The effect of time-since-burning and hand-pulling on the growth and stem density of *Centaurea stoebe* and *Linaria dalmatica*

by

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# Abstract

Prescribed burning and hand-pulling are used to manage invasive plants but treatments can differentially affect species. My objective is to determine the effect of time-since-burning and hand-pulling on stem density and growth of *Centaurea stoebe* (spotted knapweed) and *Linaria dalmatica* (Dalmatian toadflax). Prescribed burns occurred in March 2015 and 2016, while hand-pulling occurred in April and May of 2017. I conducted vegetation surveys in May, June, and July 2017. Growth rates differed among treatments and by species. *Centaurea stoebe* was not detected in the prescribed burn treatments. Hand-pulling increased stem density of *C. stoebe*, but individuals were smaller and 60% remained as basal rosettes compared to control. *Linaria dalmatica* were significantly taller in the burn treatments, and the stem density of *L. dalmatica* was greater in the prescribed burn and hand-pull treatments compared to control. The tallest *L. dalmatica* occurred in the 2-year post-burn site, indicating a time-since-burning interaction.

Keywords: Invasive plants; prescribed burning; hand-pulling; *Centaurea stoebe*; *Linaria dalmatica*;

Dedication

In loving memory of my uncle Richard and Aunt Jo-Ann: you taught me the value of hard-work and that things might not always turn out as planned. And to my mom, dad, and sister, Camille, who have always encouraged and supported me to pursue my goals.

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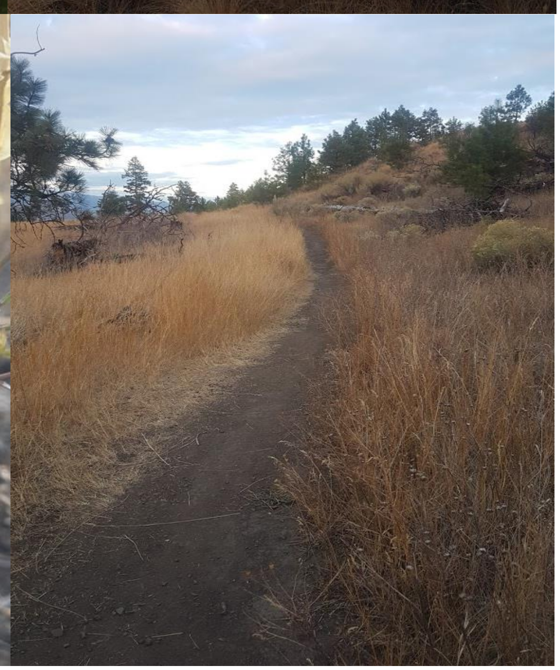
List of Acronyms

AIC	Akaike Information Criterion
BACI	Before-after-control-intervention
BCIT	British Columbia Institute of Technology
BEC	Biogeoclimatic
REML	Residual Maximum Likelihood
SFU	Simon Fraser University



1.0 Introduction

Ecological restoration aims to restore the natural processes and structure of an ecosystem. Fire is an ecosystem process that has been manipulated by humans through widespread fire suppression and prescribed burning (Allen et al. 2002). Prescribed burns are a tool in ecological restoration designed to mimic wildfire in ecosystems where wildfire plays a role as a disturbance agent (Pendergrass et al. 1999). The main objective of prescribed burning in ecological restoration is to restore native plants and natural vegetation structure; however, prescribed burns can differentially affect multiple levels of the ecosystem from entire plant communities to the individual plant (Pendergrass et al. 1999).

At the plant community level, prescribed burning can help maintain biodiversity. Areas with frequent prescribed burning have low tree densities and a high number of fire tolerant plants (Beckage and Scott 2000). Conversely, areas with infrequent burning have a higher abundance of fire-sensitive plants and woody plants (Beckage and Scott 2000). In this way, fire supports a heterogeneous plant community at the landscape scale where local plant communities fluctuate among multiple stable states depending on the site-specific fire return interval (Whisenant 1990, Province of British Columbia 1995, MacDougall et al. 2013). Within the landscape, patches of unburned areas provide sites for fire-intolerant plants to persist and to provide propagules to burned areas following disturbance (Whisenant 1990).

Prescribed burns can have distinct effects on individual plants depending on the characteristics and life history of the plant and the timing, frequency, and intensity of burning. Prescribed burning in fall results in hotter fires than spring burns because vegetation and litter are drier (Lesica and Martin 2003). This results in a greater fire severity (Lesica and Martin 2003). Therefore, timing of fire can differentially affect native and invasive plants (Lesica and Martin 2003). Fall burning is more consistent with the natural fire regime in the Pacific Northwest than spring burning (Pendergrass et al. 1999, Keeley 2006), but spring burning is more socially acceptable because spring burning results in lower severity fires (Keeley 2006). Prescribed burning requires favourable weather conditions, but hotter and drier conditions increase the likelihood of a fire becoming out-of-control (Fernandes and Botelho 2003).

In addition to the change in the timing of prescribed burns compared to wildfire, current plant communities differ from historic plant communities where wildfire functioned as a natural disturbance. Humans have facilitated the spread of exotic species across the globe resulting in changes to community composition and structure that can alter ecosystem processes and disturbance regimes (Chapin III et al. 2000). In some cases this results in a self-perpetuating positive feedback loop where the changes in ecological processes attributed to the introduction of an exotic species can promote the persistence of the exotic species (Chapin III et al. 2000). For example, invasion of cheatgrass (*Bromus tectorum* L.) in sagebrush-steppe ecosystems in the western United States alters the litter composition and the continuity of vegetation resulting in larger, more uniform fires with shorter return intervals than historically, which promote the spread and persistence of cheatgrass (Whisenant 1990).

In ecological restoration, prescribed burning is also used for removing invasive plants. Burning can aid in reducing the number of seedlings that become established or reducing number of seeds produced if timing of the prescribed burn is matched with the phenology of the invasive plant (Emery and Gross 2005, MacDonald et al. 2007). For example, burning in the spring and summer resulted in negative population growth of *Centaurea stoebe* (spotted knapweed, previously known as *C. maculosa* (S.G. Gmelin ex Gugler) Hayek) because spring and summer burning is timed with flowering of *C. stoebe* thereby reducing seed production (Emery and Gross 2005). However, prescribed burning provides opportunities for invasive plants to colonize or spread because fire removes layers of leaf litter and increases the amount of sunlight penetrating the soil (Lesica and Martin 2003). This was observed from early spring burning that resulted in an increased recruitment of the invasive plant *Potentilla recta* (Sulphur cinquefoil) (Lesica and Martin 2003).

Prescribed burning is used by land managers for various objectives, so understanding the species-specific response to prescribed burning is important (Pendergrass et al. 1999). It is necessary to investigate the effect of fire on individual plants before implementing prescribed burns as part of an ecosystem management plan (Pendergrass et al. 1999). If management objectives include reducing invasive plant abundance, then differences in life history traits of invasive and native plants must be considered (Emery and Gross 2005), and investigating how invasive plants respond to prescribed burning is necessary.

Hand-pulling is commonly suggested as an effective removal technique across a broad category of invasive plants (Vujnovic and Wein 1997, Sheley et al. 1998); however, there lacks quantitative data that supports the use of hand-pulling (MacDonald et al. 2013). Hand-pulling is labour intensive and is recommended only for small infestations (MacDonald et al. 2013), but hand-pulling is an attractive removal technique on projects that have a broad volunteer base because it is a simple, low-risk activity that volunteers can take part in. For ecological restoration practitioners, it is important to understand the effectiveness of different removal techniques for specific invasive plants to be able to appropriately assess the best technique for a particular site to achieve restoration goals.

In British Columbia (B.C.), *Centaurea stoebe* and Dalmatian toadflax (*Linaria dalmatica* (L.) Mill.) are noxious weeds (Province of British Columbia 1996). There is limited information on the success of prescribed burning in managing *C. stoebe* and *L. dalmatica* in the southern interior of B.C. since most research has been conducted in the United States, particularly in Montana and Arizona (e.g., Jacobs and Sheley 2003, Dodge et al. 2008, Pokorny et al. 2010, Pearson et al. 2012). In addition, hand-pulling over consecutive years is a commonly suggested removal strategy (Goodwin and Burch 2007, USDA 2014), but there is a lack of empirical evidence to support the use of hand-pulling (MacDonald et al. 2013). Prescribed burns conducted in March of 2015 and 2016 in Kenna Cartwright Park, Kamloops provide an opportunity to examine the effects of time-since-burning on the growth rate and stem density of *C. stoebe* and *L. dalmatica* in comparison to hand-pull and control treatments.

1.1. Species Description

1.1.1. Centaurea stoebe

Centaurea stoebe is a perennial flower in the Asteraceae family that is native to Europe and lives for three to five years (Watson and Renney 1974). In B.C., *C. stoebe* is listed as a noxious weed under Schedule A of the *B.C. Weed Control Act* and Weed Control Regulations (Province of British Columbia 1996). *Centaurea stoebe* is a pioneer species and occurs in disturbed, open areas (Watson and Renney 1974). Plants in the genus *Centaurea* produce allelopathic biochemicals that contribute to the success of *Centaurea* spp. at invading North American grassland ecosystems (Watson and Renney 1974, Callaway and Aschehoug 2000). *Centaurea diffusa* (diffuse knapweed), a close relative of *C. stoebe*, produces allelopathic biochemicals that suppress the growth of North American grasses but not Eurasian grasses from the same family (Callaway and Aschehoug 2000). This suggests Eurasian grasses that co-evolved with *C. diffusa* have adapted to the allelopathic interactions while these interactions are novel in North American grassland ecosystems (Callaway and Aschehoug 2000). Novel interactions can provide competitive advantages for exotic plants (Callaway and Aschehoug 2000).

Mature plants of *C. stoebe* alternate between two life stages: adults occur in either a basal rosette form or bolting form (i.e., grow flowering stems) (Story et al. 2001). The proportion of mature plants that bolt increases to age 4, but never reaches 100% in field conditions (Story et al. 2001). First-year individuals rarely bolt in the field and only 10% of second-year adults bolt (Story et al. 2001). Bolting occurs in June followed by flowering in July (Story et al. 2001). Only bolting adults produce seeds that disperse in August and September (Jacobs and Sheley 1998, Story et al. 2001). Individuals can persist through the winter as seeds or basal rosettes (Watson and Renney 1974).

Reproduction occurs by seeds that germinate in spring and fall, and by lateral shoots (Watson and Renney 1974). Average seed production by *C. stoebe* on B.C. rangeland near Kamloops was calculated to be 349 viable seeds per bolting adult per year, based on an average of 16.35 ± 4.44 flower heads per plant, 26.64 ± 2.88 seeds per flowering head, and assuming 80% seed viability (Watson and Renney 1974); however, seed production is variable and dependant on moisture conditions (Watson and Renney 1974, Story et al. 2001). Desiccation of seed heads results in the opening of seed bracts expelling the achenes (Watson and Renney 1974). Seeds are dispersed by a flicking motion of the seed head that can disperse the seeds up to 1 m; seeds also disperse greater distances by attaching to animals, such as birds and rodents (Watson and Renney 1974). Humans can also facilitate in seed dispersal because seed heads can become attached to vehicles or seeds can be transported in mud attached to shoes and vehicles (Sheley et al. 1998). *Centaurea stoebe* establishes in open areas of overgrazed rangeland, and *C. stoebe* is adapted to the dry climate of the southern interior of B.C. (Watson and Renney 1974).

1.1.2. *Linaria dalmatica*

Linaria dalmatica is a perennial flower in the Plantaginaceae family that is native to Europe (Vujnovic and Wein 1997). In B.C., *L. dalmatica* is listed as a noxious weed under Schedule A of the *B.C. Weed Control Act* and Weed Control Regulations (Province of British Columbia 1996). *Linaria dalmatica* is competitive in North American ecosystems because *L. dalmatica* is an early spring emergent with vegetative buds observed as early as March (Robocker 1970, Jacobs and Sheley 2003). *Linaria dalmatica* is a ruderal plant that is successful at establishing new colonies because reproduction occurs by both seeds and lateral stems: *L. dalmatica* produces a large number of seeds and seeds remain viable in the seedbank for up to 10 years (Sing and Peterson 2011).

Seeds of *L. dalmatica* germinate in March and April when soil temperatures reach 10°C (Robocker 1970). Flowering occurs from May to August and most seeds are produced from June until early September, but phenology varies with factors such as temperature (Vujnovic and Wein 1997). Reproduction occurs by seeds produced from insect-pollinated flowers or through vegetative propagation by prostrate stems (Vujnovic and Wein 1997). Seed dispersal occurs from the end of July to August (Robocker 1970). Ninety-five percent of seeds produced by *Linaria vulgaris* (a close relative of *L. dalmatica*) disperse in a 0.5 m radius (Nadeau and King 1991).

Capsules on the main stem produce an average of 250 seeds/year, while lateral branches produce an average of 140 seeds/year (Robocker 1970). Individuals of *L. dalmatica* have a high seed-production potential with larger plants producing more seeds (Robocker 1970). However, the number of branches produced is dependent on moisture availability, soil type, and growing conditions (Robocker 1970). Seeds are small and wind dispersed or dispersed when animals ingest the seeds while grazing (Robocker 1970). Seed germination can occur in the fall if moisture is available (Robocker 1970). The extensive lateral root system of *L. dalmatica* is adventitious for acquiring moisture especially during periods of drought in comparison to native grasses that typically exhibit shallow rooting (Coupland et al. 1963). The roots of *L. dalmatica* store many sugars (Vujnovic and Wein 1997).

Linaria dalmatica grows in a variety of climatic conditions and is located across North America from 33º to 56º N latitude (Vujnovic and Wein 1997). *L. dalmatica* grows in ponderosa pine (*Pinus ponderosa*) forests particularly on disturbed sites and on rangelands (Vujnovic and Wein 1997). *Linaria dalmatica* can tolerate a wide range of temperatures and soil conditions, and occurs up to 2800 m in elevation (Vujnovic and Wein 1997).

1.1.3. Negative effects of *C. stoebe* and *L. dalmatica* on Ecosystem Function

Successful invasions by exotic species can result in negative effects on ecosystem function. Exotic species that alter the energy flow in an ecosystem or alter ecosystem processes have the greatest negative effects on natural ecosystems (Chapin III et al. 2000). Invasive plants that develop into highly productive monocultures displace native species and decrease overall diversity (Vila et al. 2011), which can lead to extirpation of native species that are unable to compete (Chapin III et al. 2000). Ultimately, invasive species can change the conditions of the ecosystem that native species previously existed in (Vitousek et al. 1997).

Centaurea stoebe exhibits negative effects on several ecosystem processes including altering soils, hydrology, and forage quality. The amount of sediment yield from surface runoff is greater from stands composed of *C. stoebe* than stands composed of native bunchgrasses, resulting in increased erosion on sites dominated by *C. stoebe* (Lacey et al. 1989). Therefore, the conversion of bunchgrass to *C. stoebe* monocultures can increase the amount of sediment entering waterways and result in greater loss of topsoil (Lacey et al. 1989). In addition, *C. stoebe* is a poor forage species for ungulates because of the high fibre content and low nutrient content (Watson and Renney 1974). *Centaurea stoebe* grows into monocultures with a dense over story, thereby displacing native grasses with higher nutritional value (Watson and Renney 1974). It is suggested that monocultures of *C. stoebe* can result in decreased grazing by wildlife and reduced fitness of animals (Vila et al. 2011). *Centaurea* spp. grow into dense, monoculture stands particularly in overgrazed pastures and areas of high disturbance such as roadsides and along rights-of-way (Watson and Renney 1974). The presence of *Centaurea spp.* is used as an indicator of sites that have been degraded from a native state in the Ponderosa Pine Biogeoclimatic (BEC) zone in B.C. (Hope et al. 1991).

In a comparison between native and exotic forbs (that included both *C. stoebe* and *L. dalmatica* as exotic forbs) in bluebunch wheatgrass (*Pseudoroegneria spicata*) and rough fescue (*Festuca scabrella*) ecosystems, the exotic forbs exhibited significantly different morphology and phenology compared to native forbs (Pearson et al. 2012). In particular, exotic forbs allot greater resources into growing taller flowering stems and fewer resources into vegetative growth in comparison to native forbs that were observed to do the opposite (Pearson et al. 2012). The exotic forbs also occurred at greater densities because exotic forbs (such as *C. stoebe* and *L. dalmatica*) grow into dense monocultures (Pearson et al. 2012). This indicates that these invasive forbs can alter the vegetative structure of the community, not just the composition, and alter ecosystem functions (Pearson et al. 2012).

In addition to altering the vegetation structure and composition, *L. dalmatica* exhibits traits that make it a successful competitor in grasslands in North America. These traits include emerging in early spring before most bunchgrasses have begun growth (Robocker 1970, Jacobs and Sheley 2003) and reproducing by both seeds and lateral stems (Sing and Peterson 2011). The competitive ability of *L. dalmatica* threatens to displace native vegetation resulting in a change in vegetation composition (Sing and Peterson 2011). Grazing livestock avoid *L. dalmatica*, which might increase the grazing pressure on other vegetation and increase the competitive success of *L. dalmatica* (Vujnovic and Wein 1997).

Because of the detrimental effects of *C. stoebe* and *L. dalmatica* on ecosystem processes, there is a strong desire to remove these invasive plants from natural areas. The objective of my study is to compare the stem density and growth rate of *C. stoebe* and *L. dalmatica* following treatment by prescribed burning and hand-pulling within Kenna Cartwright Park, Kamloops.

1.2. Study Area

My study area is in the western section of Kenna Cartwright Park, which is an 800-ha park west of the City of Kamloops that encompasses most of Mount Dufferin (Figure 1). The park is operated by the City of Kamloops and is designated as a ‘nature park’ with the aim of protecting native diversity and natural characteristics (City of Kamloops 2013). The mountain pine beetle outbreak of the early 2000s resulted in high mortality of ponderosa pine trees increasing the amount of dead wood (Marrow 2016). Dead trees and high amounts of litter increase the risk of catastrophic fire, so removing the build-up of dead vegetation using prescribed burning is a high priority in the park. To reduce the available fuel, the City, in partnership with the Wildfire Branch of MFLNRO, conducted prescribed burns in Sites 1 and 2 of the study area (Figure 2). Prescribed burns were completed in March 2016 and March 2015, respectively (Marrow 2016).

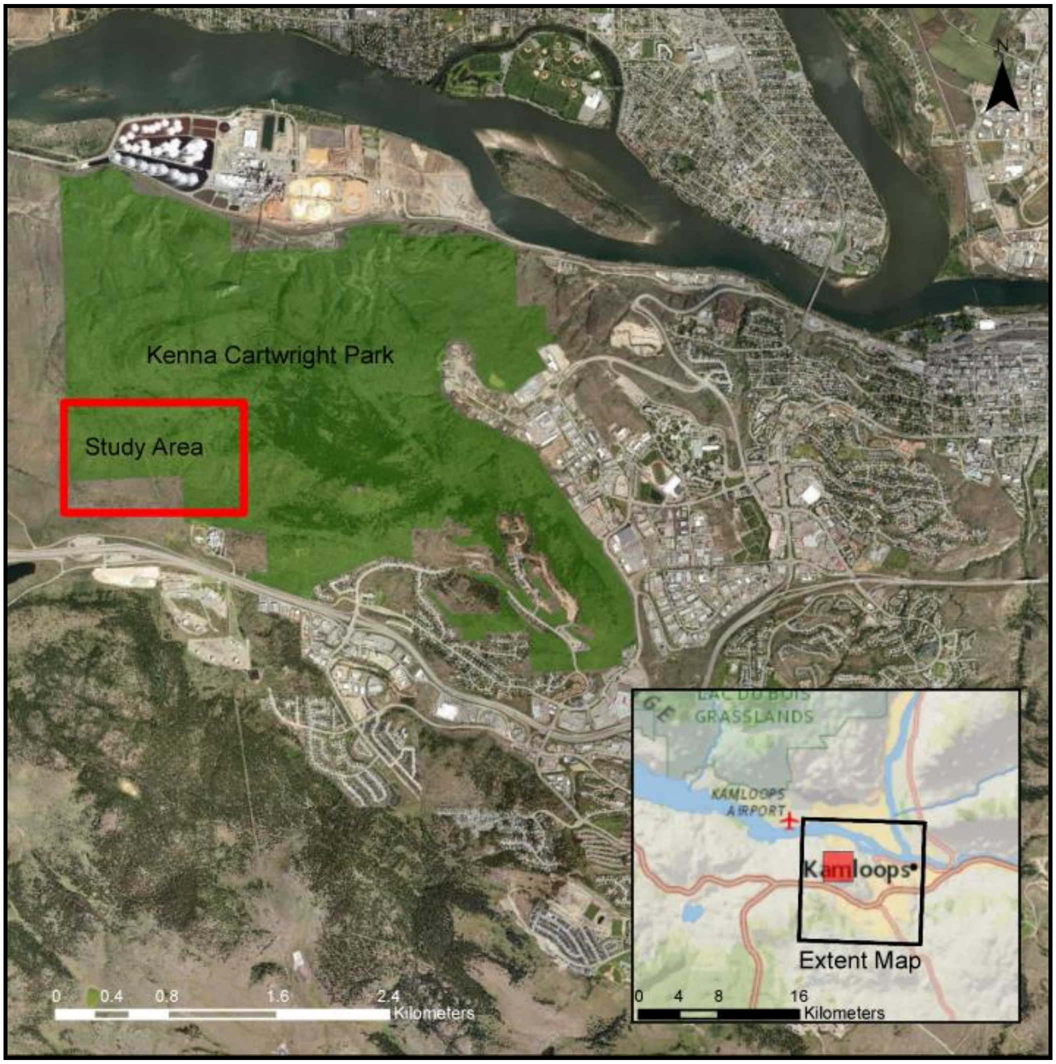


Figure 1. The study area (outlined in red) is located in Kenna Cartwright Park, west of the City of Kamloops. Kenna Cartwright Park is indicated by the red box in the inset map.

The study area is in the Ponderosa Pine biogeoclimatic (BEC) zone (very hot dry PPxh subzone variant 2) apart from the southern portion of Site 4, which is in the Bunchgrass BEC zone (very dry warm BGxw subzone variant 1) (Data BC, Province of British Columbia 2013). All sites fall into the natural disturbance type 4 (NDT4) (Data BC, Province of British Columbia 2013), which is characterized by frequent stand-maintaining fires (Province of British Columbia 1995). The Ponderosa Pine BEC zone is characterized by hot, dry summers that often result in a moisture deficit (Hope et al. 1991). In the Kamloops area, July and August have the greatest potential for natural wildfire occurrence because these months have the highest yearly temperatures, lowest relative humidity, and the most frequent lightning strikes (Klenner et al. 2008). These conditions generate optimal wildfire conditions and fire functions as a natural disturbance in the Ponderosa Pine and Bunchgrass BEC zones (Hope et al. 1991, Province of British Columbia 1995).

I designated four experimental sites in the study area (Figure 2). Site 1 was approximately 3 ha that was the location of a prescribed burn in March 2016, and Site 2 was approximately 4 ha that was burned in March 2015. Recent pre-burn data were not available for *C. stoebe* and *L. dalmatica* in Sites 1 and 2. Site 3 is 9 ha area that was intended for prescribed burning in spring 2017, but the high snowpack over the winter prevented spring burn activities from occurring. Site 4 is an area of approximately 8 ha intended to be a control site for Site 3. Hand-pulling and control treatments were crossed between Sites 3 and 4. These sites were selected based on information and mapping provided by the City of Kamloops.

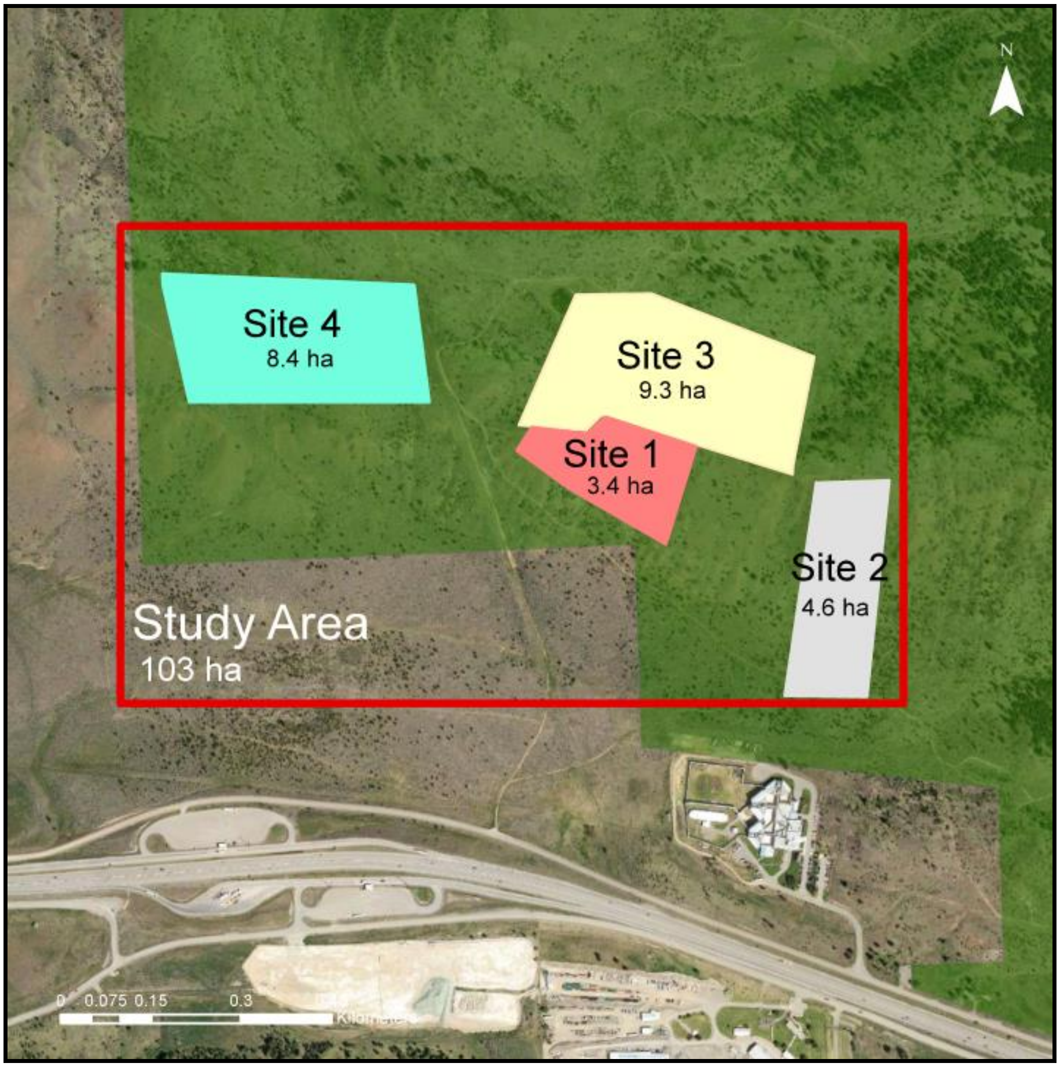


Figure 2. Site locations within the study area in Kenna Cartwright Park, Kamloops. Prescriebd burning occurred in Site 1 in March 2016 and in Site 2 in March 2015. Hand-pulling and control treatments were crossed between Sites 3 and 4.

The predominant trees present in the Ponderosa Pine BEC zone are ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) (Hope et al. 1991), but most of the trees in Kenna Cartwright Park are Douglas-fir (Marrow 2017). Soils in this BEC zone are classified as chernozems or brunisols (Hope et al. 1991). Big sagebrush (*Artemisia tridentata*), green rabbit-brush (*Chrysothamnus viscidiflorus* ssp. *lanceolatus*), and bluebunch wheatgrass (*Pseudoroegneria spicata*) are common throughout my study area. Invasive plants are widespread throughout the park including *C. stoebe* and *L. dalmatica* (Tarasoff 2002, Marrow 2017). Weed inventory mapping was available for Kenna Cartwright Park from 2001 (Figure 3).

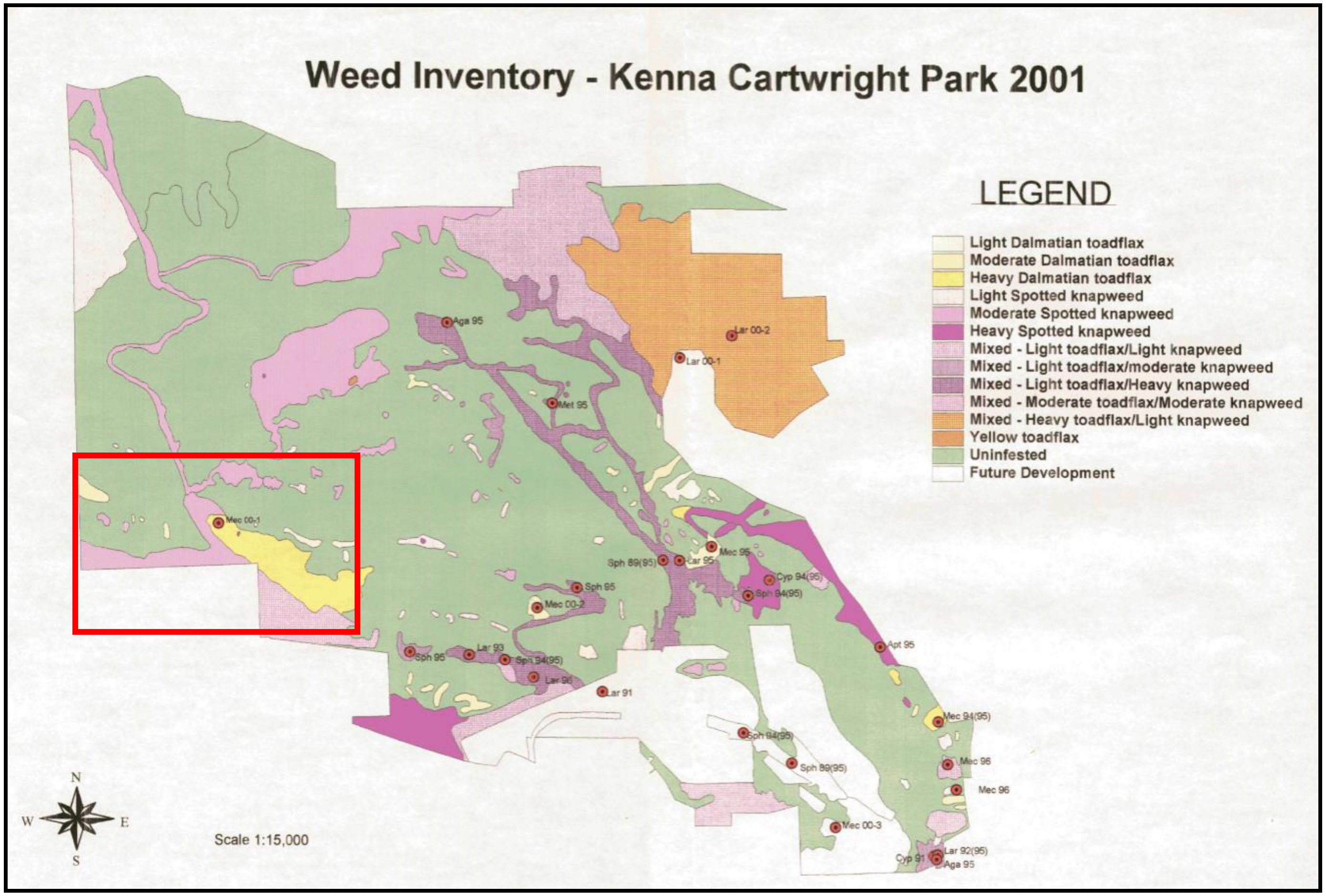


Figure 3. Invasive plant inventory mapping from 2001 (Tarasoff 2002) indicating areas of high, moderate, and light infestation of *Centaurea stoebe* (spotted knapweed) and *Linaria dalmatica* (Dalmatian toadflax) in Kenna Carwright Park. The red outline indicates my study area for comparison to Figure 1 and Figure 2. Point release locations of biological control agents are marked by red circles. The release site Mec 00-1 occurs in my study site and indicates a release site for *Mecinus janthinus,* a stem mining beetle, in the year 2000.

2.0 Methods

2.1. Plot Locations

I established 1-m^2 plots to sample vegetation in all study sites using a systematic random design. I established ten plots in each of Sites 1 and 2 to sample vegetation post-prescribed burning. Prescribed burning was nested in Sites 1 and 2 statistically since prescribed burning only occurred in these sites. In Sites 3 and 4, I randomly assigned 10 sampling plots to the hand-pull treatment and 10 plots to control treatment for a total of 20 control and 20 hand-pull treatments in my study area. Hand-pulling and control treatments were statistically nested in Sites 3 and 4. Because hand-pull and control treatments were conducted in both Sites 3 and 4 these treatments were statistically crossed between both sites.

I recorded the elevation, aspect, and slope at each plot and I summarized the range of each Site. I measured elevation using a Garmin GPS. I measured slope using a clinometer, and aspect using a compass. I did not measure aspect for plots with a slope less than 5%. I dug one soil pit in each Site, except Site 2, which is located on the same slope as Site 1, to examine the soil characteristics. I selected the soil pit locations by randomly selecting a location within 10 m of a randomly selected vegetation-sampling plot in the Site.

2.2. Burn Treatments

Prescribed burning was completed by wildfire crews from the BC Wildfire Service in Kenna Cartwright Park. The first prescribed burn was conducted in March 2015 in an area northeast of the Kamloops Correctional Facility from Ponderosa Pine Trail to Tower Trail (Site 2). The second burn was conducted in March 2016 north of the Kamloops Correctional Facility between Big Pine Trail and Tower Trail (Site 1) (Marrow 2016). Currently, there are twelve other burn management units identified within the park for future burn activities (Marrow 2017). Prescribed burns in early spring are low intensity surface fires (Marrow 2017).

2.3. Hand-pull Treatments

I used hand-pulling as a removal technique to compare to prescribed burning with respect to control plots. I conducted hand-pull treatments in Site 3 on April 25-26, 2017 and in Site 4 on May 14, 2017. I removed all individuals of *C. stoebe* and *L. dalmatica* that had emerged in the 1-m^2 sampling plots. In addition, I removed all individuals in a 2-m radius from the edge of sampling plots to create a buffer from the dispersing seeds from outside sampling plots. The hand-pulling buffer around the sampling plots ensured that individuals occurring within the sampling plots after hand-pulling are from the residual seed bank as opposed to seeds produced in the same year as hand-pulling. I used a pitch fork to remove the roots. I recorded the number of stems of *C. stoebe* and *L. dalmatica* prior to hand-pulling. The sampling plots treated with hand-pulling on May 14, 2017 were not included in the May vegetation surveys.

2.4. Data Collection

In each sampling plot, I counted the number of stems of all *C. stoebe* and *L. dalmatica*. I chose stem count as an estimate of abundance instead of the number of individuals. *L. dalmatica* and *C. stoebe* can propagate through underground rhizomes (Watson and Renney 1974, Vujnovic and Wein 1997) making it difficult to determine one individual from the next without uprooting the plant. I also measured height of flowering stems as an estimate of growth. *C. stoebe* has two adult phases: a rosette phase with only basal rosette leaves and a bolting phase where growth of flowering stems occurs (Story et al. 2001). I measured the height of stems in the bolt phase based on the height of the longest branch on the flowering stem. I attributed a value of 0.5 cm to any *C. stoebe* individuals in the rosette phase because there was no discernable flowering stem. I conducted stem count and height surveys at the end of May, June, and July.

Vegetation surveys examining the density and growth rate of *C. stoebe* and *L. dalmatica* prior to prescribe burning in March 2015 and 2016 were not available. However, I compared my study area to Weed Inventory Mapping from 2001 (Figure 3 adapted from Tarasoff 2002) and known site from the Invasive Alien Plant Program available from iMapBC (Data BC, Province of British Columbia 2013). The Weed Inventory Mapping from 2001 provides locations rated as heavy, moderate, or light infestation of *C. stoebe* and *L. dalmatica* (Tarasoff 2002)*.* Based on this mapping, light infestation of *C. stoebe* and *L. dalmatica* occurred south of Sites 1 and 2, and moderate infestation of *C. stoebe* occurred in Sites 3 and 4 (Tarasoff 2002). Heavy infestation of *L. dalmatica* occurred in the southern part of Sites 1 and 2, and patches of moderate *L. dalmatica* infestation occurred in Site 3 and 4 (Tarasoff 2002).

2.5. Data Analysis

To visualize the spatial arrangement of treatments, sampling plots, detection rates, and densities of both *C. stoebe* and *L. dalmatica*, I mapped the occurrence and densities of invasive plants in my study area using UTM coordinates of the sampling plots. I calculated survey effort for each site based on the total area sampled divided by the total area of each Site (see Study Area). I determined detection rate per treatment and Site based on detecting at least one individual within a sampling plot and determined the mean over the three survey dates.

I used descriptive statistics of means and 95% confidence intervals of the height and stem density of *C. stoebe* and *L. dalmatica* to compare the three treatments (i.e., burn, hand-pull, and control) among the three survey dates (i.e., May, June, and July). I removed sampling plots that contained no individuals of either *C. stoebe* or *L. dalmatica* when I compared height of individuals. I bound the 95% confidence interval bars so bars do not extend below zero for height and stem density. I used the same descriptive statistics to compare time-since-burning of the two burn treatments (March 2015 and March 2016) for *L. dalmatica* to the control treatments. Data collected from the burn conducted in March 2015 represents 2-years post-burn and data collected from the burn conducted in March 2016 represents 1-year post-burn. A comparison of time-since-burning could not be completed for *C. stoebe* because no *C. stoebe* occurred in the burn sampling plots when they were surveyed in the summer of 2017.

I used height over the three survey dates to determine the growth rate of *C. stoebe* and *L. dalmatica.* Sampling plots containing zero values were removed. I analyzed height data for normality. Height data for *C. stoebe* could not be transformed to achieve normality due to the high occurrence of 0.5 cm values from basal rosettes, but my data resembled a Poisson distribution; therefore, I used a generalized linear model to compare growth rates. I log transformed height data for *L. dalmatica* prior to plotting the growth rate. I used a linear model to compare growth rates among treatments. I included 95% confidence intervals to compare growth rates among treatments.

To compare differences in height among treatments statistically, I used a linear mixed-effects model. I selected a linear mixed-effects model because my experimental design was unbalanced. I used treatment (prescribed burn, hand-pull, and control) as my fixed effects and ran models using a combination of random effects for both *C. stoebe* and *L. dalmatica*. The dependent variable selected was height of invasive plants and plots without individuals detected were removed prior to generating models. Because no *C. stoebe* were detected in the prescribed burn treatments, I could not use prescribed burn as a fixed effect when analyzing the data for *C. stoebe.* The random effects I tested included survey date, aspect, elevation, slope, site, and sampling plot. All models were generated using residual maximum likelihood (REML, also known as restricted maximum likelihood). REML is recommended for linear mixed-effects models when the models being compared differ in the random effects (Gurka 2006). I used Akaike Information Criteria (AIC) values to select a 'best' model for each species based on minimizing the AIC value (Gurka 2006, Zuur et al. 2009). Comparing models using AIC enables selection of the model that is the most parsimonious and minimizes residuals (Zuur et al. 2009). I ran 13 models with different random effects for *C. stoebe* and *L. dalmatica.* Four of the models for *C. stoebe* did not reach convergence, and I did not compare these models during model selection. For *C. stoebe,* I selected the model that included the random effects of survey date and sampling plot. For *L. dalmatica,* I selected the model that included the random effects of survey date, site, and sampling plot.

I performed all analyses using R version 3.4.3 (R Core Team 2017). For data analysis, I used the 'readr' package (Wickham et al. 2017), 'plyr' package (Wickham 2011), and 'zoo' package (Zeileis and Grothendieck 2005) in R. Linear mixed-effects models were generated using the 'lmer' function from the 'lme4' package (Bates et al. 2015). All graphics were created using the 'ggplot2' package (Wickham 2009).

2.6. Climate

Snowfall in the Kamloops area typically peaks in December with an average monthly snowfall of 26 cm, and snowfall declines to March with an average monthly snowfall of 3 cm (The Weather Network 2018). December 2016 had low snowfall compared to average (13.3 cm compared to a monthly average of 26 cm); however, snowfall in February 2017 was 24.3 cm, which is high compared to the monthly average of 11cm (The Weather Network 2018, World Weather Online 2018). The late snow accumulation persisted on the ground into the early spring and provided abundant moisture in the early spring; however, the summer of 2017 was characterized by low precipitation in Kamloops compared to the 30 year monthly averages in June and July (Table 1). Total precipitation was approximately one-tenth of the average monthly precipitation in the months of June and July. Because moisture conditions can affect the growth and reproduction of *C. stoebe* and *L. dalmatica* (Watson and Renney 1974, Story et al. 2001) it is important to consider how climate in 2017 can influence my data.

Table 1. Monthly 30-year average temperature (°C) and precipitation (mm) in Kamloops, B.C, in comparison to the 2017 montly average temperature and monthly total precipitation.

		Monthly Average[1]			Average temperature and total precipitation in 2017		
		May	**June**	**July**	**May**	**June**	**July**
Temperature (°C)	High	21.3	24.8	28.3	20 [2]	25 [2]	29 [2]
	Low	7.5	11.3	13.7	11 [2]	15 [2]	14 [2]
Precipitation (mm)		24.4	35.2	29.5	31.1 [3]	3.4 [3]	3.4 [3]

[1] (The Weather Network 2018)

[2] (World Weather Online 2018)

[3] (Weather Stats 2018)

3.0 Results

3.1. Site Characteristics

Elevation of sampling plots within the study area ranged from 721 to 828 metres above sea level (Table 2). Site 4 had the narrowest elevation range. The slope at sampling plots ranged from 2 to 63%. Aspect was the greatest difference among the four sites. Site 1 exhibited the greatest range in aspect. Sites 3 and 4 were predominantly north facing, while Site 2 was predominantly south facing.

Table 2. Range of elevation, aspect, and slope of sampling plots at each Site in the study area in Kenna Cartwright Park during surveys in 2017.

Site	Elevation Range (metres above sea level)	Aspect Range in degrees (cardinal direction)	Slope Range (%)
Site 1	744 - 802	20 – 278 (NE to W)	5 – 54
Site 2	721 - 828	170 – 222 (S to SW)	21 – 63
Site 3	741 - 804	298 – 64 (NW to NE)	19 – 60
Site 4	727 - 764	294 – 60 (NW to NE)	2 – 62

Depth of the soils across the sites ranged from 11 to 34 cm below the surface, with the deepest soils occurring in Site 3 and the shallowest soils occurring in Site 4 (**Error! Not a valid bookmark self-reference.**). Burn sites had course fragments on the surface across most of the higher elevation sampling plots, scree-like slopes, and visibly more bare ground in comparison to Sites 3 and 4.

Table 3. Soil characteristics in each Site. Slope, aspect and elevation correspond to characteristics where the soil pit was dug in Kenna Cartwright Park, Kamloops.

Site	Slope (%)	Aspect	Elevation (m)	Horizon	Depth (cm)	Texture	Colour (dry)
Site 1 (Site 2)	56	SW	795	A	0 to 22	Sandy with coarse fragments	4/3 olive brown
Site 3	35	N	743	Ah	0 to 7	Fine	Very dark brown
				B	7 to 34	Fine	Very dark brown
Site 4	29	NW	746	A	0 to 11	Sandy clay	Very dark brown

3.2. *Centaurea stoebe*

Detection rate of *C. stoebe* within Sites 1 and 2 (burn treatment areas) was zero. Detection rates were greater in hand-pull than control treatments but were more similar within sites than by treatments (Table 4). Survey effort among the four sites was comparable: my survey effort in Site 1 was 0.03% of the Site surveyed, Site 2 was 0.025%, Site 3 was 0.02%, and Site 4 was 0.025%. Based on detection rates, my data suggested that Site 3 was the most heavily infested site with *C. stoebe* because it had the lowest survey effort but the highest detection rate. Conversely, Site 1 might have the lowest infestation rate because this site had the highest survey effort and no detection of *C. stoebe*.

Table 4. Mean detection rate of *C. stoebe* with 95% confidence intervals in the 4 survey Sites among treatments (prescribed burn, hand-pull, and control) in Kenna Cartwright Park, Kamloops. Detection rates are expressed as a percent of sampling plots with a minimum one individual and the mean was calculated among the three survey dates (May, June, and July, 2017).

Site	Burn	Hand-pull	Control
Site 1	0%	-	-
Site 2	0%	-	-
Site 3	-	55% (± 18%)	40% (± 18%)
Site 4	-	20% (± 18%)	10% (± 11%)

No *C. stoebe* were detected in the burn treatments that are nested in Sites 1 and 2. Hand-pull and control treatments were crossed between Sites 3 and 4. I detected *C. stoebe* at a greater number of sampling plots in Site 3 compared to Site 4. In addition, the greatest detection and density of *C. stoebe* occurred in hand-pull treatments. Based on Figure 4, *C. stoebe* exhibited a clumped distribution and appears to be limited to the north portion of the study area. However, I observed *C. stoebe* in the southern portion of the study area despite not detecting *C. stoebe* in sampling plots, particularly along trail networks. Invasive plant inventory mapping from 2001 also confirm the occurrence of light *C. stoebe* infestation in the southern portion of Sites 1 and 2 (Tarasoff 2002).

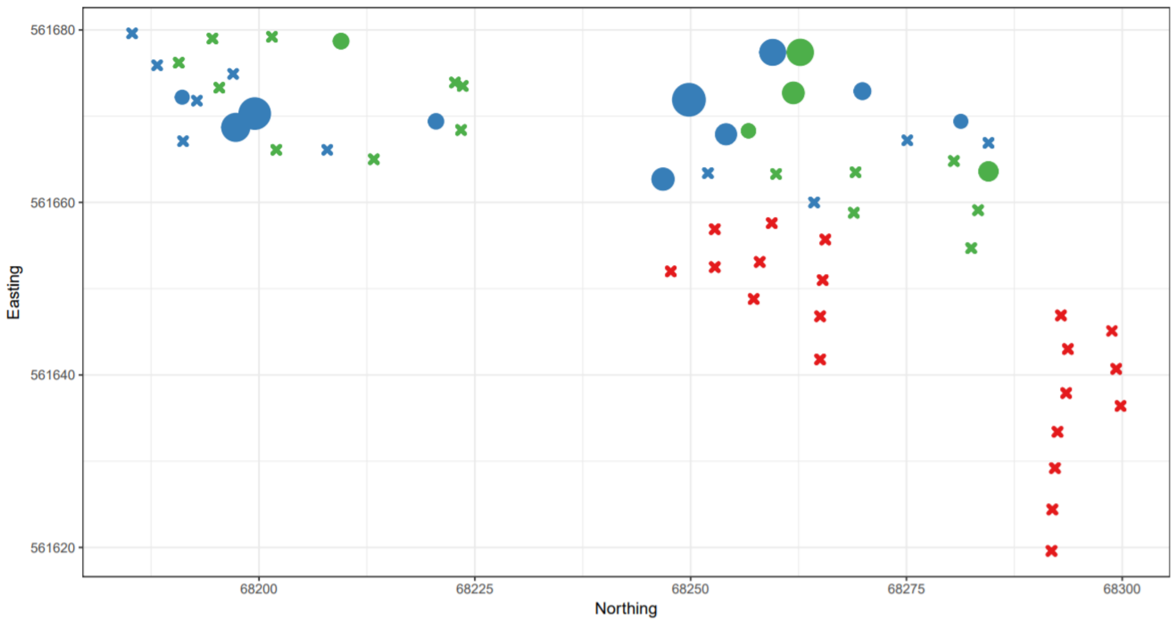


Figure 4. Detection rate of *Centaurea stoebe* and relative density among sampling plots within the study area in Kenna Cartwright Park, Kamloops. The x- and y-axis represent the Northing and Easting, respectively (UTM 10). Within the graph, an ‘x’ denotes no detection within a sampling plot and circles indicate presence of *C. stoebe.* The size of the circle corresponds to the relative density of *C. stoebe* with larger circles indicating greater density. The treatments include prescribed burning, hand-pulling, and control as indicated by red, blue, and green colours, respectively.

3.2.1. Growth Rate

Growth rate of *C. stoebe* differed between the control and hand-pull treatments as indicated by the differing slopes of the best-fit lines. In addition, the y-intercept in May 2017 of control and hand-pull treatments differ, and the control y-intercept occurs at a greater height for *C. stoebe* (Figure 5). Since hand-pulling occurred in April 2017 and all individuals within sampling plots were removed including a 2-m buffer, *C. stoebe* occurring in the hand-pull treatments were recently germinated seeds from the residual seed bank. Based on my data, 60% of *C. stoebe* in the hand-pull treatments were basal rosettes (non-bolting adults assigned a height value of 0.5 cm). Therefore, the growth rate of *C. stoebe* in the hand-pull treatments is weighted down by the high proportion of basal rosettes. In comparison, 87% of *C. stoebe* in the control treatments were bolting adults. This is biologically significant because only bolting individuals produce flowers and reproduce by seed (Jacobs and Sheley 1998, Story et al. 2001). The growth rate for *C. stoebe* in the burn treatments is zero because no individuals occurred in the sampling plots.

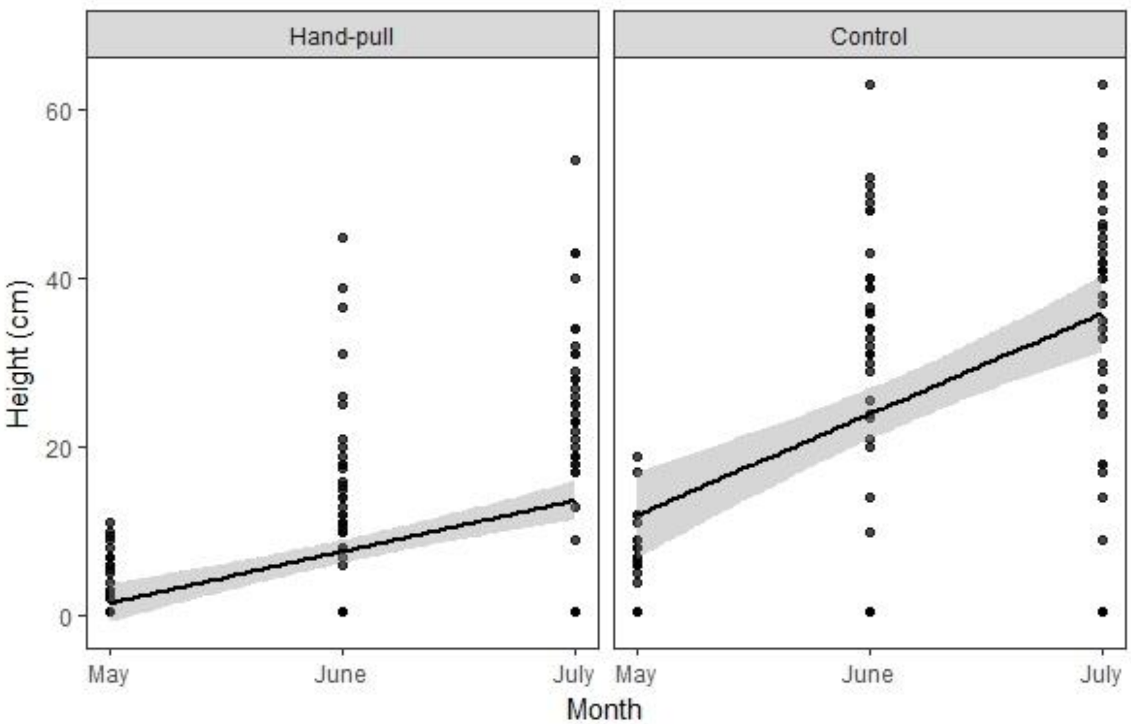


Figure 5. Growth rate of *Centaurea stoebe* from May to July 2017 between the hand-pull (N = 204 individuals, y = 6.1x – 4.6) and control (N = 102 individuals, y = 11.98x – 0) treatments in Kenna Cartwright Park, Kamloops. Lines-of-best-fit were produced using generalized linear modelling with the 95% confidence interval indicated by the shading around each line. Hand-pulling treatment occurred in April and May of 2017.

Based on the estimates for hand-pull and control, the mean height of *C. stoebe* in hand-pull and control treatments does not differ (Table 5). However, 60% of *C. stoebe* in the hand-pull treatment were basal rosettes, while 13% of *C. stoebe* in the control treatment were basal rosettes.

Table 5. Summary table of the linear mixed-effects model generated using residual maximum likelihood (REML) using the height of *Centaurea stoebe* among treatments groups (hand-pull and control) as fixed effects and survey date and sampling unit as random effects. Model selection was based on minimizing the Akaiki Information Criterion (AIC) estimates.

Fixed Effects	Estimate	Standard Error	t-value
Hand-pull	7.429	5.630	1.320
Control	12.841	5.062	2.537
Random Effects	**Variance**	**Standard Deviation**	
Survey date	61.63	7.85	
Sampling unit	60.07	7.75	
Residual	122.33	11.06	

To illustrate the model selection process, I graphed the residual error of each model tested and the AIC values for comparison. Model 4 is the model I selected (Table 5). The complexity of the models increased from model 1 to model 9, apart from model 4 which is less complex than model 3. The residual error of the model does not decrease with increasing complexity beyond model 3 (Figure 6), and the AIC value is minimized in model 4 (Figure 7). Model 3 differs from model 4 by the inclusion of Site as a random effect; however, including Site as a random effect explained a negligible amount of variance (1.978×10^{-11}) and increased the complexity.

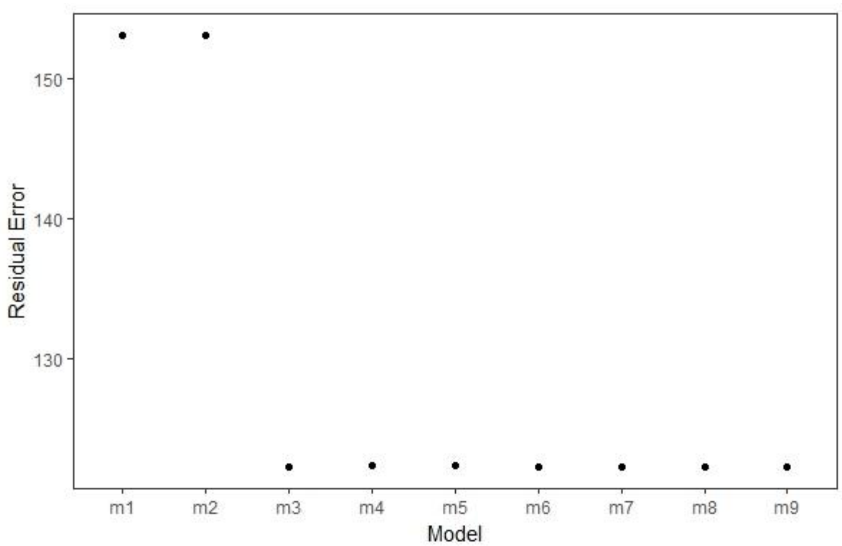


Figure 6. Residual error among the nine linear mixed-effects models tested for *Centaurea stoebe*. All models were generated using residual maximum likelihood (REML). Models differ only in the random effects tested and increase in complexity from model 1 to model 9, apart from model 4.

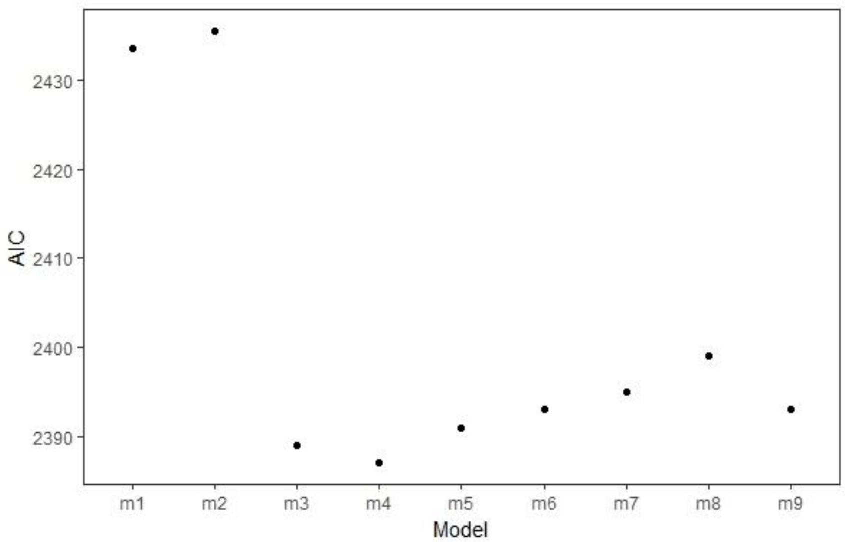


Figure 7. Aikaiki information criteria (AIC) for the nine linear mixed-effects models tested for *Centaurea stoebe*. The models were generated using residual maximum likelihood (REML). Models differ in the random effects and increase in complexity from model 1 to model 9, apart from model 4.

3.2.2. Height

The mean height of *C. stoebe* was greater in the control compared to the hand-pull treatments across all survey dates (Figure 8). At the end of the growing season (July) mean height and 95% confidence interval in the hand-pull treatment was 14.3 ± 3.64 cm compared to 32.92 ± 5.52 cm in the control. There were no individuals of *C. stoebe* detected in the burn sampling plots, and, because zero values were removed prior to plotting the mean height, the burn treatment was not plotted. Mean height of *C. stoebe* increased from May to July in both treatments. In the hand-pull treatments, the mean height of individuals was weighed down because 60% of *C. stoebe* remained in basal rosette form (recorded as 0.5 cm height) and did not bolt in the summer of 2017.

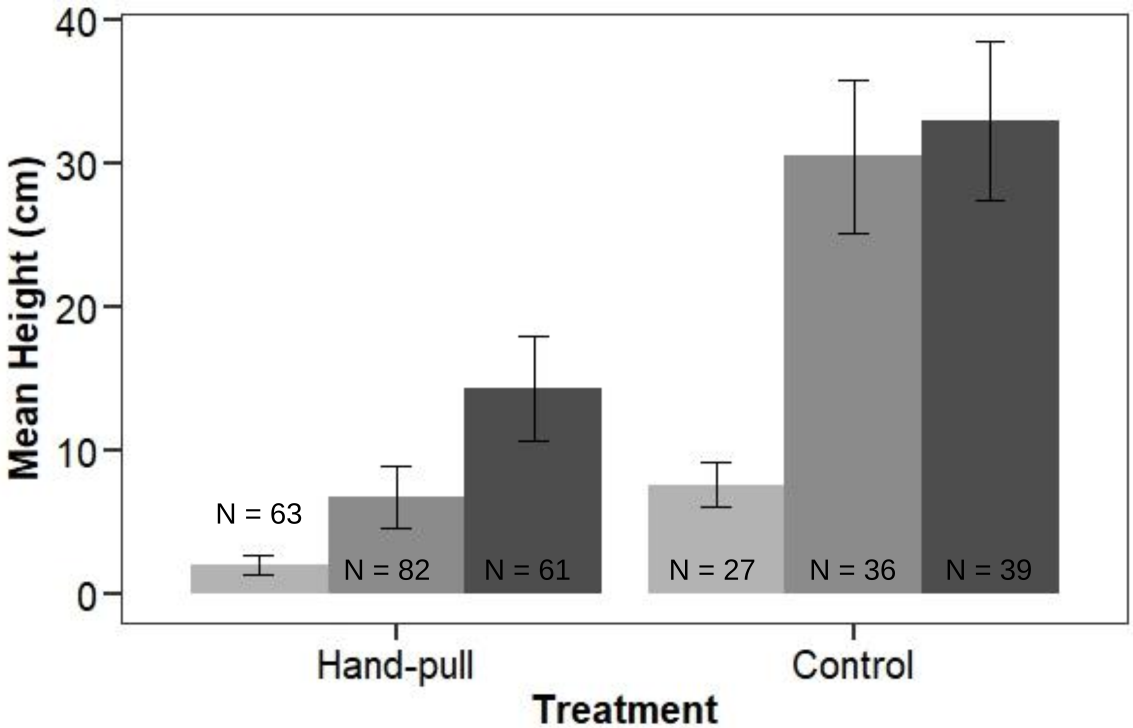


Figure 8. Mean height (cm) of *Centaurea stoebe* among treatments (Hand-pull and Control) in Kenna Cartwright Park, Kamloops. Error bars indicate the 95% confidence interval. Data was collected over three survey dates May, June, and July 2017 as indicated by the light, medium, and dark grey shaded bars, respectively. Hand-pulling was conducted in spring 2017.

3.2.3. Stem Density

Mean stem density of *C. stoebe* was greater in the hand-pull treatments compared to control (Figure 9). At the end of the growing season (July), mean stem density and 95% confidence interval in the hand-pull treatments was 6.15 ± 4.67 stems compared to 1.95 ± 2.17 stems in control treatments. Mean stem density of *C. stoebe* was zero in the burn treatments because no individuals occurred in any of the burn treatments. Greater variability in stem density was observed in the hand-pull treatments compared to control. Stem density increased in the control treatment from May to July while the lowest stem density for *C. stoebe* was observed in June for the hand-pulling treatments. High variability in mean stem density of *C. stoebe* resulted from the clumped distribution of *C. stoebe* across the study area; often sampling plots contained no individuals or several individuals, I rarely observed only one or two.

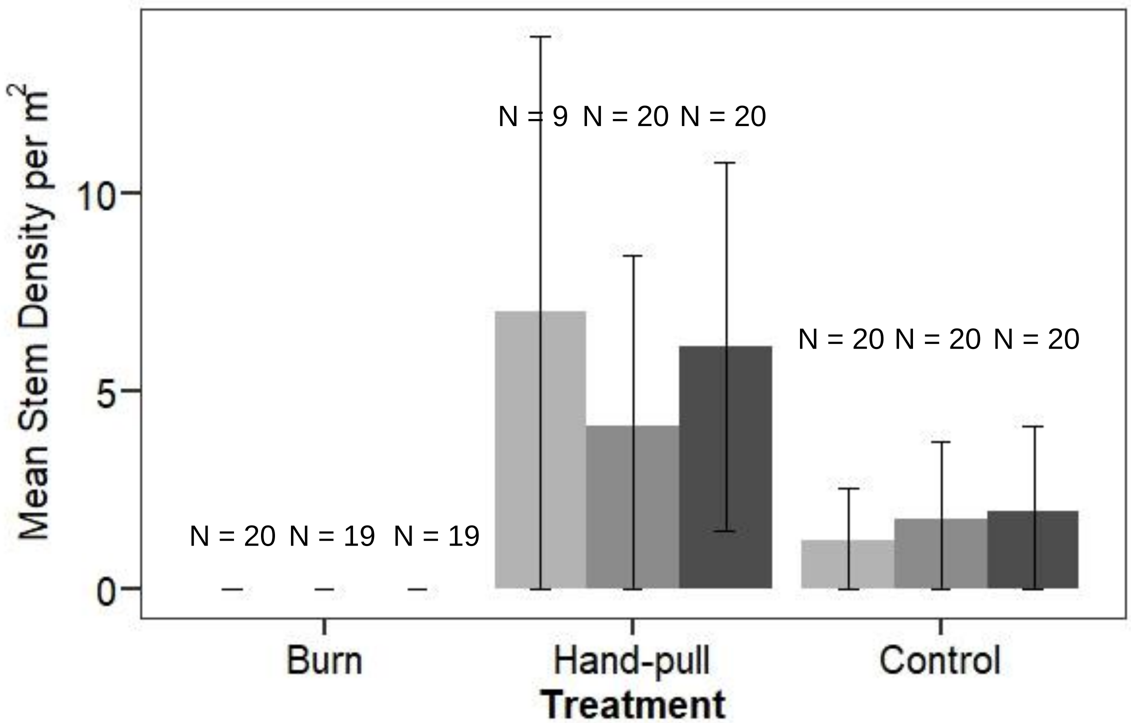


Figure 9. Mean stem density of *Centaurea stoebe* in the three treatments (burn, hand-pull, and control) in Kenna Cartwright Park, Kamloops. Survey dates occurred in May, June, and July 2017 and are indicated by the light, medium, and dark grey shaded bars, respectively. Error bars indicate the 95% confidence interval.

A time-since-burning comparison of the mean height and stem density of *C. stoebe* in the two burn treatments was not performed because *C. stoebe* was not detected in the burn treatments. *Centaurea stoebe* individuals growing in the hand-pull treatments were smaller in height with a greater proportion of basal rosettes but more numerous than the individuals growing in the control treatments.

3.3. *Linaria dalmatica*

Detection rates for *L. dalmatica* were highest in the burn treatments (Table 6). I detected more *L. dalmatica* in the hand-pull treatments compared to control treatments in both Sites 3 and 4. Survey effort was comparable among the four sites: my survey effort in Site 1 was 0.03% of the Site surveyed, Site 2 was 0.025%, Site 3 was 0.02%, and Site 4 was 0.025%. However, considering the overall size of my sites, my survey effort was low. Detection rates were more similar between treatments then between Sites for *L. dalmatica.*

Table 6. Mean detection rate of *L. dalmatica* with 95% confidence interval in the four Sites among three treatments in Kenna Cartwright Park, Kamloops. Detection rates are expressed as a percent of sampling plots that contained at minimum one individual and the mean was calculated among the survey dates (May, June, and July, 2017).

Site	Burn	Hand-pull	Control
Site 1	47% (± 18%)	-	-
Site 2	43% (± 19%)	-	-
Site 3	-	52% (± 19%)	17% (± 14%)
Site 4	-	35% (± 22%)	10% (± 11%)

L. dalmatica exhibits a clumped distribution within the study area (Figure 10). *L. dalmatica* occurs in the southern part of Sites 1 and 2, the western part of Sites 1, 3, and 4. This is similar to the 2001 Invasive Plant Inventory Mapping for Kenna Cartwright Park that indicates heavy infestation of *L. dalmatica* in the south and western areas of Sites 1 and 2, and low to moderate infestation in Sites 3 and 4 (Tarasoff 2002). I detected the greatest number of *L. dalmatica* in the prescribed burn treatments in Sites 1 and 2, followed by hand-pull, and then control. Among my study sites, I detected the fewest individuals of *L. dalmatica* in Site 4.

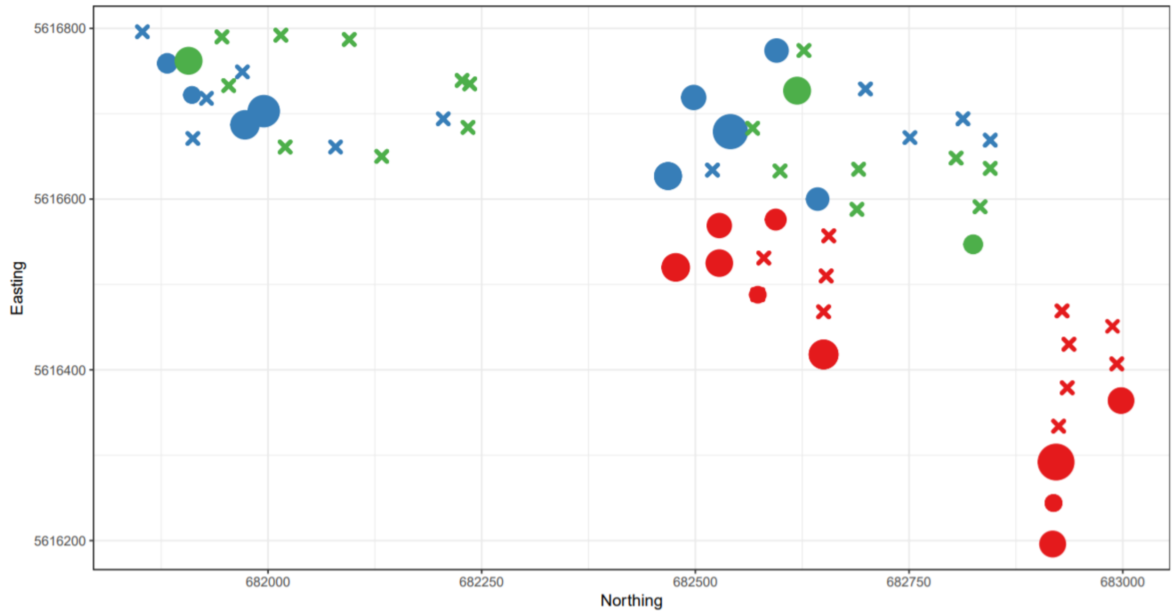


Figure 10. Detection of *Linaria dalmatica* and relative density among sampling plots within the study area in Kenna Cartwright Park, Kamloops. An 'x' denotes no detection of *L. dalmatica* within the sampling plot and circles indicate presence of *L. dalmatica.* Circle size indicates relative density of *L. dalmatica* with larger circles indicating higher density. Three treatments include prescribed burning, hand-pulling, and control as indicated by red, blue, and green colours, respectively.

3.3.1. Growth Rate

Growth rate (determined using the log of height) was greatest in the control treatments compared to the hand-pull and prescribed burn treatments for *L. dalmatica*. Growth rate was comparable between the prescribed burn and hand-pull treatments as indicated by parallel lines-of-best-fit. However, the y-intercept of the prescribed burn treatment occurs at a greater height than the hand-pull treatments, while individuals in the hand-pull and control plots have a similar y-intercept in May 2017 (Figure 11). The higher y-intercept of *L. dalmatica* in the prescribed burning treatments might be due to the predominantly southern aspects of Sites 1 and 2. Southern aspects typically become snow free earlier in the growing season enabling a longer growth period prior to May surveys. However, while I was establishing sampling plots in March of 2017, I observed *L. dalmatica* individuals in Site 3 that had germinated despite there being snow on the ground.

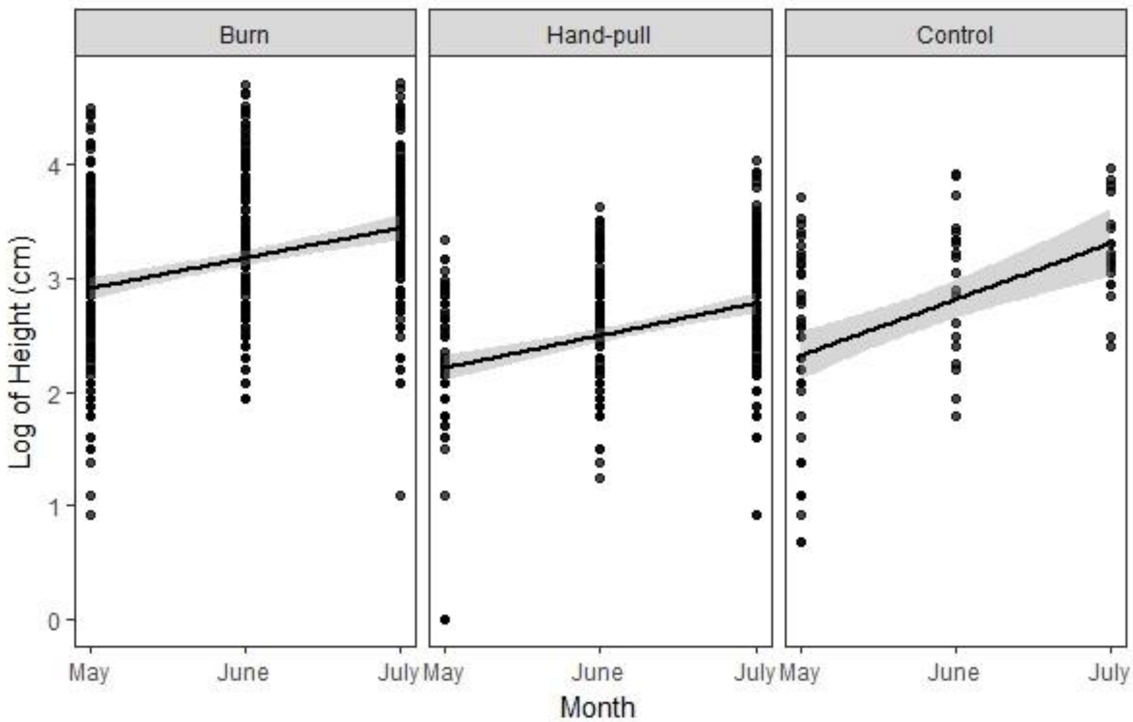


Figure 11. Growth rate of *Linaria dalmatica* from May to July 2017 among burn (N = 421 individuals, y = 0.27x + 2.6), hand-pull (N = 356 individuals, y = 0.28x + 1.9), and control (N = 82 individuals, y = 0.49x + 1.8) treatments in Kenna Cartwright Park, Kamloops using log transformed height. Lines-of-best-fit were produced using linear modelling with the 95% confidence interval indicated by the shading around each line. Prescribed burning was conducted in March 2015 and March 2016.

Using the log height of *L. dalmatica*, I selected a linear mixed-effects model that minimizes the AIC value. I included treatment (prescribed burn, hand-pull, and control) as the fixed effects and the random effects of the model I selected included survey date, sampling unit, and experimental unit (Table 7). Based on the model output, the height of individuals in the prescribed burn treatment differs significantly from the hand-pull and control treatments, but the control and hand-pull treatments do not differ from one another. In addition, a high amount of variance could not be explained by the random effects.

Table 7. Summary table of the linear mixed model fit using residual maximum liklihood (REML) using the log height of *Linaria dalmatica.* Model selection was based on minimizing the Akaiki information criterion (AIC). The best-fit model includes treatments (prescribed burn, hand-pull, and control) as fixed effects, and survey date, sampling unit, and site as random effects.

Fixed Effects	**Estimate**	**Standard Error**	**t-value**
Prescribed burn	3.0777	0.3045	10.108
Hand-pull	-0.6858	0.3798	-1.805
Control	-0.3252	0.4177	-0.778
Random Effects	**Variance**	**Standard Deviation**	
Survey date	0.06398	0.2529	
Sampling unit	0.11861	0.3444	
Site	0.11276	0.3358	
Residual	0.32732	0.5721	

To illustrate the performance of the selected model in comparison to the other models I tested, I graphed the residual error and AIC values of each model for comparison. Model 3 is the model I selected. The complexity of the models increases from model 1 to model 13. The residual error of the model decreases after model 3 only for the most complex models (models 10 to 13) that consider random slope and intercept (Figure 12). The AIC value is minimized in model 3 (Figure 13).

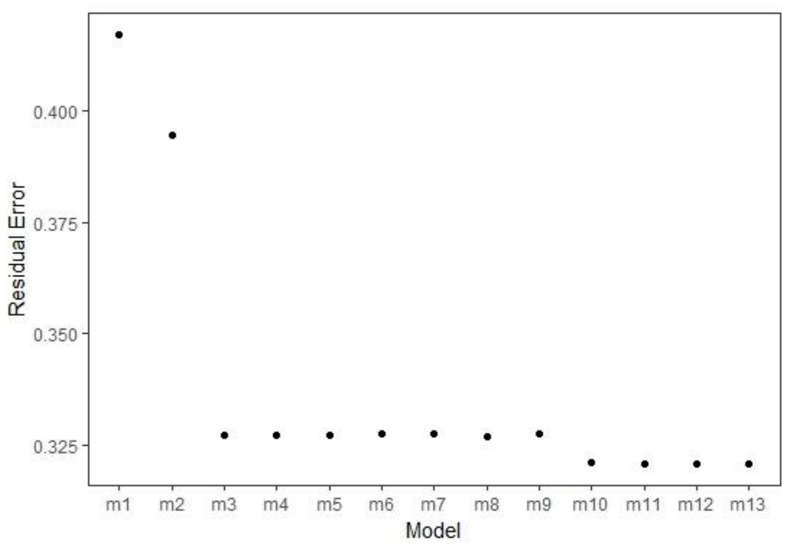


Figure 12. Residual error among 13 linear mixed-effects models tested for *Linaria dalmatica*. All models were generated using residual maximum likelihood (REML). Models differ in random effects and increase in complexity from model 1 to model 13.

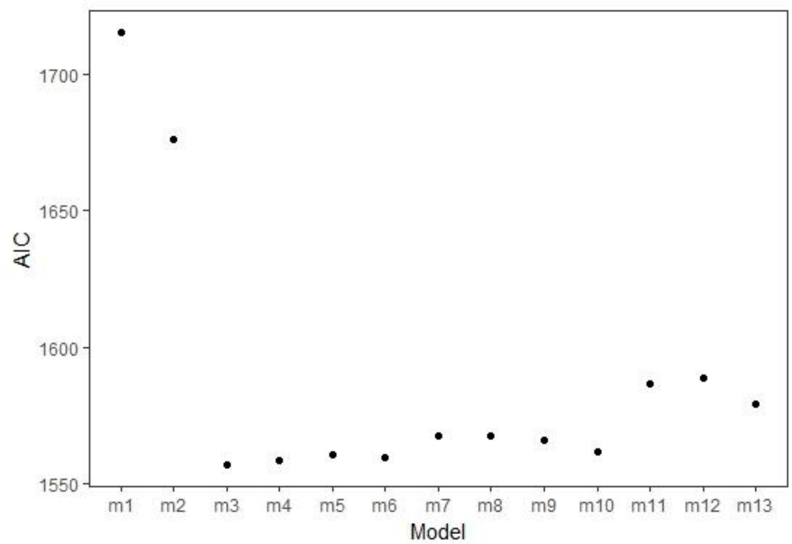


Figure 13. Aikaiki information criteria (AIC) for the 13 linear mixed-effects models tested for *Linaria dalmatica*. The models were generated using residual maximum likelihood (REML). Models differ in random effects and increase in complexity from model 1 to model 13.

3.3.2. Height

Mean height of *L. dalmatica* increased from May to July in all three treatments. Mean height of *L. dalmatica* was greatest in the burn treatments for all survey dates relative to the hand-pull and control treatments, and individuals in the control plot had a greater mean height than individuals in the hand-pull treatments (Figure 14). At the end of the growing season (July), the mean height (95% confidence interval) of *L. dalmatica* in burn treatments was 35.78 (± 3.98) cm, 18.01 (± 1.74) cm in hand-pull treatments, and 27.63 (± 5.25) cm in control treatments. Variability in the height of *L. dalmatica* was greatest in control treatments and lowest in the hand-pull treatments. One month following hand-pull treatment the mean height of *L. dalmatica* was similar to individuals in the control treatments; however, growth rate in the hand-pull treatments was slower.

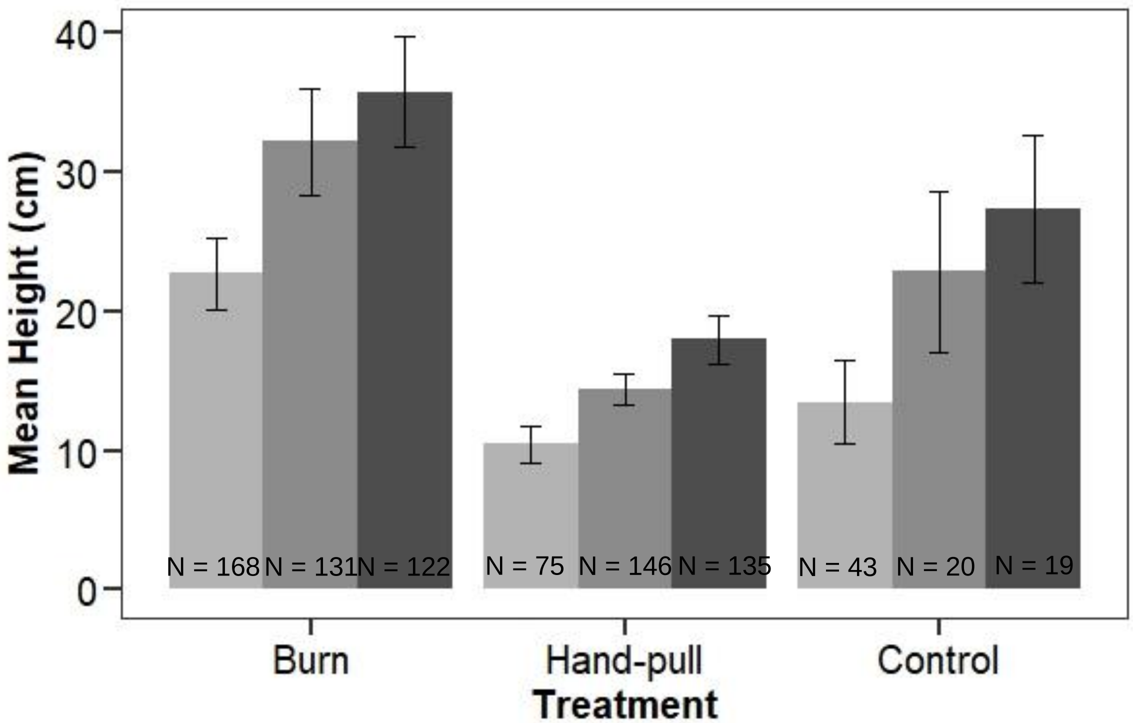


Figure 14. Mean height (cm) of *Linaria dalmatica* among three treatments (Burn, Hand-pull, and Control) in Kenna Cartwright Park, Kamloops. Error bars indicate the 95% confidence interval. Data was collected over 3 sampling periods in May, June, and July 2017 as indicated by light, medium, and dark grey shading, respectively. Burn refers to prescribed burning that was conducted in March 2015 and 2016. Hand-pulling was conducted in spring 2017.

3.3.3. Stem Density

Stem density of *L. dalmatica* was lowest in the control treatments and similar between the burn and hand-pull treatments. However, variability was high in all treatments (Figure 15). High variability resulted from the clumped distribution of *L. dalmatica* across the study area; often sampling plots contained no individuals or several individuals, rarely only one or two. Mean stem density and 95% confidence interval in the burn treatments at the end of the growing season (July) were 6.42 (± 5.53) stems per m^2, 6.75 (± 5.28) stems per m^2 in the hand-pull treatments, and 0.40 (± 0.60) stems per m^2 in the control treatments.

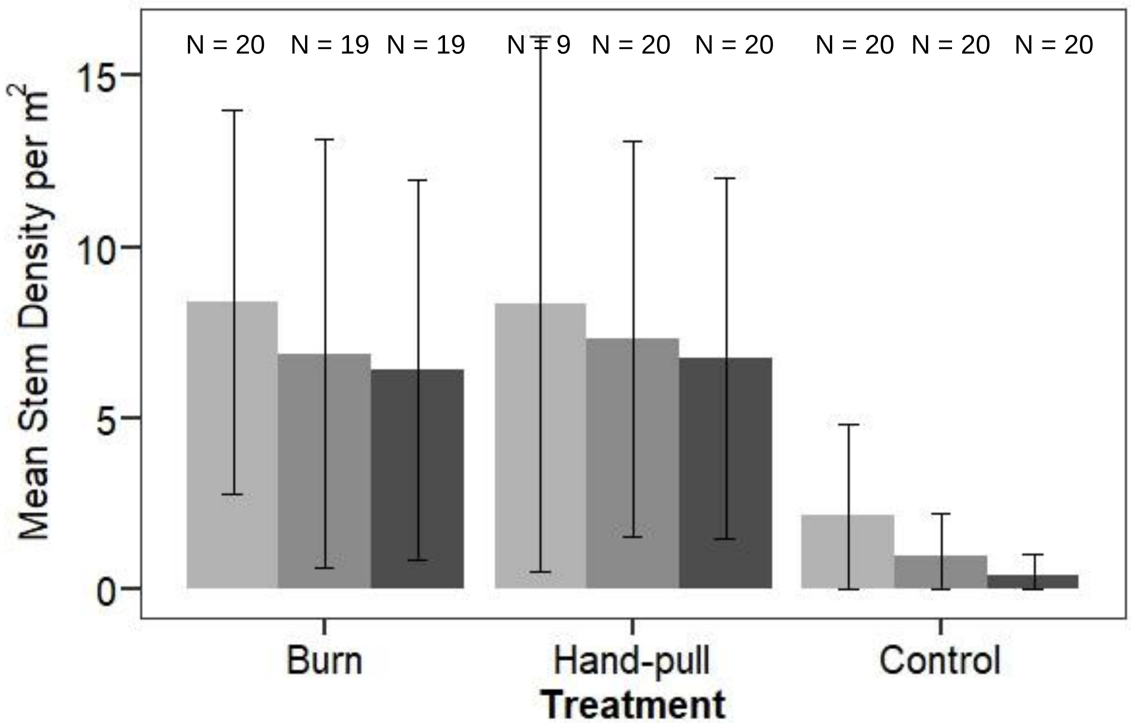


Figure 15. Mean stem density (per 1m^2) of *Linaria dalmatica* across three treatments (Burn, Hand-pull, and Control) in Kenna Cartwright Park, Kamloops. Error bars indicate the 95% confidence interval. Data was collected over 3 sampling periods in May, June, and July 2017 indicated by light, medium and dark grey shading, respectively.

3.3.4. Time-since-burning

Mean height of *L. dalmatica* was greatest in the 2-year post-burn treatment in comparison to the 1-year post-burn and control treatments across all survey dates; however, there was no difference between the 1-year post-burn and control treatments (Figure 16). At the end of the growing season (July), mean height and 95% confidence interval in the 2015 burn treatment was 39.72 ± 4.99 cm, 29.49 ± 6.23 cm in the 2016 burn treatments, and 27.37 ± 5.25 cm in the control treatments. This suggests that there might be an interaction of burning that influences the growth of *L. dalmatica* beyond the first season after burning. However, without recent information on the height and density of *L. dalmatica* in the study sites prior to prescribe burning, it is possible that individuals in the 2-year post-burn site might be taller for reasons unrelated to the prescribed burning treatment such as soil moisture or sunlight availability.

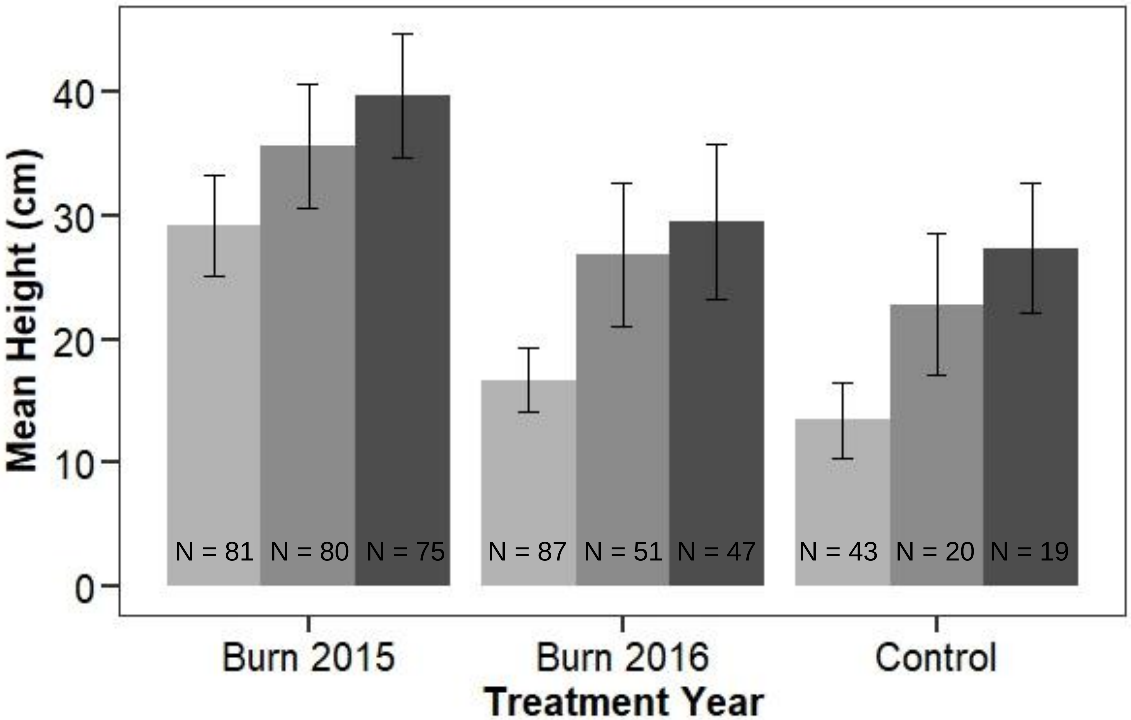


Figure 16. Comparison of the mean height (cm) of *Linaria dalmatica* in burn treatments conducted in March 2015, March 2016, and control in Kenna Cartwright Park, Kamloops. Data was collected over 3 sampling periods in May, June, and July 2017 indicated by light, medium and dark grey shading, respectively. Error bars indicate the 95% confidence interval.

Stem density of *L. dalmatica* is highly variable due to the clumped distribution across the study area. The mean stem density was greater in both burn treatments compared to control treatments; however, there is a complete overlap of the 95% confidence intervals for both burn sites and the control treatments (Figure 17). Mean density (with 95% confidence interval) at the end of the growing season (July) was 8.33 (± 10.05) stems per m^2 in the 2015 burn treatment, 4.7 (± 5.73) stems per m^2 in the 2016 burn treatment, and 0.40 ± (0.60) stems per m^2 in the control treatments. The burn site from 2015 exhibited similar mean stem density across all survey dates while the burn site from 2016 exhibited a decline in mean stem density after the May survey date. Similarly, control sites have a declining mean stem density through the growing season.

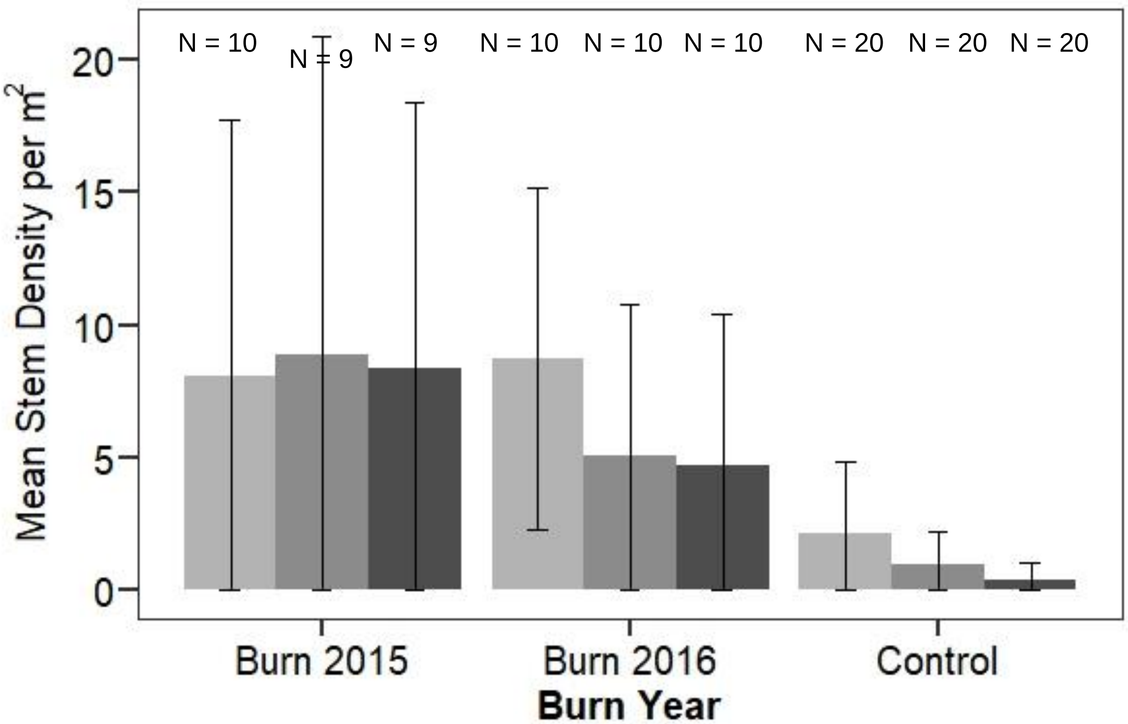


Figure 17. Comparison of the stem density (number of stem per 1m^2) of *Linaria dalmatica* in burn treatments conducted in March 2015, March 2016, and control plots in Kenna Cartwright Park, Kamloops. Data was collected over 3 sampling periods in May, June, and July 2017 indicated by light, medium and dark grey shading, respectively. Error bars indicate the 95% confidence interval.

4.0 Discussion

4.1. *Centaurea stoebe*

My results indicate that *C. stoebe* was not detected in the burn treatments, which are nested in Sites 1 and 2. Hand-pull and control treatments were crossed between Sites 3 and 4. Individuals of *C. stoebe* in the hand-pull treatments have a slower growth rate compared to individuals in the control treatments. *Centaurea stoebe* in the hand-pull treatments have a smaller mean height, but hand-pull treatments had a greater stem density compared to control treatments. Sixty percent of *C. stoebe* in the hand-pull treatments remained in the basal rosette form compared to 13% remaining as basal rosettes in the control treatments.

Centaurea stoebe was not detected in the prescribed burning sampling plots during data collection from May to July 2017. Data on the stem density, height, and growth rate of *C. stoebe* within Sites 1 and 2 prior to burning was not available. Invasive Plant Inventory mapping available for Kenna Cartwright Park from 2001 indicates low *C. stoebe* infestation south of Sites 1 and 2, and moderate infestation of *C. stoebe* in the western part of Sites 1 and 3 (Tarasoff 2002). In addition, Invasive Alien Plant layers from iMapBC indicate *C. stoebe* occurring south of Sites 1 and 2, but the records do not provide assessment dates (Data BC, Province of British Columbia 2013). Without quantifiable pre-burn data on the stem density and growth rate of *C. stoebe* it is difficult to determine whether the absence of *C. stoebe* in sampling plots was the result of prescribed burning. *Centaurea stoebe* is widespread throughout the park (Tarasoff 2002, Marrow 2017), but no individuals were detected in any of my sampling plots in the prescribed burn treatments suggesting that *C. stoebe* might be less common in Sites 1 and 2. I propose three possible theories for the low detection rate of *C. stoebe* within the burn sampling plots, including:

- reduced competitive ability of *C. stoebe* because of residual charcoal content in prescribed burn treatments,
- negative effects of prescribed burning on populations of *C. stoebe*, and
- low detection rate of *C. stoebe* due to survey effort and site conditions within the prescribed burn treatments.

Residual charcoal content in the soil from prescribed burning (though not measured directly) might affect the ability of *C. stoebe* to compete with other plants. Residual charcoal is likely greater in the prescribed burn areas compared to the hand-pull and control treatments, where no burning has occurred in recent years. Plants in the *Centaurea* genus exude unique allelopathic chemicals that contribute to their success in North America grasslands (Callaway and Aschehoug 2000). In comparison to native forbs, *C. stoebe* tissues have higher levels of phosphorus, and *C. stoebe* is capable of taking up more phosphorus from the soil; however, secondary biochemicals released by *C. stoebe* might increase phosphorous availability in the soil and alter phosphorous cycling (Thorpe et al. 2006, Weidenhamer and Callaway 2010). Studies using charcoal addition reduced the effect of *C. stoebe* secondary biochemicals on soil phosphorus availability (Weidenhamer and Callaway 2010). Without increased phosphorus uptake, the competitive ability of *C. stoebe* might be reduced. Therefore, the low detection rate of *C. stoebe* in Sites 1 and 2 might be because of reduced competitive ability following prescribed burning.

A second possibility for observing no individuals of *C. stoebe* in the burn treatments is that prescribed burning negatively affects *C. stoebe* population growth. In similar studies, low-intensity spring and summer burning was an effective tool for reducing the population size of *C. stoebe* by reducing flower and seed production (Emery and Gross 2005). Three consecutive years of spring burning reduced the density of *C. stoebe,* decreased rates of juvenile establishment, increased adult mortality, and depleted the residual seed bank in burn plots compared to control (MacDonald et al. 2007). *C. stoebe* is a widespread weed throughout the park (Tarasoff 2002, Marrow 2017), and the low detection rates within the prescribed burn area (Sites 1 and 2) compared to adjacent survey areas (Sites 3 and 4) suggests that prescribed burning might be an effective tool at reducing the dominance of *C. stoebe.* MacDonald et al. (2007) indicate that timing prescribed burns to maximize the negative effects on *C. stoebe* can reduce the dominance of *C. stoebe* in the plant community, particularly in ecosystems where fire is a natural disturbance agent. Spring burning conducted annually can reduce the density of *C. stoebe* to levels that enable the persistence and dominance of native grasses (Sheley et al. 1998). However, there is still controversy in whether prescribed burning is an effective tool for reducing *C. stoebe* populations. No effect on the density or biomass of *C. stoebe* was detected following a single prescribed burn conducted in early April in

Michigan, but burning conditions were not optimal (MacDonald et al. 2013). Therefore, the effect of prescribed burning on *C. stoebe* growth and stem density in Kenna Cartwright Park requires further investigation (see Research Implications).

Lastly, my sampling plots within the prescribed burn treatment might not have captured the individuals that occurred in Sites 1 and 2. In total I surveyed 20 1-m^2 sampling plots across a total burn area of approximately 7 ha. This represents a sampling effort of 0.029% of the burn area. It is possible my sampling plots missed the individuals that do occur in the burn areas. Using random selection to locate sampling plots within the treatment area can affect results, and a greater sampling effort will increase the accuracy of results (MacDonald et al. 2007). A greater number of sampling plots within the burn area would provide more information on whether *C. stoebe* is absent from the sites or if my sampling plots did not capture the presence of *C. stoebe.* However, sampling effort in Sites 3 and 4 were comparable to the burn treatments at 0.022% and 0.025%, respectively, and *C. stoebe* were detected in both Sites 3 and 4. Based on my field observations, individuals of *C. stoebe* occurred in Sites 1 and 2 where the burn treatment was applied, but were restricted to lower elevations where the slope is less steep. *Centaurea stoebe* were largely absent from higher elevation and steeper sampling plots. The areas in Sites 1 and 2 with more gradual slopes are also closer to pedestrian and bike trails. Areas located closer to trails might be more heavily infested because of the high use of these trails by park users who can facilitate the spread of weeds (Sheley et al. 1998). Areas close to trails might be under-represented by my sampling plots, which might affect detection of *C. stoebe.* In addition, mapping of *C. stoebe* from the Invasive Alien Plant Program indicates *C. stoebe* occurring at the south end of Sites 1 and 2 but no presence on the slope itself (Data BC, Province of British Columbia 2013). However, invasive plant points provided by the Invasive Alien Plant Program fall only along the trails in the park, so might under represent areas further from the trails.

Prescribed burning is nested in Sites 1 and 2, which are also predominantly south facing slopes. Prescribed burning in grasslands removes litter increasing soil temperature and decreasing soil moisture because of increased evaporation and transpiration (Hulbert 1969). Considering the southern aspects of Sites 1 and 2, increased soil temperature and decreased soil moisture might be exacerbated in these Sites, especially in the summer of 2017 when precipitation during the summer months was low (The Weather

Network 2018, Weather Stats 2018), resulting in dry microclimates. *C. stoebe* grows best on mesic sites and seed germination is greatest when soil moisture levels are at field capacity (Eddleman and Romo 1988). The low detection rates of *C. stoebe* might be related to soil moisture levels in Sites 1 and 2, which likely had the driest soil of the Sites. The low soil moisture availability might have inhibited germination and growth of *C. stoebe* during my survey period. Future studies should look at soil moisture levels in the different treatment areas to determine if soil moisture can explain the distribution of *C. stoebe*.

Based on my results, hand-pulling is an effective management strategy for small patches of *C. stoebe*. Hand-pulling increased stem density of *C. stoebe* compared to control, but the individuals were smaller in height throughout the growing season. Hand-pulling, after mowing and herbicide application, reduced both biomass and density of *C. stoebe* over three years to 5.8% and 7.1% in comparison to mowing and herbicide application alone (MacDonald et al. 2013). In my hand-pulling treatments I removed all individuals observed aboveground in the sampling plots, and all individuals aboveground in a 2-m buffer around the sampling plots; therefore, *C. stoebe* surveyed following treatment in the hand-pulling plots were recently germinated seeds from the seed bank. Hand-pulling of *C. stoebe* increases opportunity for more seeds to germinate from the seed bank, and, with follow-up hand-pulling treatments, this can deplete the seed bank of *C. stoebe* (MacDonald et al. 2013). Bolting of *C. stoebe* is rare in the first-year after germination; therefore, most first-year individuals do not produce flowering stems (Story et al. 2001). This is consistent with my results; 60% of *C. stoebe* in the hand-pull treatments remained in the basal rosette form compared to 13% in the control treatments. *C. stoebe* in the control treatments were also taller than individuals in the hand-pull treatments; the height of *C. stoebe* stems is correlated with the number of flowering heads (Story et al. 2001), so taller individuals produce more seeds. If hand-pulling is continued in consecutive years, hand-pulling could deplete the seed bank by reducing seed production and promoting germination from the seed bank (MacDonald et al. 2013).

Combining hand-pulling with other treatments such as mowing or prescribed burning that is timed with the flowering of *C. stoebe* can be an effective method to limit seed production by *C. stoebe* and result in negative population growth (MacDonald et al. 2007). However, testing combined treatments is necessary to determine the effectiveness of different combinations at reducing the density and growth of *C. stoebe.*

Hand-pulling of small infestations of *C. stoebe* is effective based on my data, but would require follow-up treatment over multiple years to be able to deplete the seed bank (MacDonald et al. 2013). Seeds of *C. stoebe* can remain viable in the seed bank for up to 8 years (Davis et al. 1993). Hand-pulling is also labour intensive and unrealistic for large infestations (MacDonald et al. 2013), but could be used in Kenna Cartwright Park to limit the spread from already infested areas to adjacent sites. Used in conjunction with herbicide application and mowing, hand-pulling can be effective at reducing *C. stoebe* density to enable native vegetation to re-establish (MacDonald et al. 2013).

4.2. *Linaria dalmatica*

Linaria dalmatica exhibits a faster growth rate in the control treatments compared to the burn and hand-pull treatments. Individuals of *L. dalmatica* were tallest in the burn treatment followed by control, and individuals in the hand-pull treatments were the smallest. Height of individuals is biologically significant because taller individuals have greater reproduction (Robocker 1970). Mean stem density in the burn and hand-pull treatments are greater than control treatments. My data suggests prescribed burning might affect population growth of *L. dalmatica* beyond the initial burn year, since the 2-year post-burn treatments had the tallest individuals and the highest abundance of *L. dalmatica* compared to the 1-year post-burn and control treatments.

Jacobs and Sheley (2003) examined the effects of prescribed burning on the density, biomass, and seed production of *L. dalmatica,* and concluded that spring burning increased biomass of *L. dalmatica* when measured at the end of the same growing season. This is consistent with my results; however, I examined height instead of biomass. Burning increases nutrient input, particularly nitrogen, which is typically limiting in grassland ecosystems (Jacobs and Sheley 2003). *Linaria dalmatica* might be able to maximize nutrient acquisition following early spring burning because *L. dalmatica* emerges in early spring before many native grasses begin growth for the season (Robocker 1970, Jacobs and Sheley 2003). The prescribed burns were conducted in March of 2015 and 2016. I observed *L. dalmatica* emerging as early as March when I began field surveys in 2017 in Site 3, even though snow was still on the ground.

Disturbance from prescribed burning can provide opportunities for invasive plants. Prescribed burning can reduce soil moisture availability, especially in the top soil layers, by removing litter layers exposing soil to sunlight and wind, thereby increasing evaporation (Hulbert 1969). *Linaria dalmatica* has an extensive root system that penetrates to deeper soil layers than native bunchgrasses, which enables *L. dalmatica* to maximize water acquisition from the soil (Vujnovic and Wein 1997). Acquiring water from the soil is a competitive advantage for *L. dalmatica,* especially during periods of drought. The months of June and July, 2017 received one tenth the 30-year average precipitation resulting in dry conditions (The Weather Network 2018, Weather Stats 2018). Sites 1 and 2 likely have the lowest soil moisture availability because these sites are predominantly south facing and were subject to prescribed burning. The dominance of *L. dalmatica* in Sites 1 and 2 might be a result of *L. dalmatica*'s ability to withstand dry soil conditions. My field observations also indicate these sites have well drained, steep slopes with coarse fragments. *L. dalmatica* is competitive at colonizing open, sunny areas with coarse soils (Vujnovic and Wein 1997).

My results also suggest height of *L. dalmatica* continues to increase with increased time-since-burning as indicated by the taller individuals of *L. dalmatica* occurring in the older burn treatment. In similar studies, prescribed burning significantly increased seed production by *L. dalmatica* (Jacobs and Sheley 2003). Though I did not measure seed production directly, my results are consistent with this observation. If seed production is increased in the growing season following spring burning, population increases due to increased seed production would not be observed until the following years. A higher density of *L. dalmatica* in the 2-year post-burn site is expected because of the increase in seed production. If burning increases seed production, the burn treatments will have a greater residual seed bank of *L. dalmatica,* and burning will result in more seedlings germinating in subsequent years after the initial burn. The density of *L. dalmatica* in Arizona following fire was observed to peak 2-years post-burn, with declining density after this peak (Dodge et al. 2008). This is consistent with my observations that density is greatest in the 2-year post-burn treatment. My results also suggest that the older burn sites have the tallest individuals. Larger individuals produce more seeds because larger individuals have more flowering branches in addition to the main flowering stem (Robocker 1970). This suggests a beneficial interaction of time-since-burning for *L. dalmatica*. Monitoring of the burn treatments and future prescribed burns in the park should continue to determine if the interaction of time-since-burning persists beyond 2-years post-burn, and if this interaction is observed at other sites in the park.

Stem density of *L. dalmatica* was constant in the 2015 burn treatments, but declined in the 2016 burn treatments and control treatments in June. The decline in stem density might be the result of low precipitation in June and July resulting in desiccation. However Site 2, where the 2015 burn occurred, is a predominantly south-facing slope with minimal shade cover from trees except at the base of the slope. I would expect desiccation to be most pronounced on this Site, but it appeared to be the only Site where stem density remained constant over the growing season. Smaller *L. dalmatica* and recently germinated individuals are not competitive against well-established native bunchgrasses, but once established *L. dalmatica* is highly competitive (Jacobs and Sing 2006). Individuals in the control and 2016 burn treatments were smaller than individuals in the 2015 burn treatments. Smaller *L. dalmatica* might not survive the dry conditions because of limited competitive ability, resulting in a decline in stem density.

Invasive plant inventory mapping from 2001 indicates high infestation of *L. dalmatica* in the south and western part of Site 1 and southern part of Site 2 where prescribed burning was applied (Tarasoff 2002). Patches of low infestation of *L. dalmatica* occur in Sites 3 and 4 (Tarasoff 2002). Additional information from the Invasive Alien Plant Program, accessed through iMapBC, indicate the occurrence of *L. dalmatica* in the south and north areas of Sites 1 and 2, the south and west section of Site 3, and south of Site 4 (Data BC, Province of British Columbia 2013). The Invasive Alien Plant Program data only provides locations that occur along trails in the park, so the data might under-represent areas further from trails. The rate of spread of *L. dalmatica* in the United States ranges from 8-29% annually (Duncan et al. 2004) and is greatest on coarse textured soil (Vujnovic and Wein 1997), so the infestation levels assessed in 2001 have likely spread prior to the first prescribed burn in 2015. Additional measures to reduce *L. dalmatica* have been undertaken in the park; these measures include the release of the biological control agent *Mecinus janthinus*, a root mining weevil specific to *L. dalmatica* that occurred in 2000, and goat grazing that occurred from 2013 to 2015 in Sites 1 and 2. *M. janthinus* was released at a location between Sites 1 and 3 (denoted by the red point labeled MEC 00-1 in Figure 3). Mean density of *L. dalmatica* at the release site in 2001 was 1.55 individuals per m^2 (Tarasoff 2002). Mean density (with 95% confidence intervals) of *L. dalmatica* in Sites 1 and 3 were 8.7 (± 6.43) stems per m^2 and 5.16 (± 4.34) stems per m^2 in May, respectively, and 4.7 (± 5.73) stems per m^2 and 3.75 (± 3.85) stems per m^2 in July, respectively. The goat grazing program was discontinued in 2015. For a detailed discussion see Alternative Treatments for Invasive Plants.

Based on my results, hand-pulling reduces the growth rate and height of *L. dalmatica,* and hand-pulling promotes germination of seeds from the residual seedbank by increasing the available space and increasing sunlight penetration to the soil. However, *L. dalmatica* can produce seeds in the same year as germination and seeds remain viable in the seedbank for up to ten years (Robocker 1970). Hand-pulling is labour intensive and would require multiple follow-up treatments. Therefore, hand-pulling alone is not an effective treatment for *L. dalmatica*. Stem density of *L. dalmatica* in areas treated by hand-pulling were comparable to those treated with prescribed burning and greater than control treatments. The height and growth rate of individuals in hand-pull treatments were lower than control, but not significantly. This suggests that hand-pulling can stimulate seed germination of *L. dalmatica,* but a month after hand-pulling the newly germinated individuals are similar in height to the control treatments. Hand-pulling is often suggested as an effective removal technique for small infestations, but my data suggests hand-pulling might not be worth the time and labour required to implement treatment, unless the residual seedbank is known to be minimal.

If prescribed burning is planned to continue in Kenna Cartwright Park, an integrated weed management plan is required and should include alternative treatments of *L. dalmatica* in future prescribed burn areas (Jacobs and Sheley 2003). Treatments to consider include herbicide application and seeding with native grasses (Jacobs and Sheley 2003). Future prescribed burns should continue to be low severity, as high severity fires have the most beneficial effect on the growth and density of *L. dalmatica* populations (Dodge et al. 2008). Hand-pulling alone is ineffective for treatment considering the size of prescribed burn areas in Kenna Cartwright Park and the increased density in comparison to control. Seeding with native species in combination with hand-pulling can help increase competitive stress on recently germinated *L. dalmatica* in hand-pull treatments since young *L. dalmatica* are poor competitors (Robocker 1970, Jacobs and Sing 2006).

4.3. Influence of Climate

When examining the growth and stem density of plants, it is important to consider the influence of climate over the field season(s) of study. I collected data over the summer of 2017, so my data is representative of one growing season and influenced by the climate of 2017. Precipitation in June and July were one tenth the 30-year monthly average. By July, I observed many plants desiccating. Grassland forbs under drought conditions increase root growth to acquire water and reduce shoot growth to limit transpiration and water loss (Hofmann and Isselstein 2004). My data used height of flowering stem to measure growth rates, but my data does not capture the underground growth of plants, which might have been significant in the dry conditions of the summer of 2017.

In invasive plant management, it is necessary to plan for changing climatic conditions. Change in climate can affect plant population growth rates, but the predicted effect is dependent on the characteristics of the plant, and the changes in climate. Changing climate has the potential to exacerbate the problem of invasive plants (B.C. Ministry of Environment 2016), and can differentially affect native and invasive plants. Below I summarize the current and predicted trends in climate, and how change in climate might influence *C. stoebe* and *L. dalmatica* populations in the southern interior of B.C.

In the southern interior of B.C., the temperature has risen over the past century an average of 0.9°C, with the greatest increase observed in winter months (1.5°C per century) (B.C. Ministry of Environment 2016). Increasing temperature is predicted to continue with global climate change, and will affect the form of precipitation received resulting in more precipitation falling as rain compared to snow (B.C. Ministry of Environment 2016). Increasing temperature can also increase evaporation rates and plant transpiration rates particularly in the summer (B.C. Ministry of Environment 2016). Ultimately, this might lead to hotter and drier conditions in the summer due to reduced soil moisture levels and drier fuels, conditions that are conducive to wildfire. Daily maximum and minimum temperatures have also been increasing over the past century. In particular, night-time minimum temperatures in fall, winter, and spring has been increasing, which can extend the growing season (B.C. Ministry of Environment 2016). A longer growing season will enable emergence of *L. dalmatica* even earlier in the year. Warmer temperatures might negatively affect *C. stoebe* if increasing temperatures result in drier conditions since *C. stoebe* growth is dependent on soil moisture availability (Eddleman and Romo 1988).

Precipitation is also increasing in the southern interior of B.C. with an increase of 17% observed over the past century; however, climate change is predicted to increase the year-to-year variation in precipitation and increase the number of severe storms (B.C. Ministry of Environment 2016). Larger storm events increase the amount of sediment loss (Mohamadi and Kavian 2015), and large storm events following fire can increase erosion and compromise slope stability (FAO 2007). Erosion and sediment run-off will be exacerbated in areas dominated by *C. stoebe* and *L. dalmatica* compared to areas of native grasses because of the high amount of exposed ground (Lacey et al. 1989). Ultimately, this might result in greater inputs of sediment into stream ecosystems. High year to year variability in precipitation can affect wildfire events because years of drought will result in dry conditions conducive to wildfire. If fire is beneficial to *L. dalmatica*, wildfire events in the future might enable the spread and proliferation of *L. dalmatica.* In addition, wildfire events are likely to increase in frequency and severity (FAO 2007). High severity fires has greater potential to kill native plants and destroy seed banks (Zouhar et al. 2008), which can create opportunities for invasive plants. Both *C. stoebe* and *L. dalmatica* establish on recently disturbed sites (Watson and Renney 1974, Vujnovic and Wein 1997), so large wildfire events might present opportunities for invasion.

Lastly, the amount of snow has decreased in the southern interior by 7% per decade and snow depth is declining 11% per decade (B.C. Ministry of Environment 2016). Snow stores large amounts of water, and deeper snow melts at a slower rate because of the insulating properties of snow (B.C. Ministry of Environment 2016). A decline in the amount of snow can increase melting rates, which can result in issues such as flooding and erosion. Since 1970, the extent of snow cover has declined by 10% in the early spring (B.C. Ministry of Environment 2016). Uncovered, bare ground absorbs heat more efficiently than snow covered ground (B.C. Ministry of Environment 2016). The amount of localized warming will likely be greater in areas dominated by *C. stoebe* and *L. dalmatica* because bare ground is more prevalent under these monocultures compared to areas dominated by native bunchgrasses (Lacey et al. 1989). Localized warming can increase germination rates for *L. dalmatica* earlier in the growing season because germination occurs when soil temperatures reach 10°C (Robocker 1970).

Wildfire incidence and severity are predicted to continue to increase in North America from both human caused and natural fire starts (FAO 2007). The summer of 2017 was the largest wildfire season on record in B.C. The province experienced the longest period in a Provincial State of Emergency (70 days) and broke records for the number of hectares burned (1.2 million ha), the number of people displaced (65 000), and the cost of fire suppression ($548 million) (The Province of British Columbia 2018). Seven of the most notable fires (n=18) occurred in the Kamloops Fire Centre (The Province of British Columbia 2018). Despite 2017 being a record breaking year, large wildfire seasons will become more common (FAO 2007). In the face of climate change, increased year to year variation in climatic conditions will result in a greater frequency of large wildfire seasons (FAO 2007). Prescribed burning is likely to continue to be an important method for multiple objectives including minimizing the risk of wildfire and reducing carbon dioxide output from wildfire (Wiedinmyer and Hurteau 2010); however, more information is required to understand how fire events are altering plant communities especially where high infestations of invasive plants occur. Invasive plant management should be considered in combination with prescribed burn activities.

4.4. Management Considerations

Multiple management objectives and treatments are implemented in nature parks, such as Kenna Cartwright Park. For example, to reduce the predominance of invasive plants, mainly *C. stoebe* and *L. dalmatica*, multiple treatments have been implemented within the park such as goat grazing, prescribed burning, release of biological control agents, herbicide application, and mechanical treatments. Despite implementing multiple treatments, land managers often have limited resources for adequate monitoring; however, the use of multiple treatments creates the ideal setting for active adaptive management (Walters and Holling 1990).

Active adaptive management is based on the principle that no one model can correctly predict the response of an ecosystem to treatments, so management decisions should explore alternative models to gain reliable knowledge about the short-term and long-term response of a system to different treatments (Walters and Holling 1990). Treatments within nature parks can be viewed as experiments, and, despite our best predictions, the outcomes of such treatments are largely uncertain (Walters and Holling 1990). Involving researchers in the design and implementation of treatments can improve the experimental design of treatments in nature parks to gain reliable knowledge from monitoring programs (Walters and Holling 1990). It is difficult to obtain long-term data on the response of ecosystems to treatments because monitoring programs are often planned for short time periods (Walters and Holling 1990). Maintaining a long-term partnership between the City of Kamloops nature parks and the SFU/BCIT Masters of Science in Ecological Restoration program can be mutually beneficial to managers and researchers. Researchers (i.e., Masters students) can be involved in the planning and design of experiments for invasive plant management, while managers are familiar with the social and economic risks and benefits of different treatment options (Walters and Holling 1990). Fostering these long-term relationships can advance our understanding of the long-term effects of treatments (Walters and Holling 1990).

My experimental design was conducted opportunistically after the prescribed burns in March 2015 and 2016. The lack of before data limits the certainty in the results because it is possible that the sites treated with prescribed burning differed in the stem density and growth of *C. stoebe* and *L. dalmatica* prior to prescribed burning. However, prescribed burning within Kenna Cartwright Park will continue in future years and other areas of the park. I began monitoring of future prescribed burn treatment areas during the summer of 2017 to collect data on invasive plants prior to prescribed burning. This data can provide a better understanding of the spatial and temporal variation in populations of *C. stoebe* and *L. dalmatica,* and can enable the implementation of a before-after-control-intervention (BACI) experimental design for future prescribed burns. BACI experimental designs can provide reliable data on the effects of prescribed burning on the growth rate and density of *C. stoebe* and *L. dalmatica* in comparison to site and yearly variation. BACI designs enable direct comparison of the site specific growth and density of *C. stoebe* and *L. dalmatica* before burning to the growth and density after burning in the same site. Long-term data on the variation in the density and growth rates of invasive plants is necessary to determine how effective treatments are relative to natural variation resulting from variable weather conditions, landscape variability, and change through time. Monitoring is also required post-treatment to determine the long-term effectiveness of a treatment at altering invasive plant populations. Based on my discussions with the City of Kamloops, the intent remains to conduct prescribed burning in Site 3 in early spring 2018. To date, I have established plots throughout Site 3 that were surveyed in May, June, and July of 2017 and can be used to conduct a BACI experiment to improve the current understanding of the effects of prescribed burning on *C. stoebe* and *L. dalmatica*.

Replications of experimental treatments is necessary to make conclusions about the differences between treatments and control, and for the use of inferential statistics (Hurlbert 1984). The lack of replication or statistical independence, commonly termed pseudoreplication, renders results inconclusive regarding treatment effects (Hurlbert 1984). Within my experiment, prescribed burns were applied in two different areas at two different time points, March 2015 in Site 2 and March 2016 in Site 1. The sampling plots within each burn Site are not statistically independent since they were all burned during the same prescribed burn event, while the burn Sites are independent of one another. The differences between the location of burn treatments and time-since-burning cannot be differentiated but comparative analysis of these two burns in time can still provide valuable information despite errors in design (Hurlbert 1984). For example, *L. dalmatica* are taller in the older burn site (2015) compared to the newer burn site (2016). Based on other research, *L. dalmatica* population growth peaks two years following fire activity (Dodge et al. 2008). Continued monitoring of invasive plants in the prescribed burn treatments in Kenna Cartwright Park and in future burn sites is necessary to determine if a similar trend is observed throughout my study area. Monitoring in subsequent years will also increase the number of replicates (e.g., if the 2016 burn site is monitored in 2018 it will represent a second 2-year post-burn site).

Lastly, it is important to use knowledge of the effects of prescribed burning on invasive plant density and growth, and apply learnings to wildfire rehabilitation programs. Wildfire rehabilitation programs largely focus on stabilizing slopes for human safety, but programs such was the Early Detection Rapid Response would be useful to integrate into rehabilitation programs following wildfire (Zouhar et al. 2008). In fire suppression, direct action on large wildfire incidents begins with building guard around wildfires, which often requires the use of heavy equipment to create a fuel break (i.e., removing all vegetation and soil down to the mineral soil). The guard is then used to move machinery, vehicles, and people. Disturbance from building guard can create openings for invasive plants to establish from the surrounding area. In addition, human and vehicle traffic along the guard can facilitate the introduction and spread of invasive plant propagules into areas recently disturbed by both machinery and wildfire. Occurrences of invasive plants should be documented during wildfire rehabilitation to assess new infestations and plan eradication.

4.4.1. Alternative Treatments for Invasive Plants

To successfully restore native plant communities, goals must be set on the desired native plant community to restore (Sheley et al. 1998). Once goals are established, an integrated invasive plant management plan that uses multiple treatments is the best strategy for reducing invasive plant dominance (Sheley et al. 1998). Treatments should be coordinated and timed with the phenology of the invasive plants and be in line with management objectives (Sheley et al. 1998). Prior to implementing treatments, a thorough inventory and mapping of the area is required to quantify the density of invasive plants and prioritize areas for treatment (Sheley et al. 1998).

I explored two methods for managing invasive plants. Hand-pulling is a mechanical treatment that is labour intensive but highly specific to target plants, while prescribed burning operates on the assumption that restoring natural processes to the ecosystem will enable native species to recuperate. However, a number of alternative treatments exist and have been implemented in Kenna Cartwright Park. Below, I explore alternative treatment methods for *C. stoebe* and *L. dalmatica*.

Chemical Treatments

Herbicide use is a common tool for the eradication of invasive plants, including *C. stoebe* and *L. dalmatica*. In B.C., the use of herbicides is regulated under the *Integrated Pest Management Act* (Province of British Columbia 2003). A summary of herbicides that have been tested on *C. stoebe* and *L. dalmatica* and the effect of herbicide application are included in Table 8 and Table 9, respectively.

Herbicide application has shown success at reducing populations of both *C. stoebe* and *L. dalmatica,* but repeated treatment of herbicides is required due to the residual seed bank (Jacobs and Sing 2006). In addition, herbicide application can affect native plants that are not the target for eradication. For example, treatment with 280g active ingredient / ha of aminocyclopyrachlor reduces the abundance of native Asteraceae forbs and increases the abundance of native *Eriogonum* species (Kyser and DiTomaso 2013).

Over multiple herbicide treatments, changes to the abundance of native plants can alter species interactions and change community composition.

Furthermore, herbicide application can indirectly alter arthropod communities by altering the species composition and abundance of plant species (Taylor et al. 2006). Herbicide spraying reduces the number and biomass of arthropods that are required by birds during the nesting season (Taylor et al. 2006). The amount of food availability during the breeding season can affect chick survival and growth, and herbicide application can reduce the abundance of arthropods and plant seeds used by nesting birds (Boatman et al. 2004). The negative relationship between herbicides and bird food availability can negatively affect bird survival (Boatman et al. 2004). Application of herbicides should consider timing to avoid indirect effects on breeding birds.

Table 8. Summary of the effect of herbicide application on *Centaurea stoebe* from the literature.

Herbicide	Application	Effect on *Centaurea stoebe*	Reference
Glyphosate	Single application in May at recommended rate	Initially reduce biomass, density, and dominance of knapweed, but no residual effects Increase warm-season grass biomass and dominance	(MacDonald et al. 2007)
2-4-Dichlorophenoxy	Single application in May at recommended rate	Initially reduce biomass, density, and dominance of knapweed, but no residual effects Increase warm-season grass biomass and dominance	(MacDonald et al. 2007)
Picloram	0.14 to 0.28 kg active ingredient / ha	Complete control 36 months after treatment	(Davis 1990)
	0.11 to 0.28 kg active ingredient / ha	100% control 24 months after treatment	
		Six years after treatment density of treated plots did not differ significantly from untreated plots at one site, but seven years after treatment at the second site treated plots had significantly fewer *C. stoebe* than untreated plots	

Table 9. Summary of the effect of herbicide application on *Linaria dalmatica* from the literature.

Herbicide	Application	Effect on *Linaria dalmatica*	Reference
Picloram	Granular 0.5 lb / acre 1.0 lb / acre 1.5 lb / acre In water 0.5 lb / acre 1.0 lb / acre 1.5 lb / acre	Granular application was more effective at treating *L. dalmatica* then foliar applications. Granular application of 1.0 and 1.5 lb / acre reduced the number of crowns of *L. dalmatica* to zero in the year after application.	(Robocker 1968)
Silvex	4 lb / acre	Reduced the number of *L.dalmatica* crowns after one year but no difference from control after 3 years	(Robocker 1968)
	2 lb / acre	Reduced the number of *L. dalmatica* crowns after one year but no difference from control after 3 years	
Aminocyclopyrachlor	280 g active ingredient /ha	Reduced *L. dalmatica* cover by 89% two years after treatment Increased native *Eriogonum* spp. and decreased native Asteraceae species	(Kyser and DiTomaso 2013)
	140 g active ingredient /ha	Reduced *L. dalmatica* cover by 63% two years after treatment	
Chlorosulfuron	158 g active ingredient /ha	Reduced *L. dalmatica* cover by 74% two years after treatment	(Kyser and DiTomaso 2013)
	105 g active ingredient /ha	Reduced *L. dalmatica* cover by 75% two years after treatment	
Aminopyralid	245 g active ingredient /ha	Reduced *L. dalmatica* cover by 46% two years after treatment	(Kyser and DiTomaso 2013)

L. dalmatica has a thick waxy cuticle that can reduce the effectiveness of herbicide application (Jacobs and Sing 2006). Factors such as environmental conditions and time of year of application can affect the success of herbicide application on *L. dalmatica* (Jacobs and Sing 2006, Kyser and DiTomaso 2013). Treatment using picloram on *L. dalmatica* has been used in Kenna Cartwright Park and the effectiveness of treatment on invasive plants is an active area of research. Preliminary results indicate broadcast spraying is most effective in the short-term reduction of *L. dalmatica* density (Bradshaw 2017).

Biological Control Agents

The use of biological control agents (biocontrol agents) for invasive plant management is based on the hypothesis that stress inflicted on an invasive plant by a natural enemy will be sufficient to reduce competitive ability of invasive plants, ultimately reducing the population size (Hezewijk et al. 2010). The use of biocontrol agents is controversial and involves releasing non-native predatory species to target other non-native species (Simberloff and Stiling 1996). Biocontrol agents are selected based on host-specificity, but often the effects of biocontrol agents on non-target native species are poorly studied prior to release (Simberloff and Stiling 1996). In addition, the lack of baseline data of native species prior to release often inhibits an assessment of the effects of biocontrol agents on non-target species (Simberloff and Stiling 1996). In B.C., 13 biocontrol agents have been identified for *C. stoebe,* and seven biocontrol agents have been identified for *L. dalmatica* (Ministry of Forests, Lands and Natural Resource Operations 2018).

Biocontrol agents have been released in Kenna Cartwright Park as early as 1988 for *C. stoebe* and 1994 for *L. dalmatica* (Tarasoff 2002). A summary of biocontrol agents released in the Park and their target species are summarized in Table 10. Surveys conducted in 2001 compare the change in the population of invasive plants prior to biocontrol release to post-treatment levels to determine the population size of biocontrol species in 2001 (Tarasoff 2002). Mean density of *C. stoebe* in release sites prior to release ranged from 1.72 to 76.9 individuals per m^2, while mean density of *L. dalmatica* ranged from 1.42 to 4.42 individuals per m^2 (Tarasoff 2002).

One biocontrol release site occurs in my Study Area on the border of Site 1 and 3 where *Mecinus janthinus* was released in 2000 (Tarasoff 2002). Limited monitoring data of biocontrol agents was available after 2002; however, regional-scale monitoring of *M. janthinus* in southeastern B.C. indicate that *M. janthinus* is capable of dispersing and colonizing areas up to 25 km from release locations within 13 years of release (Hezewijk et al. 2010). Peak populations of *M. janthinus* are observed 8 years after release with declining populations afterwards (Hezewijk et al. 2010). Considering the close proximity of all my sites to the *M. janthinus* release locations (Figure 3) it is expected that similar infestation rates of *M. janthinus* occur in all my sites. In addition, all release sites for *M.*

janthinus in Kenna Cartwright Park occurred greater than eight years ago, so the population of *M. janthinus* in my study area is likely on the decline. However, I did not assess the level of *M. janthinus* infestation in sampling plots.

Table 10. Biocontrol agents and target plant species released in Kenna Cartwright Park since 1988. Data adapted from Tarasoff 2002. Release location names correspond to the Invasive Plant Inventory Mapping from 2001 (Figure 3)

Biocontrol Agent	Target Species	Number of Releases	Release Years	Release Location Name
Agapeta zoegana	*Centaurea stoebe*	3	1992, 1995	Aga 95
Cyphocleonus achates	*Centaurea stoebe*	2	1991, 1994	Cyp 91, Cyp 94(95)
Larinus minutus	*Centaurea stoebe*	7	1992, 1993, 1995, 1996, 2000	Lar 91, Lar 92(95), Lar 93, Lar 95, Lar 96, Lar 00-2,
Mecinus janthinus	*Linaria dalmatica*	7	1994, 1995, 1996, 2000	Mec 94(95), Mec 95, Mec 96, Mec 00-1, Mec 00-2, Mec 00-2
Metzneria paucipunctella	*Centaurea stoebe*	1	1995	Met 95
Sphenoptera jugoslavica	*Centaurea stoebe*	8	1988, 1989, 1994, 1995	Sph 89(95), Sph 94(95), Sph 95

Based on density data of *L. dalmatica* prior to release of *M. janthinus* and surveys conducted in 2001, two general trends are observed. First, at five of the seven sites, a decline in the density per m^2 of *L. dalmatica* at release sites was observed (years-since-biocontrol-release ranged from one to seven) (Tarasoff 2002). This is consistent with other studies that suggest a peak in *M. janthinus* populations eight years after release followed by a decline in the population (Hezewijk et al. 2010). However, the total area infested by *L. dalmatica* increased at five of the seven sites, indicating a continuous spread of *L. dalmatica* throughout the park. Based on this data, and my observations that *L. dalmatica* is still an invasive plant in the park, the biocontrol release program to date has not been successful at minimizing the population of *L. dalmatica* to levels that enable co-existence with native plants. Mean density of *L. dalmatica* across my study area in 2017 ranged from 0.40 ± 0.61 to 8.4 ± 5.63 stems per m^2 which is a greater range of density than measured in 2001 that is provided in Table 11 (Tarasoff 2002).

Table 11. Biological control release sites for *Mecinus janthinus* in Kenna Cartwright Park. The site name corrosponds to the Invasive Plant Inventory Mapping from 2001 (Figure 3). A comparison of the density of *Linaria dalmatica* prior to release and in 2001 and the size of area affected by the infestation is provided. Information is adapted from (Tarasoff 2002).

Release Site and Year (Mec Yr)	Attack Rate (% plants attacked)	Density of *Linaria dalmatica* in release year per m^2	Density of *Linaria dalmatica* in 2001 per m^2	Number of insects per plant in 2001 surveys	Size of area infested by *L. dalmatica* at release (ha)	Size of area infested by *L. dalmatica* in 2001 (ha)
Mec 94/95	67 %	2 - 5	2.4	4.19	0.5 - 1.0	0.8
Mec 95	75 %	2 - 5	4.3	3.38	0.01 - 0.04	1.3
Mec 96S	65 %	6 - 10	1.61	3.09	0.01 - 0.04	0.9
Mec 97N	65 %	6 - 10	1.98	2.78	0.01 - 0.04	0.2
Mec 00-1	24 %	6 - 10	1.55	0.70	0.04 - 0.25	12
Mec 00-2	35 %	>10	1.45	0.90	0.25 - 0.5	0.7
Mec 00-3	84 %	>10	4.42	4.20	>1.0	unknown

Grazing

Target grazing for invasive plants is an increasing area of ecological restoration. Target grazing for forbs mainly use goats and sheep that preferentially target flowers and seed heads of forbs instead of vegetative parts of the plant, thereby reducing the reproductive output of forbs. Goats can be trained to target specific plants (Murphy 2017). By targeting flower heads prior to seed set, grazing can minimize contributions of seeds to the seed bank. Consecutive years of grazing can deplete the residual seed bank.

In the case of *C. stoebe,* sheep grazing has reduced density over a three-year grazing period compared to ungrazed areas (Olson et al. 1997). The number of viable *C. stoebe* seeds was significantly lower in grazed compared to ungrazed areas in Montana (Olson et al. 1997). Over the same period, the density of native Idaho fescue (*Festuca idahoensis*) increased (Olson et al. 1997). Grazing by sheep alters the age distribution of *C. stoebe* because sheep target the more palatable young *C. stoebe,* and grazing reduces the number of viable seeds in the seed bank (Olson et al. 1997). However, *C. stoebe* exhibits a compensatory response once grazing is discontinued and grazed areas have twice the number of flowering stems than ungrazed areas the first year after grazing is discontinued (Olson et al. 1997). Reducing the population of *C. stoebe* using sheep grazing would require multiple years of repeated treatment, but has the potential to reduce the dominance of this invasive plant in native systems (Olson et al. 1997).

Grazing using goats trained to target *C. stoebe* reduced the number of seed heads when measured after grazing (Murphy 2017). Goats can be managed by herders to avoid grazing native plants and goats cause minimal compaction in comparison to machines and humans (Murphy 2017). Goat grazing reduces seed production, but the long-term effects of goat grazing at reducing populations are under-studied.

Grazing increases the amount of exposed bare ground (Olson et al. 1997) and grazing animals avoid *L. dalmatica* (Vujnovic and Wein 1997, USDA 2014). Grazing might benefit *L. dalmatica* by creating opportunities for seed germination, and, because *L. dalmatica* is avoided by livestock, increased grazing pressure on native plants might reduce competitive effects on *L. dalmatica*. Grazing with sheep might temporarily reduce seed production, but follow-up treatment using herbicides is required (USDA 2014).

Goat grazing has been used as part of the invasive plant management program in Kenna Cartwright Park. Goats were used over a four-year period starting in 2012 and ending in 2015. Considering the longevity of seeds of both *C. stoebe* and *L. dalmatica*, four years would be insufficient at depleting the seed bank. Goat grazing was used in Sites 1 and 2 in the summers of 2013, 2014, and 2015.

4.5. Research Implications

Based on my study, I was able to produce linear mixed-effects models. While this is useful to determine statistical differences between my fixed effects, detailed spatial data from within the study area would enable extrapolation of models to other areas of the park beyond my sampling plots. Models can be generated to predict the occurrence and expected densities of *C. stoebe* and *L. dalmatica* throughout the study area. This type of modelling is useful in invasive plant management because it can help managers identify priority areas for treatment based on predicted areas of infestation and edges between low infestation areas and heavily infested areas.

Models are only as reliable as the data used to generate them, so acquiring more data on the spatial occurrence of *C. stoebe* and *L. dalmatica* and the effect of treatments on the occurrence of both species in the park to improve the model are necessary. In addition, the models I developed have a high amount of residual variance. Collecting data at sampling plots on other factors such as soil moisture, the distance from trails, and other factors that influence the growth and density of *C. stoebe* and *L. dalmatica* are important to test as random effects to improve models. By explaining more of the residual variance the models can perform better at predicting the occurrence of species given site conditions in the Park.

In addition to research to improve models, we also require more data on the effects of prescribed burning on invasive plants. Prescribed burning provides a natural treatment option for species such as *C. stoebe* as an alternative to herbicide application, but the effectiveness of prescribed burning depends on the timing of prescribed burning and the life history of plants (MacDonald et al. 2007). Future research in the use of prescribed burning in ecological restoration should examine the effect of timing of prescribed burning on the density and growth of invasive plants to determine the time of year to optimize negative effects on invasive plants. However, this is a contentious issue because prescribed burning during July and August is most consistent with the natural wildfire occurrence (Klenner et al. 2008), but has the highest risk of becoming out-of-control. Charcoal is suggested to inhibit the competitive ability of *C. stoebe* (Weidenhamer and Callaway 2010) . Examining the effects of charcoal additions on the growth and stem density of *C. stoebe* would be beneficial to determine whether charcoal addition alone can be used in invasive plant management.

4.6. Application to Ecological Restoration

Ecological restoration aims to re-establish natural functions and processes to degraded ecosystems. Many ecosystem functions are the result of the structure and composition of vegetation communities, and the interactions between species in plant communities (Chapin III et al. 2000). For example, loss of soil moisture due to exotic deep rooted plants in the *Centaurea* family creates more arid soil conditions in grasslands (Chapin III et al. 2000). Invasive plants alter the composition and structure of vegetation communities that often results in a change in ecosystem functions (Vitousek et al. 1997); therefore, conducting ecological restoration requires removing and managing invasive plants to a level where invasive plants are not disrupting ecological function.

Multiple invasive plant management techniques exist; however, due to the overwhelming and increasing number of invasive plants in B.C., there often lacks sufficient quantitative data on the effectiveness of techniques for specific invasive plants. Moreover, many studies are conducted within a narrow geographic range in comparison to the geographic range these techniques are being applied. For example, most studies in the management of *C. stoebe* and *L. dalmatica* using prescribed burning have occurred in the U.S. (mainly Montana and Arizona) (Jacobs and Sheley 2003, Dodge et al. 2008, Pokorny et al. 2010, Pearson et al. 2012). Testing whether similar results occur in more northern climates and vegetation communities are important if prescribed burning is being used in these areas.

C. stoebe and *L. dalmatica* are prevalent invasive plants in the southern interior of British Columbia, where the dominant BEC zones include the Ponderosa Pine, Bunchgrass, and Interior Douglas-fir. These BEC zones are characterized by dry, hot summers that result in a moisture deficit, creating conditions that are conducive to wildfire (Hope et al. 1991). Wildfire has functioned as a natural disturbance in many of these landscapes enabling a mosaic of plant communities at the landscape scale depending on the site-specific wildfire return interval (Whisenant 1990, MacDonald et al. 2013). Prescribed burning can be used as a tool in ecological restoration to return fire disturbance to the landscape, and enable the use of more natural methods for restoration compared to other management techniques like herbicide use. For example, three years of consecutive prescribed burning reduced the residual seed bank of *C. stoebe* to similar levels as seven years of herbicide application, without the adverse effects of herbicide use on native plants (Weidenhamer and Callaway 2010). Herbicide application can negatively affect native plants by reducing soil microbial diversity and altering soil chemistry (Weidenhamer and Callaway 2010). Furthermore, the herbicide picloram used to treat *C. stoebe* infestation inhibits flowering and seed set in the native forb arrowleaf balsamroot (*Balsamorhiza sagittata*) for up to four years after application and can result in population decline in the species (Crone et al. 2009). However, more consideration of the specific effects of fire on target invasive plants is necessary to determine whether burning is effective at reducing populations of invasive plants to enable co-existence with native plants.

My study demonstrates the importance of considering the life history characteristics of multiple invasive plants present in plant communities because management practices that are beneficial in the removal of one species might be beneficial to the spread and proliferation of another. This is important in the field of ecological restoration since treatments are often applied to benefit native species that we value and remove invasive species that are detrimental to ecosystem services and functions. For example, my results suggest both prescribed burning and hand-pulling might be effective techniques in the removal of *C. stoebe,* but hand-pulling treatments had little effect on *L. dalmatica* and increased stem density by promoting seed germination*.* Early spring burning also appears ineffective at reducing the stem density of *L. dalmatica,* and trends suggest prescribed burning might actually contribute to population growth of *L. dalmatica*. Early spring prescribed burning might increase the competitive ability of *L. dalmatica* by providing opportunities for *L. dalmatica* to invade in recently disturbed areas or to provide opportunities for seeds in the seed bank to germinate because prescribed burning removes litter layers and increases sunlight reaching the soil (Lesica and Martin 2003). Because *L. dalmatica* emerges earlier in the spring than the dominant native bunchgrasses (Robocker 1970, Jacobs and Sheley 2003), early spring burning might provide a competitive advantage to *L. dalmatica.* The community and the individual species in the community must be considered when we plan treatment programs for invasive plant removal. If prescribed burning is continued in Kenna Cartwright Park, alternative treatments such as herbicide application and goat grazing should be used to prevent the growth of *L. dalmatica* populations.

5.0 Conclusion

Managing for invasive plants is a growing challenge in the field of ecological restoration. Understanding the specific life history and characteristics of invasive plants is necessary to select treatments that will negatively affect population growth. Treatment methods are likely to have different effects on different invasive plants.

The effect of prescribed burning differed between the two invasive plants. *C. stoebe* were not detected in the burn treatments. Prescribed burning might reduce the population of *C. stoebe* or site conditions might not be conducive to *C. stoebe* growth prior to burning. Conversely, *L. dalmatica* was abundant and larger in the burn treatments than hand-pull and control treatments. However, *L. dalmatica* might have occurred at a higher density prior to burning because Sites 1 and 2 were assessed as high infestation in 2001 (Tarasoff 2002). Despite pre-burn conditions, the areas treated with prescribed burning remained highly infested by *L. dalmatica* post-burn, and prescribed burning in the early spring is likely beneficial. Future prescribed burns should incorporate alternative treatments for *L. dalmatica* such as herbicide application.

The effect of hand-pulling was similar between species, but, because of the life history of *C. stoebe,* hand-pulling is likely more effective at removing *C. stoebe.* A greater density of *C. stoebe* occurred in sampling plots treated with hand-pulling; however, 60% of *C. stoebe* in the hand-pull treatments remained as basal rosettes. Therefore, hand-pulling over consecutive years can deplete the seed bank of *C. stoebe*. *L. dalmatica* also occurred at a higher stem density in hand-pull treatments and were smaller compared to control, but *L. dalmatica* that germinated after hand-pulling were still capable of producing seeds. Hand-pulling is only recommended for small infestations or in combination with other treatments because of the labour cost.

Partnerships between institutions and industry can improve long-term monitoring. Establishing a BACI experiment design in future prescribed burn sites in Kenna Cartwright Park is essential to further investigate the effects of prescribed burning on the growth and density of *C. stoebe* and *L. dalmatica.* Continued monitoring of previous burn sites will inform the long-term effects of prescribed burning and hand-pulling on populations of *C. stoebe* and *L. dalmatica* in the park.

References

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

Bates, D., M. Maechler, B. Bolker, and S. Walker. 2015. Fitting linear mixed-effects models using lme4. Journal of Statistical Software 67:1–48.

B.C. Ministry of Environment. 2016. Indicators of Climate Change for British Columbia 2016 Update. Page 61. Government of B.C.

Beckage, B., and I. J. Scott. 2000. Effects of repeated burning on species richness in a Florida Pine Savanna: A test of the intermediate disturbance hypothesis. Journal of Vegetation Science 11:113–122.

Boatman, N. D., N. W. Brickle, J. D. Hart, T. P. Milsom, A. J. Morris, A. W. A. Murray, K. A. Murray, and P. A. Robertson. 2004. Evidence for the indirect effects of pesticides on farmland birds. Ibis 146:131–143.

Bradshaw, J. L. 2017. Non-conventional control of Linaria genistifolia ssp. dalmatica in a semi-arid grassland of British Columbia’s southern interior. Thompson River University, Kamloops, B.C.

Callaway, R. M., and E. T. Aschehoug. 2000. Invasive plants versus their new and old neighbours: A mechanism for exotic invasion. Science 290:521-523.

Chapin III, F. S., E. S. Zavaleta, V. T. Eviner, R. L. Naylor, P. M. Vitousek, H. L. Reynolds, D. U. Hooper, S. Lavorel, D. E. Sala, S. E. Hobbie, M. C. Mack, and S. Diaz. 2000. Consequences of changing biodiversity. Nature 405:234–242.

Coupland, R. T., S. Zilke, and G. W. Selleck. 1963. Spread of toadflax in Saskatchewan. Canadian Journal of Plant Science 43:214–221.

Crone, E. E., M. Marler, and D. E. Pearson. 2009. Non-target effects of broadleaf herbicide on a native perennial forb: a demographic framework for assessing and minimizing impacts. Journal of Applied Ecology 46:673–682.

Data BC, Province of British Columbia. 2013. iMapBC. https://maps.gov.bc.ca/ess/hm/imap4m/.

Davis, E. S. 1990. Spotted knapweed (Centaurea maculosa L.) seed longevity, chemical control and seed morphology. Master of Science, Montana State University, Bozeman, Montana.

Davis, E. S., P. K. Fay, T. K. Chicoine, and C. A. Lacey. 1993. Persistence of spotted knapweed (Centaurea maculosa) seed in soil. Weed Science 41:57–61.

Dodge, R. S., P. Z. Fule, and C. Hull Sieg. 2008. Dalmatian toadflax (Linaria dalmatica) response to wildfire in a southwestern USA forest. Ecoscience 15:213-222.

Duncan, C. A., J. J. Jachetta, M. L. Brown, V. F. Carrithers, J. K. Clark, J. M. DiTomaso, R. G. Lym, K. C. McDaniel, M. J. Renz, and P. M. Rice. 2004. Assessing the economic, environmental, and societal losses from invasive plants on rangeland and wildlands. Weed Technology 18:1411–1416.

Eddleman, L. E., and J. T. Romo. 1988. Spotted knapweed germination response to stratification, temperature, and water stress. Canadian Journal of Botany 66:653–657.

Emery, S. M., and K. L. Gross. 2005. Effects of timing of prescribed fire on the demography of an invasive plant, spotted knapweed Centaurea maculosa. Journal of Applied Ecology 42:60–69.

FAO. 2007. Fire management - global assessment 2006. Food and Agriculture Organization of the United Nations, Rome.

Fernandes, P. M., and H. S. Botelho. 2003. A review of prescribed burning effectiveness in fire hazard reduction. International Journal of Wildland Fire 12:117–128.

Goodwin, K., and D. Burch. 2007. Watch out for knapweed. Montana State University.

Gurka, M. J. 2006. Selecting the best linear mixed model under REML. The American Statistician 60:19–26.

Hezewijk, B. H. V., R. S. Bourchier, and R. A. D. Clerck-Floate. 2010. Regional-scale impact of the weed biocontrol agent Mecinus janthinus on Dalmatian toadflax (Linaria dalmatica). Biological Control 55:197–202.

Hofmann, M., and J. Isselstein. 2004. Effects of drought and competition by a ryegrass sward on the seedling growth of a range of grassland species. Journal of Agronomy and Crop Science 190:277–286.

Hope, G. D., D. A. Lloyd, W. R. Mitchell, W. R. Erikson, W. I. Harper, and B. M. Wikeem. 1991. Chapter 9: Ponderosa Pine zone. Pages 139–151 *in* D. V. Meidinger and J. Pojar, editors. Ecosystems of B.C. Ecosystems of B.C. Research Branch, Goverment of B.C., Victoria, B.C.

Hulbert, L. C. 1969. Fire and litter effects in undisturbed bluestem prairie in Kansas. Ecology 50:874–877.

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

Jacobs, J. S., and R. L. Sheley. 1998. Observation: Life history of spotted knapweed. Journal of Range Management 51:665–673.

Jacobs, J. S., and R. L. Sheley. 2003. Prescribed fire effects on Dalmatian toadflax. Journal of Range Management 56:193–197.

Jacobs, J., and S. Sing. 2006. Ecology and management of Dalmatian toadflax (Linaria dalmatica (L.) Mill.). Natural Resources Conservation Service, United States Department of Agriculture, Montana.

Keeley, J. E. 2006. Fire management impacts on invasive plants in the western United States. Conservation Biology 20:375–384.

Klenner, W., R. Walton, A. Arsenault, and L. Kremsater. 2008. Dry forests in the southern interior of British Columbia: Historic disturbance and implications for restoration and management. Forest Ecology and Management 256:1711–1722.

Kyser, G. B., and J. M. DiTomaso. 2013. Effects of timing on chemical control of Dalmatian toadflax (Linaria dalmatica) in California. Invasive Plant Science and Management 6:362–370.

Lacey, J. R., C. B. Marlow, and J. R. Lane. 1989. Influence of spotted knapweed (Centaurea maculosa) on surface runoff and sediment yield. Weed Technology 3:627–631.

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

MacDonald, N. W., L. M. Martin, C. K. Kapolka, T. F. Botting, and T. E. Brown. 2013. Hand pulling following mowing and herbicide treatments increases control of spotted knapweed (Centaurea stoebe). Invasive Plant Science and Management 6:470–479.

MacDonald, N. W., B. T. Scull, and S. R. Abella. 2007. Mid-spring burning reduces spotted knapweed and increases native grasses during a Michigan experimental grassland establishment. Restoration Ecology 15:118–128.

MacDougall, A. S., K. S. McCann, G. Gellner, and R. Turkington. 2013. Diversity loss with persistent human disturbance increases vulnerability to ecosystem collapse. Nature 494:86–89.

Marrow, B. 2016, April 1. Kenna Cartwright Park 2016 prescribed fire activities.

Marrow, B. 2017. Prescribed Fire Burn Plan. Page 12 pp. City of Kamloops, Kamloops, B.C.

Ministry of Forests, Lands and Natural Resource Operations. 2018. Biocontrol agent on invasive plant matrix. https://www.for.gov.bc.ca/hra/plants/biocontrol/Agent-plant_matrix.htm.

Mohamadi, M. A., and A. Kavian. 2015. Effects of rainfall patterns on runoff and soil erosion in field plots. International Soil and Water Conservation Research 3:273–281.

Murphy, N. K. E. 2017. The efficacy of goat browsing on invasive plant species in British Columbia. Simon Fraser University, Burnaby, B.C.

Nadeau, L. B., and J. R. King. 1991. Seed dispersal and seedling establishment of Linaria vulgaris. Canadian Journal of Plant Science 71:771–782.

Oasis of activity: City of Kamloops parks master plan. 2013. City of Kamloops.

Olson, B. E., R. T. Wallander, and J. R. Lacey. 1997. Effects of sheep grazing on a spotted knapweed infested Idaho fescue community. Journal of Range Management 50:386–390.

Pearson, D. E., Y. K. Ortega, and S. J. Sears. 2012. Darwin's naturalization hypothesis up-close: Intermountain grassland invaders differ morphologically and phenologically from native community dominants. Biological Invasions 14:901–913.

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

Pokorny, M. L., J. M. Mangold, J. Hafer, and M. K. Denny. 2010. Managing spotted knapweed (Centaurea stoebe)-infested rangeland after wildfire. Invasive Plant Science and Management 3:182–189.

Province of British Columbia. 1995. Biodiversity Guidebook. Forest Practices Code of British Columbia Act, Victoria, B.C.

Province of British Columbia. 1996. Weed Control Act and Weed Control Regulation.

Province of British Columbia. 2003. Integrated Pest Management Act.

R Core Team. 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Robocker, W. C. 1968. Control of Dalmatian toadflax. Journal of Range Management 21:94–98.

Robocker, W. C. 1970. Seed characteristics and seedling emergence of Dalmatian toadflax. Weed Science 18:720–725.

Sheley, R. L., J. S. Jacobs, and M. F. Carpinelli. 1998. Distribution, biology, and management of diffuse knapweed (Centaurea diffusa) and spotted knapweed (Centaurea maculosa). Weed Technology 12:353–362.

Simberloff, D., and P. Stiling. 1996. How risky is biological control? Ecology 77:1965–1974.

Sing, S. E., and R. K. D. Peterson. 2011. Assessing environmental risks for established invasive weeds: Dalmatian (Linaria dalmatica) and yellow (L. vulgaris) toadflax in North America. International Journal of Environmental Research and Public Health 8:2828–2853.

Story, J. M., L. Smith, and W. R. Good. 2001. Relationship among growth attributes of spotted knapweed (Centaurea maculosa) in western Montana. Weed Technology 15:750–761.

Tarasoff, C. 2002. Restoration plan for Kenna Cartwright Municipal Park. Terrestrial Ecosystem Restoration Program.

Taylor, R. L., B. D. Maxwell, and R. J. Boik. 2006. Indirect effects of herbicides on bird food resources and beneficial arthropods. Agriculture, Ecosystems and Environment 116:157–164.

The Province of British Columbia. 2018. Wildfire Statistics. https://www2.gov.bc.ca/gov/content/safety/wildfire-status/wildfire-statistics/wildfire-season-summary.

The Weather Network. 2018. The Weather Network. https://www.theweathernetwork.com/ca/api/sitewrapper/index?b=%2Fstatistics%2F&p=%2Fforecasts%2Fstatistics%2Findex&url=%2Fstatistics%2Fcabc0146%2Fkamloops%2F%2F%2F%3F.

Thorpe, A. S., V. Archer, and T. H. DeLuca. 2006. The invasive forb Centaurea maculosa increases phosphorous availability in Montana grasslands. Applied Soil Ecology 32:118–122.

USDA. 2014. Field guide for managing Dalmatian and yellow toadflaxes in the southwest. United States Department of Agriculture Forest Service, Albuquerque, New Mexico.

Vila, M., J. L. Espinar, M. Heida, P. E. Hulme, V. Jarosik, J. L. Maron, J. Pergl, U. Schaffner, Y. Sun, and P. Pysek. 2011. Ecological impacts of invasive alien plants: a meta-analysis of their effects on species, communities, and ecosystems. Ecology Letters 14:702–708.

Vitousek, P. M., C. M. D'Antonio, L. L. Loope, M. Rejmanek, and R. Westbrooks. 1997. Introduced species: a significant component of human-caused global change. New Zealand Journal of Ecology 21:1–16.

Vujnovic, K., and R. W. Wein. 1997. The biology of Canadian weeds. 106. Linaria dalmatica (L.) Mill. Canadian Journal of Plant Science 77:483–491.

Walters, C. J., and C. S. Holling. 1990. Large-scale management experiments and learning by doing. Ecology 71:2060–2068.

Watson, A. K., and A. J. Renney. 1974. The biology of Canadian weeds 6. Centaurea diffusa and C. maculosa. Journal of Plant Science 54:687–701.

Weather Stats. 2018. Kamloops Weather Stats. https://kamloops.weatherstats.ca/charts/precipitation-monthly.html.

Weidenhamer, J. D., and R. M. Callaway. 2010. Direct and indirect effects of invasive plants on soil chemistry and ecosystem function. Journal of Chemical Ecology 36:59–69.

Whisenant, S. G. 1990. Changing fire frequencies on Idaho’s Snake River Plains: Ecological and management implications. Pages 4–10 *in* E. D. McArthur, E. M. Romney, S. D. Smith, and P. T. Tueller, editors. USDA Forest Service, Intermountain Research Station, General Technical Report INT-276, Las Vegas, NV.

Wickham, H. 2009. ggplot2: Elegant graphics for data analysis. Springer-Verlag, New York, NY.

Wickham, H. 2011. The split-apply-combine strategy for data analysis. Journal of Statistical Software 40:1–29.

Wickham, H., J. Hester, and R. Francois. 2017. readr: Reach rectangular text data.

Wiedinmyer, C., and M. D. Hurteau. 2010. Prescribed fire as a means of reducing forest carbon emissions in the western United States. Environmental Science Technology 44:1926–1932.

World Weather Online. 2018. World Weather Online. https://www.worldweatheronline.com/kamloops-weather-averages/british-columbia/ca.aspx.

Zeileis, A., and G. Grothendieck. 2005. zoo: s3 infrastructure for regular and irregular time series. Journal of Statistical Software 14:1–27.

Zouhar, K., J. Kapler Smith, and S. Sutherland. 2008. Chapter 2: Effects of fire on nonnative invasive plants and invasibility of wildland ecosystems. Pages 7–32. Gen. Tech. Rep., USDA Forest Service.

Zuur, A. F., E. N. Ieno, N. J. Walker, A. A. Saveliev, and G. M. Smith. 2009. Mixed effects models and extensions in ecology with R. Springer, New York, NY.