Invasive Species Removal and Changing Fire Regimes in a lək̓ʷəŋən Garry Oak Ecosystem

by

Cole Lysgaard
B.S., University of Washington, 2018

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We acknowledge and respect the lək̓ʷəŋən peoples on whose traditional territory the university stands and the Songhees, Esquimalt and W̱SÁNEĆ peoples whose historical relationships with the land continue to this day.

Supervisory Committee

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Abstract

This thesis examines restoration of Garry oak ecosystems in Southwestern British Columbia, Canada. Before the arrival of European settlers, Coast Salish peoples practiced intensive stewardship and cultivation practices that heavily shaped Garry oak ecosystems. These long-standing stewardship practices are responsible for the abundance of culturally important plants found in Garry oak ecosystems today. In addition to their cultural value to Coast Salish peoples, Garry oak ecosystems also support unique biodiversity, including numerous at-risk species. These ecosystems and the values they embody came under threat with the arrival of European settlers, who introduced non-native plants and excluded Coast Salish peoples and their stewardship practices from these ecosystems. Today, Garry oak ecosystems have been reduced to a fraction of their pre-colonial distribution and remaining patches are typically heavily invaded by both native and non-native plants. Their cultural and biological values coupled with ongoing degradation has motivated both Indigenous and non-Indigenous land managers to implement restoration programs in Garry oak ecosystems. To inform future restoration efforts, this thesis examines ecological impacts of a long-term restoration program and a wildfire in a lək̓ʷəŋən Garry oak ecosystem at Mill Hill Regional Park near Langford, British Columbia. In Project 1, vegetation responses to a 13-year invasive species removal program were quantified to determine if native plant populations were successfully bolstered by the removal efforts. In Project 2, impacts of an unintended wildfire on the relative cover of native and non-native plants were examined. This attempted to explore potential ecosystem shifts that may occur as wildfires increase in frequency and severity as predicted by climate models. In Project 1, the greatest change observed after invasive species removal was an increase in other introduced species, while increases in native species were not statistically significant. In Project 2, introduced *Anthoxanthum odoratum* was facilitated by fire while native *Camassia* spp. were reduced by it. Taken together, these results demonstrate the complexity of restoring Indigenously managed ecosystems where multiple introduced species have existed for long periods. Invasive species, specifically *Anthoxanthum odoratum*, showed greater responses to removal efforts and wildfire than native species. Intensive, long-term restoration programs that utilize multiple tools, including low-intensity fire, invasive removal, herbicide, and seeding of native species appear necessary to bolster native species without unintentional facilitation of introduced species. Coast Salish peoples and stewardship practices were integral in maintaining these ecosystems before

the arrival of European settlers and should play a key role in their restoration today, though traditional practices will likely need adapted to account for environmental changes caused by colonization. Furthermore, to avoid continuing the cultural damage that began with colonization, it is vital that Coast Salish First Nations lead or be directly involved in restoration of these ecosystems, which continue to hold irreplaceable cultural value.

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I am grateful to have lived and worked in lək̓ʷəŋən territory for two years following the start of my graduate program. I sincerely hope that my work will be of some use to the Songhees, Esquimalt, and W̱SÁNEĆ Nations in ecocultural restoration of their territories.

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Chapter 1: Introduction

This thesis was composed within the traditional unceded territory of the lək̓ʷəŋən peoples, including the Songhees and Esquimalt Nations. My office at the University of Victoria, field research site, and home at this time were all located in lək̓ʷəŋən territory. Completion of this work would not have been possible without access to these lands. lək̓ʷəŋən relationships to the land continue today and will not be ceded or extinguished. I acknowledge the traumatic history of colonization in Coast Salish territory, a process that continues to dispossess First Nations of their land and culture. As a descendent of European ancestry, though I do not intentionally perpetuate it, I acknowledge that I benefit directly from this history of colonization. Throughout this thesis, when referring to regional Indigenous peoples, I use the term Coast Salish, and when specifically rereferring to southeast Vancouver Island, including Mill Hill, I use the names lək̓ʷəŋən- and W̱SÁNEĆ-speaking peoples.

Thesis overview

Prior to European colonization, Coast Salish peoples shaped the landscapes around them through intensive stewardship and cultivation. The iconic Garry oak ecosystems of Vancouver Island, British Columbia (BC), were shaped by these processes, particularly through the cultivation of blue camas (*Camassia quamash* and *C. leichtlinii*) and the use of controlled burning. Colonization has caused widespread degradation of Garry oak ecosystems through the cessation of traditional stewardship practices, the introduction of exotic plants, and agricultural and urban development. The loss and degradation of Garry oak ecosystems is of great concern to First Nations and non-indigenous land managers due to the culturally and ecologically important plants that depend on them. This thesis discusses two research projects designed to answer pressing questions in the preservation and restoration of Garry oak ecosystems.

Two research projects were conducted, both studying the threat of invasive species in a lək̓ʷəŋən Garry oak ecosystem at Mill Hill Regional Park in Langford, BC (Figure 1). Project 1 examined a 13-year Scotch broom (*Cytisus scoparius*) removal program at Mill Hill and evaluated the program’s efficacy in restoring native species. Project 2 examined vegetation response to a wildfire at Mill Hill with the aim of determining whether native or exotic species benefited more from the fire. These two projects address four central research questions:

1) Did repeated removal of a dominant invasive species, Scotch broom, over several years facilitate increases in native abundance and richness? (Project 1)
2) Was the presence of Scotch broom a greater constraint for native or exotic plant abundances? (Project 1)
3) Was precipitation important in determining species abundances and how might this have influenced responses to Scotch broom removal? (Project 1)
4) Will an increased presence of fire in Garry oak ecosystems facilitate the recovery of native species or accelerate the spread of invasive species? (Project 2)

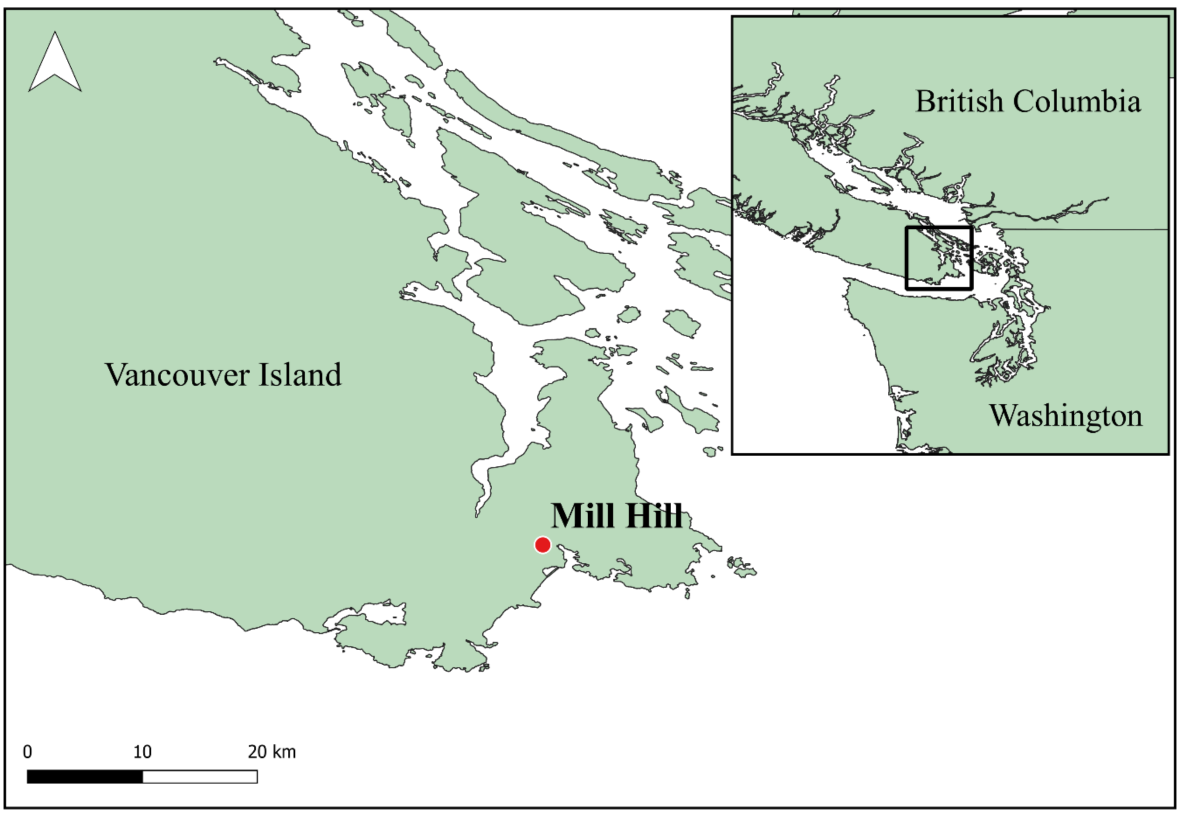

Figure 1. The location of the study site, Mill Hill Regional Park, where field sampling for both projects was conducted. Mapping data sourced from GADM (https://gadm.org/index.html).

Thesis organization

The remainder of Chapter 1 provides theoretical context and introduces the two research projects. This begins with background on Garry oak ecosystems, discussing their ecology, biodiversity, history of Coast Salish management, and present-day threats to their persistence. Next, Project 1 is introduced, including background on invasive species, challenges in the restoration of invaded Garry oak ecosystems, and research objectives. Chapter 1 ends with an introduction to Project 2, which discusses contemporary interest in reintroducing fire to Garry oak ecosystems, forecasted increases in wildfires due to climate change, uncertainties regarding vegetation response to fire, and research objectives. Chapter 2 contains methods used for Projects 1 and 2, including a shared site description, sampling designs, and data analyses used. Chapter 3 reports results from Projects 1 and 2. Chapter 4 discusses results for both projects, including how they address my research questions, how they relate to other research, and restoration implications. Chapter 5 concludes the thesis with recommendations for future research, overall management takeaways, and comments on the role that Coast Salish peoples, stewardship practices, and cultural values should play in the restoration of Garry oak ecosystems.

Garry oak ecosystems

Distribution, climate, biodiversity, and cultural significance

The Garry oak ecosystems of southwestern BC are a unique and iconic part of the region’s diverse natural landscapes. These ecosystems stretch along the Pacific coast from southern California northward, finding their northernmost distributions throughout the Strait of Georgia (Pellatt and Gedalof 2014). In contrast to BC’s rugged coastal mountains that draw recreationalists away from urban centers, Garry oak ecosystems are often a part of the urban environment. Their open canopies, dramatically gnarled trees, and stunning spring wildflowers make them some of the region’s most visited parks. Indeed, many of Victoria’s most popular urban parks—Beacon Hill Park, Uplands Park, and Mt. Tolmie are all preserved Garry oak woodland. Even residents of Southeastern Vancouver Island who may not know these ecosystems by name likely frequent them. Yet these ecosystems are valued for much more than their recreational use alone. The Garry oak ecosystems we know today have been shaped by a deep history of land stewardship and cultivation by Coast Salish peoples. They hold great cultural value and are home to many culturally important species. Garry oak ecosystems are also hotspots of biodiversity and host species found nowhere else in Canada.

Garry oak (*Quercus garryana*; ĆEN̲¸IŁĆ in SENĆOŦEN), the species for which these ecosystems are named, is a deciduous broadleaved hardwood tree and the only species of oak native to BC (Pellatt and Gedalof 2014). Garry oak is often associated with two evergreen tree species, broad-leaved Pacific madrone (*Arbutus menziesii*) and coniferous Douglas-fir (*Pseudotsuga menziesii*) (Gedalof et al. 2006). Canopy cover in Garry oak ecosystems is variable and can range from open savanna to shady woodland (Fuchs 2001; Gedalof et al. 2006). These ecosystems show higher floral biodiversity than any other terrestrial ecosystem in BC, including several endemic species (Fuchs 2001; GOERT 2002b). Understory communities are dominated by forbs (Shackelford et al. 2019), including obligates such as blue camas (*Camassia leichtlinii* and *C. quamash*), Henderson's shooting star (*Primula hendersonii*), western buttercup (*Ranunculus occidentalis*), field chickweed (*Cerastium arvense*), and wholeleaf saxifrage (*Micranthes integrifolia*) (MacDougall and Turkington 2007). Grasses are also common, including native species such as California brome (*Bromus carinatus*), California oatgrass (*Danthonia californica*), blue wildrye (*Elymus glaucus*), Idaho fescue (*Festuca idahoensis*), and Junegrass (*Koeleria macrantha*) (Proctor 2013). Garry oak ecosystems of BC fall within the Coastal Douglas-fir Biogeoclimatic zone and typically experience mild, wet winters and dry summers (Beckwith 2004; MacDougall 2005). Garry oak ecosystems may exist on either deep or shallow soils, including coastal bluffs with exposed bedrock (Beckwith 2004).

Since the arrival of European settlers in the late 18th century, Garry oak ecosystems have been devastated by land development, encroachment of late-successional species, and invasion by introduced plants (Beckwith 2004; MacDougall and Turkington 2007; Pellatt and Gedalof 2014). Due to these impacts, Garry oak ecosystems occupy less than 2000 hectares today—only ten percent of their distribution prior to European colonization (Dunwiddie and Bakker 2011; Vaino 2011; Clements 2013). Much of that remaining fraction is severely degraded by diversity loss and invasion by both exotic and native species so that less than five percent of pre-colonial Garry oak distribution remains with intact native species assemblages (Figure 2)(GOERT 2002b; Dunwiddie and Bakker 2011). This decline has resulted in concerning decreases in many species that depend on these ecosystems. Including all taxa, 104 species associated with Garry oak ecosystems are designated as at-risk within BC (Vaino 2011), including 70 plant species

(Clements 2013). Across the entirety of Canada, Garry oak ecosystems account for ten percent of the country's total at-risk species despite their relatively small distribution (Vaino 2011). The threats to Garry oak distribution and health are on-going, and without concerted intervention associated species will likely face local extirpation and global extinction (GOERT 2002b).

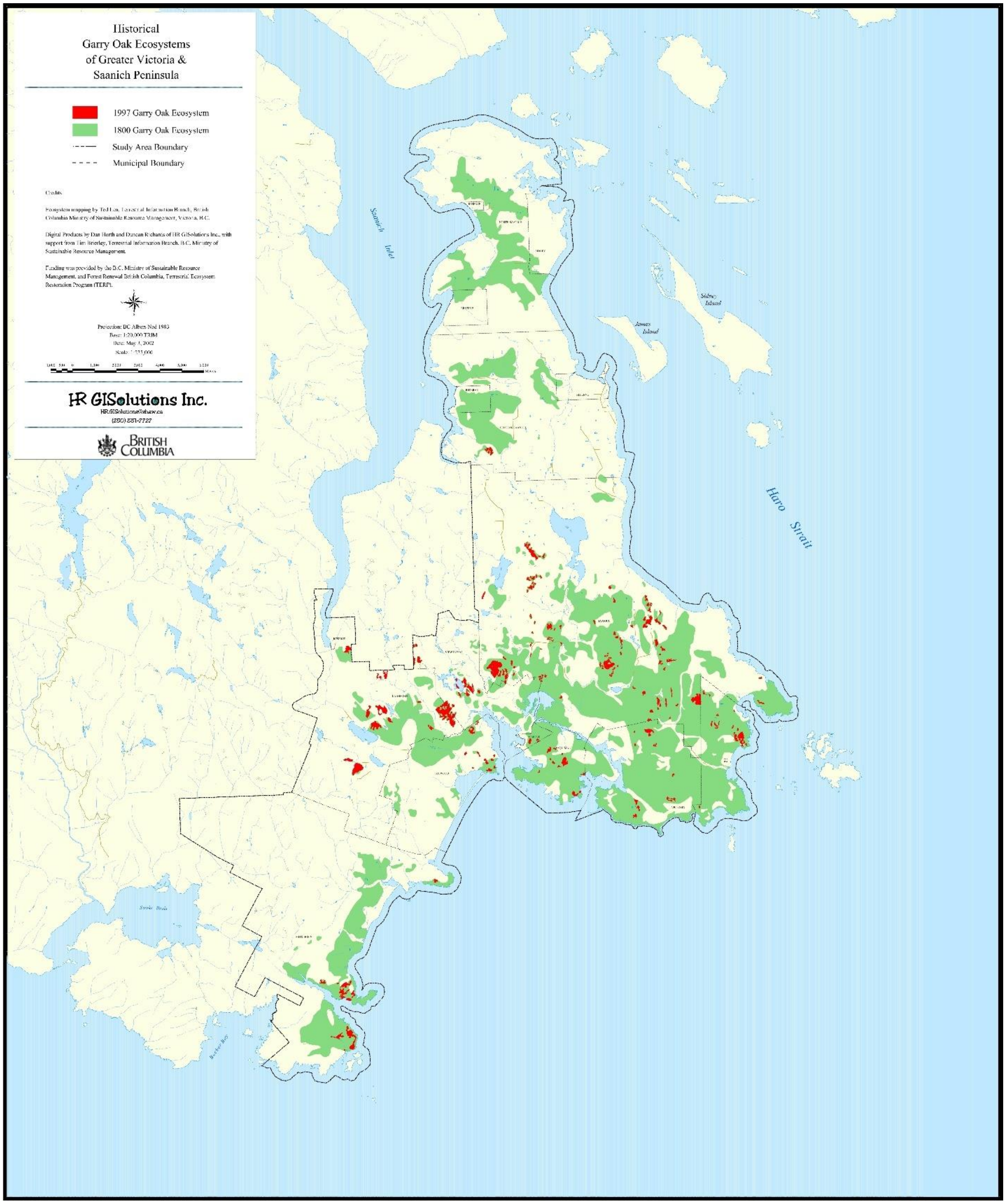

Figure 2. Garry oak ecosystem distribution in 1997 (red) and 1800 (green) in greater Victoria and the Saanich Peninsula (GOERT 2021).

Garry oak ecosystems have been vital to Coast Salish cultures for millennia and continue to hold unique value for First Nations. Before the arrival of Europeans, these ecosystems were the center of intensive cultivation practices that provided vital dietary staples, especially camas (discussed in detail below). Culturally important species including camas, barestem desert parsley (*Lomatium nudicaule*), and chocolate lily (*Fritilaria affinis*) depend on these ecosystems for survival. As stated by Garibaldi and Turner (p1, 2004), "These are the species that become embedded in a people's cultural traditions and narratives, their ceremonies, dances, songs, and discourse." Burial practices were also tied to these landscapes, and ancient Coast Salish burial cairns are often numerous in remaining Garry oak systems (Mathews 2016). Connections to these ecosystems through culturally important plants and burial practices are jeopardized by the widespread habitat loss and degradation that has occurred over the last century—yet another impact of colonization (Garibaldi and Turner 2004). If culturally important species that depend on these ecosystems are lost, so are the defining cultural connections to them. First Nations seek to restore these ecosystems and to reestablish connections to them that were intentionally broken by colonial authorities (Darcy Mathews pers. comm. 2021).

Garry oak ecosystems are Indigenous ecosystems

Garry oak ecosystems are not well suited to the climate of southwestern BC. These ecosystems expanded throughout coastal BC during a period of relatively warm weather then began to decline with the onset of cooler temperatures around 3,800 years before present (Fuchs 2001; Pellatt and Gedalof 2014). Since that date, the climate has favored infilling of these open ecosystems with trees and the transition to coniferous forest (Pellatt and Gedalof 2014). Yet despite a less-than-suitable climate, Garry oak ecosystems persisted extensively in BC and the Pacific Northwest until the arrival of Europeans. The cause for their persistence in spite of these conditions is unequivocal; tending and cultivation by Indigenous peoples, particularly through the use of controlled burning, maintained these ecosystems until such practices were repressed by European settlers (Suttles 1974; Turner 1999; Fuchs 2001; Beckwith 2004; Gedalof et al. 2006; Proctor 2013). Coast Salish peoples intensively cultivated vast areas along the coastline and surrounding suitable inland areas, providing regular and exacting disturbance regimes that promoted edible plants and ultimately shaped the unique ecosystems we know today.

The most influential form of management within these ecosystems was the cultivation of camas (ḰŁO¸EL; SPÁÁNW̱ in SENĆOŦEN). Camas is a perennial geophyte in the lily family highly valued by Indigenous peoples throughout northwestern North America for its edible and nutritionally important bulbs (Turner and Bell 1971; Proctor 2013). Two species are present in BC, collectively referred to as blue camas (*Camassia leichtlinii* and *C. quamash*) and both are predominantly found in Garry oak ecosystems (Beckwith 2004; Proctor 2013). Camas was an integral staple in Coast Salish diets and was also an important commodity for trade throughout kin-based trade networks across the Northwest Coast (Turner and Kuhnlein 1983; Beckwith 2004; Proctor 2013). Camas was cultivated, harvested, consumed, and traded on a massive scale in Southeastern Vancouver Island and in the Gulf Islands. Estimates reckon that a single Coast Salish family could harvest approximately 260 kilograms (8,000-10,000 individual bulbs) each year (Beckwith 2004; Proctor 2013). Beckwith (2004) puts the cultural value of camas into perspective by stating that its significance to Coast Salish cultures has been equal to that of cedar and salmon. In fact, the historian John Lutz (p66, 2008) states that “Although usually remembered as the “salmon people,” the Straits Salish [lək̓ʷəŋən and W̱SÁNEĆ] and other Coast Salish groups could just as accurately be called the “camas people.””

Coast Salish peoples, especially the lək̓ʷəŋən- and W̱SÁNEĆ-speaking peoples of Southeastern Vancouver Island, employed a number of strategies to harvest and maintain the productivity of extensive camas beds. Camas harvest took place during the summer after flowering was complete and seeds had begun to set (Turner and Kuhnlein 1983; Proctor 2013). Camas bulbs were traditionally dug using specialized digging sticks typically made of oceanspray (*Holodiscus discolor*) or Pacific yew (*Taxus brevifolia*) (Turner and Kuhnlein 1983; Proctor 2013). Large bulbs were collected while smaller bulbs were replaced in the soil to be collected in subsequent years (Turner and Kuhnlein 1983; Proctor 2013). The very largest bulbs were also likely replaced, so that they might continue to reproduce (Proctor 2013). Another practice designed to ensure a renewable harvest was the scattering of camas seeds into the freshly churned soil (Turner and Kuhnlein 1983; Proctor 2013). Rocks and weeds—particularly camas’ toxic look-alike, death camas (*Toxicoscordion venenosum*)—were removed to further increase productivity (Turner and Bell 1971; Turner and Kuhnlein 1983; Proctor 2013). Digging, tilling, and weeding increased soil aeration and water-holding capacity while also providing disturbance and cumulatively would have played an important role in shaping structure and species assemblages of Garry oak ecosystems (Beckwith 2004; Turner and Peacock 2005; Proctor 2013).

One of the most impactful methods used to maintain camas productivity was the use of fire. Typically, in late summer or early fall, once harvest was complete, camas beds would be burned (Turner and Bell 1971; Turner and Kuhnlein 1983; Proctor 2013). Interviews with First Nations elders and accounts from early European settlers indicate that these fires were typically of low intensity and occurred every few years at a given camas bed (Turner 1999; Beckwith 2004; Proctor 2013). A precisely executed low-intensity fire would have combusted plant materials above ground while leaving soil organics intact and belowground bulbs and rhizomes alive and capable of resprouting in the freshly cleared land (Antos et al. 1983; Turner 1999; Beckwith 2004). Achieving a low-intensity fire, however, was not a simple task and required detailed knowledge of how vegetation humidity, fuel quantity, landscape slope, wind speed, interval between burns, and predicted weather would influence a fire’s intensity (Turner 1999; Knapp and Keeley 2006).

Fire influenced these ecosystems through altered soil chemistry and regular disturbance. Burning may have bolstered plant growth by increased soil nutrient availability in the form of alkaline ions such as phosphorus, calcium, magnesium, and potassium, also resulting in increased pH (Turner 1999; Turner and Peacock 2005). However, even at low intensity, fire has been noted to reduce soil moisture and nitrogen, both important in vegetation recovery (Beckwith 2004). Finally, regular disturbance from fire was likely one of the most important drivers of Garry oak ecosystem structure. Even low-intensity burns would have caused high tree and shrub seedling mortality, preventing infilling and transition to closed-canopy climax forest (Turner 1999; Gedalof et al. 2006; Proctor 2013; Pellatt and Gedalof 2014). Burning every few years would have maintained camas beds in early- and mid-seral stages dominated by understory species. In a climate that favored shaded forest, this would have provided unique habitat for species dependent on disturbance. These impacts on disturbance and succession are why fire, through its use in promoting and sustaining camas harvest, has been the most important element in shaping and maintaining the Garry oak ecosystems that exist today.

Threats to Garry oak ecosystems: Invasive species and absence of Indigenous peoples

As stated above, Garry oak ecosystems have been reduced to a fragment of their former distribution and those that remain have been greatly degraded. Two of the main causes for this are invasion of non-native species and the exclusion of Indigenous peoples and their management practices (Dunwiddie and Bakker 2011). Both of these issues began with the arrival of European settlers in BC and both are responsible for widespread declines in native Garry oak ecosystem species.

Remaining Garry oak ecosystems have experienced declines in native species diversity due to rapid invasion by non-native plants (Fuchs 2001; Dunwiddie and Bakker 2011; Rook et al. 2011; Shackelford et al. 2019). In addition to compromising ecosystem health, these declines put culturally important plants at risk of extirpation. Invasive plants can out-compete and displace native species and with ongoing presence can alter fundamental ecosystem structure and disturbance regimes (Mack et al. 2000; Fuchs 2001; Gaertner et al. 2012). Scotch broom (*Cytisus scoparius*), perhaps the most notorious invasive species in these ecosystems, can drastically transform open savanna ecosystems into monoculture thickets in which native understory species are excluded (Fuchs 2001; Rook et al. 2011). Several species of exotic grasses are also widespread in Garry oak ecosystems and similarly out-compete and displace native species (Fuchs 2001; MacDougall and Turkington 2007; Rook et al. 2011). Exotic forbs are also common, though they tend to have a lesser impact on native flora in comparison to shrubs and grasses (Fuchs 2001). Together these taxa of introduced species have come to make up a major portion of Garry oak ecosystem plant communities. Exotic plants in BC have been found to occupy 59-82% of herbaceous cover in Garry oak meadows (Fuchs 2001). In Washington State, more recent work found that an average of 40% of plant species in Garry oak ecosystems were introduced (Dunwiddie and Bakker 2011). MacDougall et al. (2004) state that 151 exotic species are now naturalized in Garry oak systems. Furthermore, invasive species issues are only likely to become more complex with time, as proximity to roads and urban centers facilitates introduction of new species to Garry oak systems (Dunwiddie and Bakker 2011). Much time, energy, and research has gone into controlling the spread of invasive species, yet results have been mixed (Fuchs 2001). Developing restoration strategies that remove exotic species and return native ones without demanding prohibitively large investments from land managers is an ongoing challenge.

Colonial suppression of Coast Salish camas cultivation practices has also enabled native trees and shrubs to encroach on and replace Garry oak ecosystems and associated species. These practices, particularly the use of controlled burning, were quickly opposed and restricted by early European settlers (Turner 1999; Beckwith 2004; Proctor 2013). This has resulted in over a century of disrupted fire regimes in Garry oak ecosystems throughout BC (MacDougall 2005; Gedalof et al. 2006; Pellatt and Gedalof 2014). A direct consequence of this has been the infilling of these open ecosystems with trees and shrubs, resulting in a gradual replacement of Garry oak-associated species with late seral species and ultimately complete ecosystem conversion to climax coniferous forest (Fuchs 2001; Gedalof et al. 2006; Proctor 2013). These processes are a primary cause of Garry oak ecosystems' reduced distribution and continue to push them and obligate species towards extirpation today (Dunwiddie and Bakker 2011).

Another confounding consequence of excluding Indigenous burning is the accumulation of flammable materials in the form of both living plants and plant debris (Gedalof et al. 2006; Knapp and Keeley 2006; Ryan et al. 2013). The shrubs and trees responsible for infilling of Garry oak ecosystems provide ample combustible fuel, as do dead plant materials such as downed wood, dry grasses, and leaf litter (Knapp and Keeley 2006; Ryan et al. 2013). These abundant fuels can propel fires beyond the intensity of the quick-burning low-temperature fires traditionally used to clear camas beds after harvest (MacDougall 2005; Knapp and Keeley 2006; MacDougall and Turkington 2007). Uncontrolled high-intensity fires caused by heightened fuel loads will likely inflict considerable environmental damage, including mortality of belowground plant structures, mature trees, and soil organics, all of which can survive low-intensity burns (Turner 1999; Knapp and Keeley 2006). In effect, cessation of regular burning creates a feedback cycle in which burning becomes increasingly difficult to reintroduce the longer it is absent. The short return interval of Indigenous burning of camas beds not only encouraged growth of culturally important plants, but also guaranteed repeatability of low-intensity fires year after year.

Suppression of Indigenous burning practices by Europeans had a third consequence: loss of traditional knowledge. While ethnographic records contain many accounts from First Nations elders and knowledge holders regarding the use of fire in land stewardship, much knowledge has since been lost concerning the specific details of implementing a successful burn (Turner 1999; Beckwith 2004). Traditional burning practices required a precise and intimate understanding of the biotic and environmental variables that influenced the success or failure of a fire. This understanding was undoubtedly developed over millennia of carefully burning and attentively observing outcomes. This loss of knowledge, in addition to accumulated fuels, invasive species, and a changing climate, prevents a simple reintroduction of traditional burning to Garry oak ecosystems. These knowledge gaps will likely become apparent as burning is reintroduced today, and many lessons regarding a successful burn will likely need to be re-learned.

Project 1: Impacts of invasive species removal

One of the most recognized cultural and environmental consequences of plant invasion is a reduction in native species diversity (Mack et al. 2000; Gaertner et al. 2012; Simberloff et al. 2013). This is a concern for more than biodiversity alone: loss of native species can have cultural repercussions when culturally important species are lost (Corntassel and Bryce 2012). There are several mechanisms through which invasive plants can suppress and displace native flora. Competition is one, wherein invaders out-compete native species for resources such as light, water, pollinators, and soil nutrients (Gaertner et al. 2012). Another is the ability of invaders to incite fundamental changes in ecosystem structure and disturbance regimes (Mack et al. 2000; Gaertner et al. 2012). For instance, introduced grasses in Northern Australia produce flammable materials that cause fires to burn at much higher intensity than in uninvaded areas (Buckley et al. 2007). Likewise, the structure of an open-canopied grassland may shift with infilling of invasive shrubs and trees (Nowacki and Abrams 2008). Additionally, invasive plants may alter a variety of soil characteristics. Common means of soil modification include nitrogen increases, water depletion, and changes to the microbial community, all of which can challenge the persistence of native species (Rook et al. 2011; Gaertner et al. 2012; Davis et al. 2021). These ecosystem-level changes can put the invaders at a competitive advantage and at extremes these altered ecosystems may no longer be suitable for the native species that lived there prior to invasion. This can result in a positive feedback cycle in which invasive dominance encourages favorable conditions for invasion and vice versa (Mack et al. 2000; Buckley et al. 2007; Gaertner et al. 2012). If left unchecked, heightened competition and ecosystem alteration caused by invasion can ultimately result in extirpation of native species.

It is imperative that land managers be equipped to combat the negative impacts of plant invasion. Ideally, introduced species are dealt with early on, before they have the opportunity to become wide-spread (Mack et al. 2000; Simberloff et al. 2013). In many ecosystems, however, land managers must address invasive species that are already well-established across a wide range. This issue becomes compounded when multiple invasive species are present. In such situations—when multiple well-established invaders co-exist—effective restoration becomes extremely complex and challenging. Land managers often address advanced invasion through removal of the most dominant invader (Hazelton et al. 2018; Shackelford et al. 2019). When effective, this approach is desirable for its efficiency. Ideally, removal of the dominant invader would allow native species to repopulate in its place without additional intervention (Hazelton et al. 2018; Shackelford et al. 2019). This is not guaranteed, however, and even if removal of the dominant invader is feasible, say through cutting, herbicide, or biological control, it may not have the desired effects (Cordell et al. 2016). The invader may quickly repopulate after removal, especially in ecosystems where it has been present long enough to alter soil characteristics in its favor (Gaertner et al. 2012; Simberloff et al. 2013). Furthermore, when multiple introduced species are present, the dominant invader may constrain other potential invaders as much as or more than native species. In such cases, removal of one invader may only facilitate its replacement with another (Buckley et al. 2007; Gorman 2011; Cordell et al. 2016). Such setbacks can be extremely costly, negating the often-monumental investments of time and money required for removal. Given these risks, and the fact that dominant invasive removal continues to be a popular and seemingly intuitive solution, land managers need ecosystem-specific evidence that will allow them to predict whether removal is likely to achieve desired outcomes and whether it should be undertaken.

In Project 1, I examined the complexities of invasive species removal in a forb-dominated Garry oak savanna in BC, Canada. As discussed above, their cultural value, numerous associated at-risk plants, and vastly diminished range make Garry oak savannas a primary concern for First Nations and non-Indigenous land managers (Fuchs 2001; MacDougall and Turkington 2007; Dunwiddie and Bakker 2011). Invasive species are one of the greatest threats to these ecosystems, and in many cases account for large percentages of vegetation cover (Fuchs 2001; Dunwiddie and Bakker 2011). Intensive and long-term management of Garry oak savannas that combines manual invasive removal, herbicide application, prescribed fire, and seeding of native species is recommended in the literature, but is likely to be prohibitively costly in many scenarios (MacDougall and Turkington 2007; Dennehy et al. 2011; Gorman 2011; Hamman et al. 2011; Gaertner et al. 2012; Tahiliani 2019). Consequently, many restoration projects focus on removal of the dominant invader alone (Shackelford et al. 2019). This assumes that the dominant invader primarily constrains native species rather than other introduced species, and that native species will show the greatest increases following removal. This assumption seems tenuous, considering the ability of invasive species to quickly disperse into and colonize disturbed areas, such as those caused by clearing (Seabloom et al. 2003; Buckley et al. 2007). Furthermore, climatic factors such as precipitation may be just as important as competition with a dominant invader in determining species abundances in these drought-prone ecosystems. If this is true, then removal of a dominant invader may not achieve native recovery if climate is unfavorable. The outcomes of these removal projects are rarely documented, and their efficacy is uncertain (Shackelford et al. 2019).

To determine the effectiveness of removing a dominant invader at facilitating passive recovery of native species in Garry oak savannas, I examined vegetation response to a 13-year invasive species removal program at Mill Hill Regional Park near Victoria, BC. This work attempts to answer my first three research questions:

1) Did repeated removal of a single dominant invader, Scotch broom (*Cytisus scoparius*), facilitate passive increases in native abundance and richness?

2) Was the presence of Scotch broom a greater constraint for native or exotic species abundances? This question considered the possibility that Scotch broom limited exotic species as much as native species and questioned the assumption that removal solely benefited native plants.

3) Were growing season precipitation and drought important in determining yearly species abundances during a time of active Scotch broom removal? Answering this allowed me to compare the importance of climate in shaping species abundance to the importance of Scotch broom removal. These forb and grass-dominated communities respond quickly to annual climate, making this an ideal ecosystem in which to assess the role of precipitation in shaping species dynamics during and after invasion control.

Together, these questions were intended to address uncertainty around invasive species removal in Garry oak savannas, and to determine whether land managers can safely rely on passive native recovery following removal of a single dominant invader.

Project 2: Impacts of changing fire regimes on Garry oak ecosystems

Fire is returning to Garry oak ecosystems in the form of intentional burns set by land managers and in the form of unintentional wildfires. As discussed above, Garry oak ecosystems experienced regular low-intensity burns set by Coast Salish peoples centuries before the arrival of European settlers (Turner 1999; Fuchs 2001; Beckwith 2004). These long-standing practices created ecosystems that depend on regular disturbance, especially in the form of fire (Gedalof et al. 2006). Upon their arrival, settlers suppressed Coast Salish burning practices and effectively removed fire from these landscapes (Beckwith 2004; Pellatt and Gedalof 2014). In addition to the cultural damage inflicted, this removed a disturbance regime necessary for the persistence of these early successional Garry oak communities. Without fire, late successional plants, including shrubs and trees, have encroached on Garry oak woodlands and displaced their biodiverse and culturally important flora (Pellatt and Gedalof 2014). Today, many First Nations are actively pursuing plans to perform controlled burns in these ecosystems to revitalize an integral cultural practice while simultaneously restoring these ecosystems (Darcy Mathews pers. comm. 2021). The scientific community has also shown an intense interest in reintroducing fire to Garry oak ecosystems as a means of reversing encroachment and restoring biodiversity (Hamman et al. 2011; Nemens et al. 2019). Land managers and restoration scientists see unique potential in using controlled burning as a tool in the fight against invasive species, an endeavor that has few clear solutions (discussed above; Hamman et al. 2011). Plants native to Garry oak ecosystems are adapted to regular low-intensity fire and so it is often assumed that they will benefit from fire's return (Fuchs 2001; MacDougall and Turkington 2007).

Yet, since the arrival of European settlers, Garry oak ecosystems have experienced important changes that may complicate the reintroduction of fire. The first is the introduction of exotic species. Many introduced plants thrive in Garry oak savannas, displacing native flora and altering ecosystem conditions. In addition, introduced plants may respond positively to fire (Agee 1996; Fuchs 2001; Lesica and Martin 2003). If this is the case, then reintroducing fire may worsen conditions by accelerating the spread of exotic species at the expense of native species. Studies on the reintroduction of controlled burning show mixed results: in some instances, fire reduced exotic species while in others they were stimulated (Agee 1996; Dunwiddie 2002; MacDougall and Turkington 2007; Dennehy et al. 2011; Hamman et al. 2011; Livingston et al. 2016). This variation in results may be due to differences in the species being monitored, how fire was conducted, how long vegetation was monitored post-fire, other management strategies applied, and climatic conditions (Hamman et al. 2011). Continued research is needed to determine whether and how fire can be reintroduced without accelerating invasion.

Another challenge in the reintroduction of fire is an accumulation of combustible fuel. Fire suppression over the last century has enabled infilling of shrubs and trees and accumulation of flammable plant debris that supply fuel to potential fires (MacDougall 2005; Livingston et al. 2016; Nemens et al. 2019). With these heightened fuel sources, fires today have the potential to burn at higher intensities and for longer periods than the low-intensity fires traditionally used by Coast Salish peoples to clear camas beds (Tveten and Fonda 1999; Nemens et al. 2019). High-intensity fires may lack the ecosystem benefits of low-intensity burns and may instead be detrimental to ecosystem health (Hamman et al. 2011; Ryan et al. 2013). Mortality of belowground plant structures (roots, bulbs) and soil organics caused by high-intensity fires could drastically slow post-fire regeneration of target species (Turner 1999; Beckwith 2004). The consequences of high fuel loads can be mitigated by removing trees, shrubs, and litter prior to burning or by burning when plants have higher moisture content (Agee 1996; Knapp and Keeley 2006; Hamman et al. 2011). In pre-planned controlled burns, therefore, the risks associated with heightened fuel loads can often be mitigated. This is not the case with unintended wildfires, which can be propelled by heightened fuels to intensities capable of causing significant environmental damage (Ryan et al. 2013).

Naturally occurring and unintentional human-caused wildfires are likely to increase in the Pacific Northwest. Summer temperatures and droughts are projected to increase over the next century, leading to an extended fire season and increased severity of wildfires (Mote and Salathé 2010; Bachelet et al. 2011; Gaur et al. 2021). Wildfires are more likely than intentional prescribed burns to reach environmentally harmful intensities, especially with heightened fuel loads that are difficult to reduce preemptively. Firefighting techniques may also have environmental impacts that will become more widespread as wildfires become more frequent. As with prescribed burns, it is uncertain whether increased wildfires will facilitate fire-tolerant native species or accelerate invasion in Garry oak ecosystems (Bachelet et al. 2011).

In Project 2, I examined vegetation responses to a wildfire that occurred in July 2020 at Mill Hill Regional Park. Photo-point monitoring captured vegetation growth from April through May 2021, the first growing season following the fire. By comparing vegetation growth in burned and unburned areas, I addressed my fourth research question: will an increased presence of fire in Garry oak ecosystems facilitate the recovery of native species or accelerate the spread of invasive species? While this project examined impacts of an unintended wildfire, its results also addressed some uncertainties around native and exotic vegetation responses to prescribed fires.

Chapter 2: Fieldwork and Data Analysis Methods

This chapter discusses the research site, sampling designs, and data analyses used for Projects 1 and 2. Sampling design and data analyses are discussed under separate subheadings for the two projects. Research questions, the projects they correspond to, and the sampling and data analyses methods used to answer those questions are summarized in Table 1.

Table 1. Research questions and their corresponding projects, data sampling methods, and data analyses used.

Research Question	**Project**	**Data sampling method**	**Data analyses used**
1. Did repeated removal of a dominant invasive species, Scotch broom, over several years facilitate increases in native abundance and richness?	Project 1 Part A	Vegetation surveys recording percent cover of all plants in control and treatment quadrats (Scotch broom removed)	T-tests comparing native and exotic species abundances and richness between control and treatment quadrats
2. Was the presence of Scotch broom a greater constraint for native or exotic plant abundances? 3. Was precipitation important in determining species abundances and how might this have influenced responses to Scotch broom removal?	Project 1 Part B	Vegetation surveys recording percent cover of all plants in control and treatment quadrats (Scotch broom removed)	Models of six focal species—three native and three exotic. Focal cover was modeled as a function of Scotch broom cover and precipitation metrics
4. Will an increased presence of fire in Garry oak ecosystems facilitate the recovery of native species or accelerate the spread of invasive species?	Project 2	Vegetation cover collected through photo-point monitoring of burned and unburned plots throughout the growing season	Abundances of five focal groups were modeled as a function of treatment (burned or unburned) and survey duration

Mill Hill research site—Projects 1 and 2

Mill Hill Regional Park is located in Southeastern Vancouver Island in the city of Langford, BC. It is managed by the Capital Regional District (CRD) and comprises 71.33 hectares of primarily Garry oak savanna that falls within traditional territory of lək̓ʷəŋən-speaking Coast Salish peoples (Capital Regional District; Beckwith 2004). Like others of their kind, these open grassland ecosystems abundant in culturally important species are undoubtedly the legacies of periodic burning, harvesting, propagation, and other management techniques practiced by ləkʷ̓əŋən peoples prior to colonization (Suttles 1974; Turner 1999; Turner and Peacock 2005). Even today, the agricultural history of this site is strikingly apparent in the form of extensive fields occupied almost exclusively by edible blue camas (*Camassia leichtlinii* and *C. quamash*; Figure 3). Archaeological features also attest to long-term ləkʷ̓əŋən presence here. Several stone burial cairns were identified during an initial visit to the site by archaeologist Dr. Darcy Mathews. These cairns can be found around the periphery of formerly cultivated camas fields, a pattern that has been observed in Garry oak savannas throughout the region (Mathews 2016). Multiple shell middens are also found within park boundaries (Beckwith 2004).

Figure 3. Abundant blooming camas at Mill Hill in late April.

Like other Garry oak savannas, Mill Hill has a sparse overstory and an understory composed of forbs and grasses. Garry oak (*Quercus garryana*) is the most abundant tree species, while Douglas-fir (*Pseudotsuga menziesii*) and Pacific madrone (*Arbutus menziesii*) occur in lower densities. Common native understory species include blue camas, blue wildrye (*Elymus glaucus*), long-stoloned sedge (*Carex inops*), and hairy honeysuckle (*Lonicera hispidula*). Exotic species are abundant throughout the park and include Scotch broom (*Cytisus scoparius*), sweet vernal grass (*Anthoxanthum odoratum*), orchardgrass (*Dactylis glomerata*), Kentucky bluegrass (*Poa pratensis*), barren brome (*Bromus sterilis*), hairy cat's ear (*Hypochaeris radicata*), and dandelions (*Taraxacum* spp.). See Appendix 1 for a full list of species mentioned in this thesis, as well as their origin, scientific, common, and SENĆOŦEN names. As with all Garry oak ecosystems, Mill Hill contains species and communities of high conservation value, including two red-listed plant communities: *Quercus garryana/Bromus carinatus* (Garry oak/California brome) and *Quercus garryana/Holodiscus discolor* (Garry oak/oceanspray), both considered critically imperiled (S1) within BC (Beckwith 2004; B. C. Conservation Data Centre 2021). One red listed species, California buttercup (*Ranunculus californicus*), was found regularly during the study period, as well as blue listed whitetop aster (*Sericocarpus rigidus*).

In July 2020, 4.6 hectares on the east slope of Mill Hill Regional Park were impacted by wildfire (Figure 4). The most intensive firefighting efforts took place on the first day of the fire and included construction of handguards (shallow trenches dug to remove vegetation and other flammable materials) to prevent further fire spread, spraying with hoses, helicopter bucketing (water dumped from helicopters), and fire retardant dropped from a plane. Fire retardant was only applied to areas of anticipated spread and not on burning areas. Fire suppression activities continued at a lower intensity over the next four days, primarily through continued water spraying using hoses. Spraying was pressurized to dislodge soil and reach hot spots below the surface that might otherwise reignite. Firefighting details were provided by personal communication with Stuart Walsh, Senior Park Ranger with the CRD, 2021.

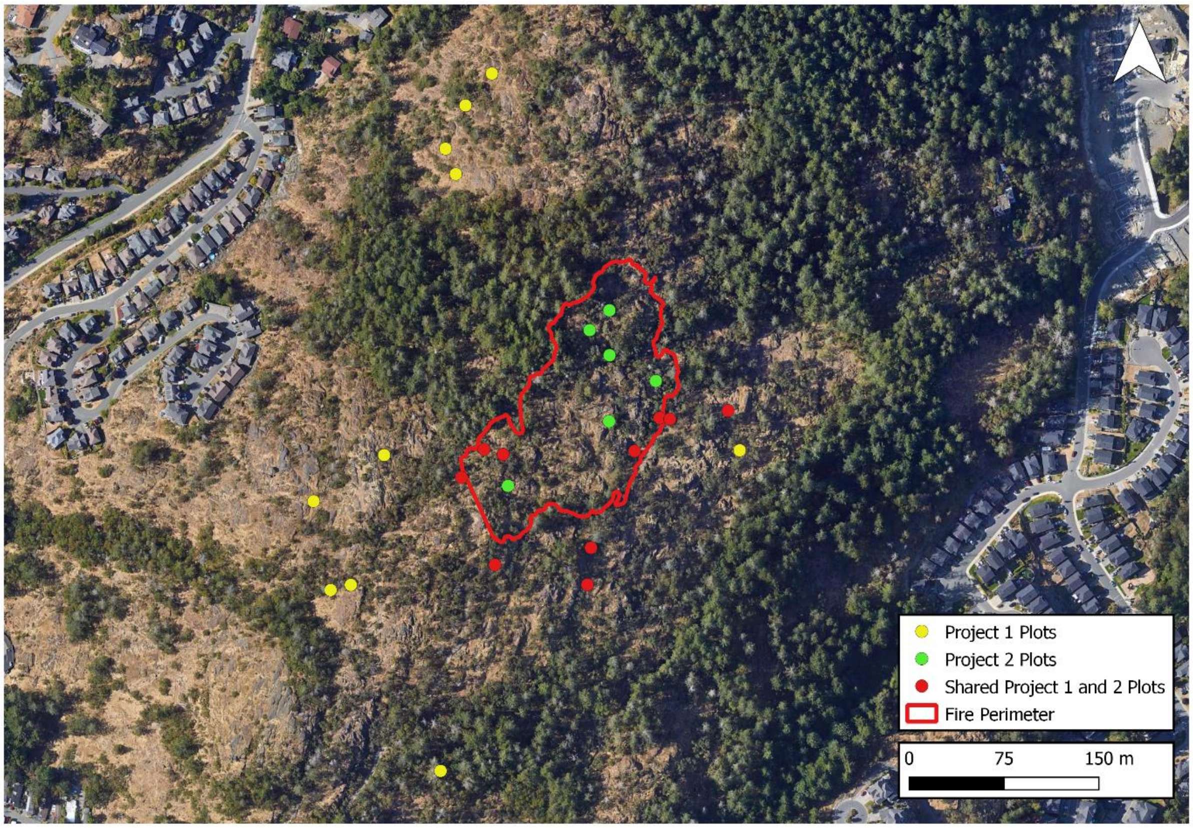

Figure 4. Satellite imagery of Mill Hill, including plot locations and area impacted by wildfire. Note that some plots were shared between Projects 1 and 2. In these plots, vegetation sampled by the two projects may have been adjacent but did not overlap.

Project 1 sampling design: Impacts of invasive species removal

Project 1 builds on a previous study by the CRD designed to examine the effects of invasive species removal at Mill Hill. The CRD study, including plot set-up and sampling design, is outlined by Maslovat (2008). As part of this study, twenty plots were established in June 2008 and spanned a range of elevations (150-192m) and aspects (0-330º; Figure 4). Plots were intentionally established in areas of uniform Scotch broom cover and moderate soil depth to make vegetation responses comparable. Twelve plots were in areas where Scotch broom had regrown from previous cutting, and eight were in stands of unmanaged Scotch broom. At each plot were two paired 3x3 meter quadrats (40 quadrats total), one randomly selected as a treatment quadrat and the other as a control. Baseline surveys were conducted for all quadrats in June 2008, in which all plants were identified and estimated for percent cover. After baseline surveys, Scotch broom and daphne (*Daphne laureola*) were removed in treatment quadrats by cutting stems at ground level while minimizing disturbance to other species. In 2009, 2010, 2012, 2013, and 2015, plots were resurveyed in June and Scotch broom and daphne were cleared from treatment quadrats thereafter. In 2019, Scotch broom was removed from quadrats in both treatments. As part of the present study, I conducted a final round of vegetation surveys in June 2021 on all but one of the plots which could not be relocated.

Project 1 data analyses: Impacts of invasive species removal

Two main analyses were conducted. Part A quantified overall impacts of Scotch broom removal on cover and richness of native and exotic species. Part B examined the influence of Scotch broom cover and climate on abundances of six focal species. All analyses were conducted in R and figures were produced using ggplot2 (Wickham 2016; R Core Team 2020).

Project 1 Part A analyses: Overall impacts of Scotch broom removal on cover and richness of native and exotic species

These analyses address my first research question by assessing to what degree passive native recovery occurred following Scotch broom removal and how it differed between control and treatment quadrats. Overall impacts of Scotch broom removal were assessed through two analyses, one of changes in vegetation cover and one of changes in species richness. Changes in cover and richness across the study period were calculated and compared between control and treatment quadrats using t-tests, and results were separated by species origin. Data used in t-tests were normally distributed, and Welch's t-test was used in one instance where variances were not equal. Data from 2021 were not used in these analyses since all quadrats—both control and treatment—were cleared of Scotch broom in 2019. These analyses therefore examined changes in cover and richness between 2008 and 2015.

For the first analysis, native species cover values were summed in each quadrat for both 2008 and 2015. Summed cover values for 2008 were subtracted from cover values in 2015 to calculate change for each quadrat. The same calculation was made for change in exotic species cover, but Scotch broom was intentionally left out to examine responses of species that were not part of the removal treatment. T-tests were used to compare cover change between control and treatment quadrats for native and exotic species.

For the second analysis, native and exotic richness were calculated in each quadrat. Next, changes in species richness were calculated for each quadrat from 2008 to 2015. Changes in richness for native and exotic species were compared between control and treatment quadrats using t-tests.

Project 1 Part B analyses: Influence of Scotch broom and climate on cover of six focal species

To address my second and third research questions, the abundances of six focal species were modeled to understand how Scotch broom cover and climate drove these communities. Focal species included the three most abundant native species—*Camassia* spp. (*C. quamash* and *C. leichtlinii* grouped), *Carex inops*, and *Elymus glaucus*—and the three most abundant exotic species—*Anthoxanthum odoratum*, *Hypochaeris radicata*, and *Poa pratensis*. This analysis used data from all survey years, including 2021, since treatment was not an explicit focus. Predictor variables were scaled and centered and included Scotch broom percent cover, total annual spring precipitation, and consecutive rain-free days in the spring. Spring precipitation and rain-free days were calculated using local climate data accessed online from Environment and Climate Change Canada. Spring precipitation and rain-free days were not correlated and had a correlation coefficient of -0.033. Random effects included treatment nested within plot and year. The glmmTMB package (Brooks et al. 2017) was used to model focal species cover. Models used a beta family distribution and logit link with adjustment for zero-inflation for all focal species except *Anthoxanthum odoratum* (Damgaard et al. 2019). Quantile-quantile plots, plots of simulated residuals vs. predicted values, and dispersion tests were generated through the DHARMa package (Hartig 2021) and used to validate model structure for each focal species.

Project 2 sampling design: Impacts of changing fire regimes on Garry oak ecosystems

In March 2021, 16 photo-point monitoring plots were established at Mill Hill, with half inside the area burned in 2020 and half in adjacent unburned areas (Figure 4; plot coordinates listed in Appendix 2). Plots were visited and photographed bi-weekly from early April to late May for a total of five visits each. Downward-facing photos were taken and each captured one square meter of surface area. The April to May monitoring period allowed photographs to capture the start through the end of the first growing season after the fire. Fire retardant was not applied to any burned plots during fire suppression efforts. Some burned plots did, however, show signs of soil disturbance from pressurized water spraying.

Following protocols established by Hall (2002), Eastman and Engelstoft (2011), and Lucey and Barraclaugh (2001), plot set-up and monitoring protocol were designed so that images taken during successive visits captured the same view as all previous visits at a given plot. To do this, the camera needed to be in the same position and focused on the same point for each photo. This was achieved by setting a tripod's height to 70 cm and placing its legs on permanent stake markers. Then the tripod center column was positioned at a 90-degree angle and oriented to face a predetermined azimuth (Appendix 2). Finally, the camera's crosshairs were placed on a permanent stake marking the center of each photograph (Figure 5). Ensuring that all photos at a given plot captured the same surface area was essential to make comparisons between plots and treatments meaningful.

Figure 5. Four sequential photographs of a lightly burned plot taken on April 21st, May 5th, May 18th, and May 30th (from left to right, top to bottom).

For consistency, all photographs were taken with a Canon PowerShot SX70 HS with 35mm equivalent zoom lens. Aperture and shutter speed were set to automatic with ISO manually set to 100. An auto-exposure bracket was used to automatically take three replicates of each image, one with higher, one with lower, and one with unaltered exposure (Eastman and Engelstoft 2011). Images were saved as both RAW and low-compression JPG files. Also, for consistency, a single tripod was used throughout the study.

Project 2 image analysis: Impacts of changing fire regimes on Garry oak ecosystems

SamplePoint image analysis software was used to measure vegetation cover in all photos (Booth et al. 2006). This software enabled images to be analyzed in a way that mimics conventional vegetation cover measurements in the field. For each photo, a grid of 100 points was overlaid on the image. Each point was then identified according to species or terrain type (soil, rock, litter, etc.). Identifying plants in images presented several challenges: images were occasionally blurred and it was impossible to touch or examine plants from different angles. Due to this, plants were identified to growth form when species were unclear. Once an image had been processed and all 100 points identified, the end product was an automatically generated Excel spreadsheet containing the total number of points classified as a given species, terrain type, or growth form. Since 100 points were classified for each image, this total also represented percent cover for that classification. These percent cover values were used in the analyses described below.

Project 2 data analyses: Impacts of changing fire regimes on Garry oak ecosystems

To answer my fourth research question, the relative abundances of five species and species groupings were modeled to determine how they were impacted by fire across the duration of the survey period. Focal species differed from Project 1 and included individual species and groups of species. The five most abundant groups were examined regardless of origin: *Camassia* spp. (including *C. quamash* and *C. leichtlinii*), *Anthoxanthum odoratum, Cytisus scoparius*, introduced asters (including *Hypochaeris radicata* and *Taraxacum* spp.), and *Bromus* spp. (including *B. sterilis*, *B. tectorum*, and *B. hordeaceus*). The introduced asters and *Bromus* spp. groups were used due to the difficulty of distinguishing species within these groups in photos and also due to their similarities in terms of origin and phylogeny. The response variable, relative percent cover, was calculated by dividing focal percent cover by total vegetation cover. This was important, since total vegetation cover was, on average, lower in burned plots and as low as zero for some observations. Predictor variables included treatment (burned or unburned), days since the first survey on April 7th (hereafter referred to as duration), and their interaction. Including duration in the models was important in accounting for large changes in vegetation throughout the growing season, changes that were not always synchronized between burned and unburned plots. Duration was scaled and centered. Plot was included as a random effect. Models were specified using the glmmTMB package (Brooks et al. 2017) with beta distribution and logit link (Damgaard et al. 2019). An argument for zero-inflation was included in models for *Cytisus scoparius*, *Bromus* spp., and introduced asters, but was not included for *Anthoxanthum odoratum* and *Camassia* spp. Quantile-quantile plots, plots of simulated residuals vs. predicted values, and dispersion tests were generated through the DHARMa package (Hartig 2021) and used to validate model structure for each focal group. All analyses were conducted in R and figures produced using ggplot2 (Wickham 2016; R Core Team 2020).

Chapter 3: Results

This chapter contains results obtained from data analyses described in Chapter 2. Results are reported with separate headings for Projects 1 and 2.

Project 1: Impacts of invasive species removal

Project 1 Part A results: Overall impacts of Scotch broom removal on cover and richness of native and exotic species

The dominant trend from 2008 to 2015 was an increase in vegetation cover across the study site—mean cover in control and treatment quadrats increased for both native and exotic species (Figure 6). However, increases in control quadrats were relatively low, with an average increase of 1.20 percent cover per quadrat for native species and 4.18 for exotic species. Treatment quadrats showed greater increases, especially for exotic species. In treatment quadrats, native species increased by an average of 4.73 percent cover per quadrat and exotics by 26.4 percent cover per quadrat—an increase over five times greater than that of native species. Average cover increases for exotic species were significantly different between control and treatment quadrats ($p = 0.0021$). The difference between control and treatment quadrats was not significant for native species, however ($p = 0.49$). The data further show that exotic increases in treatment quadrats were primarily due to a single species, *Anthoxanthum odoratum*, which accounted for 86 percent of total exotic increases in treatment quadrats.

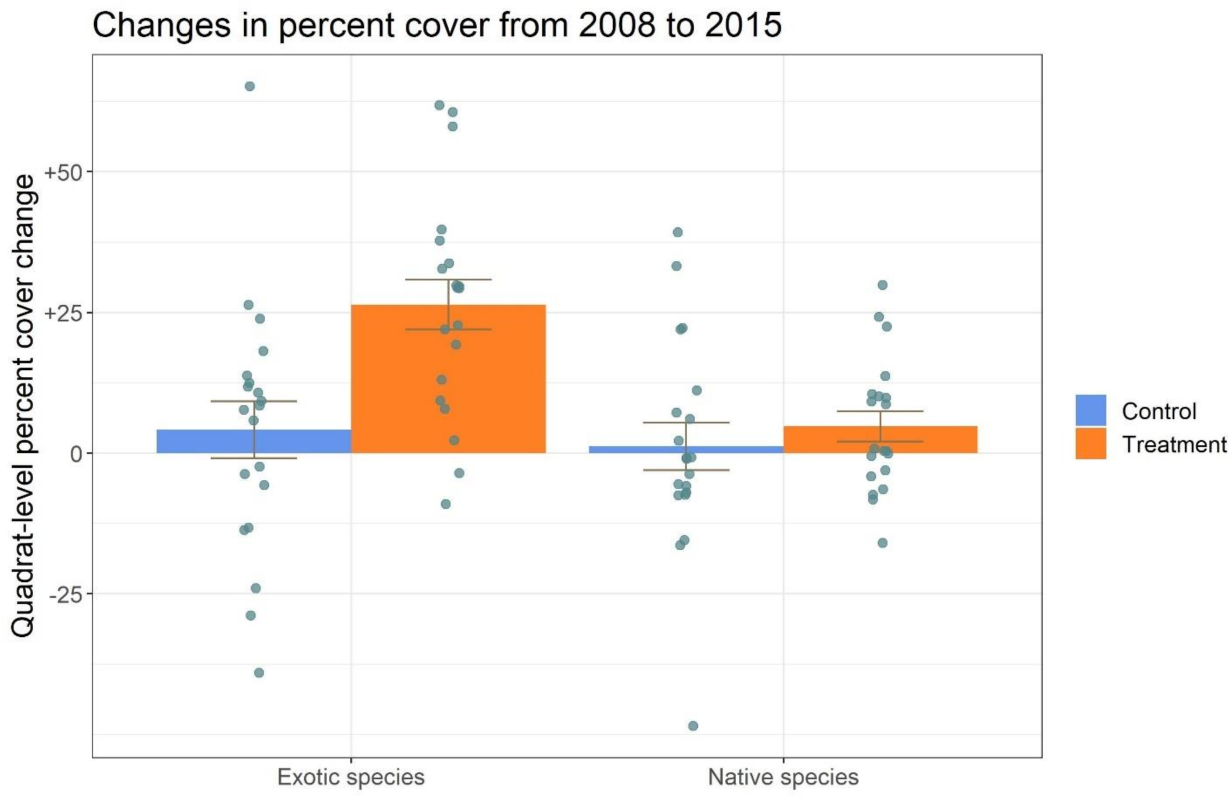

Figure 6. Changes in cover from 2008 to 2015 for all quadrats grouped by species origin and treatment. Dots indicate increases and decreases in percent cover at individual quadrats, bars indicate means ± SE. Means for control and treatment quadrats differed significantly for exotic species ($p = 0.0021$) but not for native species ($p = 0.50$).

Species richness also showed overall increases from 2008 to 2015 (Figure 7). Average species counts increased in both control and treatment quadrats for both native and exotic species. As with cover values, richness increases were greater for treatment quadrats for both native and exotic species. Control quadrats increased by a mean of 0.95 species for native species and by 2.1 for exotic species, while treatment quadrats increased by a mean of 2.1 for natives and 3.3 for exotics. Average richness change in control and treatment quadrats did not differ significantly for either native or exotic species, however. While exotic species did show a greater average increase in richness, especially in treatment quadrats, this was not nearly as pronounced as with cover increases.

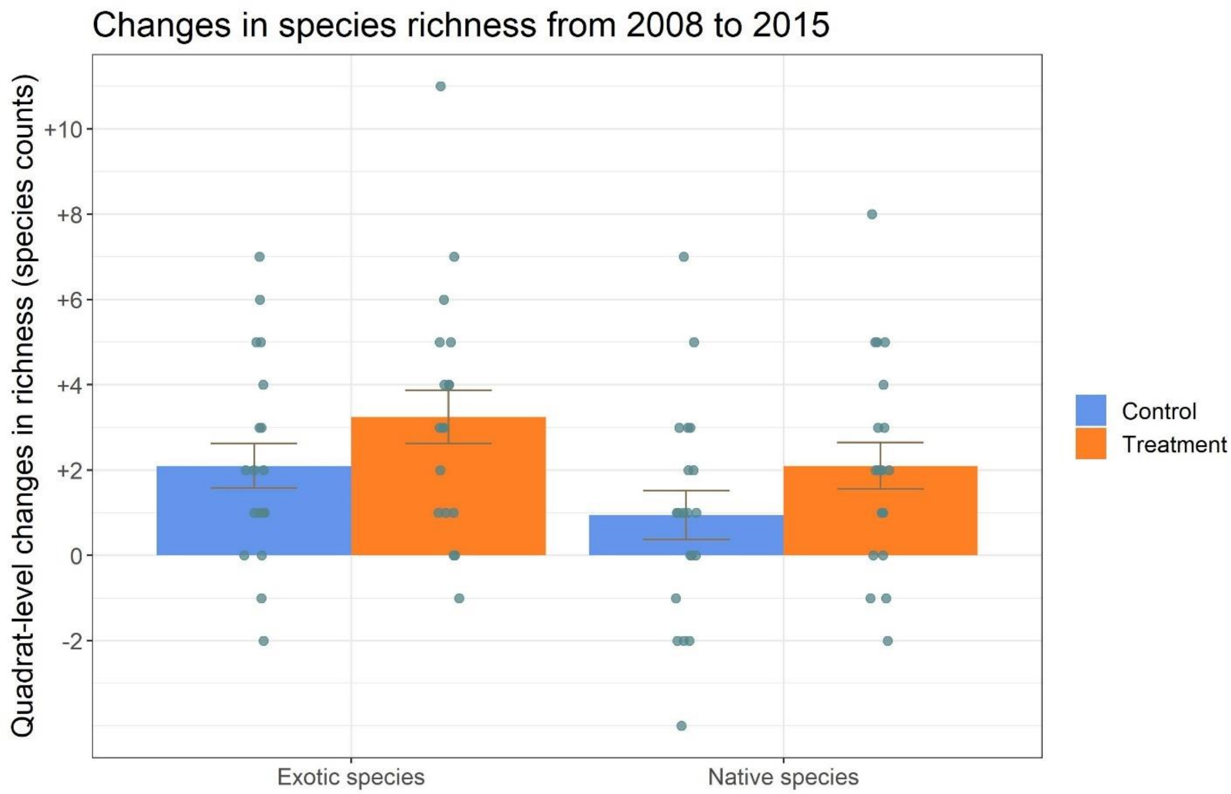

Figure 7. Changes in richness (species counts) from 2008 to 2015 for all quadrats grouped by species origin and treatment. Dots indicate species gained or lost at individual quadrats, bars indicate means ± SE. Means for control and treatment quadrats did not differ significantly for either exotic or native species ($p = 0.13$ and 0.21, respectively).

Project 1 Part B results: Influence of Scotch broom and climate on cover of six focal species

Scotch broom cover had a negative influence on all focal species and was significant for three exotic species and two native species (Figure 8 and Table 2). This influence was much greater in magnitude (more negative) for exotic species and greatest for *Anthoxanthum odoratum*. Spring precipitation had a significant negative relationship with *Carex inops*, *Anthoxanthum odoratum*, and *Hypochaeris radicata*. Consecutive rain-free days had a significant positive relationship with *Anthoxanthum odoratum* and a significant negative relationship with *Poa pratensis*.

Table 2. Model outputs for six focal species. Response variables are focal species percent cover. All predictor variables were scaled and centered and include Scotch broom cover, total annual spring precipitation, and consecutive spring rain-free days. Values indicate coefficient estimates and p-values (estimate / p-value). Bold font indicates significance at α = 0.05.

Species	**Native/ Exotic**	**Scotch broom**	**Precipitation**	**Rain-free days**
***Camassia* spp.**	Native	**-0.22 / < 0.001**	-0.049 / 0.69	0.080 / 0.52
Carex inops	Native	**-0.21 / 0.0055**	**-0.26 / 0.0076**	0.043 / 0.64
Elymus glaucus	Native	-0.073 / 0.51	0.14 / 0.17	-0.12 / 0.25
Anthoxanthum odoratum	Exotic	**-0.60 / < 0.001**	**-0.12 / 0.0051**	**0.14 / 0.0013**
Hypochaeris radicata	Exotic	**-0.50 / < 0.001**	**-0.38 / 0.0065**	0.24 / 0.091
Poa pratensis	Exotic	**-0.27 / 0.0051**	0.085 / 0.58	**-0.31 / 0.037**

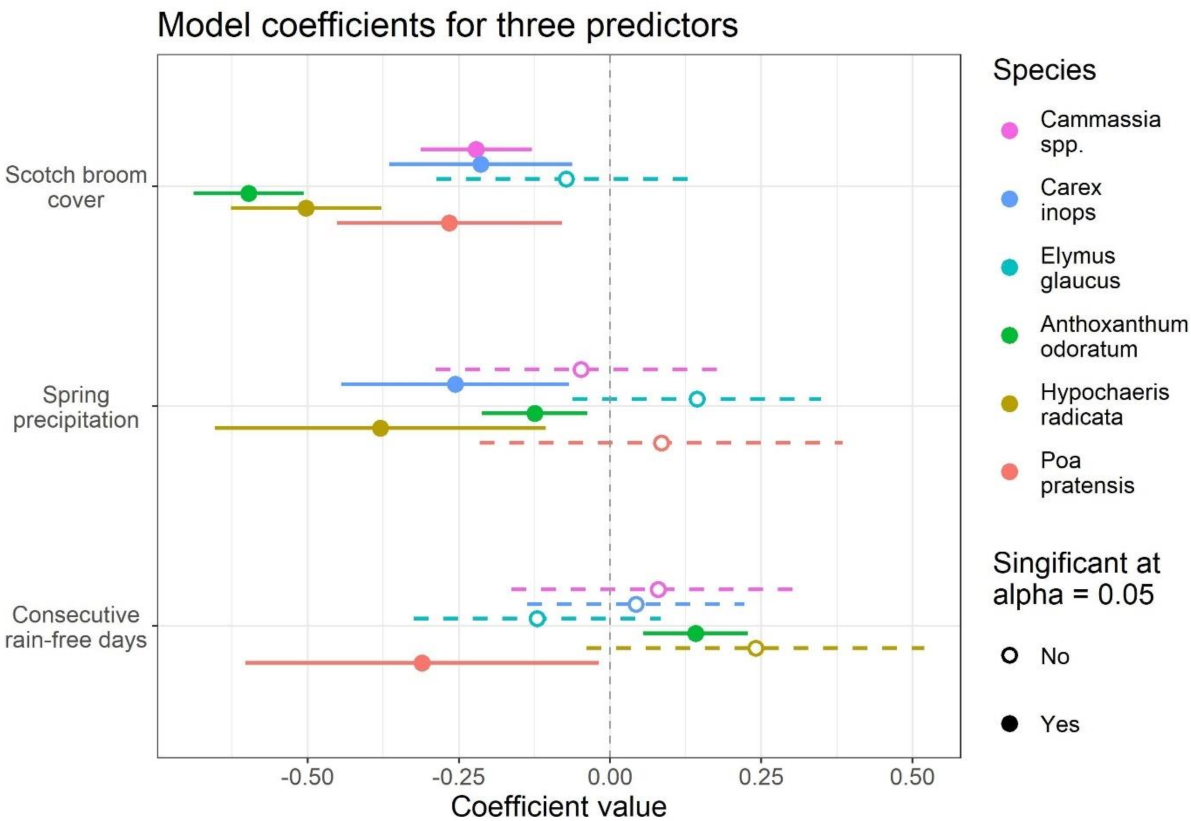

Figure 8. Model coefficients for six focal species. Points represent coefficient estimates and lines represent 95% confidence intervals. Predictor significance at α = 0.05 is represented by a closed point and solid line, while non-significant predictors have open points and dashed lines. All predictors were scaled and centered.

Project 2: Impacts of changing fire regimes on Garry oak ecosystems

Three of the five focal groups showed a significant relationship with predictor variables: *Camassia* spp., *Anthoxanthum odoratum*, and introduced asters (Table 3). *Camassia* spp. and introduced asters showed significantly higher relative abundance in unburned plots. This trend was reversed for *Anthoxanthum odoratum*, which had significantly higher abundance in burned plots. Duration had a significant negative relationship with *Camassia* spp. and a significant positive relationship with *Anthoxanthum odoratum* (Figure 9). The interaction between treatment and duration was also significant for both *Camassia* spp. and *Anthoxanthum odoratum* with both species showing a negative relationship with unburned plots over time. *Camassia* spp. decreased more rapidly over time in unburned plots while *Anthoxanthum odoratum* increased less rapidly in unburned plots. *Cytisus scoparius* and *Bromus* spp. did not show a significant relationship with either predictor or their interaction.

Table 3. Model outputs for *Camassia* spp., *Anthoxanthum odoratum*, and introduced asters. Relative percent cover was the response variable for both species. Predictor variables include treatment, duration (scaled and centered), and their interaction. Values indicate coefficient estimates and p-values (estimate / p-value). Interaction coefficients represent unburned plots. Bold font indicates significance at α = 0.05. Models for *Camassia* spp. and *Anthoxanthum odoratum* were not adjusted for zero inflation while the model for introduced asters was.

Species	**Origin**	**Treatment**	**Duration**	**Interaction**
***Camassia* spp.**	Native	**1.3 / 0.0068**	**-0.31 / 0.036**	**-0.44 / 0.014**
Anthoxanthum odoratum	Exotic	**-1.2 / < 0.001**	**1.0 / < 0.001**	**-0.53 / 0.017**
Introduced asters	Exotic	**1.1 / 0.0026**	-0.19 / 0.68	0.90 / 0.065

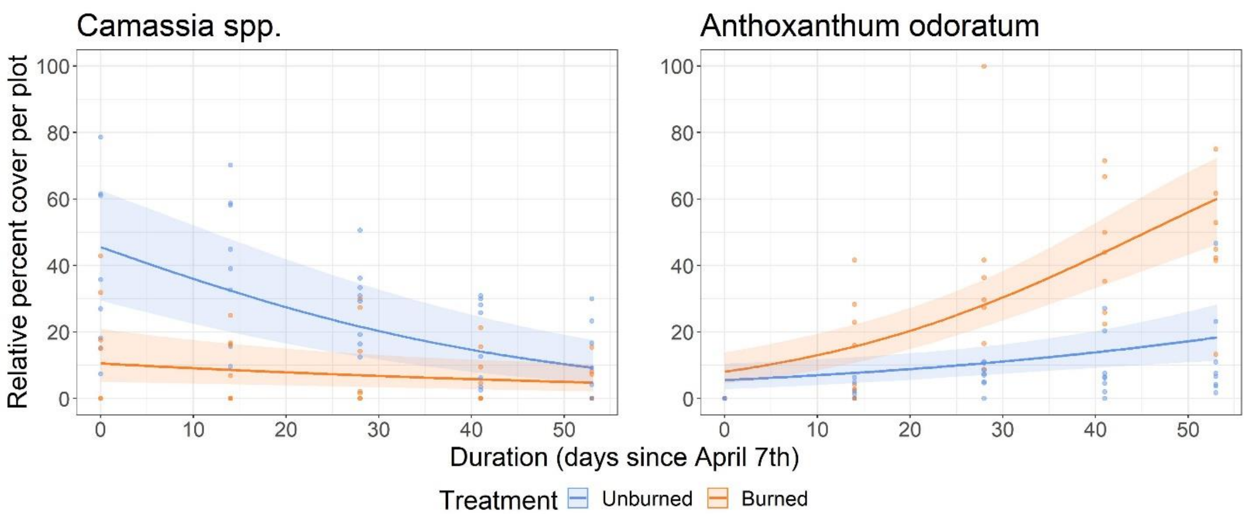

Figure 9. Model predictions and 95 percent confidence intervals for native *Camassia* spp. (left) and exotic *Anthoxanthum odoratum* (right), the two most abundant species in this study. Relative percent cover represents focal cover divided by total vegetation cover. Duration represents the number of days since the first survey on April 7th, 2021. Dots represent observed values.

Chapter 4: Discussion

Project 1: Impacts of invasive species removal

Unfortunately, these results show that Scotch broom removal did not lead to substantial passive recovery of native species. Instead, the only statistically significant change caused by removal was an increase in abundance of introduced species. My models explain this observation by showing that Scotch broom cover was a greater constraint for introduced species than it was for native species, since all three exotic focal species experienced stronger negative responses to increased Scotch broom cover than did native species. Cutting Scotch broom and removing that constraint benefited introduced species more than natives and facilitated their expansion. Overall, results from Project 1 show that Scotch broom removal alone was not adequate in restoring native species and suggest that additional interventions are necessary to do so.

It is important to consider why exotic species responded so strongly to Scotch broom removal, as the answer informs how restoration should respond. A likely explanation is that invasive exotic species were better able to colonize areas cleared of Scotch broom (Seabloom et al. 2003; Buckley et al. 2007). Traits such as abundant seed production, long-distance dispersal, and rapid growth have been associated with invasive species worldwide and may have contributed to rapid post-removal establishment at Mill Hill (van Kleunen et al. 2010). Furthermore, native Garry oak species tend to be dispersal limited, especially when adult abundances are low and habitats are highly fragmented (MacDougall and Turkington 2005). Results from Project 2 indicate that invasive *Anthoxanthum odoratum* was uniquely able to disperse seeds into and grow in areas impacted by wildfire and fire suppression tactics (see following section). The same ability to colonize disturbed areas likely explains *Anthoxanthum odoratum's* increases in treatment plots in Project 1.

A second explanation is that soil changes caused by Scotch broom favored exotic species over native species. Studies in Pacific Northwest prairie ecosystems show that Scotch broom modifies soil properties in ways that may promote exotic species and inhibit native species. Several years after removal, areas where Scotch broom was once present have been found to support lower native richness and relative abundance when compared to adjacent sites that have never been invaded by Scotch broom (Rook et al. 2011; Martin 2020). Furthermore, restoration of sites formerly occupied by Scotch broom has been much less effective at restoring native abundance than in sites where it was never present (Martin 2020). Other studies found that exotic species out-performed native species in areas where Scotch broom was currently present compared to uninvaded sites (Shaben and Myers 2010; Carter et al. 2019). Since these studies examined sites where Scotch broom was currently present, it is unclear whether exotic species did better due to altered soil conditions or instead due to resource partitioning or other means of persisting under a Scotch broom canopy.

There are several mechanisms that potentially explain why native species might be disadvantaged in Scotch broom legacy soils long after Scotch broom has been removed. As a leguminous nitrogen-fixing shrub, increases in soil nitrogen are often hypothesized to contribute to native species declines in Scotch broom-invaded areas. The verity of this mechanism is currently unclear, however, with some studies finding increases in soil nitrogen in Scotch broom-invaded areas (Haubensak et al. 2004; Haubensak and Parker 2004) while others found no difference (Shaben and Myers 2010; Carter et al. 2019). Shaben and Myers (2010) suggest that rapid nitrogen uptake might explain the lack of soil nitrogen increase they observed in Scotch broom-invaded sites and recommend measuring foliar nitrogen to determine its abundance. Scotch broom invasion has also been shown to result in depleted soil phosphorus and increased concentrations of calcium and magnesium in soil water (Shaben and Myers 2010; Carter et al. 2019). Studies have yet to directly link these nutrient changes to diminished native richness and productivity in Garry oak savannas.

Alterations to biotic soil conditions may also be influencing the success of native and exotic species. Davis et al. (2021) found that Scotch broom increased symbiotic nitrogen-fixing bacteria and arbuscular mycorrhizal fungi in soils of Pacific Northwest forests over a relatively short time frame. Arbuscular mycorrhizal fungi tend to have low host specificity and are equally likely to form relationships with native and exotic species (Smith and Read 2010; Bunn et al. 2015). Exotic species may be limited by a lack of fungal mutualists away from their country of origin, but increases in arbuscular mycorrhizal fungal generalists caused by Scotch broom may reduce that limitation and make invasion more likely. Furthermore, it is possible that exotic species originating from the same ecosystem as Scotch broom will uniquely benefit from these biotic soil alterations if they mimic conditions in their ecosystems of origin (Martin 2021). Surprisingly, Davis et al. (2021) also found that Scotch broom exhibited stunted growth in soils where it had previously grown when compared to uninvaded soils despite increases in soil mutualists. This may have been due to nutrient depletion or accumulation of soil-borne pathogens (Davis et al. 2021). Just as increased mutualists may benefit other exotic species, nutrient depletion and pathogen accumulation may negatively affect them. Whether or not these soil changes promote exotic over native species is a matter of which group experiences greater impacts from them. If native species experience greater negative impacts from nutrient depletion or soil pathogens than exotic species, then invasion will be promoted. Invasion will also be promoted if exotic species experience greater benefits from increased mutualists than native species.

A third explanation for exotic dominance in treatment quadrats is herbivory that negatively impacted native species while facilitating exotic species. Due to predator removal and reduced hunting in the region, black tailed deer (*Odocoileus hemionus*) and exotic fallow deer (*Cervus dama*) now occur at high enough abundances in Garry oak ecosystems to cause considerable ecosystem shifts (Martin et al. 2011; Arcese et al. 2014). In addition to causing reductions in floral richness and diversity, abundant deer have also been shown to preferentially browse native rather than exotic species (Gonzales and Arcese 2008; Martin et al. 2011; Arcese et al. 2014). Camas, the most abundant native species observed at Mill Hill, is specifically noted to be preferred by deer (Gonzales and Arcese 2008). Deer are likely present at Mill Hill in high abundances and could be influencing the relative success of native and exotic species. Preferential grazing of native plants may have restricted native species regeneration in treatment quadrats while simultaneously reducing competition for exotic species. Exotic earthworms may also preferentially depredate native plants and could be contributing to exotic plant dominance in this study (Fleri et al. 2021).

A single species, *Anthoxanthum odoratum*, was overwhelmingly responsible for exotic species increases in Scotch broom removal quadrats. In my models, *Anthoxanthum odoratum* also showed a greater negative relationship with Scotch broom cover than all other focal species. *Anthoxanthum odoratum* is a rhizomatous, perennial, cool season grass that was introduced for grazing (Dennehy et al. 2011; Livingston and Varner 2016). Originating in Eurasia, it can be found across the globe, including the east and west coasts of North America and in Australia (Weber 2003; Pickering and Hill 2007; Dennehy et al. 2011). Other research in Garry oak ecosystems has documented a relationship between *Anthoxanthum odoratum* and Scotch broom. Carter et al. (2019) and Shaben and Myers (2010) found that *Anthoxanthum odoratum* had increased cover in sites where Scotch broom was present. In addition to soil alterations, Carter et al. (2019) suggest that *Anthoxanthum odoratum's* shallow roots and cool season growth may explain its ability to perform well in Scotch broom's presence through resource partitioning. These traits may allow *Anthoxanthum odoratum* to uptake water from higher in the soil than Scotch broom and use it earlier in the year. It is also possible that the Scotch broom soil modifications discussed above facilitate *Anthoxanthum odoratum's* co-occurrence with Scotch broom. In this study, the ability of *Anthoxanthum odoratum* to persist beneath dense Scotch broom canopies may have positioned this species to react quickly to the removal of those canopies and to achieve the observed cover increases. The tendency of deer to avoid herbivory of exotic grasses may also have helped this species to achieve dominance post-removal (Arcese et al. 2014).

The influence of climatic factors varied in direction and across species origins. Climate's importance was comparable to that of Scotch broom cover for three focal species: *Carex inops*, *Hypochaeris radicata*, and *Poa pratensis*. These species were also significantly influenced by Scotch broom cover. The combined influence of Scotch broom and precipitation metrics suggests that how these species respond to invasive species removal may depend on climate. For instance, *Hypochaeris radicata* and *Poa pratensis* show opposite responses to the two precipitation metrics. In years of low spring precipitation, *Hypochaeris radicata* might proliferate more rapidly in removal sites while *Poa pratensis* may be impeded. For the other three focal species, the influence of climate was either not significant or was lower in magnitude than that of Scotch broom cover. Overall, the significant influence of climate and the variation in species responses to it may explain yearly fluctuations in vegetation responses to removal. Precipitation changes caused by climate change may impact several focal species. Spring precipitation is expected to increase in the Pacific Northwest, though these predicted changes are less certain and lower in magnitude than the expected decreases in summer and increases in winter precipitation (Mote and Salathé 2010; Bachelet et al. 2011). Nevertheless, these changes may impact several focal species, namely the two most abundant exotic species, *Anthoxanthum odoratum* and *Hypochaeris radicata*, which may experience a decline in their competitive advantage in a future of increased spring precipitation.

Project 2: Impacts of changing fire regimes on Garry oak ecosystems

Results from Project 2 suggest that the return of fire to Garry oak ecosystems has potential to accelerate invasion. Invasive *Anthoxanthum odoratum* had higher abundance in burned plots while native *Camassia* spp. were reduced. *Anthoxanthum odoratum* has been shown to benefit from fires in other Garry oak communities (Clark and Wilson 2001; Dennehy et al. 2011) while responses for *Camassia* spp. have varied (Dunwiddie 2002; Beckwith 2004). Throughout the first growing season following the fire at Mill Hill, fire adaptations did not seem to put native species at an advantage over exotic species. However, due to the characteristics of this wildfire and the firefighting strategies used, it is reasonable to expect different outcomes from controlled burns. Nevertheless, these results demonstrate the ability of fire to negatively impact native species and benefit exotic species regardless of whether they were intended or not.

Several characteristics of the Mill Hill fire differentiate it from controlled burns intended to restore ecosystem health. First, its lack of intention made it difficult to control—it was started during one of the driest times of the year, a burn crew was not in place to control its spread until after it started, and plant fuels could not be preemptively removed. This difficulty, in addition to flammable plant debris that had accumulated due to a lack of fire in the area, likely contributed to a fire of greater intensity than pre-colonial fires set by Coast Salish peoples. Controlled burns have a much greater ability to control conditions and achieve low intensities (Hamman et al. 2011).

In addition, firefighting efforts may have had substantial impacts on vegetation regrowth, especially the use of hoses to spray soil with pressurized water. Several studies demonstrate that Scotch broom seed banks are stimulated by fire, resulting in abundant germination post-burn (Agee 1996; GOERT 2002a; Dennehy et al. 2011). This was not observed in any of the burned plots. It seems likely that pressurized spraying washed away topsoil, and with it seed banks and belowground plant structures that otherwise would have sprouted during the survey period. These effects were evident in several burned plots where soil was bare and the organic layer lacking. In addition, exposed, charred camas bulbs were visible during an early visit to the site, suggesting that perennial belowground structures of native species were unable to survive the combined impacts of high intensity fire and firefighting as they would have with a traditional Indigenous fire (Turner 1999; Fuchs 2001; Beckwith 2004). Considering this, *Anthoxanthum* *odoratum's* dominance in burned plots was likely due to its ability to disperse into burn areas once firefighting had ceased at greater rates than competitors. Once there, it may have benefited from reduced competition due to the absence of seed banks and belowground structures. As summers in the Pacific Northwest become hotter and dryer (Mote and Salathé 2010; Bachelet et al. 2011), the environmental toll and invasion facilitation seen in the Mill Hill fire and necessary firefighting efforts are likely to impact more and more land as fires increase in frequency (Gaur et al. 2021).

Controlled burns set by First Nations and non-Indigenous land managers in Garry oak ecosystems have the potential to avoid many of the negative effects seen in the Mill Hill fire. With control over a variety of factors that influence fire intensity and a reduced need for firefighting, these burns can more accurately mimic the seasonal timing and intensity of pre-colonial Coast Salish burns. However, multiple studies document vegetation responses to controlled burns, with varied results. Few studies show decreases in exotic species and increases in native species from burning alone (MacDougall and Turkington 2007), while many others show mixed results—either increases in both native and exotic species or inconsistent responses across replicates (Dunwiddie 2002; Beckwith 2004; Stanley et al. 2011; Livingston et al. 2016). Also, responses are likely to be species-specific without regard for origin (Agee 1996; Dennehy et al. 2011). Though responses to burning alone are inconsistent, management recommendations from the scientific community are not: carefully executed burning can be very effective when repeated over a three- to five-year interval and combined with other interventions, especially seeding of native species, use of grass-specific herbicide, use of broad-spectrum herbicide, and preemptive fuel reduction (Tveten and Fonda 1999; MacDougall and Turkington 2007; Hamman et al. 2011; Stanley et al. 2011).

A short time frame and a limited number of observations in this study reduced my ability to quantify impacts from the Mill Hill fire. Focal group observations were low due to the number of replicates and difficulty identifying to species level during image analysis. This likely caused the lack of statistical significance seen in several models of focal group abundance. Many of the studies cited above recommend monitoring an area for multiple years post-burn. By monitoring only once shortly after the fire, more relevant long-term impacts were likely missed. The photo-point monitoring plots remain at Mill Hill (Appendix 2) and survey protocols have been finalized in the hopes that graduate students, CRD officials, or other researchers will continue to monitor the fire for several years. In addition, deer herbivory was not controlled for and may have contributed to exotic dominance due to preferential browsing of native species (discussed above). Despite these caveats, Project 2 results show statistically significant responses of a dominant native and exotic species and grant insight into future reintroductions of fire.

Chapter 5: Conclusion

Restoration implications and need for future work

This thesis outlines the value of Garry oak ecosystems and demonstrates the complexity of restoring them in a post-colonial landscape. In regard to Project 1, traits that enable invasive species to rapidly colonize disturbed areas—such as high fecundity, long-distance dispersal, and rapid growth—may be responsible for reinvasion of invasive species removal sites. In addition, it seems possible that long-term Scotch broom presence has modified soils in ways that benefit exotic species and impede the reestablishment of native species. Alternatively, preferential grazing of native species by deer may be responsible for the dominance of exotic species observed. Future research is needed to identify the relative importance of these mechanisms in facilitating exotic regeneration and dominance. If invasive species traits are primarily responsible for reinvasion, then native dominance might be achievable through native seeding and continued low-disturbance invasive removal. If soil alterations are in fact impeding native growth, then post-removal efforts may be necessary over longer time frames to establish native dominance. Understanding specific mechanisms behind problematic soil alterations—such as changes to soil nutrients, mutualists, or pathogens—may enable land managers to directly modify soils in ways that allow more rapid native establishment. Traditional Coast Salish management of camas meadows involved tilling, fertilizing, weeding, and burning of Garry oak ecosystems—all processes that alter soil makeup and that have potential to return soils to pre-invasion condition (Turner 1999; Beckwith 2004; Proctor 2013). If herbivory is the main cause of exotic success, then deer population control may be effective at restoring native plants.

In regard to Project 2, understanding how various plant reproductive strategies enable post-fire regeneration may prove useful if controlled burns can be tailored to assist species with certain strategies. Its ability to widely disperse seeds after the Mill Hill fire seems to have allowed *Anthoxanthum odoratum* to establish quickly in a scenario where soil had been washed away by firefighting efforts. *Anthoxanthum odoratum* may not experience this advantage after quick, low-intensity controlled burns where soil organics remain intact. In this case, species with seed banks and belowground structures capable of sprouting post-burn may win out. Native species, especially camas, are noted for their ability to resprout from belowground structures that can survive low-intensity burns (Antos et al. 1983; Turner 1999; Fuchs 2001; Beckwith 2004).

Conversely, invasive Scotch broom is notorious for seed banks that germinate aggressively following fire, but that are susceptible to repeated burns (Agee 1996; Dennehy et al. 2011). Future work is needed to determine whether a correlation exists between post-fire reproductive strategies and species origin: do most native species sprout from belowground structures? Are most introduced species reliant on seed banking or dispersal? If such a correlation exists, then fire treatments may be designed to benefit reproductive strategies typical of native species.

Both projects and the restoration studies cited above consistently demonstrate that there is no one treatment that can effectively restore invaded Garry oak ecosystems on its own. Long-term combinations of burning, invasive species removal, seeding of native species, herbicide application, and fuel reduction are needed to address the complexities of invaded systems. The cost of such actions is certain to be high, but will be better invested than the costs of less intensive management efforts that do not achieve restoration goals.

Coast Salish cultural values and the future of Garry oak ecosystems

It is crucial that Coast Salish peoples and land stewardship practices play a role in the future of Garry oak ecosystems. Traditional Coast Salish cultivation was remarkably effective, yielding vast quantities of camas and other resources for millennia while maintaining sustainability and enhancing biodiversity. Considering their past efficacy, the reintroduction of those practices seems promising for Garry oak restoration today. However, due to changes caused by invasive species, long-standing fire suppression, and climate change, these ecosystems may not respond to those practices like they did before the arrival of European settlers. Yet, even if these practices cannot be reinstated exactly as they were before colonization, they still have a role to play in the future of Garry oak ecosystems. It is a mistake to consider these practices incapable of change and relevant only to the past. Coast Salish management strategies have never been static: they continue to evolve, as they have always done, and can adapt past practices to present day challenges (Turner and Clifton 2009). Garry oak ecosystems are Indigenous landscapes dependent on human intervention; therefore, it seems unlikely that these ecosystems will return to pre-colonial conditions without reintroducing some form of Coast Salish management. Interestingly, the suite of traditional practices used by Coast Salish peoples has a striking resemblance to the combination of restoration tools recommended by the scientific community, as discussed in the beginning of this chapter.

Garry oak ecosystems hold unique cultural value to Coast Salish First Nations, values that are threatened by the decline of these ecosystems. It is important that these values be considered when determining restoration goals. For example, cultural goals such as the return of culturally important species or the maintenance of burial sites associated with Garry oak ecosystems should be primary restoration outcomes alongside measurements of ecological health such as native species diversity and richness. Furthermore, it is vital that Coast Salish First Nations lead or be directly involved in developing these goals, implementing the restoration practices seeking to attain them, and assessing whether or not goals are achieved. If these goals are adopted without First Nations’ involvement and oversight, Coast Salish values are at risk of being mis-characterized, altered, or appropriated by the non-Indigenous community. Unless it comes directly from First Nations, any alteration of these values, especially in an attempt to align them with values of western science, has the potential to perpetuate cultural harm. Inclusion of Coast Salish values in the broader world of ecological restoration has the potential to redress some of the cultural damage done through colonization, but only if First Nations maintain the ability to define and pursue those values.

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Appendix 1: Species List

Table 4. A list of all plants mentioned in this thesis, including their origin and scientific, common, and SENĆOŦEN names (Montler 2015).

Scientific name	Common name	SENĆOŦEN name	Origin
Anthoxanthum odoratum	sweet vernal grass	NA	introduced
Arbutus menziesii	Pacific madrone	ḰEḰEYIȽĆ	native
Bromus carinatus	California brome	NA	native
Bromus hordeaceus	common soft brome	NA	introduced
Bromus sterilis	barren brome	NA	introduced
Bromus tectorum	cheatgrass	NA	introduced
Camassia leichtlinii	great camas	ḰȽO¸EL; SPÁÁNW̱	native
Camassia quamash	small camas	ḰȽO¸EL; SPÁÁNW̱	native
Carex inops	long-stoloned sedge	NA	native
Cerastium arvense	field chickweed	NA	native
Cytisus scoparius	Scotch broom	NA	introduced
Dactylis glomerata	orchardgrass	NA	introduced
Danthonia californica	California oatgrass	NA	native
Daphne laureola	daphne	NA	introduced
Elymus glaucus	blue wildrye	NA	native
Festuca idahoensis	Idaho fescue	NA	native
Fritilaria affinis	chocolate lily	NA	native
Holodiscus discolor	oceanspray	KÁT̸EȽĆ	native
Hypochaeris radicata	hairy cat's ear	NA	introduced
Koeleria macrantha	Junegrass	NA	native
Lomatium nudicaule	barestem desert parsley	KEXMIN	native
Lonicera hispidula	hairy honeysuckle	NA	native
Micranthes integrifolia	wholeleaf saxifrage	NA	native
Poa pratensis	Kentucky bluegrass	NA	introduced
Primula hendersonii	Henderson's shooting star	NA	native
Pseudotsuga menziesii	Douglas-fir	JSȺ¸IȽĆ	native
Quercus garryana	Garry oak	ĆEṈ¸IȽĆ	native
Ranunculus californicus	California buttercup	NA	native
Ranunculus occidentalis	western buttercup	NA	native
Sericocarpus rigidus	whitetop aster	NA	native
Taraxacum spp.	dandelion	NA	introduced
Taxus brevifolia	Pacific yew	ṮEṈ¸KIȽĆ	native
Toxicoscordion venenosum	death camas	NA	native

Appendix 2: Photo-point monitoring details

Table 5. Photo-point monitoring plot details, including locations in UTMs using WGS 84 coordinate reference system.

Plot number	Zone	Easting	Northing	Burned/Unburned	Tripod Azimuth (degrees)
1	10 U	464693	5367200	Unburned	290
2	10 U	464688	5367166	Unburned	200
3	10 U	464728	5367276	Unburned	230
4	10 U	464748	5367302	Unburned	274
5	10 U	464756	5367299	Unburned	212
6	10 U	464798	5367304	Unburned	352
7	10 U	464617	5367187	Unburned	290
8	10 U	464591	5367256	Unburned	170
9	10 U	464609	5367278	Burned	162
10	10 U	464624	5367274	Burned	278
11	10 U	464627	5367247	Burned	170
12	10 U	464720	5367383	Burned	212
13	10 U	464707	5367359	Burned	210
14	10 U	464693	5367367	Burned	218
15	10 U	464747	5367326	Burned	216
16	10 U	464709	5367324	Burned	196