P	Iv	E		Av	O (22)
	Bromus japonicus	6	A	I	E		In	U (9)
	Bromus tectorum	6	A	Iv	E		In	U (13)
	Calamagrostis canadensis	6	P		N		In/En	U (7)
	Calamagrostis rubescens	6	P		N		En	C (95)
	Dactylis glomerata	6	P	I	E	Yes	Av	O (16)
	Deschampsia cespitosa	6	P		N		En	U (2)
	Deschampsia cespitosa ssp. cespitosa	6	P		N		En	U (1)
	Deschampsia elongata	6	P		N		En	U (2)
	Elymus glaucus	6	P		N		Av	U (6)
	Elymus glaucus ssp. glaucus	6	P		N		Av	U (3)
	Elymus repens	6	P	Iv	E		Av	U (1)
	Elymus trachycaulus	6	P	I	E	Yes	Av	O (28)
	Elymus trachycaulus ssp. subsecundus	6	P		N		Av	U (1)
	Elymus x albicans	6	P		N		Av	U (4)
	Festuca campestris	6	P		N		In/En	U (2)
	Festuca idahoensis	6	P		N		In/En	O (20)
	Festuca occidentalis	6	P		N		In/En	U (5)
	Festuca rubra	6	P	I/N	E	Yes	In/En	U (8)
	Festuca saximontana	6	P		N		In/En	U (1)
	Festuca brevipila	6	P	I	E	Yes	Av	C (40)
	Hordeum jubatum	6	P		N		In	U (4)
	Koeleria macrantha	6	P		N	Yes	Av	U (7)
	Lolium multiflorum	6	A/B	I	E	Yes	Av	O (15)

Table 4. Summary of understory species attributes and trait groups (1 = coniferous tree, 2 = broad-leaved tree, 3 = evergreen shrub, 4 = deciduous shrub, 6 = graminoid, 7 = forb, 9 = moss, 10 = hepatic, 11 = lichen, 12 = dwarf woody plant, A = annual, B = biennial, P = perennial, I = introduced, Iv = invasive, Av = avoider - fire sensitive species, In = invader - highly dispersive, ruderal species, Ev = evader - long lived species with propagules stored in the soil or canopy and germinate post-fire, En = endurer - sprouts from above or below ground structures post-fire, C = common >33%, O = occasional 10-33%, U = uncommon <10% of plots).

Family	Species	Life Form	Life History	Weed Type	Native/ Exotic	Agronomic	Fire Response	Occurrence (104 plots)
	Pascopyrum smithii	6	P		N	Yes	Av/En	U (12)
	Phalaris arundinacea	6	P		N		Av	U (1)
	Phleum pratense	6	P	I	E	Yes	Av	C (38)
	Poa compressa	6	P	Iv	E	yes	Av/En	O (30)
	Poa fendleriana	6	P		N		Av/En	U (1)
	Poa palustris	6	P		N		Av/En	U (2)
	Poa pratensis	6	P	Iv	E	yes	In	C (42)
	Pseudoroegneria spicata	6	P		N		Av/En	U (8)
	Unknown Grass	6	?				?	U (1)
Polemoniaceae	*Collomia linearis*	7	A		N		In	U (6)
	Phlox gracilis	7	P		N		Ev/En	U (1)
	Phlox gracilis ssp. gracilis	7	P		N		Ev/En	U (1)
	Phlox longifolia	7	P		N		Ev/En	U (1)
Polygonaceae	*Polygonum douglasii ssp. douglasii*	7	A		N		Av	U (10)
Polytrichaceae	*Polytrichum juniperinum*	9	P		N		In	C (45)
	Polytrichum piliferum	9	P		N		In	U (4)
Pottiaceae	*Tortula ruralis*	9	P		N		In	U (2)
Pyrolaceae	*Chimaphila umbellata*	12	P		N		Av/En	U (1)
Ranunculaceae	*Delphinium nuttallianum*	7	P		N		Av	U (1)
	Ranunculus uncinatus	7	A/P		N		Av	U (1)
Rhamnaceae	*Ceanothus sanguineus*	4	P		N		Ev/En	C (42)
Rosaceae	*Amelanchier alnifolia*	4	P		N		En	C (37)
	Fragaria vesca	7	P		N		En	U (10)
	Fragaria virginiana	7	P		N		En	U (14)
	Potentilla gracilis var. fastigiata	7	P		N		Av	U (2)
	Potentilla recta	7	P	Iv	E		Av/In	U (3)
	Prunus pensylvanica	2	P		N		Ev/En	U (1)
	Prunus virginiana	2	P		N		Ev/En	U (1)
	Rosa acicularis	4	P		N		En	C (69)

Table 4. Summary of understory species attributes and trait groups (1 = coniferous tree, 2 = broad-leaved tree, 3 = evergreen shrub, 4 = deciduous shrub, 6 = graminoid, 7 = forb, 9 = moss, 10 = hepatic, 11 = lichen, 12 = dwarf woody plant, A = annual, B = biennial, P = perennial, I = introduced, Iv = invasive, Av = avoider - fire sensitive species, In = invader - highly dispersive, ruderal species, Ev = evader - long lived species with propagules stored in the soil or canopy and germinate post-fire, En = endurer - sprouts from above or below ground structures post-fire, C = common >33%, O = occasional 10-33%, U = uncommon <10% of plots).

Family	Species	Life Form	Life History	Weed Type	Native/ Exotic	Agronomic	Fire Response	Occurrence (104 plots)
	Rosa gymnocarpa	4	P		N		En	U (3)
	Rubus idaeus	4	P		N		In/Ev/En	O (23)
	Rubus leucodermis	4	P		N		In/Ev/En	U (4)
	Rubus parviflorus	4	P		N		In/Ev/En	O (17)
	Spiraea betulifolia	4	P		N		En	C (97)
Rubiaceae	*Galium boreale*	7	P		N		In	U (3)
	Galium trifidum	7	P		N		In	U (1)
Salicaceae	*Populus balsamifera*	2	P		N		Av	U (1)
	Populus tremuloides	2	P		N		In/En	C (38)
	Salix scouleriana	2	P		N		En	C (44)
Saxifragraceae	*Lithophragma parviflorum*	7	P		N		Av	U (2)
Scrophulariaceae	*Castilleja hispida*	7	P		N		Av	U (2)
	Collinsia parviflora	7	A		N		Av	O (25)
	Penstemon fruticosus	7	P		N		Av	U (3)
	Verbascum thapsus	7	B	Iv	E		In	U (12)
	Veronica sp.	7	P		N		In	U (1)
?	Soil Crust	11	?		N		Av	U (1)
?	Unknown Herb		?		N		?	U (3)

Table 5. Results of multi-response permutation procedures (MRPP) to detect differences in total species composition among treatments.

Factor	Treatments Compared	T	A	p-value[+]	
Site Type	All Treatments	-7.85	0.067	< 0.001	*
Site Type	Seeded Wildfire vs. Wildfire	-2.42	0.021	0.128	
Site Type	Seeded Wildfire vs. Salvage Wildfire	-5.66	0.051	< 0.001	*
Site Type	Seeded Wildfire vs. Salvage Seeded Wildfire	-4.78	0.045	0.005	*
Site Type	Wildfire vs. Salvage Wildfire	-3.89	0.041	0.015	*
Site Type	Wildfire vs. Salvage Seeded Wildfire	-6.86	0.082	0.001	*
Site Type	Salvage Wildfire vs. Salvage Seeded Wildfire	-3.76	0.040	0.016	*
Site Type	Salvage vs. Non-salvage	-7.94	0.038	< 0.001	*
Site Type	Seeded vs. Non-seeded	-5.24	0.025	0.001	*
Fire Severity	Low, Medium and High Severity	-0.83	0.006	0.382	

[+]Bonferroni adjusted significance, * = significant value (<0.05)

note: no significant factor interactions found

Table 6. Indicator species for each site type (treatment). Only species with an indicator value greater than 20 and a p-value less than 0.05 are shown (A = seeded species, E = exotic species, Iv = Invasive species).

Primary Disturbance	Secondary Disturbance	Multiple Secondary Disturbances
	Seeded Wildfire	**Seeded Salvaged Wildfire**
Wilfire	*Phleum pratensis* (A)	*Arenaria serpyllifolia* (E)
Amelanchier alnifolia	*Bromus inermis* (Iv)	*Dactylis glomerata* (A)
Arnica cordifolia	*Festuca idahoensis*	*Elymus trachycaulus* (A)
Aster conspicuus	*Poa pratensis* (Iv)	*Festuca rubra* (A)
Bryum caespiticium	*Pohlia nutans*	*Festuca trachyphylla* (A)
Calamagrostis rubescens		*Lactuca serriola* (E)
Ceratodon purpureus		*Lolium multiflorum* (A)
Erigeron speciosus	**Salvaged Wildfire**	*Medicago sativa* (A)
Funaria hygrometrica	*Conyza canadensis*	*Pascopyrum smithii* (A)
Prosartes hookeri	*Epilobium angustifolium*	*Poa compressa* (Iv)
Rosa acicularis	*Filago arvensis* (E)	*Sisymbrium loeselii* (E)
Rubus parviflorus	*Paxistima myrsinites*	*Verbascum thapsus* (Iv)
	Tragopogon dubius (Iv)	*Vicia americana*

Table 7. Summary of select biotic and abiotic characteristics (+ SD) of the McLure and McGillivray fires and the four site types by plot. (WF = wildfire only, WFSE = seeded wildfire, WFSA = salvage logged wildfire, WFSESA = seeded and salvaged wildfire, All SA = all salvage logged sites, All SE = all grass seeded sites)

	McLure	McGillivray	WF	WFSE	WFSA	WFSESA	All SA	All SE
Mean Slope (%)	35.4 (9.9)	32.4 (8.8)	39.9 (9.9)	36.4 (9.2)	30.5 (8.4)	27.47 (7.9)	29.95 (8.1)	33.5 (9.2)
Mean Elevation (m)	821 (141)	908 (111)	859 (95)	835 (140)	938 (115)	755 (147)	845 (160)	802 (147)
Mean Aspect (degrees)	207 (37.9)	188 (30)	183 (35.2)	210 (35.4)	204 (30.6)	211 (40)	207 (35.5)	210 (37.0)
Mean % Downed Woody Material	5.9 (6.6)	7.9 (9.1)	3.1 (2.8)	2.21 (2.0)	13.13 (8.8)	10.17 (8.3)	11.62 (8.6)	5.5 (6.8)
Mean % Substrate Rocks	9.5 (10.3)	3.2 (3.7)	6.8 (7.3)	8.3 (12.0)	9 (10.7)	6.7 (5.6)	7.9 (8.5)	7.7 (9.8)
Mean % Substrate Mineral Soil	64.9 (13.3)	71.8 (8.4)	69.8 (14.2)	66.6 (12.7)	64.8 (13.1)	65.6 (9.1)	65.2 (11.1)	66.2 (11.2)
Mean % Substrate Organics	18.8 (12.8)	16.7 (4.6)	20.1 (14.7)	21.31 (11.2)	12.04 (4.3)	17.73 (8.9)	14.8 (7.5)	19.7 (10.4)
Total Number of Species	149	116	130	121	90	103	123	27.2
Mean Understory Species Richness (S)	26.3 (5.3)	27.8 (6.8)	28.8 (4.8)	27.3 (5.0)	22.5 (6.4)	26.9 (5.5)	24.7 (6.3)	27.2 (5.2)
Mean Shannon's Diversity (H')	2.91 (0.21)	2.92 (0.31)	2.97 (0.18)	2.96 (0.17)	2.72 (0.31)	2.94 (0.23)	2.84 (0.29)	2.95 (0.20)
Mean Evenness	0.896 (0.02)	0.886 (0.02)	0.887 (0.02)	0.899 (0.02)	0.884 (0.03)	0.899 (0.02)	0.892 (0.02)	0.899 (0.02)
Simpson's index (D)	0.93 (0.02)	0.93 (0.03)	0.93 (0.01)	0.93 (0.01)	0.91 (0.03)	0.93 (0.02)	0.92 (0.02)	0.93 (0.01)
Beta Diversity (Bw)	121.7	87.2	100.2	92.7	66.5	75.1	97.3	109.8
Coefficient of Variation (%)	17.5	19.53	12.54	13.16	13.71	19.85	20.33	16.12
Total Among Plot Variance	310.12	356.72	193.33	183.74	123.24	387.70	337.07	267.06

Figures

Figure 1. Map of the two study areas showing plot locations in the McLure (A) and McGillivray (B) fires and the approximate area covered by the 2004 and 2005 aerial seeding efforts.

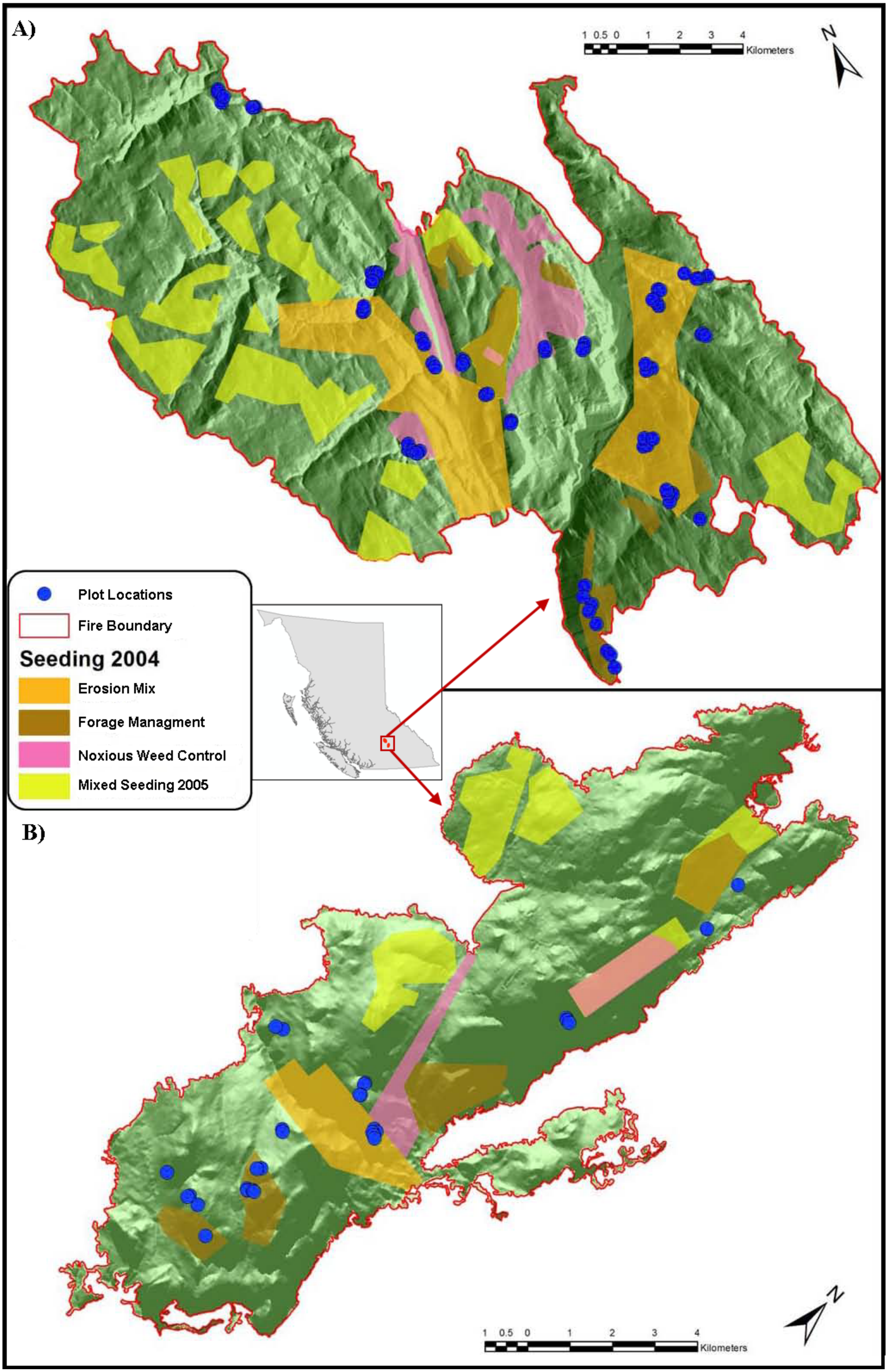


Figure 2. Layout of understory sampling for all sampled site types. Each GIS selected stand had up to three 400 m^2 plots. Plots contained 12 randomly selected 1 m^2 vegetation quadrats, 6 soil sample sites and one 50 m^2 silviculture plot recording tree seedling data.

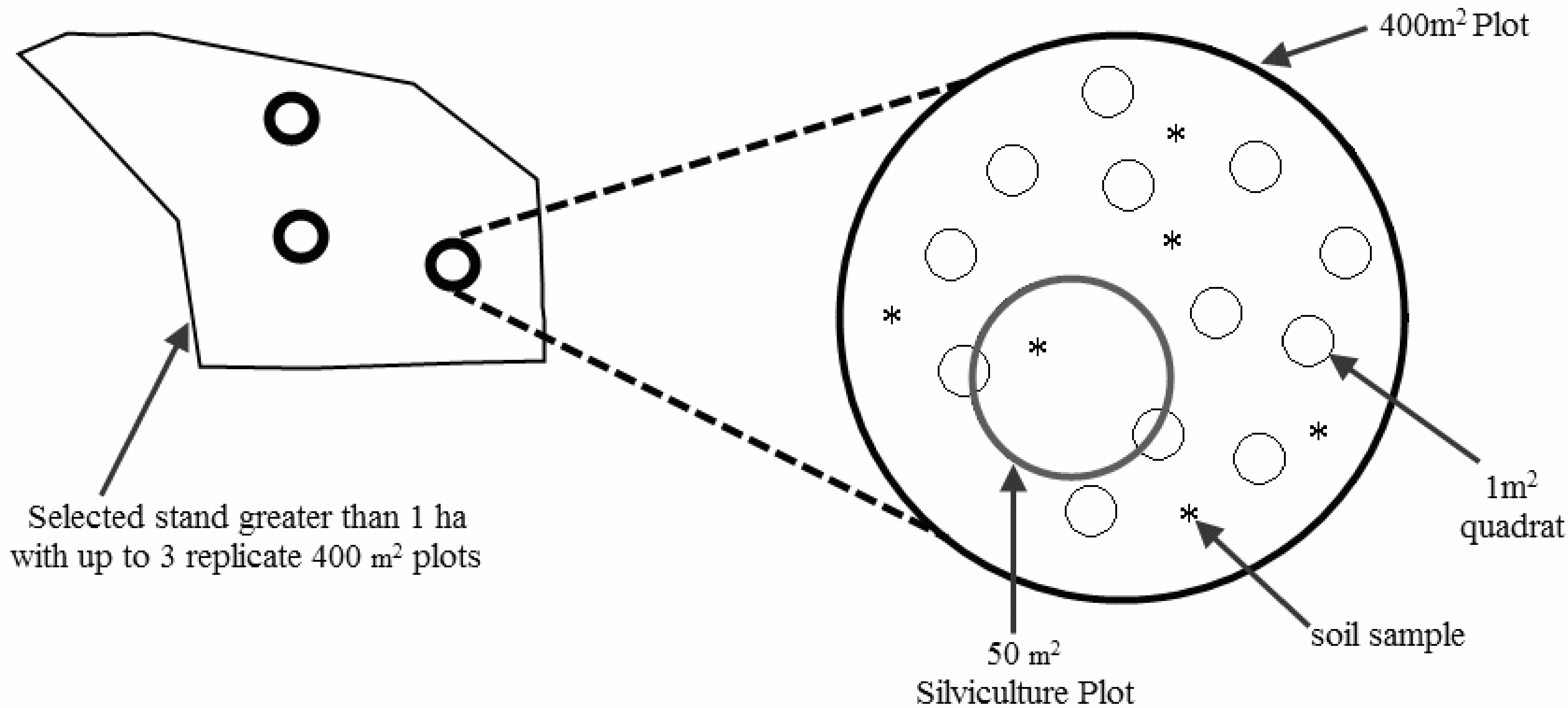


Figure 3. Mean annual temperature at Kamloops, BC, from late 1800's through 2007

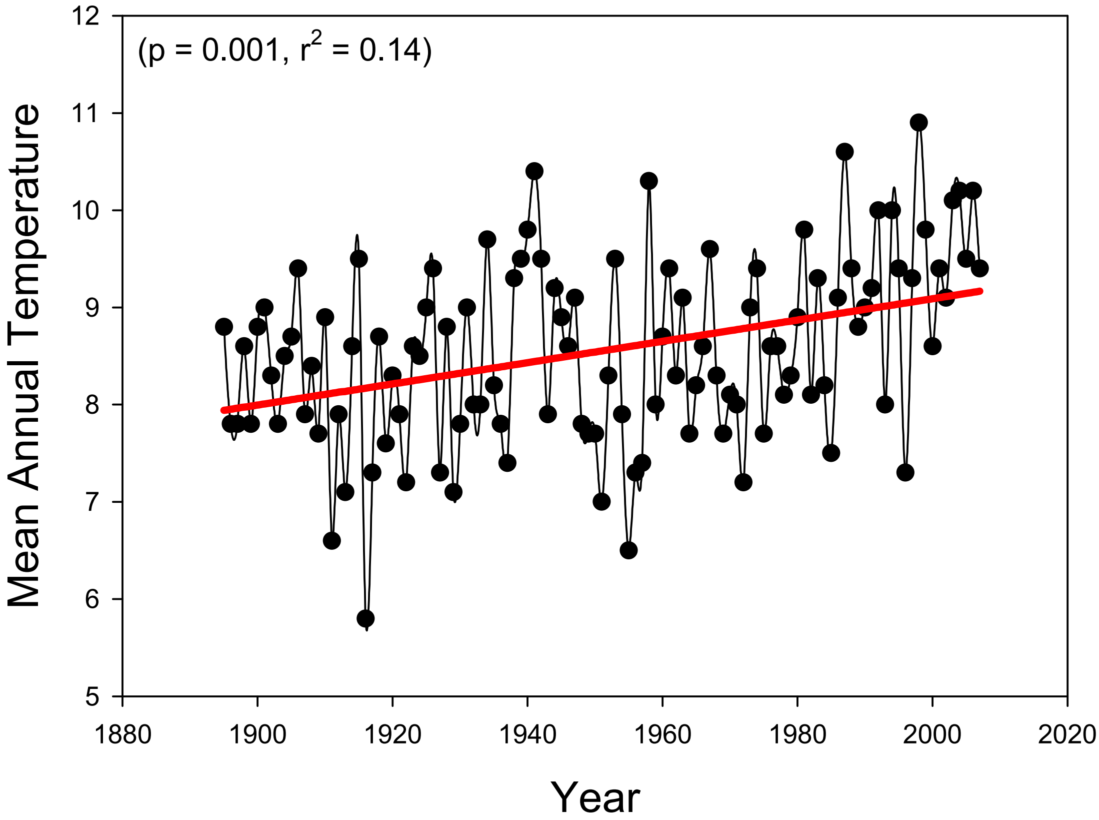


Figure 4. Venn diagram showing species overlap by site type (WF = wildfire, WFSE = seeded wildfire, WFSA = salvaged wildfire, WFSESA = seeded and salvaged wildfire).

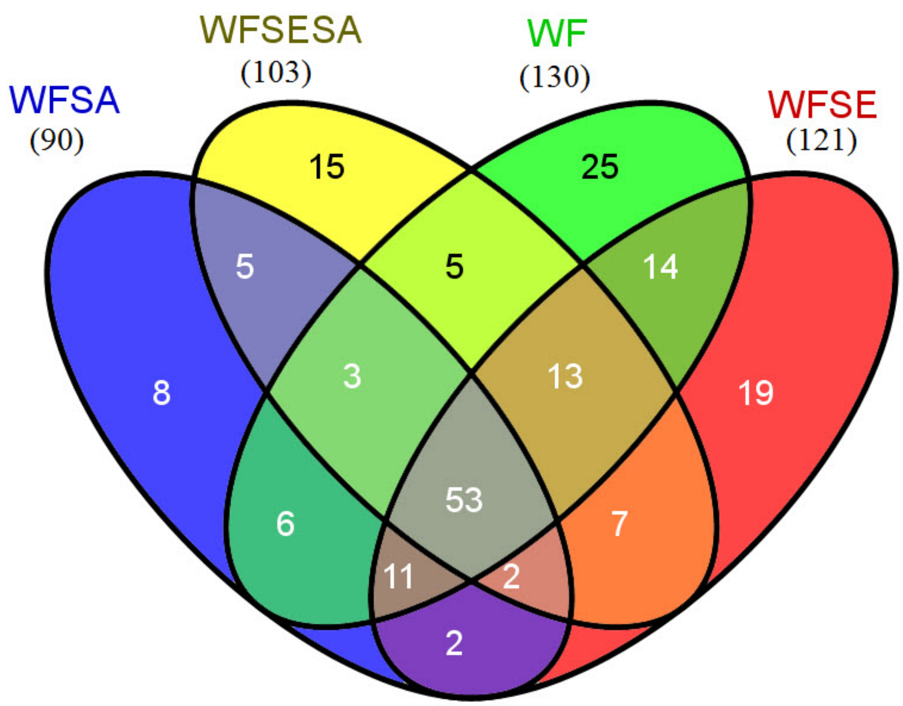


Figure 5. Species accumulation across sampled subplots of total richness by site type and estimated total species richness for each accumulation curve based on first order jackknife estimates at an unknown number of quadrats.

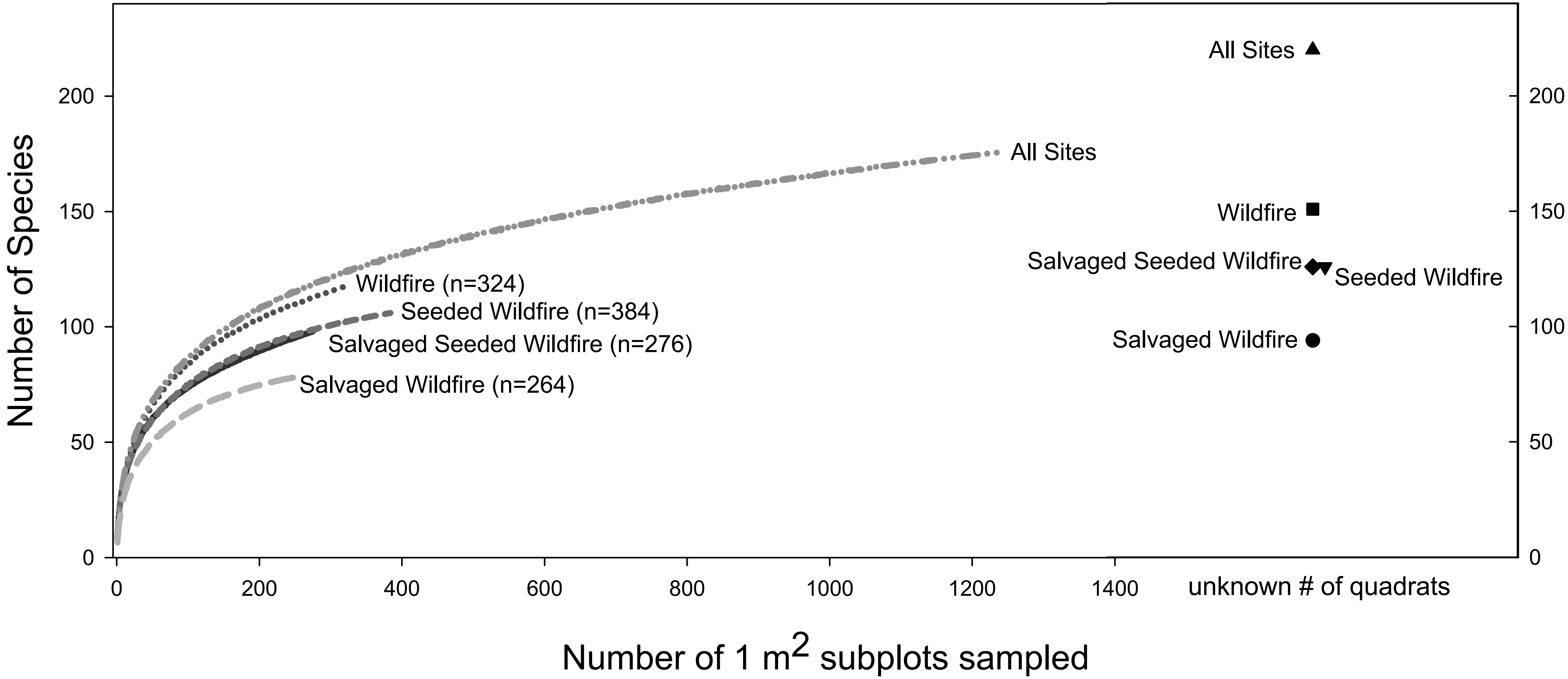


Figure 6. Two dimensional NMS ordination based on understory plant species composition showing which species groups and life forms are driving the variation of 1st and 2nd axes among the four site types at the plot scale (stress = 10.02)

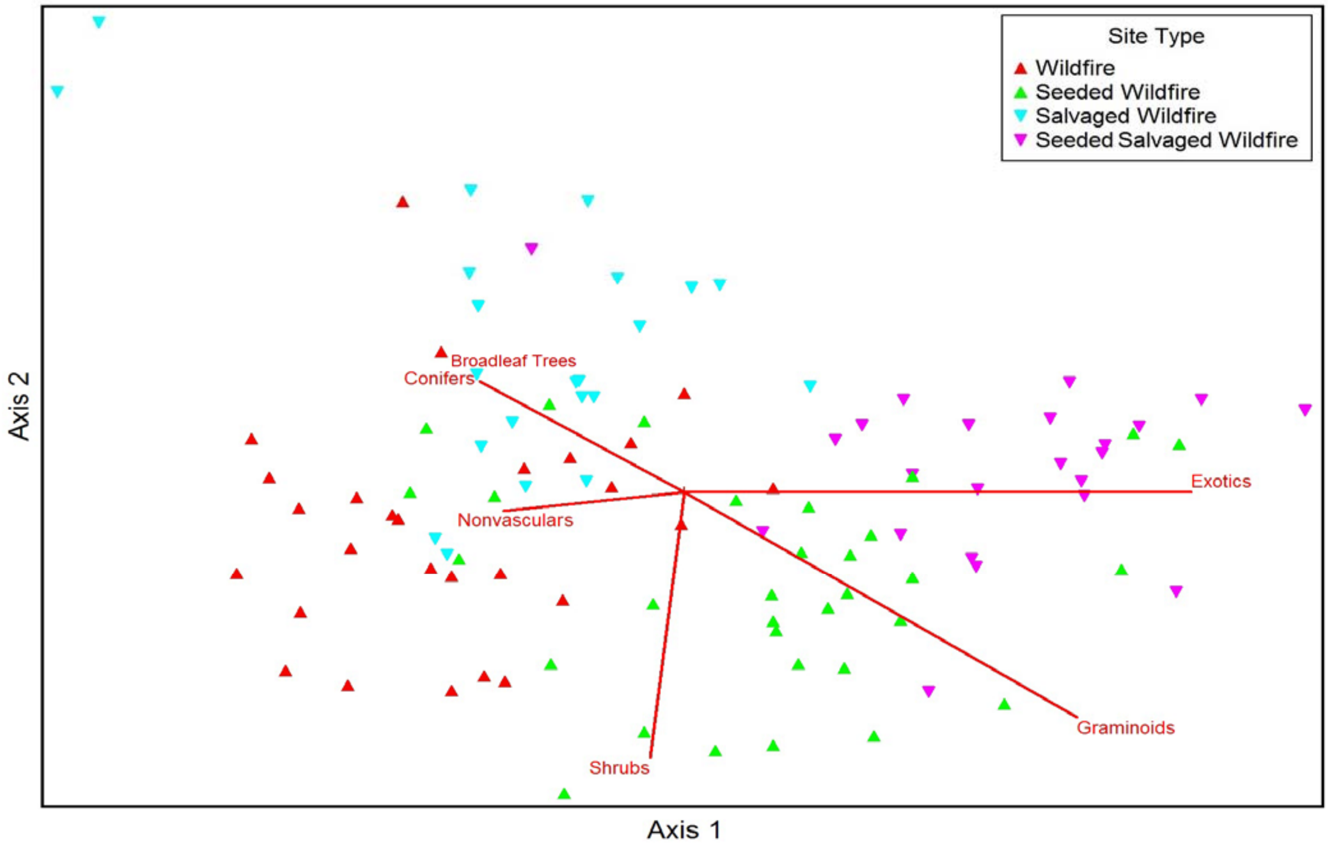


Figure 7. Two dimensional NMS ordination based on understory plant species composition showing which species are driving the variation of the axes among the four site types at the stand scale (stress = 14.0).

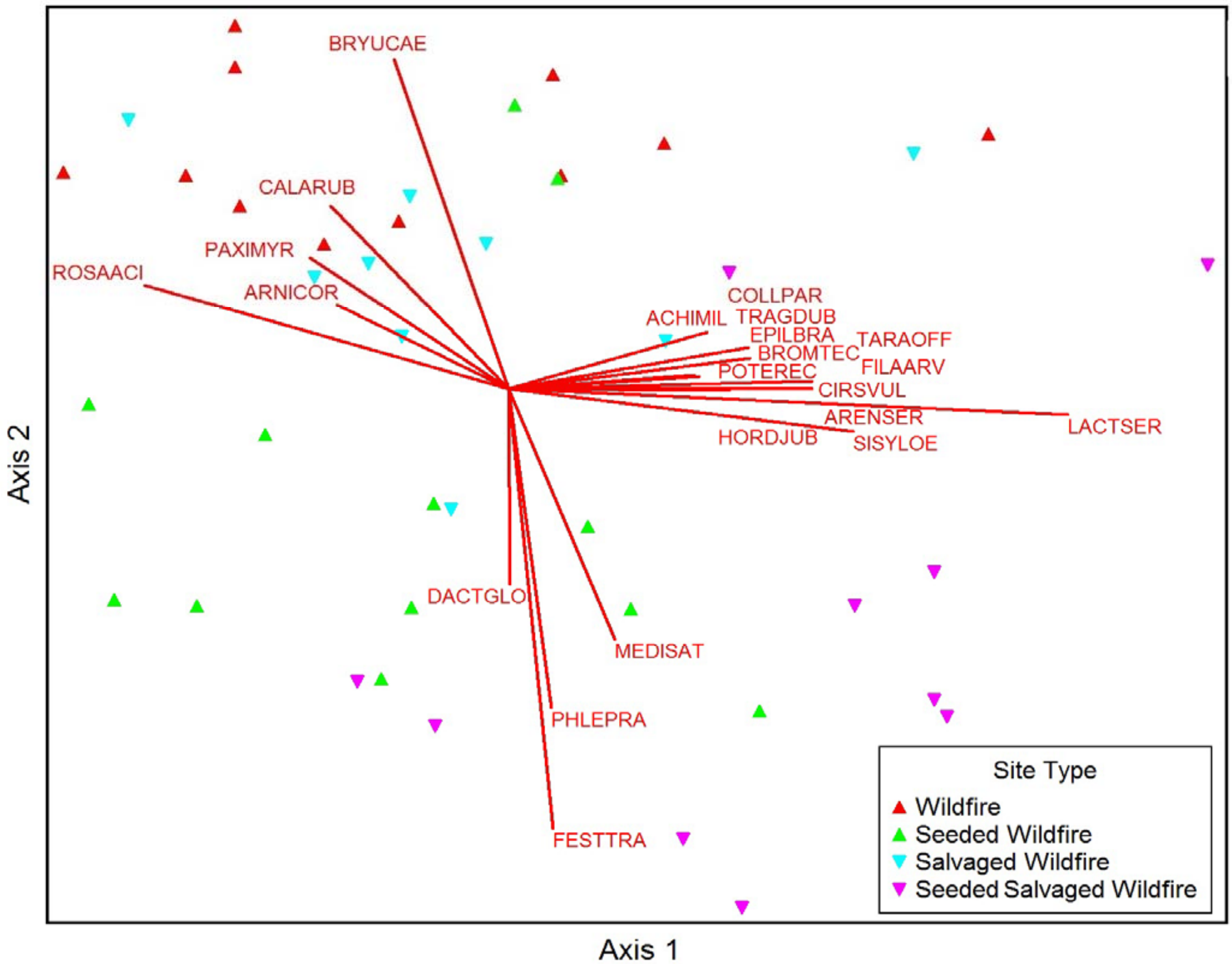


Figure 8. Mean species richness (showing upper 95% C.I.) for each site type by species group at the plot scale. Site types with same letters are not significantly different (Tukey's HSD,p = <0.05) within that species group.

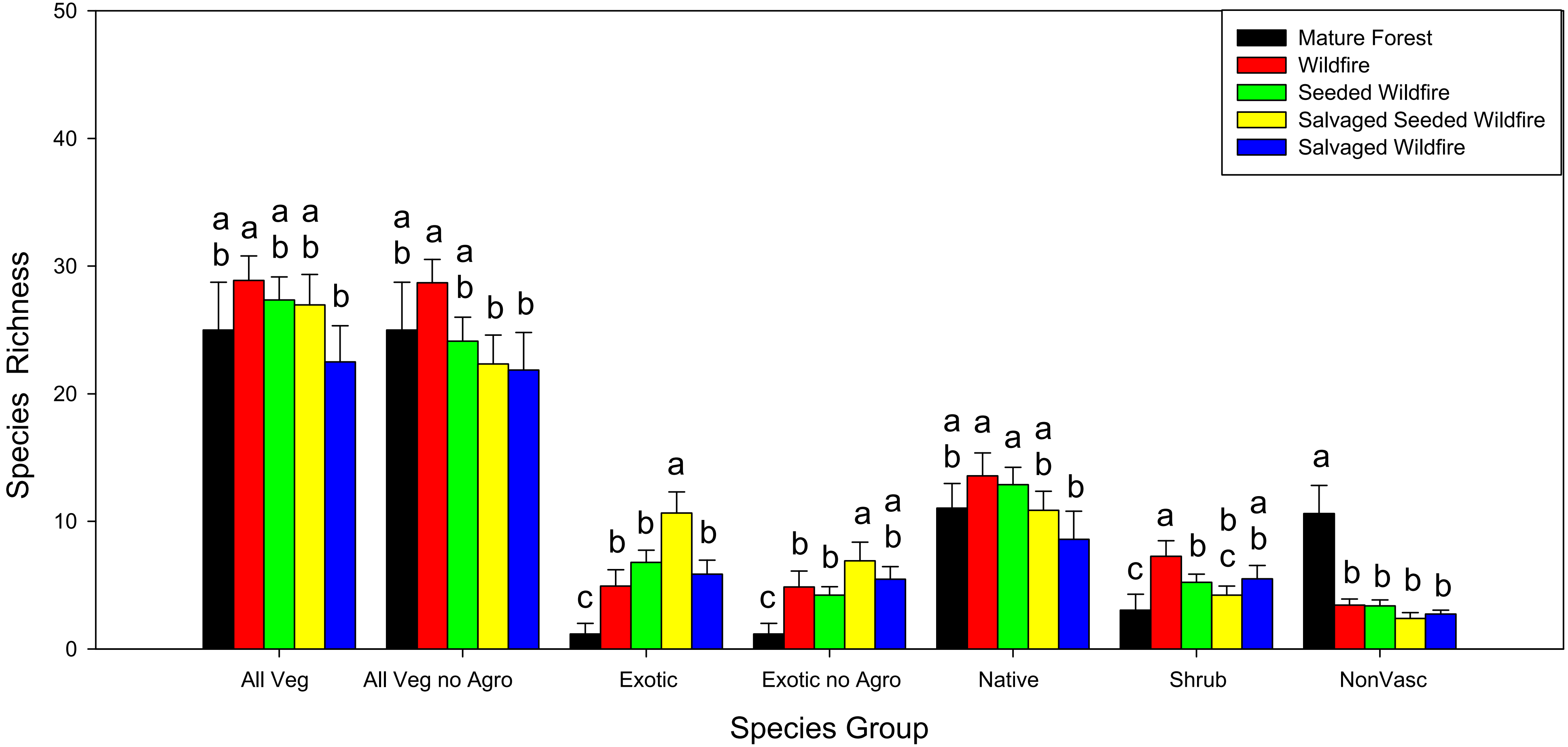


Figure 9. Mean species richness (showing upper 95% C.I.) at the stand scale between site types within species groups. Site types with the same letters are not significantly different (Tukey's HSD test, p = <0.05) between site types for that species group.

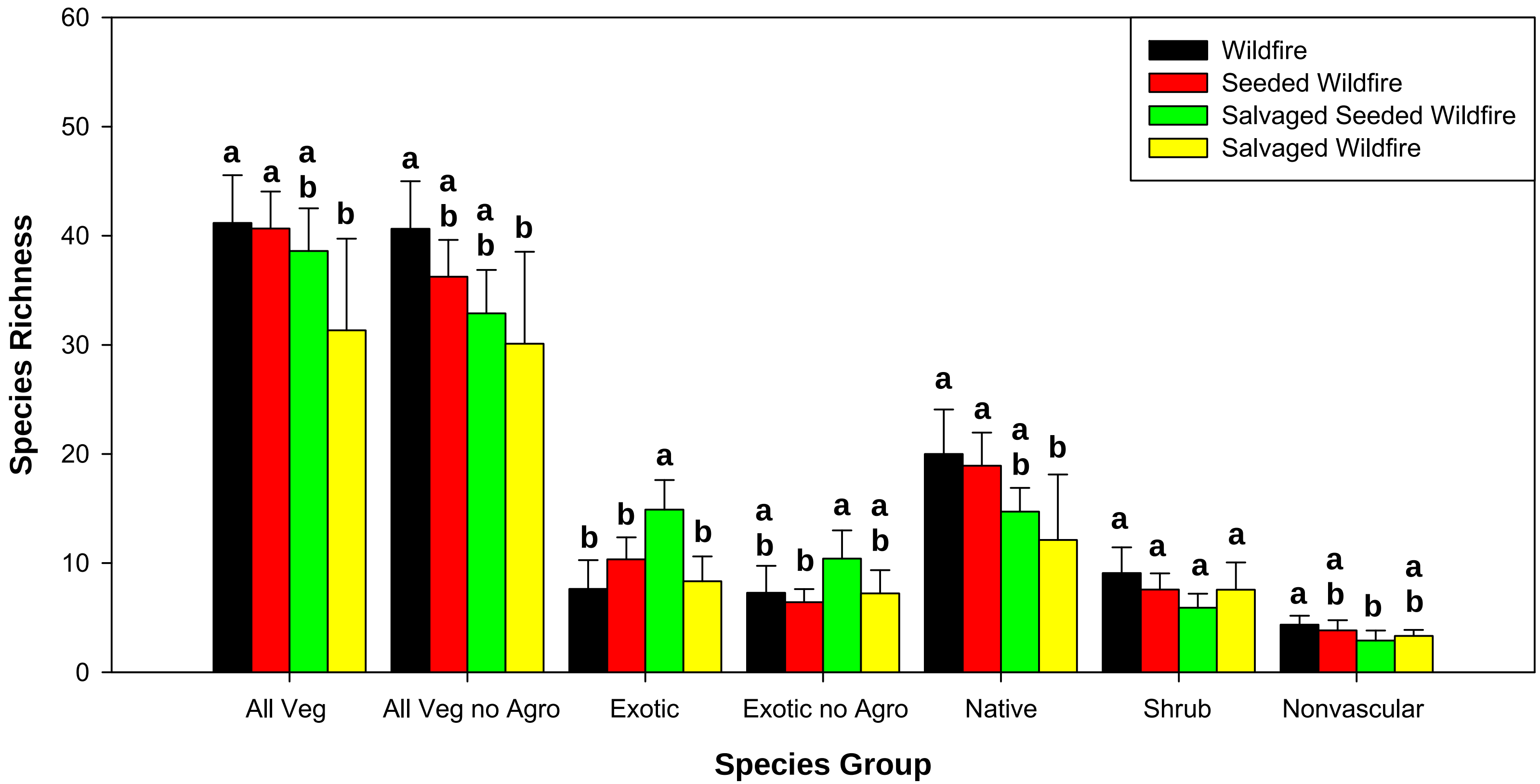


Figure 10. Ratio of mean species richness (log) at the plot scale (agronomic included). included). Site types with the same letters are not significantly different (Tukey's HSD test p = <0.05).

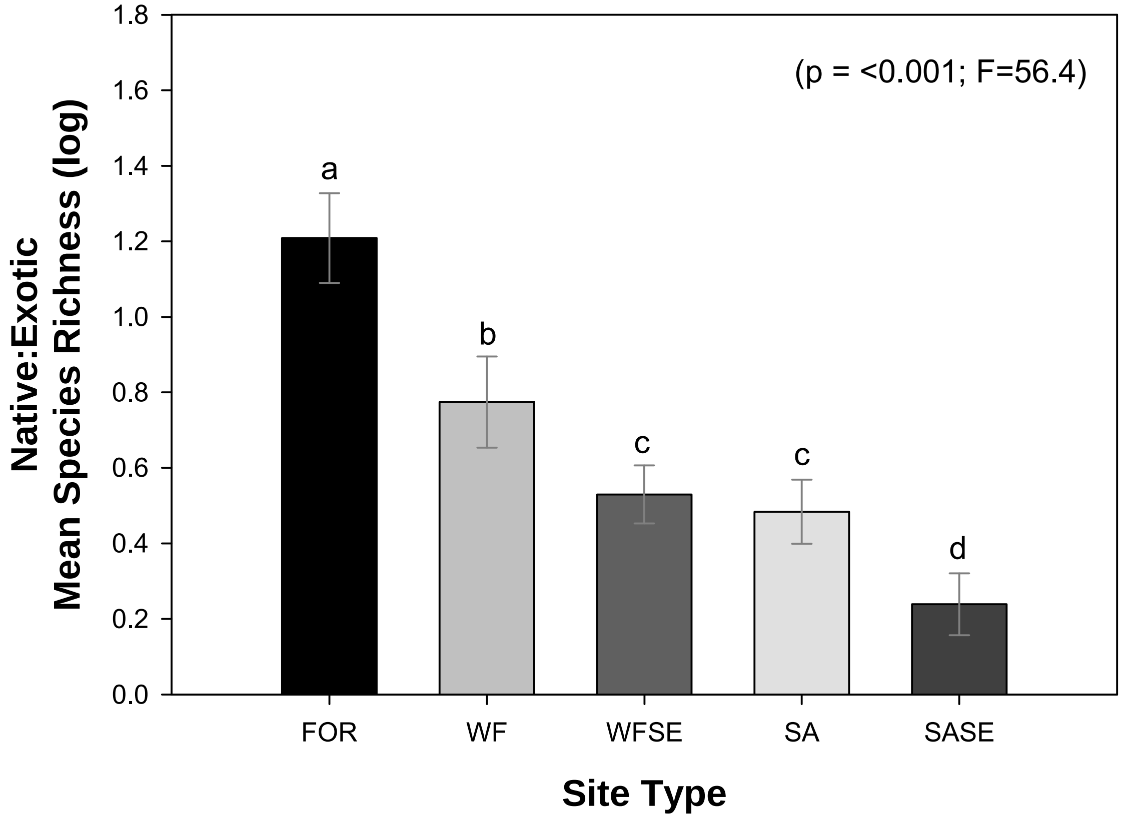


Figure 11. Ratio of mean species richness (log) at the stand scale (agronomics included). Site types with the same letters are not significantly different (Tukey's HSD, test p = <0.05).

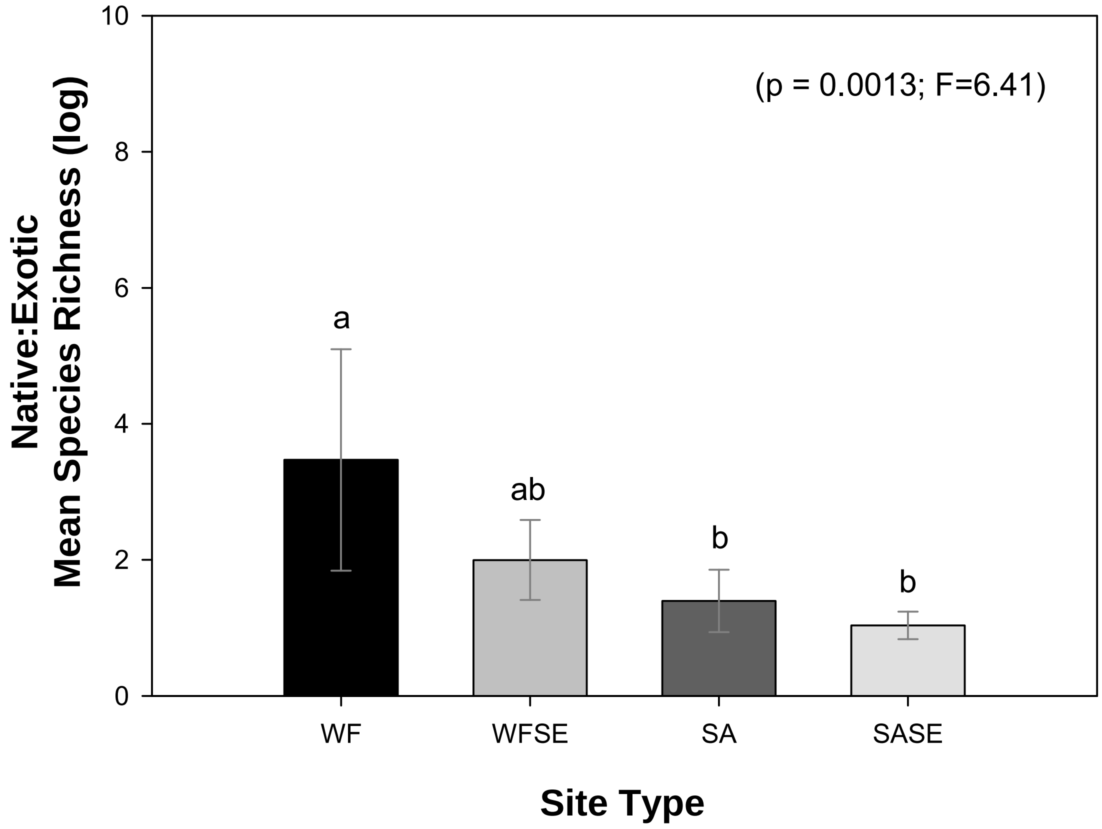


Figure 12. Mean values (+ 95% C.I.) for common (>33%), occasional (10-33%) and uncommon (<10%) native species for each site type at the plot scale. Site types with the same letters are not significantly different (Tukey's HSD test, p = <0.05). (Common: p = 0.0002; F = 7.2, Occasional: p = 0.0001; F = 11.2, Uncommon: p = 0.0001; F = 8.75).

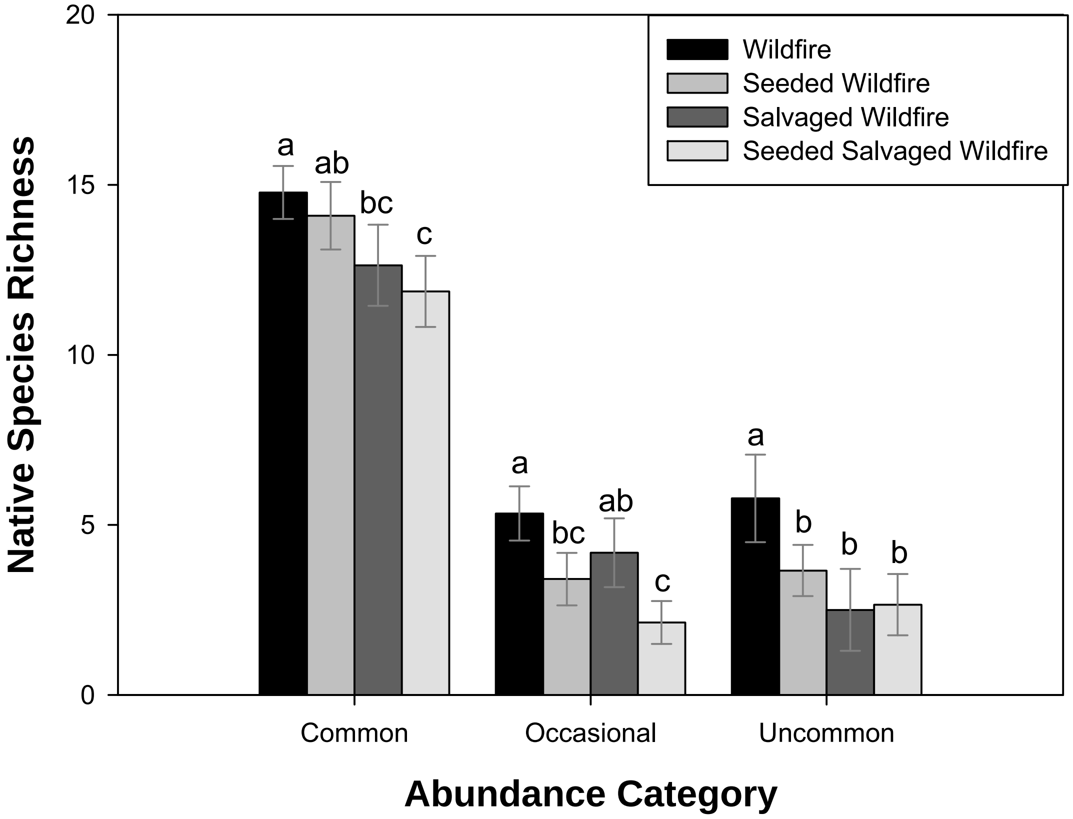


Figure 13. Mean richness values (± 95% C.I.) for common (>33%), occasional (10-33%) and uncommon (<10%) native species for each site type at the stand scale. Site types with the same letters are not significantly different (Tukey's HSD test, p = <0.05). (uncommon: p = 0.01; F = 4.3).

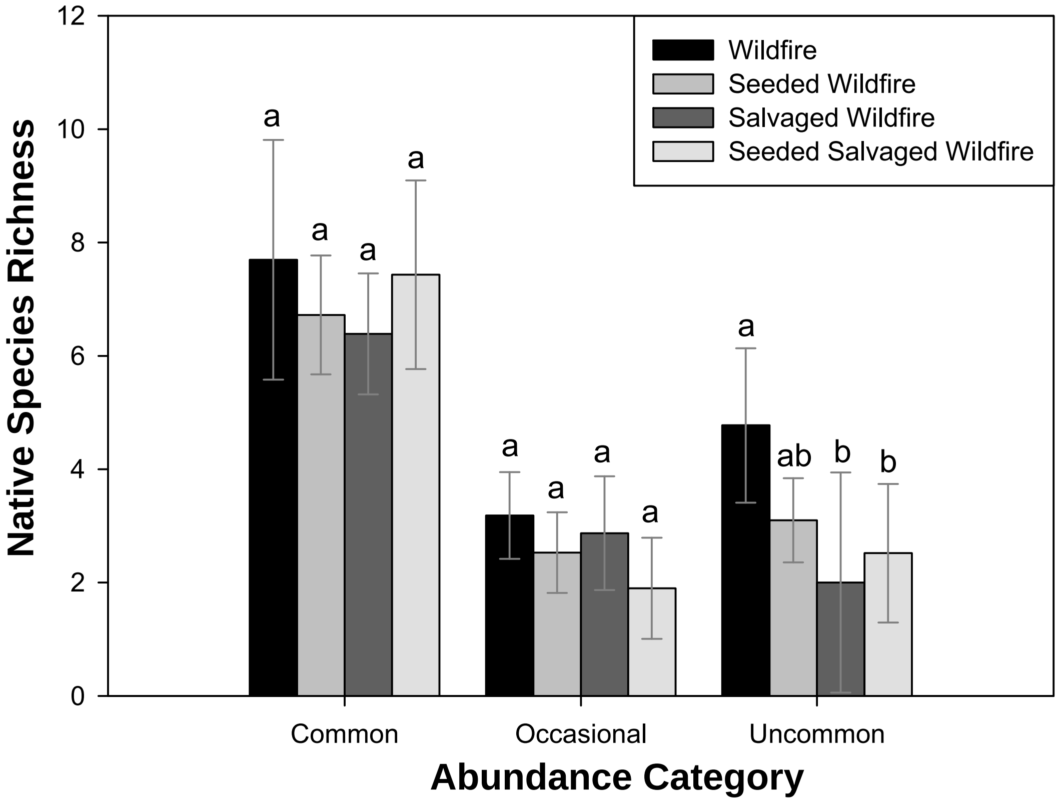


Figure 14. Linear ANCOVA model of native versus exotic species richness showing intercepts for each site type factor at the plot scale ($p = <0.001$, $r^2 = 0.597$, $F= 36.7$).

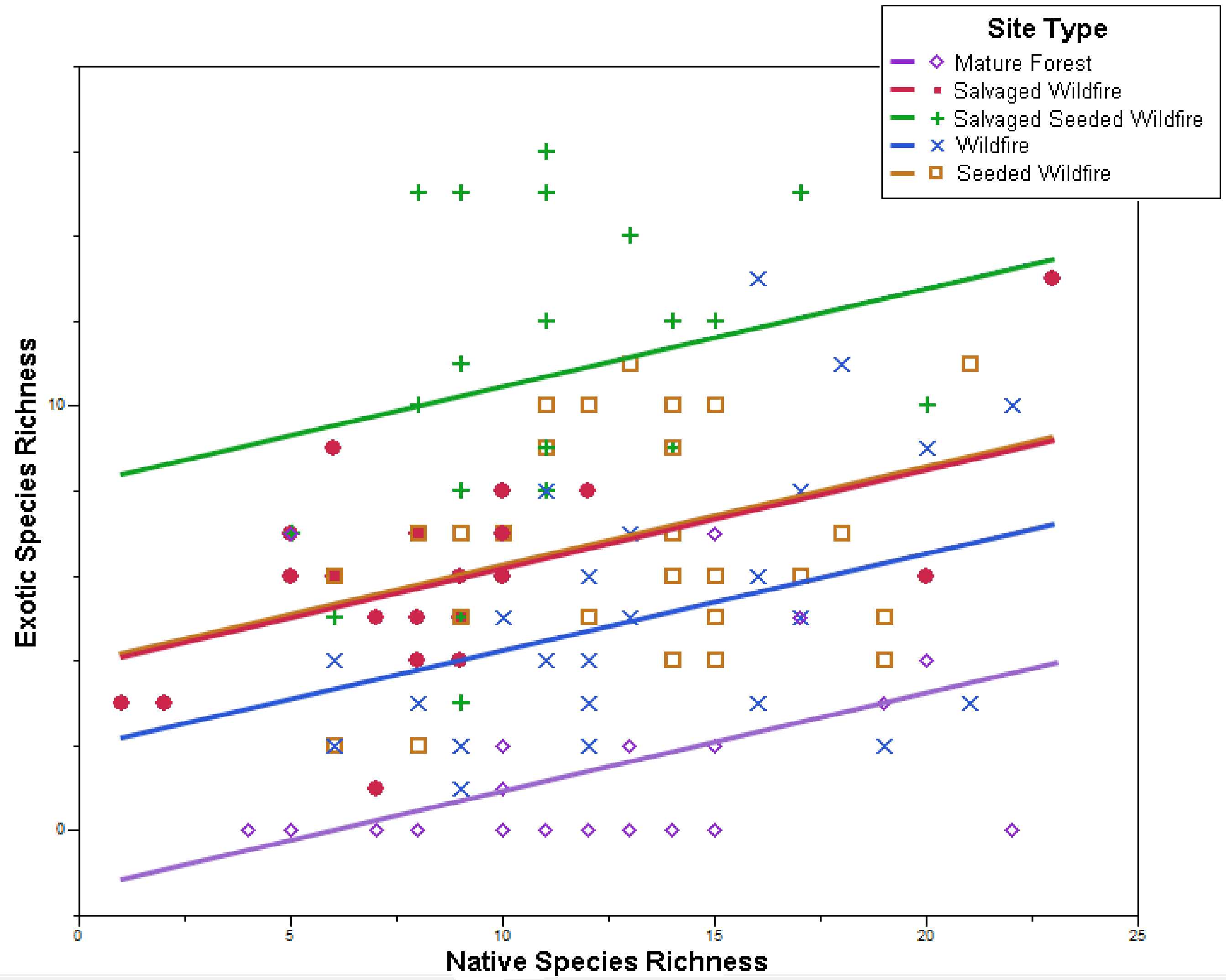


Figure 15. Covariance model with different slopes for native versus exotic species richness showing intercepts for each site type factor at the quadrat scale (p = <0.001, r^2 = 0.335, F= 89.3).

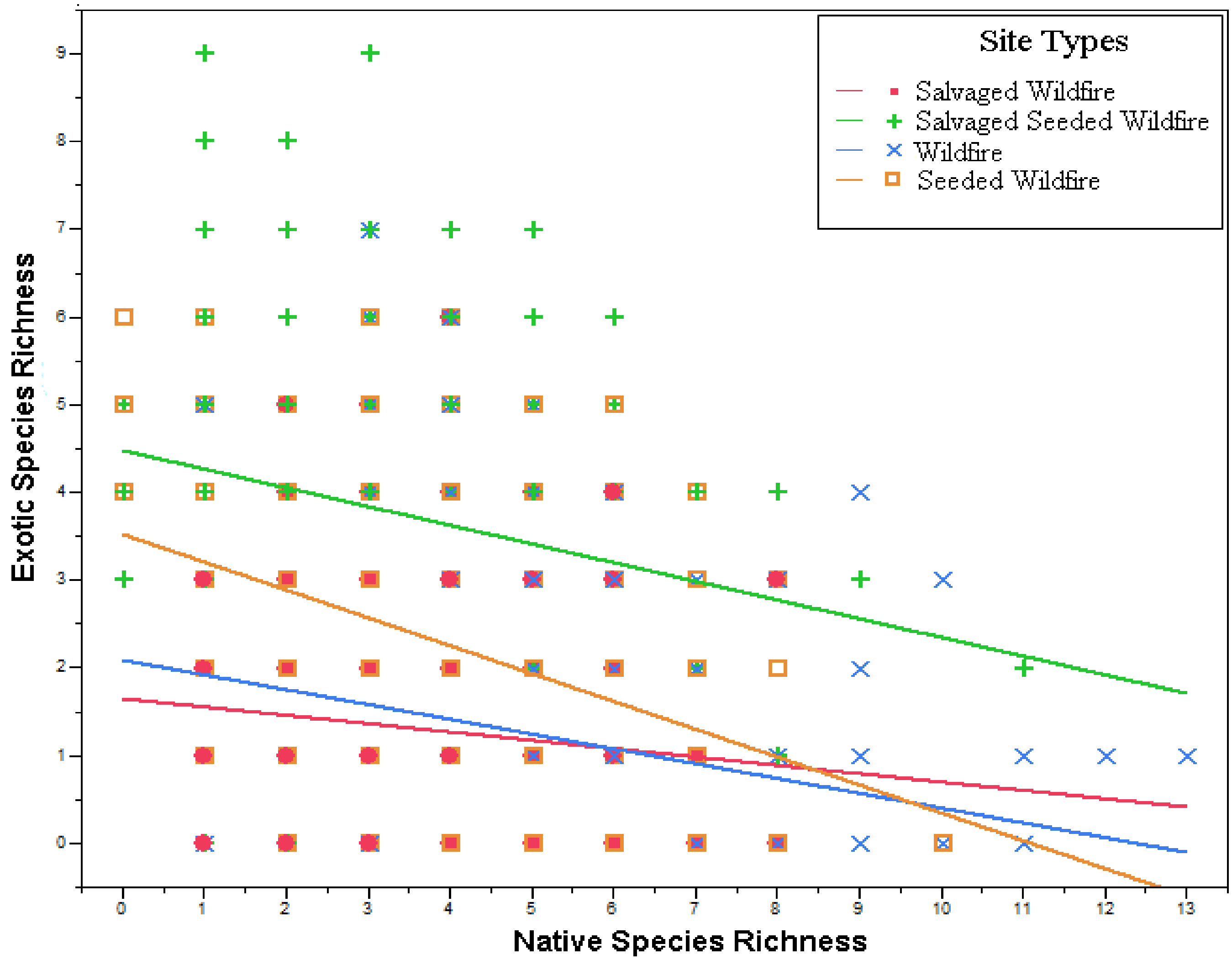


Figure 16. Linear regression between mean Shannon's diversity for stand scale understory vegetation and mean % cover of downed woody material with site type as a factor ($p = <0.001$; $r^2 = 0.56$).

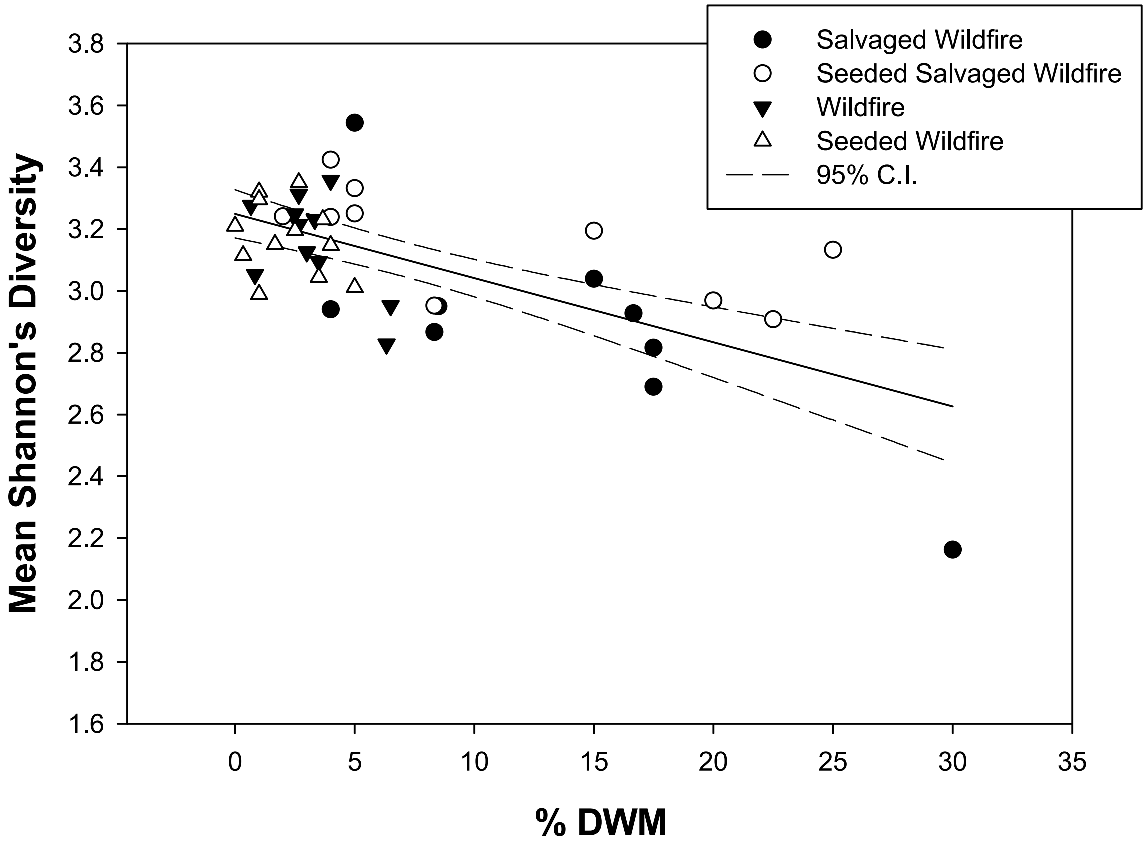


Figure 17. Mean number of well spaced naturally regenerated tree seedlings (± 95% C.I.) by site type four years after fire at the plot scale.

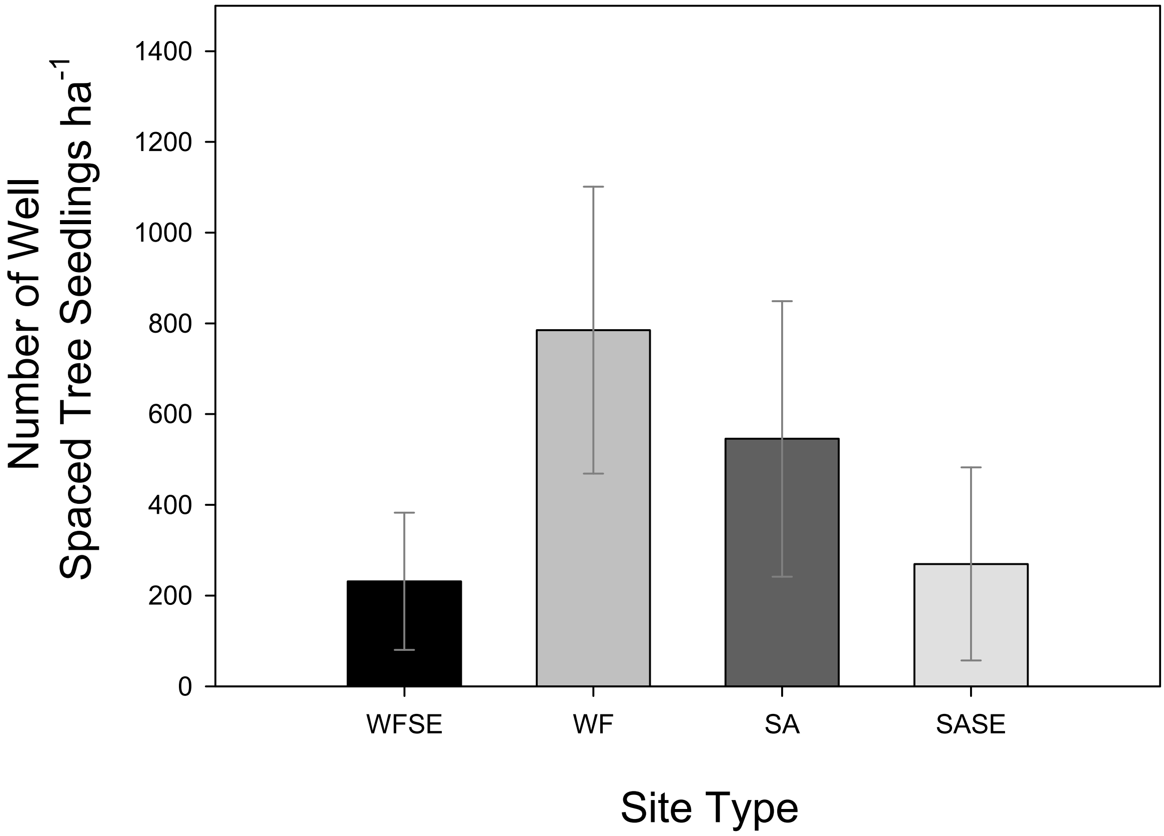


Figure 18. Mean number (± 95% C.I.) of natural well spaced tree seedlings by seeded and non-seeded site type four years after fire at the plot and stand scale.

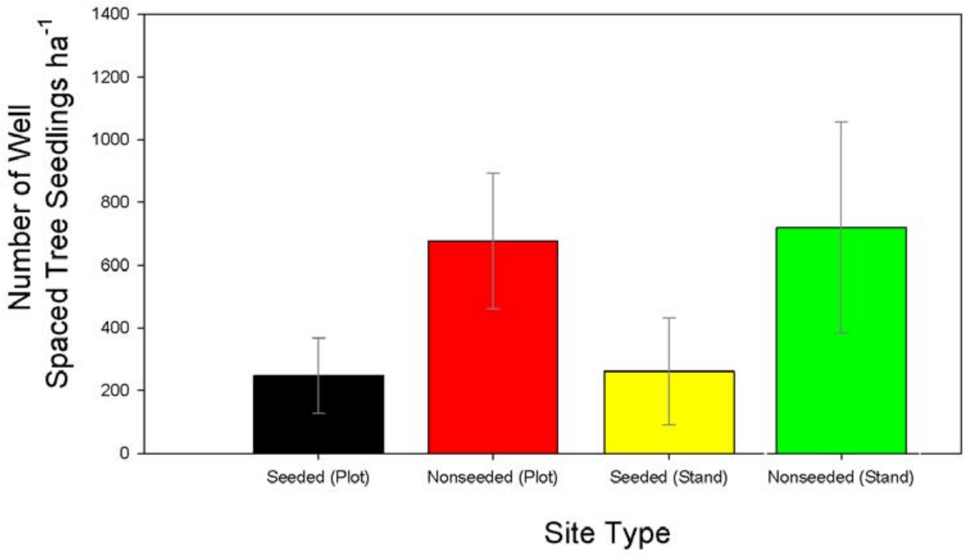


Figure 19. Log mean number of well spaced naturally regenerated tree seedlings (± 95% C.I.) by site type at plot scale four years after fire showing main tree species encountered (Fd = *Pseudotsuga menziesii*, Pl = *Pinus contorta*, At = *Populus tremuloides*, Bp = *Betula papyrifera*).

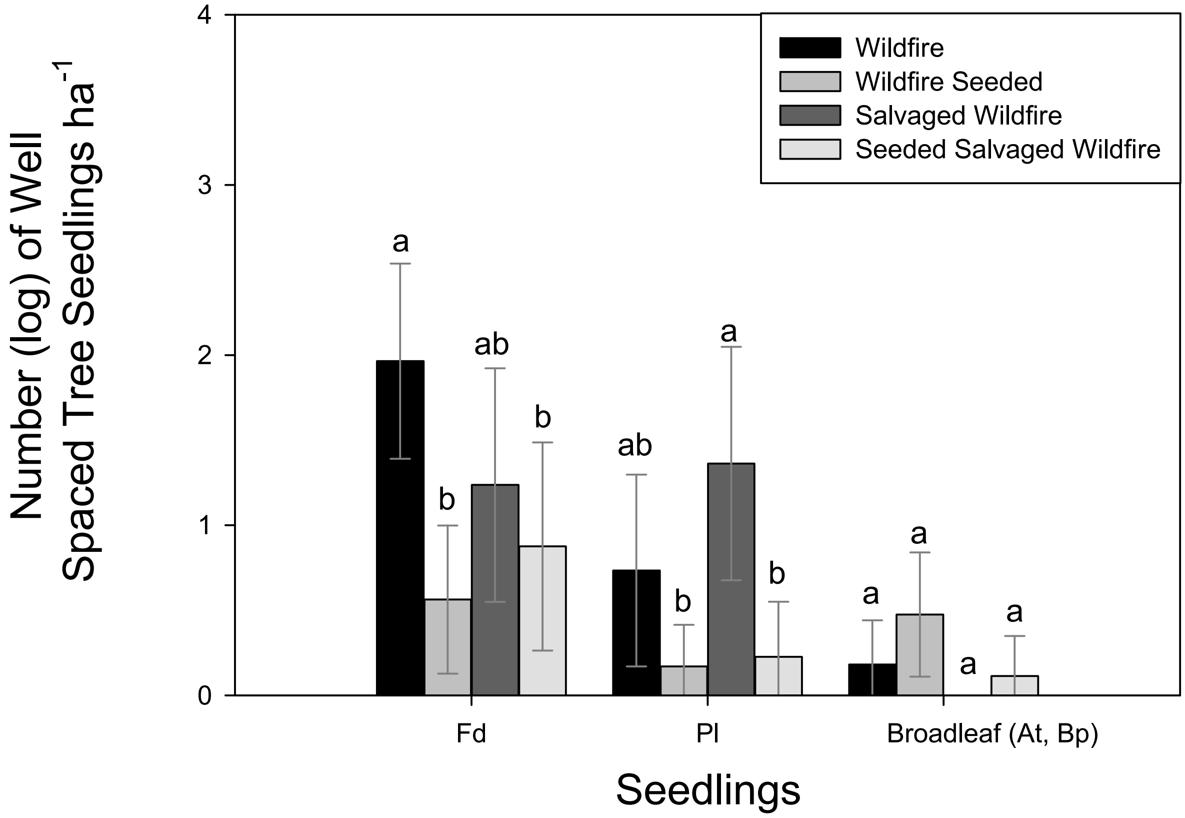


Figure 20. Mean number of well spaced naturally regenerated tree seedlings (± 95% C.I.) by seeded and non-seeded type at plot scale four years after fire showing main tree species encountered (Fd = *Pseudotsuga menziesii*, Pl = *Pinus contorta*, At = *Populus tremuloides*, Bp = *Betula papyrifera*).

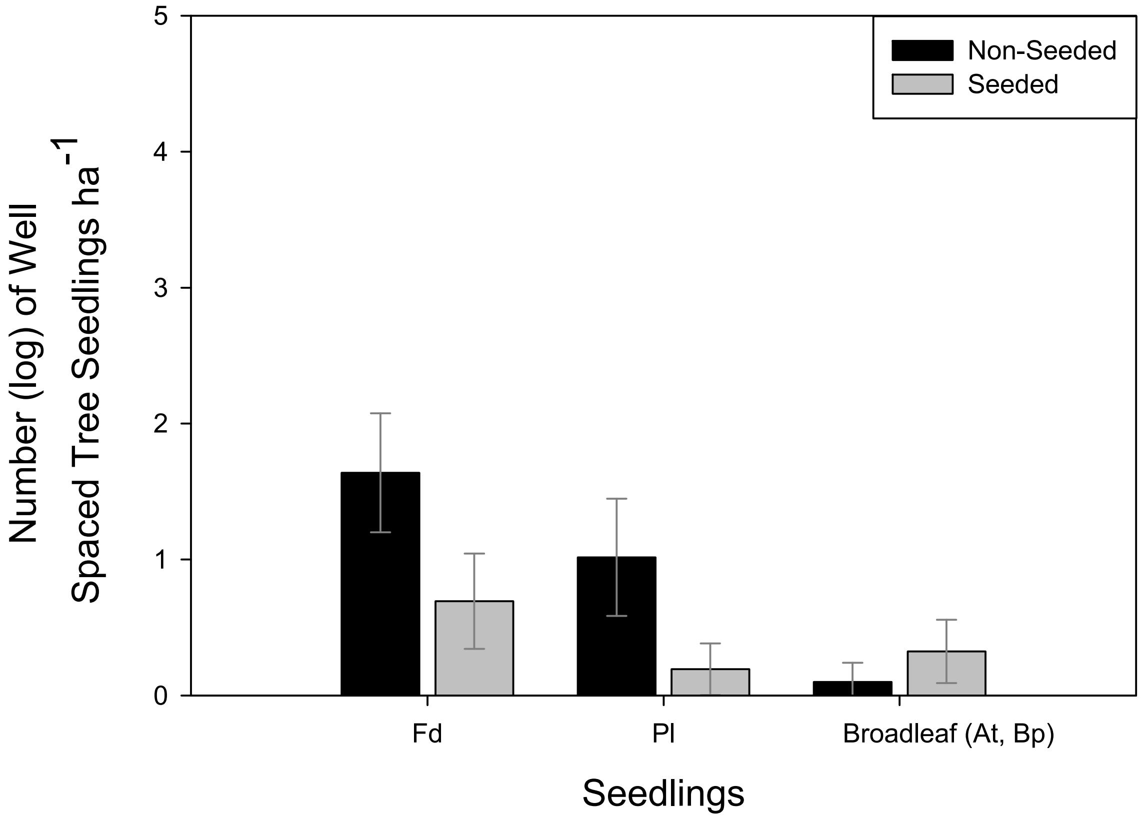


Figure 21. Mean number of well spaced (including planted trees) seedlings (± 95% C.I.) by site type four years after fire at stand scale.

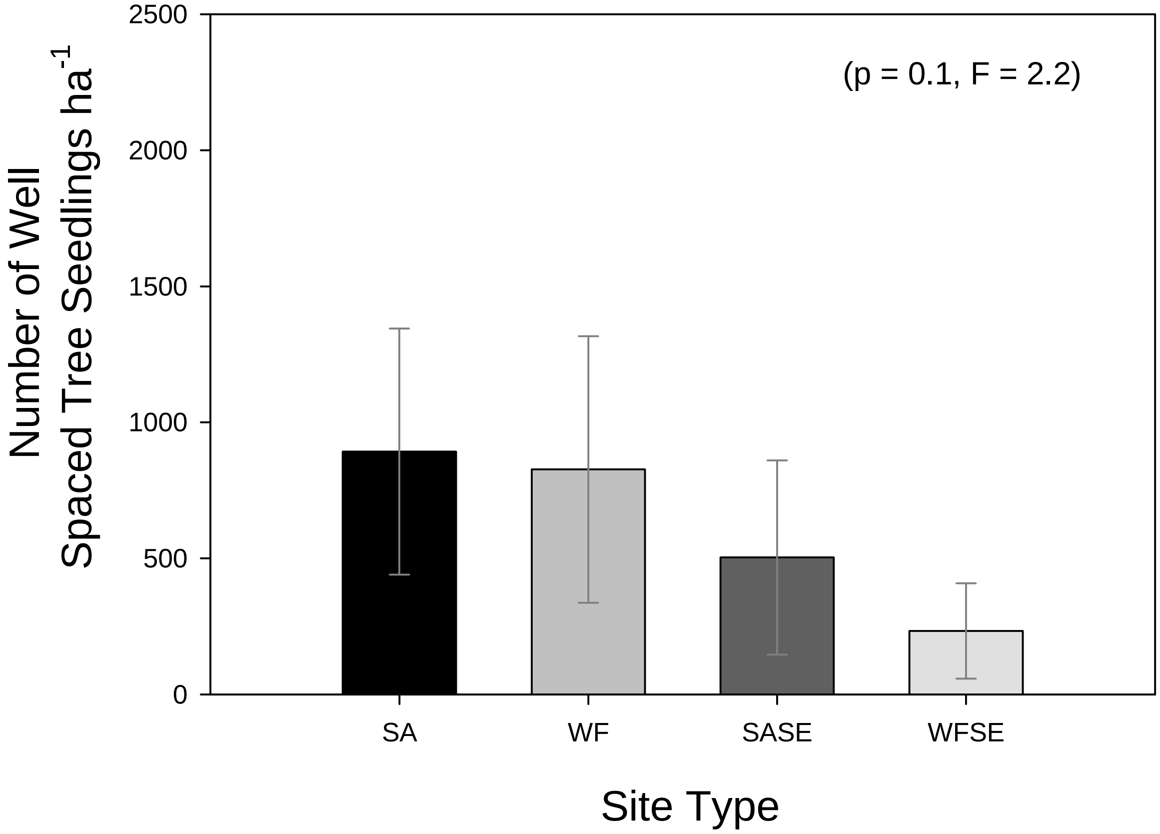


Figure 22. Mean number (log) of well spaced naturally regenerated tree seedlings (± 95% C.I.) by site type at stand scale four years after fire showing main tree species encountered (Pl = *Pinus contorta*, Fd = *Pseudotsuga menziesii*, At = *Populus tremuloides*, Bp = *Betula papyrifera*)

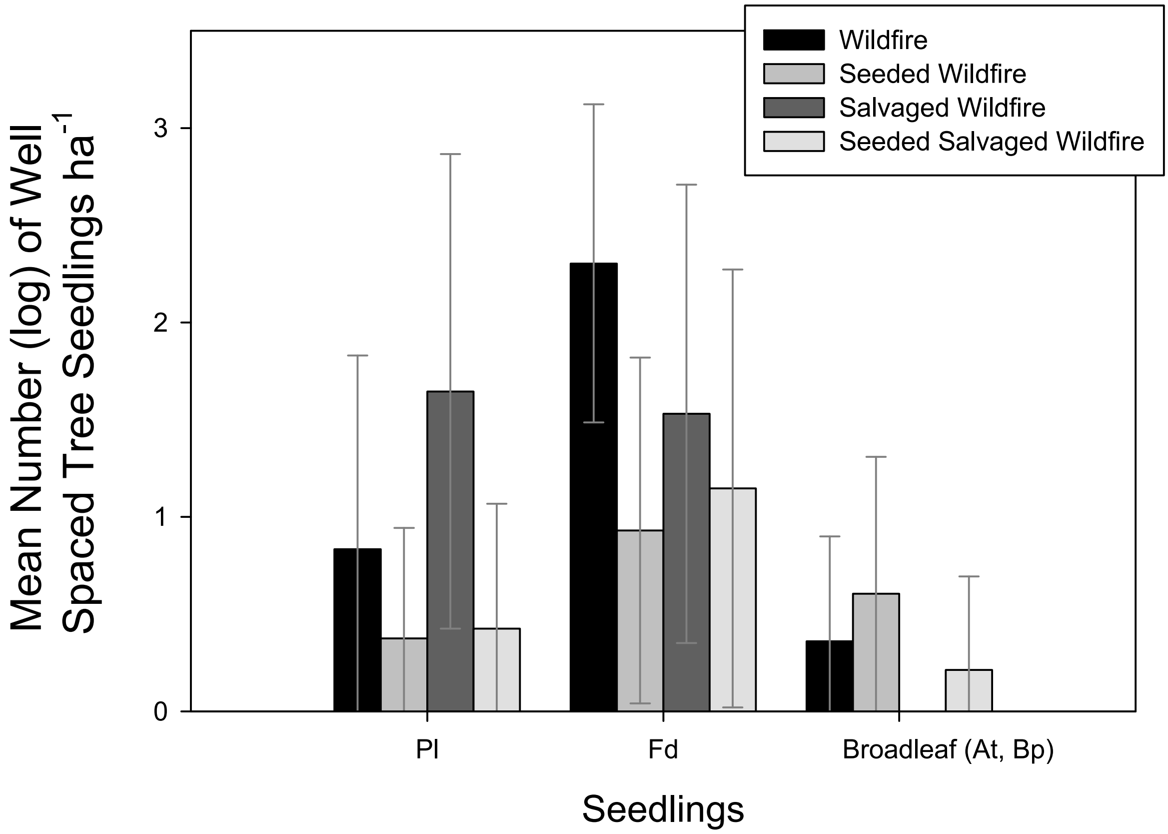
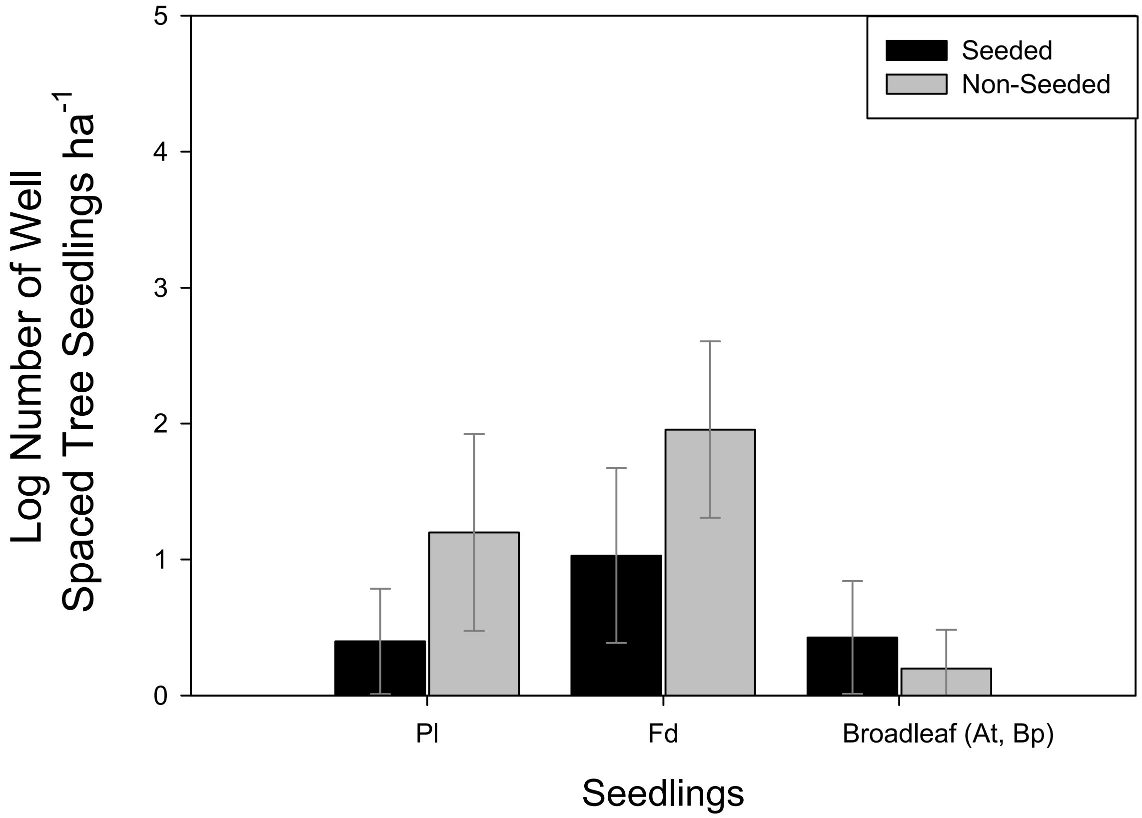


Figure 23. Mean number (log) of well spaced naturally regenerated tree seedlings (± 95% C.I.) by seeded and non-seeded type at stand scale four years after fire showing main tree species encountered (Pl = *Pinus contorta*, Fd = *Pseudotsuga menziesii*, At = *Populus tremuloides*, Bp = *Betula papyrifera*)



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Appendix I. Locations of plots sampled in the McLure and McGilivray fire areas (Datum = Nad 83, WF= wildfire only, WFSE = seeded wildfire, WFSA = salvage logged wildfire, WFSESA = Seeded and salvaged wildfire).

Plot Number	Fire	Site Type	UTM Zone	UTM Easting	UTM Northing
C01SD1	McLure	WFSE	10	693746	5671341
C01SD2	McLure	WFSE	10	693802	5671348
C01SD3	McLure	WFSE	10	693853	5671480
C0SD1	McLure	WF	10	696433	5666188
C0SD2	McLure	WF	10	696409	5666155
C0SD3	McLure	WF	10	696383	5666120
C15CC1	McLure	SA	10	692766	5679659
C15CC2	McLure	SA	10	692713	5679602
C15CC3	McLure	SA	10	692763	5679589
C15SD1	McLure	WF	10	692690	5679237
C15SD2	McLure	WF	10	692830	5679372
C16CC1	McLure	SA	10	694543	5672310
C16CC2	McLure	SA	10	694669	5672280
C16CC3	McLure	SA	10	694709	5672215
C16SD1	McLure	WF	10	694452	5672078
C16SD2	McLure	WF	10	694438	5672063
C16SD3	McLure	WF	10	694390	5672072
C17CC1	McLure	SA	10	695915	5668595
C17CC2	McLure	SA	10	695926	5668512
C17CC3	McLure	SA	10	695831	5668453
C17SD1	McLure	WFSE	10	696080	5667299
C17SD2	McLure	WFSE	10	696062	5667283
C17SD3	McLure	WFSE	11	696145	5667272
C20CC1	McLure	SA	10	693146	5666765
C20CC2	McLure	SA	10	693111	5666809
C20CC3	McLure	SA	10	693183	5666979
C20SD1	McLure	WFSE	10	693217	5666720
C20SD2	McLure	WFSE	10	693413	5666617
C20SD3	McLure	WFSE	11	693292	5666589
C35CC1	McLure	SASE	10	700406	5660767
C35CC2	McLure	SASE	11	649979	5661888
C35CC3	McLure	SASE	10	699964	5661829
C35SD1	McLure	WFSE	10	699796	5661675
C35SD2	McLure	WFSE	10	699851	5661934
C35SD3	McLure	WFSE	10	699868	5662053
C38CC1	McLure	SASE	10	698368	5667684
C38CC2	McLure	SASE	10	698408	5667785
C38SD1	McLure	WF	10	699600	5667342
C38SD2	McLure	WF	10	699514	5667204
C38SD3	McLure	WF	10	699452	5667192
C39SD1	McLure	WFSE	10	701172	5665703
C39SD2	McLure	WFSE	10	700965	5665712
C39SD3	McLure	WFSE	10	701044	5665912

Appendix I. Locations of plots sampled in the McLure and McGilivray fire areas (Datum = Nad 83, WF= wildfire only, WFSE = seeded wildfire, WFSA = salvage logged wildfire, WFSESA = Seeded and salvaged wildfire).

Plot Number	Fire	Site Type	UTM Zone	UTM Easting	UTM Northing
C44CC1	McLure	SASE	10	695048	5669754
C44CC2	McLure	SASE	10	695040	5669561
C44SD1	McLure	WFSE	10	695001	5668925
C44SD2	McLure	WFSE	10	695034	5668769
C47CC1	McLure	SASE	10	696144	5659863
C47CC2	McLure	SASE	10	695993	5659240
C47CC3	McLure	SASE	10	695916	5658328
C47CW1	McLure	SASE	10	695964	5658173
C47CW2	McLure	SASE	10	695904	5657758
C47SD1	McLure	WFSE	10	696202	5660483
C47SD2	McLure	WFSE	10	695960	5659703
C47SD3	McLure	WFSE	10	696002	5660189
C50SD1	McLure	WF	10	693572	5678647
C50SD2	McLure	WF	10	693521	5678650
C50SD3	McLure	WF	11	693493	5678670
C51SD1	McLure	WF	10	703027	5665957
C51SD2	McLure	WF	10	703078	5665954
C51SD3	McLure	WF	10	703147	5665858
C52CC1	McLure	SASE	10	702453	5667797
C52CC2	McLure	SASE	10	702438	5667841
C52CC3	McLure	SASE	10	702486	5667781
C53CC1	McLure	SASE	10	702228	5667366
C53CC2	McLure	SASE	11	702210	5667489
C53CC3	McLure	SASE	10	702131	5667622
C54CC1	McLure	SASE	10	699822	5663609
C54CC2	McLure	SASE	10	699949	5663591
C54SD1	McLure	WFSE	10	699960	5663825
C54SD2	McLure	WFSE	10	700202	5663694
C55SD1	McLure	WF	10	704043	5667518
C56SD1	McLure	WFSE	10	703742	5667581
C56SD2	McLure	WFSE	10	703678	5667598
C56SD3	McLure	WFSE	10	703405	5667915
G06CC1	McGillivray	SA	11	299437	5622894
G06CC2	McGillivray	SA	11	298399	5623071
G06SD1	McGillivray	WF	11	299153	5622949
G06SD2	McGillivray	WF	11	299130	5622900
G09SD1	McGillivray	WF	11	298591	5627401
G09SD2	McGillivray	WF	11	298414	5627345
G10CC1	McGillivray	SASE	11	301714	5626805
G10CC2	McGillivray	SASE	11	301798	5626782
G10SD1	McGillivray	WFSE	11	301730	5626715
G10SD2	McGillivray	WFSE	11	301789	5626678
G10SD3	McGillivray	WFSE	11	301854	5626622

Appendix I. Locations of plots sampled in the McLure and McGilivray fire areas (Datum = Nad 83, WF= wildfire only, WFSE = seeded wildfire, WFSA = salvage logged wildfire, WFSESA = Seeded and salvaged wildfire).

Plot Number	Fire	Site Type	UTM Zone	UTM Easting	UTM Northing
G11CC1	McGillivray	SA	11	300897	5627547
G11CC2	McGillivray	SA	11	300896	5627507
G11SD1	McGillivray	WF	11	300983	5627256
G11SD2	McGillivray	WF	11	300963	5627244
G12SD1	McGillivray	WF	11	303809	5631585
G12SD2	McGillivray	WF	11	303851	5631560
G12SD3	McGillivray	WF	11	303921	5631536
G13CC1	McGillivray	SA	11	300149	5623883
G13CC2	McGillivray	SA	11	300173	5623858
G13SD1	McGillivray	WFSE	11	300275	5623937
G13SD2	McGillivray	WFSE	11	300320	5623924
G14CC1	McGillivray	SASE	11	300022	5622409
G15CC1	McGillivray	SA	11	299975	5625504
G15CC2	McGillivray	SA	11	300023	5625467
G15SD1	McGillivray	WFSE	11	300127	5624476
G15SD2	McGillivray	WFSE	11	300051	5624404
G20CC1	McGillivray	SA	11	305216	5635250
G20CC2	McGillivray	SA	11	305188	5636511

Appendix II. Soil chemistry analysis results for composite soil samples (in PPM except where noted).

Plot Number	Al	B	Ca	Cu	Fe	K	Mg	Mn	Na	P	S	Zn	pH	C%	N%
C01SD1	878.05	0.01	1578.27	1.16	318.56	128.99	118.49	79.95	12.27	51.18	6.03	3.21	6.11	1.68	0.08
C01SD2	634.34	0.74	1746.07	0.66	288.92	245.70	107.21	305.39	7.75	102.49	7.30	14.37	7.49	1.32	0.07
C01SD3	1188.06	0.01	1195.10	0.70	497.29	284.48	107.36	141.30	8.65	420.81	8.17	5.28	6.71	1.37	0.07
C0SD1	729.69	0.01	2105.77	1.98	213.39	286.49	312.64	170.72	9.97	40.92	5.84	4.94	6.21	2.03	0.10
C0SD2	751.95	0.32	3854.27	2.85	278.39	618.00	423.77	280.60	16.65	98.70	8.12	8.64	7.19	2.71	0.13
C0SD3	761.30	0.32	5310.08	5.56	248.21	927.35	531.04	182.41	16.32	105.96	6.79	6.00	7.11	3.36	0.16
C15CC1	1422.66	0.14	3230.59	2.26	351.49	209.26	119.47	202.30	7.94	317.56	13.76	30.34	6.88	4.10	0.15
C15CC2	729.55	0.01	2228.05	1.39	361.40	170.11	99.30	161.43	10.34	125.13	8.73	7.23	7.39	2.78	0.09
C15CC3	1309.52	0.16	2705.07	2.14	400.31	231.83	111.12	132.10	7.29	307.20	8.43	11.90	7.18	3.33	0.12
C15SD1	1150.68	0.22	2688.21	0.82	419.84	262.95	170.46	130.03	6.59	358.76	8.77	10.04	6.83	2.97	0.12
C15SD2	783.45	0.01	694.48	1.01	423.46	186.70	103.07	71.77	5.06	93.44	2.17	1.84	6.64	0.88	0.04
C16CC1	673.19	0.05	622.93	0.76	383.15	156.90	68.71	56.02	7.03	125.20	4.55	2.94	6.32	0.81	0.04
C16CC2	880.19	0.01	1455.76	0.70	297.02	164.20	106.31	119.33	6.05	134.04	6.02	5.93	6.81	1.65	0.08
C16CC3	1348.66	0.01	786.53	0.72	233.23	147.18	81.66	54.75	6.32	91.78	4.69	2.28	6.32	1.26	0.06
C16SD1	631.20	0.01	404.26	0.50	387.98	99.44	68.87	41.83	5.06	84.71	3.33	1.33	5.43	0.98	0.04
C16SD2	496.49	0.01	520.62	0.43	329.95	69.71	51.75	51.06	4.87	42.48	2.29	1.17	6.01	0.67	0.03
C16SD3	1927.02	0.08	528.53	0.62	256.25	194.30	68.99	58.47	7.68	170.72	5.82	2.79	6.36	1.07	0.04
C17CC1	1012.79	0.30	1780.67	1.18	478.36	296.52	182.52	111.33	7.65	226.10	6.83	4.44	7.00	1.66	0.08
C17CC2	952.42	0.01	1797.30	0.88	496.48	371.36	204.67	85.67	5.21	254.55	6.85	2.99	7.42	1.10	0.06
C17CC3	947.79	0.11	1326.50	1.24	416.77	242.49	153.29	120.19	8.72	147.59	6.07	2.58	6.52	1.35	0.07
C17SD1	682.83	0.36	1804.98	3.07	359.58	229.90	256.05	178.25	6.96	79.11	6.20	3.99	6.93	1.71	0.07
C17SD2	642.90	0.46	2097.48	4.02	367.65	282.14	291.66	224.88	7.15	101.52	5.80	5.74	6.97	1.72	0.07
C17SD3	592.26	0.24	1889.93	2.50	261.15	215.93	226.89	202.18	7.25	46.24	7.16	5.95	7.00	1.55	0.06
C20CC1	1276.73	0.01	1743.54	1.17	319.78	212.87	117.86	96.92	8.55	79.01	7.43	2.25	6.88	1.94	0.08
C20CC2	761.79	0.01	1220.34	1.29	289.59	114.95	132.22	58.10	8.33	30.03	3.89	0.83	6.56	1.33	0.05
C20CC3	1436.69	0.01	1803.43	1.44	267.69	157.70	116.60	125.61	11.74	106.83	6.79	3.26	6.48	2.69	0.10
C20SD1	984.29	0.26	2290.17	2.39	354.63	258.02	150.36	205.19	10.60	60.25	12.04	6.09	6.74	2.88	0.12
C20SD2	1377.72	0.57	2317.76	2.65	354.49	361.86	116.96	153.47	9.96	192.34	10.39	6.80	7.23	2.26	0.10
C20SD3	738.62	0.01	1923.83	3.78	366.11	161.23	197.03	72.57	6.85	40.31	4.94	1.53	6.73	1.63	0.08
C35CC1	824.68	0.23	2814.13	1.91	415.76	477.04	212.59	140.65	14.58	169.12	8.41	9.55	7.35	2.08	0.10
C35CC2	864.29	0.07	2640.88	6.17	346.86	528.69	302.77	137.18	13.26	100.08	10.18	8.19	7.25	1.93	0.11
C35CC3	957.14	0.21	2022.81	3.30	313.78	447.52	223.88	96.89	10.45	85.60	14.31	8.01	6.62	1.81	0.09
C35SD1	869.65	0.16	2431.96	3.39	287.74	423.41	233.95	178.24	7.21	111.09	8.60	11.37	7.35	2.05	0.09
C35SD2	675.73	0.39	2008.70	2.74	258.10	352.81	191.60	188.31	11.08	93.67	10.70	17.39	6.95	2.22	0.12
C35SD3	778.28	0.64	2445.27	4.70	322.73	549.30	222.26	201.98	8.56	157.50	13.39	13.50	7.09	2.36	0.14
C38CC1	978.71	0.12	2558.65	2.02	303.89	465.84	218.72	185.65	10.90	120.09	6.35	14.73	6.93	3.12	0.14
C38CC2	902.58	0.16	2982.38	1.33	275.57	316.43	260.35	224.73	7.61	102.73	8.97	11.83	7.07	2.41	0.14

Appendix II. Soil chemistry analysis results for composite soil samples (in PPM except where noted).

Plot Number	Al	B	Ca	Cu	Fe	K	Mg	Mn	Na	P	S	Zn	pH	C%	N%
C38SD1	1072.13	0.56	2575.04	2.71	316.43	626.05	295.37	238.23	14.30	139.96	9.95	10.64	7.21	2.15	0.13
C38SD2	845.55	0.78	3521.40	2.16	278.99	648.66	296.52	235.08	9.00	119.90	11.82	27.31	7.04	3.70	0.25
C38SD3	853.98	1.32	4006.40	3.50	323.89	931.56	268.55	296.05	6.09	269.20	13.74	18.22	7.63	2.42	0.14
C39SD1	673.96	0.01	1088.27	1.84	329.23	161.86	148.79	36.49	11.38	41.36	2.89	0.96	5.89	1.09	0.05
C39SD2	876.30	0.01	1149.85	2.34	350.07	206.16	161.12	88.26	9.83	48.40	5.73	1.63	5.75	1.29	0.07
C39SD3	869.29	0.01	1376.25	3.39	372.64	311.40	228.00	70.51	11.42	107.31	5.54	2.24	6.04	1.44	0.07
C44CC1	838.51	0.01	1675.77	1.62	337.90	255.02	165.61	121.03	6.33	81.47	4.31	2.55	6.62	1.64	0.08
C44CC2	679.35	0.01	1496.13	1.64	377.50	180.36	139.49	99.71	5.18	98.90	4.96	1.77	6.79	1.36	0.06
C44SD1	806.96	0.01	1916.39	1.82	311.36	223.12	239.39	81.58	6.25	51.03	4.56	0.71	6.86	1.26	0.06
C44SD2	793.94	0.01	2201.18	0.71	299.84	275.46	227.23	128.78	7.22	71.71	5.12	2.28	6.74	1.67	0.09
C47CC1	1371.50	1.16	4036.41	3.53	328.79	673.55	264.31	300.40	15.52	187.55	17.41	4.43	7.39	3.13	0.16
C47CC2	926.10	0.67	2946.61	4.12	290.42	447.88	341.72	280.46	10.12	109.05	9.65	12.08	6.90	2.92	0.15
C47CC3	915.06	1.51	3772.64	5.35	291.23	485.90	457.76	253.15	12.54	104.06	9.98	10.71	7.04	3.88	0.21
C47CW1	1240.23	0.01	1893.36	1.66	342.31	278.97	227.75	192.21	10.63	120.66	9.17	8.55	5.93	2.54	0.15
C47CW2	945.31	0.01	2178.58	2.34	326.58	279.67	221.77	236.58	6.63	299.52	5.88	6.17	6.17	2.20	0.11
C47SD1	895.47	0.01	1431.17	1.58	234.41	246.05	152.59	86.40	5.45	19.65	4.36	1.55	6.59	1.41	0.07
C47SD2	725.59	0.11	3183.09	2.64	262.80	183.56	207.17	224.25	8.68	100.42	8.08	11.30	7.14	2.86	0.14
C47SD3	842.61	0.34	2697.71	6.43	323.71	350.31	333.18	194.93	7.67	86.36	8.37	9.72	6.93	2.40	0.09
C50SD1	702.71	0.01	1531.88	0.78	456.25	280.81	164.86	102.31	4.00	154.36	6.36	2.74	7.04	1.13	0.05
C50SD2	600.65	0.01	2003.81	1.16	364.17	223.26	143.54	82.72	3.85	113.99	5.85	3.52	7.25	1.32	0.06
C50SD3	546.68	0.23	1862.39	2.36	376.86	264.65	201.70	83.09	4.55	68.48	5.32	2.55	7.23	1.50	0.06
C51SD1	709.43	0.01	1624.42	1.79	257.66	228.32	275.55	79.77	11.28	55.27	5.04	2.01	6.67	1.40	0.07
C51SD2	785.65	0.26	2445.68	2.67	313.18	492.94	423.32	126.59	7.17	131.98	7.48	5.34	7.67	1.86	0.09
C51SD3	695.34	0.36	2522.83	3.89	311.85	564.63	343.79	166.74	6.04	108.79	5.60	2.73	7.54	1.44	0.08
C52CC1	742.95	0.01	1876.66	2.34	217.54	278.29	347.23	148.82	10.44	52.13	5.65	3.59	6.78	1.69	0.09
C52CC2	675.69	0.11	2162.93	2.31	239.66	364.85	305.39	198.34	9.06	81.71	5.78	4.56	7.33	1.88	0.10
C52CC3	745.44	0.12	2696.29	1.54	290.16	258.82	300.35	252.38	11.32	72.02	7.76	6.86	7.00	2.49	0.12
C53CC1	805.42	0.21	2304.06	3.83	361.47	496.28	296.55	178.62	4.92	154.38	5.66	4.04	7.21	1.56	0.11
C53CC2	591.64	0.05	1991.74	5.44	214.44	418.76	302.85	86.50	6.79	63.02	4.31	1.40	7.40	1.39	0.08
C53CC3	700.51	0.28	1890.63	2.89	300.03	369.25	358.03	158.58	7.66	73.23	5.24	4.88	7.05	1.91	0.09
C54CC1	955.37	0.01	1665.15	2.64	253.88	308.26	215.25	97.93	9.40	25.68	7.20	3.44	6.01	1.81	0.11
C54CC2	926.53	0.46	3035.85	3.77	321.31	613.91	250.23	201.74	9.12	163.81	9.33	20.15	7.19	2.43	0.14
C54SD1	834.10	0.01	3116.29	4.11	228.78	456.54	248.61	141.23	11.62	18.09	8.54	11.22	6.83	3.37	0.21
C54SD2	904.68	0.01	3231.55	5.37	276.04	335.53	279.59	91.88	14.72	59.64	9.97	13.15	6.79	3.00	0.18
C55SD1	631.94	0.19	1393.98	4.15	243.99	212.76	266.96	111.41	4.80	45.81	3.86	1.56	7.04	1.09	0.06
C56SD1	626.32	0.38	2263.16	3.18	239.31	530.49	362.62	179.51	5.67	69.44	4.44	2.62	7.66	1.49	0.08
C56SD2	663.83	0.44	2327.58	3.55	248.33	588.71	387.76	218.31	8.46	59.28	5.85	2.92	7.45	1.77	0.11

Appendix II. Soil chemistry analysis results for composite soil samples (in PPM except where noted).

Plot Number	Al	B	Ca	Cu	Fe	K	Mg	Mn	Na	P	S	Zn	pH	C%	N%
C56SD3	798.72	0.01	2183.01	3.09	265.54	360.03	408.51	110.94	7.69	68.44	5.59	3.05	6.78	2.01	0.09
G06CC1	833.28	0.16	2649.89	5.44	237.55	275.99	232.88	155.11	10.12	73.18	4.30	3.10	7.07	1.88	0.09
G06CC2	1035.76	0.19	2404.19	2.60	359.84	272.71	260.14	94.29	8.74	155.57	4.43	7.58	6.27	2.21	0.09
G06SD1	708.04	0.29	2079.34	5.29	273.39	334.94	245.29	143.13	5.89	80.12	3.91	4.47	7.16	1.69	0.08
G06SD2	495.63	0.69	2994.03	7.61	238.98	443.05	215.35	186.05	4.60	122.86	8.24	5.78	7.56	2.14	0.11
G09SD1	576.40	0.12	974.70	0.87	184.69	243.10	129.60	112.92	3.51	58.54	3.35	2.69	6.86	0.91	0.05
G09SD2	467.08	0.64	2610.73	2.96	282.12	635.78	226.40	113.26	6.01	137.92	6.90	3.85	7.45	1.59	0.09
G10CC1	961.73	0.20	2032.89	3.14	275.87	283.57	148.57	176.60	6.16	71.50	4.65	3.78	7.02	1.61	0.09
G10CC2	528.65	0.33	2836.75	6.50	339.72	215.07	155.03	136.88	4.43	77.46	5.48	5.30	7.26	2.20	0.10
G10SD1	510.37	0.03	1360.55	3.78	231.50	184.33	122.89	105.33	4.02	26.88	3.86	1.63	7.00	1.02	0.05
G10SD2	421.21	0.09	1627.65	5.31	293.68	187.60	116.06	91.53	3.56	31.22	3.28	2.23	7.04	1.56	0.07
G10SD3	847.03	0.21	1532.40	2.39	286.56	343.52	148.01	139.60	4.31	74.14	2.88	3.06	6.90	1.25	0.06
G11CC1	1329.96	0.37	1810.57	1.02	355.55	374.55	148.15	131.42	7.62	287.05	8.08	11.42	6.95	1.85	0.09
G11CC2	1049.40	0.42	1767.37	1.11	376.13	361.53	153.79	138.61	8.13	286.09	9.25	10.44	7.40	1.33	0.06
G11SD1	1517.18	0.07	1309.47	0.77	314.89	226.42	100.96	96.69	5.74	291.23	6.63	2.60	6.39	1.83	0.08
G11SD2	1001.79	0.35	2099.53	1.47	272.66	254.59	111.26	134.92	6.15	141.56	9.04	10.58	6.88	1.83	0.09
G12SD1	1562.88	0.26	2253.82	1.19	271.45	386.60	182.58	103.51	8.27	275.01	8.73	4.27	7.09	2.18	0.12
G12SD2	1214.04	0.72	1864.65	1.17	348.23	514.89	170.13	184.83	10.60	208.70	7.96	7.52	6.99	1.74	0.10
G12SD3	1604.80	0.49	2308.24	1.39	301.97	383.77	200.12	129.64	9.65	252.94	13.68	7.45	6.83	2.45	0.16
G13CC1	1089.22	0.26	2412.27	2.22	284.47	444.81	342.14	209.69	9.40	94.83	8.26	4.15	7.05	2.45	0.17
G13CC2	1336.16	0.21	1952.79	2.30	317.64	422.16	258.38	174.13	9.04	188.74	8.55	4.09	7.04	1.89	0.12
G13SD1	1426.62	1.35	3612.33	2.40	286.19	760.78	329.41	128.79	10.33	441.41	12.42	6.22	7.07	2.30	0.15
G13SD2	895.98	0.79	3248.55	2.01	192.34	632.33	450.68	155.81	8.27	57.91	10.11	4.30	7.12	4.12	0.30
G14CC1	904.74	0.90	3245.42	6.50	296.25	710.82	274.33	207.66	10.12	134.52	8.87	8.79	7.33	2.56	0.15
G15CC1	1612.52	0.13	1370.14	1.51	359.51	247.99	160.89	92.80	8.87	247.76	7.05	1.22	6.74	1.35	0.07
G15CC2	1397.92	0.25	1664.20	1.32	361.50	282.25	159.22	132.60	9.71	223.20	8.55	3.12	6.50	2.61	0.11
G15SD1	1639.76	0.28	2500.02	2.95	252.96	331.21	152.73	100.88	10.61	146.53	9.66	3.02	6.71	2.94	0.18
G15SD2	1694.94	0.55	2460.18	2.23	257.38	447.12	135.59	124.44	10.72	261.32	10.97	2.54	6.93	2.83	0.18
G20CC1	1884.84	0.04	504.34	0.58	194.10	161.48	67.68	14.16	7.10	133.76	6.01	0.51	6.52	1.40	0.05
G20CC2	2046.17	0.12	453.19	0.63	206.99	92.42	48.00	37.84	7.54	110.97	6.41	0.76	5.99	1.72	0.06