Evaluating the Effects of a Fire Disturbance on Native Alvar Vegetation: Implications for Woody Species Management and Conservation of Open Alvar Habitat

by

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Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

The decline and loss of certain habitat types can be attributed in some part due to a change of disturbance mechanisms or fire exclusion. This is of particular concern for globally-rare open alvar community types, and the restricted, specialist, and rare species found only on alvar habitats. North Bear Alvar, a parcel of alvar in the Carden Alvar located in the Kawartha Lakes region in Ontario, represents a system that, due to anthropogenic influences and habitat degradation, has been encroached by problematic woody species which have changed the structure, composition, and function of the open alvar. This has resulted in an increase in woody percent cover and a decrease in native herbaceous species richness and cover, two characteristics to open alvar habitat community types. It is hypothesized that to some degree, alvars necessitate periodic fire disturbance to maintain open alvar habitat attributes. However, a lack of contemporary disturbance has led to a decrease in open alvar sizes on North Bear Alvar. A lack of fire disturbance research on certain alvars has left conservation managers without specific answers to post-disturbance habitat brought on by fire. Here, I aim to comparatively assess various conservation interventions representing forms of disturbance to determine which yield more desirable open habitat conditions following disturbance. Specifically, I address the questions 1) What are the differences in post-treatment vegetation compositions following treatment by prescribed burning, glyphosate application, and manual lopping? 2) How do each of these treatments contribute to facilitating nutrient availability and soil conditions with respect to NO_3, P, and K? And 3) Can a mesocosm prescribed burn device (the 'burn box') replicate fire effects on a small 4 m^2 scale? To determine this, I collected abundance, cover, and richness data on the plant community, and soil nutrient samples on 32 permanent plots on North Bear to assess change over the three treatment types in comparison to control plots. I analysed these data for significant differences between treatments and control plots using ANOVAs. I mapped fire severity and vegetation structure to determine the efficacy of the burn box device at informing conservation management using fire. Using these results, I found prescribed burning using the burn box resulted in significant increases ($p < 0.05$) in the cover and abundance of native herbaceous species and significant decreases ($p < 0.05$) in the cover of regenerating woody vegetation, though richness remained unchanged. In addition, using the burn box device, I found replicable fire effects and dynamics observed in non-confined prescribed burns, opening the utility of the burn box to testing fire effects on small scales before initiating prescribed burning in novel, exploratory, or uncertain conditions. In order to conserve open alvar habitat on North Bear Alvar, a habitat excluded from fire for over 100 years, I recommend prescribed burning 30% of the habitat to stimulate native biodiversity and reduce woody cover in early seral stages.

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1. General Introduction

Ecosystems are experiencing some of the greatest stressors in their history by pressures in the current geological epoch, highlighting the importance for informed conservation and restoration strategies to preserve ecological functions and provisioning of habitat, especially for species at risk. Regime shifts have altered species compositions and functioning in systems, as a result, they have had profound consequences on species availability and provisioning of habitat and resources, including ecosystem services to humans (De Groot, Wilson, & Boumans, 2002; Hobbs & Huenneke, 1992; Scheffer & Carpenter, 2003). Local stressors and climate change have either directly (e.g. land-use change, exotic species introductions) or indirectly (e.g. species composition changes, altered disturbance regimes) affected many of the natural landscape processes responsible for maintaining a matrix of biodiverse habitat and features, and as a consequence have affected ecosystem integrity and resilience. Alvars, globally rare and sensitive ecosystems, are experiencing changes brought on by a loss of disturbances responsible for maintaining a matrix of resilient and biodiverse open grassland and meadow habitat. These will be further elaborated upon in the Literature Review section below.

2. Literature Review

2.1 Ecological restoration as a response to ecosystem stresses

Increasing attention and resources have been expended on ecological restoration as a means to reconcile with factors associated with human-induced degradation and climate change, including attention paid to globally rare and unique sites to preserve mosaic landscape diversity and habitat provisioning. The widespread degradation of ecosystems globally has instigated the implementation of a range of restoration practices to restore valued ecosystem functions and attributes. Restoration is a means to reconcile with these adverse environmental impacts and is guided by both evolving informed science and ethics. Historical precedent in restoration, attaining historicity, sought to closely align systems to a previous state, with a recovery of some idealised former composition or function (Desjardins, 2015; Hanberry, Jones-Farrand, & Kabrick, 2014) More contemporary notions are aimed at acknowledging the role and effect of climate change on systems and using a more adaptive management approach, reinforcing integrity and resilience within systems (Falk, Palmer, & Zedler, 2006). Overall, the field is incredibly adaptive and contextualised to local applications of restoration, assuming a dynamic nature to restore systems in question, guided by management goals, ethics, and capacity. They may assume a number of different approaches to attain restoration targets to advance and better the relationship between humans and nature.

The ultimate goal of restoring and managing ecosystems is the attainment of resilient systems with a particular composition, function, or trajectory pathway with desired states (Canadian Parks Council, 2010). Resilient systems retain their set of characteristics through repeat disturbance events that promote abiotic and biotic processes (Walker & Salt, 2012). These processes include magnitudes of geomorphic activity, hydrologic flux and storage, biogeochemical cycling, nutrient storage, and biological activity, and production (Peterson, Allen, Peterson, & Holling, 1998). In their optimal condition, resilient ecosystems are able to absorb certain types and magnitudes of disturbance and return to pre-disturbance conditions, or closely aligned trajectories within a reasonable amount of time (Holling, 1973). In a similar sense, they are also relatively resistant to change from the arrival of new potentially damaging exotic species (Hobbs & Huenneke, 1992; Holling, 1973). Resistance and resilience can both be enhanced and manipulated to prevent or avoid invasion (Elmqvist et al., 2004; Johnstone et al., 2016; Peterson et al., 1998), or be scaled up in response to invasive species by maintaining structural properties and ecological processes known to favour the persistence of desirable native species (Hobbs & Humphriest, 2013; Palmer, Ambrose, & LeRoy Poff, 1997).

Ecologists seeking to restore a system with defined goals can proceed along two restoration gradients: top-down or bottom-up approaches, or a blend of both. Top-down approaches involve more active species manipulation through the removal of the damaging invader and reducing its abundance to more acceptable levels (Falk, Palmer, & Zedler, 2006). These primarily involve manual removal with herbicides, biological controls, and identifying and controlling vectors through which unwanted propagules arrive to the site (Elmqvist et al., 2004; Peterson, Cumming, & Carpenter, 2003). The action would then be to prescribe a treatment option within reason to target the problem, and the treatment could also contribute to other ecosystem functions (i.e. space clearing, nutrient fluxes with fire, or grassland stimulation by cattle grazing). Bottom-up approaches on the other hand emphasize the restoration of properties or processes that contribute to sustainability, resilience, and resistance within a system (Falk, Palmer, & Zedler, 2006). These are more directed at the amelioration of ecosystem stressors that affect the status of desirable species, manipulation of disturbance regimes, the alteration of soil conditions to, or direct seeding or planting to yield a competitive advantage for native species. Employing a filter-based assembly model to target invader or problematic species would highlight suites of actions that practitioners could take from a mix of bottom-up and top-down approaches. The blended option employs an ecological model to filter a desirable species assembly using a disturbance-altering event and adding native species by planting (Hulvey & Aigner, 2014). Choices for restoration must also consider the null option for restoration, the ‘do nothing’ approach and allow nature to proceed along its own self-organised trajectory (Falk, Palmer, & Zedler, 2006; D'Antonio & Chambers, 2006). If applied to alvar communities, the do nothing approach would inevitably lead to a substantial loss of globally rare and significant communities, certain types of successional habitat, and nested species (Brownell, 2000; Catling & Brownell, 1998; Catling, 2009; Reschke et al., 1999).

2.2 Ecology and ecological restoration of rare habitats – Alvars

Alvars are among some of the most unique species-rich yet imperilled habitats in the world (Claudia & Douglas, 1997; Lundholm & Larson, 2003; Neufeld, Friesen, Hamel, & Dow, 2012; Reschke et al., 1999a; Taylor & Catling, 2011). Alvars are only known to occur in the Baltics (Bengtsson et al., 1988; Krahulex et al., 1986; van de Maarel, 1988), surrounding the Great Lakes region in Canada and the U.S. (Stephenson, 1983; Stephenson & Herendeen, 1986; Reschke, 1990, Gilman, 1995; Catling et al., 1975; Catling & Brownell, 1992), and in central Canada (Belcher et al., 1992; Catling & Brownell, 1992). They are ecologically defined as areas of naturally open dry grassland communities in humid and sub-humid climates, positioned over areas of glaciated horizontal limestone or dolostone with only a thin (10-15 cm) discontinuous soil mantle (Brunton & Catling, 2017; Catling, 2016; Rosén, 1995). Generally, alvar communities are characterised by a distinctive blend of elements from southern and northern prairie flora and fauna with less than 60% tree cover which is maintained by some combination of drought, flooding, fire, frost heaving, wind erosion, or mammalian grazing (Jones & Reschke, 2005a; Rosén, 1995; Schaefer & Larson, 1997). Periodic natural disturbances are responsible for maintaining open situations (less than 25% of both tree and shrub cover) and high biodiversity in alvar habitats, with up to 40 species/m^2, and up to 400 plant species occurring on certain alvar landscapes in Canada (Bakker, Bakker, Rosén, Verweij, & Bekker, 1996; Brunton & Catling, 2017). Alvars contain a number of globally rare species (G1-G3, see NatureServe, 2015 for more information on conservation ranking criteria), restricted species, significant species, and species at risk (Bakker et al., 2012; Brunton & Catling, 2017). The vegetation on alvars has been classified into three main ecosites: open alvars (grasslands and pavement), shrub alvars, and treed alvars (savannahs and woodlands), and have been differentiated into 13 community types within Ontario ecoregions (Reschke et al., 1999a).

The open habitat structure on alvars in Central Ontario is key to provisioning of critical breeding and fledging sites for migratory breeding birds and open grassland specialist species (Reid, 2011) in light of the disappearance of breeding habitat by the development surrounding the Greater Toronto Area. Their significance is further captured in sources of genes of wild crop relatives for crop improvement in stressed environments, areas of natural history and education, areas of biological research and monitoring for climate change, a canvass for the representation and preservation of biodiversity, and a source of native plants adapted to fire and drought (Brownell, 2000).

As is the case with grasslands, open alvar community types are dependent on some sort of periodic disturbance to avoid succession and colonisation by problematic woody species, especially *Juniperus communis* (Belcher, Keddy, & Catling, 1992; Claudia & Douglas, 1997). Encroachment by *J. communis* has the potential to radically transform the structure and composition of previously open habitat, facilitating lasting changes which extirpate rare, endemic native plant species, cause wide-spread changes in the use of habitat by various species of fauna, [e.g. Lepidoptera, Odonata (Taylor & Catling, 2011)], and changes in aboveground net primary productivity and biomass accumulation (Ansley & Rasmussen, 2005; Romme et al., 2009). Increases in woody cover in open alvar communities is accompanied by a decrease in native and herbaceous species cover and richness along with the disappearance of characteristic alvar species (Ratajczak, Nippert, & Collins, 2012). Encroachment eventually leads to the conversion of open alvar grasslands into shrublands and forests and a loss of phytogeographic uniqueness, posing a serious conservation problem (Rejmének & Rosén, 1992).

Many alvar community types are characterized and distinguished by a history of fire (Brunton & Catling, 2017; Reschke et al., 1999), leading to the idea that maintenance by fire is critical, at some temporal interval, to maintain ecological integrity of the ecosystem. Fire has been proven to be an effective management tool in conserving and restoring various plant communities by promoting productivity and reducing competition and encroachment in grassland systems (Quinlan, Dale, & Gates, 2003; Sherman & Brye, 2009; Stubbendieck & Volesky, 2007; Witt, 2006; Wolf, 2008). However, there has been considerable debate on the role of fire in alvar systems, and there is a further need to understand what effects fire might have on certain alvar community composition in post-fire succession habitats.

2.3 Introduction to Ontario Alvars

The alvars in the Ontario Great Lakes region are classified as a mix of boreal, eastern mixed wood forest plant, meadow, and prairie species and are located mainly on Pelee Island, Napanee Plain, in the Kawarthas, and around the Bruce Peninsula including the Manitoulin Island archipelago (cf. pers comm. Laura Robson, 2018). Alvars on Pelee Island off Ontario's south tip are comprised mainly of mixed wood forest types including Chinquapin oak savannahs and ~25% boreal species. In the Kawarthas in eastern-central Ontario, the Carden Plain is characterised by ~40% mixed wood forest species, and approximately equal contributions of boreal, and southern prairie species, and is known for its open plain habitat characteristics (Catling and Brownell, 1995), providing critical habitat for grassland breeding birds. The Bruce Peninsula alvars have a strong northern element presence including 25% boreal species affinity and are dominated by open alvar pavement communities including *Juniperus horizontalis* and *J. communis* surrounding boreal conifer savannahs (Brownell, 2000).

Three main structural alvar ecosites have been classified by Lee et al., (1998) and NHIC (1996) within Ontario, they are: open alvar, shrub alvar, and treed alvar. Open alvars are defined as having less than 25% tree cover and <25% shrub cover and are further divided by open alvar pavements, dominated by lichen-moss and dry annual forbs with over 50% exposed bedrock, and alvar grassland, dominated by over 50% cover by herbaceous species. Shrub alvars exhibit over 25% shrub and over 25% tree cover and are mainly composed of *J. communis* and *Dasiphora fruticosa L.* (Lee, et al., 1998). Two distinct subsets of treed alvars are present. Alvar savannahs are defined by 11-35% of sparsely intermixed trees with a vibrant herbaceous layer, forming a discontinuous canopy and may be positioned overtop pavement (Lee, et al., 1998). Alvar woodlands are defined by 36-60% continuous tree cover; both types containing a mix of southern and boreal tree species (Lee, et al., 1998). Open alvar grasslands, are among some of the most species-rich terrestrial communities globally, with roughly 350 species of herbaceous plant and 370 bryophytes recognised on Ontario alvars, including a number of rare and isolated species (i.e., *Agalinis gattingeri, Gentiana flavida, Cirsium pitcheri, Morus rubra, Carex juniperorum, Hymenoxys herbacea, Cystopseris laurentiana, Valerianella umbilicata, Cyprededium arietinum, Cirsium hillii, Iris lacustris, Astragalus neglectus,* and *Solidago houghtonii*), some of which are remnants of post-glacial times (Brownwell & Riley, 2000). In Ontario, 54 of the total native plant species are found mainly on alvar habitat, 43 of these are of global, regional, or provincial significance and/or rare (Reschke et al., 1999). Some species, such as *Hymenoxis herbacea* evolving to survive only in alvars (Reschke et al., 1999).

2.4 Carden Alvar Overview

The Carden Alvar (known also as the Carden Plain) is a large 2151 ha natural area containing a matrix of wetland and terrestrial habitat, including 678 ha of documented alvar habitat (NHIC, 1998). The Plain occurs on flat limestone bedrock of the Bobcaygeon formation with silty clay loam deposits. Surrounding forested areas contain more mineral rich Farmington loam and deeper soils on top of bedrock. The original survey in the mid-1800s describe the northern part of the site as “plains” or “spruce plains”, and southern areas as “prairie”, with prevalence of *Picea sp*., *Pinus sp., Thuja occidentalis*, *Betula papyrifera* (Jones & Reschke, 2005). A comparison of aerial photos from 1930 indicate that areas that were once juniper shrubland are succeeding into young forest (Jones & Reschke, 2005a). Some localised portions of alvar exhibit over 5% cover of non-native herbaceous plant cover, including the exotic grass *Poa compressa* (Brownwell & Riley, 2000). There is relatively low impact from current cattle grazing in the north end, though historically it may have been heavier due to the presence of broken down fences throughout (Brownwell & Riley, 2000). There is evidence of current and past logging practices in some areas (Schaefer, 1996), and multiple parcels of alvar in the Carden Alvar are owned and operated by mainly Lafarge (among others) for quarrying purposes (pers. obs).

Carden Alvar has secured great scientific interest and is classified as an Area of Natural and Scientific Interest (ANSI), largely due to its diversity, phytogeographical uniqueness, global rarity, imperilment, and the presence of bird species of risk (Catling, 2016; Jones & Reschke, 2005b; Nature Conservancy of Canada, 2013; Reschke et al., 1999). Part of the Plain has been classified as a Provincial Park in 2015, with parcels being owned and managed by the Nature Conservancy of Canada (NCC), Ontario Ministry of Natural Resources and Forestry – Ontario Parks Branch (OMNRF), and Couchiching Conservancy (CC). A vast majority of the site is classified as an Important Bird Area (IBA) due to the over 230 species of grassland and migratory breeding birds that utilise the complex matrix of habitat structure, including endangered, threatened, and special concern species (e.g. Henslow’s Sparrow, Loggerhead

Shrike, Bobolink, Eastern Meadowlark etc.). There is a mixture of fair to good quality communities and excellent quality, predominantly native alvar communities on Carden Alvar (Reschke et al., 1999). Biological inventories have documented over 400 plant species, many of which are rare, or restricted to alvar habitat, 130 species of Lepidoptera and Odonata, and various mollusc species (Brownwell & Riley, 2000). Plant species such as *Carex juniperorum*, *Hymenoxys herbacea, Cirsium hilli*, *Cypripedium arietinum*, *Iris lacustris, Solidago houghtonii* have high confinement to alvar habitat in various alvars in southern Canada (Brownwell & Riley, 2000).

As is common with most alvar communities around the Great Lakes Region, woody encroachment is prevalent into open areas on the Carden landscape. *J. communis* shrubland is dense, with individuals encroaching inward from treed conditions into open alvars forming thickets, with 25-50% cover of *J. communis,* and 17% of *Picea glauca* and *Thuja occidentalis* (Reschke et al., 1999). An inventory from Reschke et al. (1999) found over 110 ha of *Deschampsia cespitosa*, *Danthonia spicata*, and *Schizachyrium scoparium* alvar grasslands in Carden Alvar, with a high abundance and diversity of herbs and forbs, though open habitat types are in decline since that survey with an increase in woody species encroaching previously open areas. It presents an undesirable outcome for protecting and maintaining open alvar community types and an array of habitat structures and composition for significantly alvar-confined species and community types. Though, processes of woody encroachment are an inevitable trajectory until some succession-altering disturbance resets it to some earlier seral stage.

2.5 Alvar soils

Most literature relevant to soils on alvars has come from extensive studies of Ontario alvars (see Stark, Lunholm & Larson, 2003; Stark & Larson, 2003; Stark, Lundholm, & Larson, 2004; Jones

& Reschke, 2005; Catling & Brownell, 1999). A study on the nature of alvar soils by Belcher et al. (1992) and Shaefer & Larson (1997) revealed that alvars contained a “non-soil” soil. They defined the “non-soil” as an aggregation of surface material that did not meet the Canadian Soil Survey Committee’s (1978) classification of soils. Instead they classify it as lithic udorthent material, a type of rocky aggregation of particles with an under represented amount of actual “soil” that performs the same ecological function by supporting plant growth but comprised mainly of silicaceous sand (Stark, Lundholm, & Larson, 2003). The accumulation of this material is dependent on pre-existing depressions and crevices in the bedrock (Stark, Lundholm & Larson, 2003). Some alvars do exhibit the black nature of soils, and there are areas of deeper material that contain less siliaceous sand and more organic material (pers. obs).

Variations in subsurface limestone bedrock (generally ranging from 18m – 90m) contributes to the average soil pH of 8, with potentially more acidic soil occurring in sandy conditions (Stark et al., 2004), or those overgrown by long-lived juniper thicket. Because of the presence of micro-topographic variations in the subsurface limestone bedrock (caused by cracks, potholes, depressions, soil depth and chemistry) conditions can substantially change over small distances, in some cases within 1 meter, and dictate a patchwork of growing and environmental condition within a single alvar (Catling, 2016). Soil depth forms the basis for the mosaic of plant communities that colonise and establish on the site.

Limiting nutrients (primarily nitrate-nitrogen, but also phosphorus, and potassium) are found in significantly lower abundances on alvars, especially in areas of thin siliaceous sand, in comparison to tallgrass prairies and meadows (Sherman & Brye, 2009; Stark et al., 2003; Stark et al., 2004). This was found mainly due to significant weathering and runoff, in addition to poor holding capacity of the lithic undorthent substrate material (Stark, Lundholm, & Larson, 2004). Mean bulk densities of alvar soils, enriched with organic matter, have bulk densities generally lower than 1.0c/cm^3 (Stark, Lundholm, & Larson, 2004). Most of the substrate material, especially in open grassland alvars and patchy pavement situations, would be subjected to rapid erosion following intense rain and wind events. It is believed that cryptogamic crusts, products of lichens, mosses, and fungi, provide attenuations and stabilising functions from massive soil-eroding events (Claudia & Douglas, 1997). I initially set out to measure bulk density in all my plots and did so with moderate success (the results of which are not published in this thesis, but are available). Factors that confounded my sampling and analysis were mainly due to the rocky nature of alvar soils where some plots had a great percent cover of exposed bedrock, or others had a significant composition of small fragmented rocks on the surface and throughout the small soil profile horizons. It was possible to attain bulk density measurements in some plots but the varied samples would not have yielded any ecologically significant results with respect to the effect of the particular treatment.

2.6 Alvar conservation

Increasing research (Catling, 2016; Jones & Reschke, 2005; Neufeld et al., 2012; Reschke et al., 1999b) has highlighted the conservation importance of alvars on several different fronts. Alvars are highly diverse ecosystems, containing endemic and rare floral elements that represent relics of historic ranges. They contain globally significant elements, worth protecting as landscapes continue to experience degradation. In 2000, the Swedish alvar on the island of Öland was designated as a UNESCO world heritage site due to its intrinsic natural and historic values. In England and Ireland, limestone barrens are conserved under the Areas of Outstanding Natural Beauty Orders (1949) or Sites of Special Scientific Interest in order to preserve wildlife, wildlife habitat, physiographic features, cultural heritage, and geology (Catling, 2016; Joint Nature Conservation Committee, 2014). In North America, The Nature Conservancy (TNC), and the

Nature Conservancy of Canada (NCC) have listed alvars as globally imperilled ecosystems, and a collaborative effort between Canada and the US installed the International Alvar Conservation Initiative (IACI) to locate, preserve, research, and protect alvar habitats (Schaefer & Larson, 1997). Many alvars in Ontario are designated Areas of Natural and Scientific Interest, much like systems in Europe, due to elements of interest, plant adaptations, and habitat provisioning for rare and endemic species (e.g. Reid, 2011; Reschke et al., 1999). As a product of the harsh growing conditions on alvars, some alvar plants have developed drought tolerant adaptations to them that make them highly sought after for gene development and advancement of drought-tolerant agricultural crops in the future (Sjogren, 1988). The compositional plant structure of open alvar grasslands habitat provides cover and utility for seasonal sites for migratory grassland breeding birds, the fastest declining bird group, including a number of rare and at risk birds such as the Loggerhead Shrike, Bobolink, Eastern Meadowlark, Savannah Sparrow, Grasshopper Sparrow, Eastern Kingbird, and Eastern Bluebird (among others) (Reid, 2011).

One of the main threats to alvars globally is quarrying. Due to the easily accessible and shallowly-positioned bedrock, alvars are opportune sites for minimal removal of topsoil and maximum yield of bedrock (Catling, 2016). Additionally, anthropocentric decision making on the landscape scale may have profound influences for natural disturbance regimes (i.e., choices to suppress fire) and other environmental characteristics that are fundamental to the maintenance of highly distinct and diverse alvar communities (Collins & Calabrese, 2012; Varner, Gordon, Putz, & Kevin Hiers, 2005). To a lesser extent, logging, adjacent land development, and grazing also impact species composition and successional dynamics (Taylor & Catling, 2011).

Processes of natural succession also affect open alvar habitat by decreasing the size of natural openings through encroachment of woody species from adjacent stands, seed dispersal, or vegetative reproduction (Van Auken, 2009). Although these processes occur along natural gradients, the successional dynamics, including disturbances such as fire (Jones & Reschke, 2005b), may be disrupted as a result of landscape processes, human agency, or climate change. Increased research has focused on abating common juniper shrub encroachment, and also what constitutes a healthy matrix of alvar habitat types (see: Bakker et al., 2012; Jones & Reschke, 2005b; Leppik, Jüriado, Suija, & Liira, 2015; Reschke et al., 1999). Woody encroachment and natural succession of alvars will be further discussed in subsequent sections below.

2.7 Disturbance Ecology

The role of disturbances in ecological systems was once viewed largely as an insult to the balance of nature and synonymous with destruction of habitat (Hobbs, 1989). However, in recent decades, research has shown that certain types of disturbance constitute fundamental and creative roles in maintaining natural habitat heterogeneity in habitat conditions that organisms experience through space and time (Brawn, Robinson, & Thompson, 2001; Delong, Burton, & Geertsema, 2013; Hobbs, 1989; Johnstone et al., 2016). Ecological systems, regardless of scale, depend on some sort of disturbance frequency or magnitude to maintain productivity and resilience (Delong et al., 2013). Disturbances can be defined as "any discrete event in time that disrupts ecosystem, community, or population structure, whereby it changes resources, substrate availability or the physical environment" (Hobbs, 1989; Hobbs et al., 2006; Hobbs & Huenneke, 1992). Disturbances maintain habitat heterogeneity and diversity, and in the case of small cyclical disturbances, reinforce ecological integrity and resilience, influence above ground net primary productivity, nutrient cycling, and community structure (Copeland, Sluis, & Howe, 2002; Hobbs et al., 2006; Seifan, Seifan, Jeltsch, & Tielbörger, 2012).

Recent discourse has acknowledged the role of disturbances in the context of the 'intermediate disturbance hypothesis' (IDH), which suggests that local species diversity is maximised when ecological disturbances are neither too rare nor too frequent (Catford et al., 2012), a parody on Goldilocks' porridge. Low or reduced levels of disturbance will lead to low diversity, low productivity through competitive exclusion and dominance of long-lived or exotic species, while high or increased levels of disturbance will eliminate species unable to re-colonise, reorganise, disperse, or grow between disturbance events (Seifan, Seifan, Jeltsch, & Tielbörger, 2012b). The IDH suggests that at intermediate levels of disturbance, species richness and diversity are maximised because native species in that system thrive both at early and late successional stages brought on by a particular disturbance (Barnes, Sidhu, & Roxburgh, 2006; Catford et al., 2012; Shea, Roxburgh, & Rauschert, 2004). Intermediate disturbances feed into a non-equilibrium model which aids in the understanding of species richness and diversity. IDH is based off three central tenets: First, ecological disturbances have major effects on species richness and composition within the area of disturbance. Second, interspecific competition from one species becoming dominant in the system results in excluding a competitor. Third, moderate environmental disturbances prevent interspecific competition through species tolerance thresholds to the disturbance (Catford et al., 2012; Kalacska et al., 2004; Shea et al., 2004).

As disturbance regimes are foundational in shaping diversity, richness, as well as spatial and temporal extents of various habitat types, the loss of disturbance mechanisms is similarly accompanied by a decline of richness and function in communities (Johnstone et al., 2016; Walker & Salt, 2012). Losses of disturbance directly impact native diversity, alter the community structure, integrity and resilience, as well as increase susceptibility to exotic invasion (Falk, Palmer, & Zedler, 2006), particularly so in fire-adapted or fire-dependent communities (i.e. sage-scrubs, forests, grasslands, prairie-meadow) (Anderson, 2006; Limb, Fuhlendorf, Engle, & Kerby, 2011). In recognition of the role of disturbances in shaping and maintaining many community processes, the restoration of natural disturbance regimes to disturbance-adapted and dependent communities has been a critical component to active ecological restoration and is evidenced by prescribed burning and reintroduction of fire and large herbivore grazing to prairie grassland communities (Copeland et al., 2002; Hartley, Rogers, Siemann, & Grace, 2007).

2.7.1 Disturbance Ecology of Alvars

Floristic composition and vegetation communities vary geographically and locally within sites (Catling and Brownell 1995), primarily on the basis of disturbance and environmental conditions including soil depth and moisture regimes (Rejmének & Rosén, 1992; Rosén, 1995). The key feature of alvars, openness, may be maintained by one or a combination of disturbance mechanisms including, extreme mid-summer soil temperatures upwards of 53 °C, seasonal and periodic drought and the availability of soil moisture, mammalian grazing or fire (Brownwell & Riley, 2000). The relative importance of these factors varies from site to site depending on the bio- and physiogeography (Catling, 2016; Jones & Reschke, 2005; Reschke et al., 1999; K Stark et al., 2004; Sullivan & Sullivan, 2003). Disturbances on alvars act to virtually arrest succession through frequent events that favour stress-tolerant species and alvar-adapted species (Sjogren, 1988). In areas where disturbance does not occur, shade tolerant woody species and hardwoods such as *Quercus* sp. and *Acer* sp. start to take over (Sherriff & Veblen, 2007). Without disturbance, processes of succession pose a threat to open alvar and savannah alvar habitats (Claudia & Douglas, 1997; Jones & Reschke, 2005; Nature Conservancy of Canada, 2008; Reschke et al., 1999a; Kaeli Stark et al., 2003). Soil moisture, fire, and mammalian grazing are noted as the main contributors to facilitating the establishment and persistence of alvar-adapted species. These will be further discussed in the sections below.

2.7.2 Soil moisture

Several factors related to soil moisture are hypothesized as being influential in excluding woody species from open alvars including: drought, inundation, and the winter frost cycle. The general physical composition of alvars with shallowly positioned bedrock considerably influences water dynamics, which is mediated by soil depth, vegetation cover, the position of the water table, and surface water runoff (Eviner, Garbach, Baty, & Hoskinson, 2012; Wilcox & Davenport, 1995). Alvars experience a significant seasonal variation in wetness, with a high propensity of flooding and water pooling from March to May and sometimes again from September to November (Rosén, 1995; Brownwell & Riley, 2000). In mid-summer conditions, alvars experience desiccation and drought with soil temperatures reaching extreme temperatures of 53 °C. In some cases, periodic catastrophic drought effects, rather than the seasonal cycle of droughts, may be the most important factor in limiting woody encroachment (Rosén, 1995). Research on juniper alvar shrubland from La Cloche Alvar indicates that juniper shrublands have a lower water storage capacity and greater rates of desiccation than alvar grasslands or adjacent woodlands (Schaefer, 1999; Brownwell & Riley, 2000).

The winter frost cycle may be an equally important seasonal disturbance mechanism in open and shallow-soil alvars (Brownwell & Riley, 2000). The development of frost crystals has long been evidenced to disrupt bryophyte cover on alvar pavements, stress and kill emerging seedlings, and change vegetation patterns at the local scale. Equally, the lack of snow cover in harsh winter environments can result in ground freezing, having negative effects on seedling survival and germination in the following spring (Gilman, 1995).

2.7.3 Fire

There is considerable debate in current literature on the role of fire in alvars as a disturbance in maintaining highly distinct alvar habitat and preserving open alvar situations from woody encroachment. Alvars on the Bruce Peninsula have a particularly well-documented fire history (Suffling et al., 1995; Kelly and Kischak, 1992). Evidence of charred stumps and bleached limestone bedrock was found from Tobermory south to Cameron Lake in 1903 (Jones & Reschke, 2005b), indicating an intense fire which likely combusted most vegetation and reset succession to an earlier dominated state. Other alvars on the Peninsula may not have burned for 500 years based on ancient trees present (Jones & Reschke, 2005). Nearshore alvars have not burned in centuries due to water upwind (Schaefer, 1996). Some alvar types show a high percentage of burn evidence, such as: Bur Oak limestone savannah, White Cedar-Jack Pine alvar savannah, Creeping Juniper-Shrubby Cinquefoil alvar shrubland, and alvar non-vascular (bryophyte dominant) pavements (Jones & Reschke, 2005). For example, some alvars on Manitoulin island used to be maple hardwood forest in 1863. Following a fire in 1864, the maple-dominated forest was succeeded by Bur Oak savannah community type, and is slowly filling in with *Acer sp.* and *Ostrya virginiana* (Jones & Reschke, 2005). Over the past century, fires in other areas on Manitoulin have been reported, especially on Foxy-Gore Bay alvar, and have likely contributed to its open condition (Jones & Reschke, 2005). Research and information pertinent to the extensive alvar landscapes on Manitoulin Island, suggest that periodic fires are an important process in alvar ecology, particularly in maintaining a collage of habitat types and productivity (Jones & Reschke, 2005; Reschke et al., 1999; Schaefer & Larson, 1997). Alvar grasslands generally showed little burn evidence (Jones & Reschke, 2005), though this would be expected even if a fire did occur as physical markings may be absent due to a low abundance of rocks and other scar-bearing features present in forest stands and similar systems (Van Sleeuwen, 2006).

Other alvars have an array of old trees present and serve as a good indicator of fire history from tree rings and scarification evidence. In areas around Scugog Lake, Bear's Rump Island and parts of Dorcas Bay North, trees don't exhibit any evidence of fire scarification or charring (Schaefer & Larson, 1997). Scugog Lake and Great Cloche in Ontario both have trees >400 years old, with no burn evidence (Jones & Reschke, 2005). The sparseness of vegetation and accumulation of biomass atop the graminoid dominated site have signaled an exceptionally slow rate of growth. Fire may be extremely rare or not occur at all in some alvar situations. Therefore, Reschke (1999) as well as Catling and Brownell (1998) acknowledge that fire may not explain the existence of all alvars, though it has contributed to the creation of some and is a fundamental process on most alvars. However, there is also evidence of fire elsewhere on these sites leading to the conclusion that a heterogeneous arrangement of various fuel loads led to the creation of a mosaic of burned and unburned patches of the alvar (Schaefer & Larson, 1997), and physiographical characteristics may be the strongest determining factor to facilitate vegetation and corresponding fire patterns.

In some cases, alvars have been created by fire. Jones (1997) documented the formation of Silverwater Tower alvar by a fire in 1925 leading to the creation of a juniper alvar shrubland with no fire intervening since. Some alvars on the Bruce Peninsula and Manitoulin Island were created by fire and persisted for a long period after (Jones, 1997), reinforcing that in some cases, alvars can be created by fire, though they may or may not persist for extended periods of time after (Rejmének & Rosén, 1992). This would be dependent on various interacting forces facilitating community colonisation and recruitment. Successional alvar burns may be open for hundreds of years and are related to a number of late successional alvar communities such as shrublands, savannahs, and open woodlands. Transitionary phases may be species rich and contain various rare species that can only tolerate conditions brought on by mid-successionary stages (Jones & Reschke, 2005). Alvar creation is probably due to some combination of intense surface-vegetation cleaning fires which may occur at a time when soil moisture is significantly lower causing soil erosion because ground cover is removed and plant growth is delayed (Stubbendieck & Volesky, 2007).

Depending on the management goals for the alvar in question, which should incorporate goals at the landscape level, Jones and Reschke (1999) argue that species-poor shrubland dominated by common juniper should not detract from biologists' attention for conservation initiatives, especially those aimed at preserving critical habitat for specialist fauna species (e.g. catbird). *J. communis* in a successional timeline is still a consequence of fire and part of an ecological process that provides ecosystem services to a number of animals including Cedar and Bohemiam waxwings (Catling & Brownell, 1998). Conversely, others (e.g., Catling & Brownell, 1998; Taylor & Catling, 2003; Claudia & Douglas, 1997) suggest that fire is an intrinsic part of alvar succession gradient, and shrublands bring about more negative consequences (e.g., species richness declines, changes is soil and soil function, habitat loss). It seems appropriate to evaluate the site history and existing species to determine the significance of fire and consider alvars as dynamic openings which expand and contract as a result of time, species interactions, and their physical environment. Conserving and maintaining an array of alvar habitat (open, shrub and treed), as a result of patch dynamics created by natural or prescribed fires, other disturbances, and prevailing physical and environmental conditions is a desirable outcome in recognition that alvars represent a matrix of habitat types. Attaining this at some spatial level is necessary in conservation planning and initiatives.

2.7.4 *Mammalian grazing*

The International Alvar Conservation Initiative identified several types of grazing activities that, have an impact on the composition of alvar vegetation and structure. Graze and browse by small mammals, such as rabbits and voles, are considered part of the natural process and have a fairly negligible effect on vegetation, and may even act as important vectors in seed dispersal (Reschke et al., 1999). Some aspects of grassland and rangeland conservation are particularly concentrated on the role of mammalian grazing in supporting the restoration of grassland community from unwanted woody or exotic invaders, and stimulating native herbaceous species through a mild and intermediate disturbance (Catford et al., 2012; Henning, Lorenz, von Oheimb, Härdtle, & Tischew, 2017; Ratajczak, Nippert, & Collins, 2012). Conversations have shifted from 'do herbivores have an effect' to 'why do effects differ spatially?'. Increases in grassland diversity have been evidenced in some studies while in others it adds a vector for exotic invaders either due to fecal seed transport or through opening patches for invasion (Anderies, Janssen, & Walker, 2002). The acute effects of herbivores on plant species composition and richness is dependent on the type and abundance of herbivore species, selectivity, and composition in a particular environment (Henning et al., 2017). Natural populations of herbivores may increase plant diversity, although artificially high or introduced populations have weak or negative effects on herbaceous vegetation (Olff & Ritchie, 1998).

Some alvars within the Great Lakes region, along with those in in the Baltics, have a long history of livestock grazing influence by cattle, horse, and sheep (Sjogren, 1988). Historical grazing was typically opportunistic. Open alvars provided the perfect medium to graze cattle and livestock. However, in contemporary times, grazing by livestock raises some concerns about the effects grazing has on reducing woody encroachment (Claudia & Douglas, 1997). Cattle are selective in the herbs that they graze, and as a result, exotic and some native species have adaptations that make them unpalatable (i.e. thorns on *Echium vulgare*), and therefore cattle will select against it, leaving areas of alvar dominated by exotic species (Limb et al., 2011). A blend of prescribed burning and grazing has surprisingly negative consequences on the local alvar plant community, especially when burns are patchy and non-homogenous over the habitat type (Reschke et al., 1999). Cattle have been found to select for burnt areas due to new vigorous growth and higher palatability of herbs and forbs (Rosen & van der Maarel, 2000). They over graze these areas and leave other patches that escaped fire un-grazed, leading to imbalance and a decreased carrying capacity of the grassland alvar to support livestock (Limb et al., 2011). Intensity of grazing is also an important factor. A comparison between un-grazed, moderately grazed and, over grazed alvar grassland exhibited a large decline in biomass and floristic changes in the overgrazed area, with perennial and annual ruderal species replacing dominant alvar herbaceous species (Rosén & Bakker, 2005). There is anecdotal evidence from several Great Lakes alvar sites that elude to cattle maintaining openings from invasion by woody shrubs and trees, which may be important factors for maintaining suitable nesting sites for grassland breeding birds. However, grazing by cattle, from an ecocentric view is generally detrimental to alvar communities as it affects native species diversity and may act as a vector for exotic invasions (Brownwell & Riley, 2000).

2.8 Natural Succession of Alvars

The classical succession models, such as those proposed and developed by pioneering scholars like Clements (1916) and Gleason (1910) aren't readily applicable to alvars due to the dynamic nature of their ecosystems, patchy distribution of microhabitats within the ecosystem, and the susceptibility alvars have to frequent disturbances (e.g. flooding, drought, fire, frost heave). Some of the most comparable notions of succession and climax communities related to alvars would be the development of alvar savannahs or treed alvars. However, alvar savannahs are not common, and may not exist for a very long time (Connell & Slatyer, 1977; Taylor & Catling, 2011). Many savannah types are restricted to edge habitats or patches on the alvars without becoming a dominant community at the local site, and many are influenced by prevailing soil conditions (Taylor & Catling, 2011).

The underlying geology and soil depth is influential in the establishment of graminoid, herbaceous and woody species; with the former two selecting generally for areas of shallower soil, and woody species having a general restriction to areas of deeper soil, increased soil moisture, and cracks in the rock (Catling, 2016). There are analogues between rock barren succession and plant community development on alvars. Frequently, rock barrens lack long-term stability due to successive total clearing disturbance events (Swengel, 2001). Whereas some portions of alvars are buffered from total biomass clearing disturbances (Wentworth, 1981). These exclusions are mainly attributed to habitat heterogeneity and patchiness (Wentworth, 1981; Gilman, 1995). Both alvars and rock barrens exhibit a virtual arrest of succession in some cases; through successive disturbance events, maintenance of plant communities is kept in a generally narrow window of development (Belcher, Keddy, & Catling, 1992). In the case of alvars, patches of different habitat co-occurring together in a relatively small space aids in regeneration and preserving some habitat types from disturbance, while disturbances clear others (Bakker et al., 1996). Succession in post-disturbance alvar habitat proceeds along succession gradients similar to grasslands and prairies. These start with a high presence of ruderals, exotics, and native herbaceous species, and are determined by edaphics and recurrent disturbances (Bakker et al., 2012; Limb et al., 2011; Romme et al., 2009). The variability of environmental conditions within alvar ecosystems lead to a variety of succession stages observed within a single habitat since succession and vegetation development might proceed at faster paces in some areas of more favourable growing conditions [increased soil depth, moisture, and light (Catling, 2016)]. Additionally, vegetation development might be protracted in other areas of dry, thin soil, or areas where succession is delayed due to legacy effects (Lett & Knapp, 2005).

Various environmental conditions, produced by the physical geology of alvars, affect the ability of plants to colonise and persist in the harsh growing conditions, and act as primary filters to succession (Wentworth, 1981). Among them, edaphics, bedrock type, and surface hydrological characteristics can restrict encroachment and succession of alvars to ‘treed’ areas by limiting the extent where and which species can establish (Catling, 2016). For example, the high pH content found in calcareous limestone and dolostone rock may act as a filter to only allow high pH tolerant species to occur there, species which are more globally rare and unique (Wentworth, 1981; Gilman, 1995). In communities dominated by limestone pavement or extremely shallow soils, competition is reduced leading to many stress-tolerant endemic species occupying harsher niches (Gilman, 1995). Weedy species may grow faster in higher levels of moisture and nutrients brought on by certain disturbances. However, they may be quickly replaced by stress-tolerant species when moisture and nutrient levels are low (Meiners, Pickett, & Cadenasso, 2002). These factors allow alvars to be relatively biodiverse with migration and succession restricted within a well defined set of patches with multiple factors influencing structure and composition.

In open grassland communities in North America, late succcesional communities can be defined by an abundance of graminoid cover and their associated shrubs [e.g., hawthorn, elms, etc.; (Olff & Ritchie, 1998)]. These associations of grasses and a few shrubs have been classified as late successional vegetation communities in grassland systems, particularly when time between disturbances is long and the community reaches its later seres. Late successional communities may be dominated by just a few species with relatively high cover (Catling, 2016;

Copeland et al., 2002). Many sites within alvars may be uninhabitable for tree species and a long lasting sere in an open community may resemble a grassland composed of dominant characteristic open alvar species, or a shrubland dominated by juniper thicket (Brownwell & Riley, 2000). Certain species may only exist in early successional stages brought on by a disturbance event. Pavement alvars, although containing a diverse abundance and cover of cryptogams, don't develop further due to the lack of available rooting substrate material (Leppik et al., 2015). However, alvars are generally considered open situations with less than <25% successive tree or shrub cover (Jones & Reschke, 2005; Lee et al., 1998). Further, open alvar communities represent some of the best alvar habitat in a successional timeline, have the greatest and most diverse habitat utility, and are a desirable state to attain and maintain (Reid, 2011; Reschke et al., 1999). Without a resetting disturbance, sun tolerant opportunistic species begin the successional process, with increasingly shade tolerant species colonising until the canopy closes and soil recruitment occurs (Nature Conservancy of Canada, 2008).

2.8.1 Woody encroachment on open alvars

The quickly colonising shrub *J. communis* is rapidly displacing open alvar vegetation in alvars around the Great Lakes region (Bakker et al., 2012; Belcher et al., 1992; Schaefer & Larson, 1997; Stark et al., 2003). Woody scrub encroachment into open alvar grassland habitat has the potential to threaten habitat heterogeneity, function, and structure. Further, it is accompanied by a decline in characteristic alvar species richness and herbaceous richness (Rejmének & Rosén, 1992b). Woody encroachment correlates with increases in net primary productivity far beyond existing levels, and corresponds to a decline in plant species diversity in herbaceous-dominated communities (Harpole & Tilman, 2007; Ratajczak et al., 2012). As a result of cessation of certain land-uses (i.e., fire wood harvesting, logging, grazing) or disturbances (moisture regimes, fire) opportunistic woody species establish and eventually form homogenous scrublands over vast areas (Lett & Knapp, 2005). This increase in shrub cover in open alvars attenuates light at the surface level and changes habitat composition and structure, two elements of which are the basis for open alvar habitat. Without open characteristics, the function of the habitat changes as well. Once established, juniper is not readily eliminated. There are legacy effects in the seedbank and prolonged establishment hampers the ability for native characteristic vegetation to re-establish.

Specific site factors facilitate or hinder establishment; micro-topographic changes, soil depth, and moisture availability have distinct effects on patterns and assembly of vegetation at the local site scale leading to a patchy array of alvar communities within the ecosystem. Additionally, they allow encroachment and succession by woody species from edges into grasslands, and is part of the natural process (Ho & Richardson, 2013; Lett & Knapp, 2005; Ratajczak et al., 2012; Rejmének & Rosén, 1992; Van Auken, 2009). However, a woody species can be considered problematic when it assumes any of the following traits in a given system: it is not a native, has expanded outside of its natural range though some vector, it is absent from natural predation or mechanisms that would otherwise control its extent, forms mono-cultures or homogeneous patches excluding previous species and changing composition, structure, and function, or is a locally dominant native that prevents the growth and establishment of native species to that area (Zavaleta, Hobbs, & Mooney, 2001). Those woody species that do can interfere with the maintenance of particular vegetation types by outcompeting more desired characteristic species. Additionally, they threaten the persistence of rare species, and at the same time other trophic levels have an influence of post-disturbance succession, and keep communities in a persistent undesirable state (D'Antonio & Chambers, 2006).

Open grassland habitat comprises the vast majority of the species rich extent of alvars, harbouring populations of rare and endemic species. The main invasive threat to alvars globally is the rapid expansion of *J. communis* and *Potentilla fruticosa* into open dry habitats, often rooting in fissures created in the bedrock (Rosén, 1995). This is considered a natural succession process, proceeding in places where traditional land use practice has changed, or where some ecological process responsible for maintaining community composition has changed (Kalacska et al., 2004). Woody invasions into open alvar habitat are most profound when clusters of woody species associate together, or when juniper thickets occur and persist for a long time. Additionally, they cause significant declines in the richness and abundance of native herbaceous species. This is most apparent when shrub cover exceeds 75% leading to light attenuation, a change in available soil nutrients and chemistry, and a change in the local competitive structure (Bakker et al., 2012). Although some alvar species may persist for some time in the soil seedbank, the germination viability of seeds declines the longer the shrubs persist (Bakker et al., 2012). A persistent and high percent cover of juniper corresponds to changes at the micro-habitat scale, including higher soil acidity, a denser and more persistent litter layer, low herbaceous plant survival, and a thick moss carpet (Bakker et al., 2012). The presence of the moss carpet and the long-term presence of juniper scrub may prevent some species from establishing through seed dispersal or other mechanisms and also prevent seeds in the soil seedbank from germinating (Bakker et al., 2012). Long-lived scrub vegetation and accompanying changes act as filters which may limit the likelihood of alvar grassland species establishing by seed and persisting after a biomass clearing event (Bakker et al., 2012). Catling and Brownell (2000) compared aerial photography from 1987 to 1930 of open alvars and noted considerable infilling of open alvars in the Great Lakes from tree and shrub edges of adjacent habitats. These edge systems started off as savannah-like, with some sections dominated by juniper thickets and scattered trees, though have since experienced an increase of continuous tree cover to greater than 50% (Catling & Brownell, 2000; Belcher et al., 1992). These findings suggest that juniper shrubland is a precursor and facilitator for later treed alvar situations, or a climax alvar shrubland community.

2.9 Fire Ecology

Many natural communities and landscapes in Ontario necessitate fire as a key ecological process, providing a disturbance which augments and alters species composition, community structure, increases in habitat heterogeneity, and changes in biodiversity (Van Sleeuwen, 2006). Meanwhile, it seems logical to acknowledge the capacity for fires to cause widespread significant damage to values associated with ecological or economical elements. Historically, the prevailing long-term idea of conservation has been through fire suppression, which has negatively impacted ecological integrity and health brought on by shifts in species composition, accumulations of biomass, insect infestations, poor productivity and regeneration, and degradation of wildlife habitat utility.

Fire has long been evidenced as a natural process soon after the emergence and establishment of vegetation, some 420 million years ago (Scott and Glasspool, 2006). In many grassland, meadow, and forest systems, the forces of climate, fire, and herbivores have been the principal interacting forces that form and maintain distinctive habitats (Stubbendieck & Volesky, 2007). The legacy of fire in grassland systems supports the notion that fire is an important evolutionary factor (Bowman & Murphy, 2010). Moreover, plants and animals have coevolved with fire and are considered to be fire-adapted or fire-dependent (Brownell, 2000; Jones & Reschke, 2005). In grassland systems, broad climatic factors permit fires to burn extensively and aid in the suppression of fire-intolerant species and woody dominance, which are typically not native to the local system (Collins & Calabrese, 2012; Hood & Miller, 2007).

2.9.1 Plant response to fire

Plant species response to fire depends on several factors including the above ground height of its growing points, a function of plant maturity and plant-growth characteristics. Woody plants typically grow from their twig tips (Eastern white cedar, juniper) and are killed easily by fire (Stubbendieck & Volesky, 2007). Native perennial grass response is different as they grow from their bases and the reproductive rhizomes are unharmed during a surface burn. Generally, only the top ¼ inch of soil rises in temperature momentarily (Dudley & Lajtha, 1993). Annual grasses and broadleaf plants are damaged when burned during the active growing season (Limb et al., 2011). Biennials also differ in their response to fire. Some can be damaged during intense fires if their growing points are raised or can be spared depending on their reproductive strategy (Bailey & Anderson, 1980). Several ways in which plants tolerate or exploit post fire conditions include increased productivity (e.g., prairie and grassland species), increased inflorescence (e.g., poa species), improved post-fire seed dispersal (due to increased wind and surface water flow), and stimulated seed release in fire serotinous species (Bowman & Murphy, 2010). Not all fires are equal, and post-fire successional vegetation patterns are dependent on numerous factors, including intensity, season, soil moisture, and pre-fire vegetation (life history traits, available seed supply) (Keeley, 2009; Kral et al., 2015; Romme et al., 2009; Stubbendieck & Volesky, 2007). Fire often resets successional sequence in a secondary succession context. High intensity fires can be stand replacing and completely reset succession, while low intensity surface and ground fires may remove herbaceous layers and leave the canopy intact (Keeley, 2009).

Catling and Brownell (1998) acknowledge the importance of fire in alvar plant communities as a key factor for maintaining highly distinct plant-community associations, especially by stress-tolerant species and restricted rare and endemic species not found elsewhere. They found that increased burning coincided with an increased diversity of characteristic open grassland alvar plants (Catling & Brownell, 1998). These burns may be open for hundreds of years after, or may infill quickly and proceed through various transitions that may be species rich and contain rare species (Catling, 2009; Catling & Brownell, 1998). Taylor and Catling (2011) highlight the importance of post-fire succession habitats in globally rare systems that act as sinks for certain species, where post fire vegetation richness and availability of open habitats contribute to the increase in the richness of *Hymenoptera* and *Lepidoptera*. Other studies by Claudia & Douglas (1997), Jones & Reschke (2005), Nature Conservancy of Canada (2008), Reschke et al. (1999), Rosen (2000), Stark & Lundholm (2004), similarly acknowledge the role of fire in maintaining highly distinct alvar plant communities and associations, though this is a geographically dependent phenomenon.

2.9.2 Fire Management on Alvar Ecosystems

Fire suppression, a lack of a return interval, is recognised as one of the key contributing factors to the loss of certain diverse and mosaic habitats in fire-dependent ecosystems in Ontario (Van Sleeuwen, 2006). Prescribed burning has been used to reconcile factors associated with human-caused degradation, and to maintain a number of alvar communities including alvars on Carden Plain (Cameron Ranch, Bluebird Ranch), alvars on Pelee Island (Stone Road Alvar), alvars near Ottawa (Burnt Lands), and those along the Bruce Peninsula (Brunton & Catling, 2017; Nature Conservancy of Canada, 2008; Reschke et al., 1999). Fire can be one of the most efficient and resource effective methods for vegetation management employed by conservation authorities on alvar systems. Fire intolerant species will decline, yielding space for characteristic fire tolerant species of alvar grasses and forbs to flourish.

It may be uncommon in Eastern North America for fires to self-ignite as a result of lightning due to the generally humid and moist conditions (Nature Conservancy of Canada, 2008). Furthermore, factors associated with climate change will lead to unpredictable fire events in the future, although other systems may ignite easily due to environmental factors (Moritz, Hurteau, Suding, & D'Antonio, 2013; Van Sleeuwen, 2006). It is hard to establish exact temporal intervals of fire return in many of the landscapes without exact markers, and this is influenced somewhat by First Nations use of fire for space clearing (Van Sleeuwen, 2006). However, it is widely accepted that the processes of fire are necessary at some temporal interval and spatial extent to maintain native diversity and productivity in North American ecosystems (Van Sleeuwen, 2006).

In 1992, Stone Road Alvar was burned in order to maintain and enhance the savannah alvar communities and rare species, to supress fire-intolerant woody species, and to control invasive species (Nature Conservancy of Canada, 2008). The results of prescribed burning on Stone Road Alvar led to the creation of more savannah like habitat which quickly grew in with shrub thickets and uncharacteristic maple and beech hardwoods within 10 years. In the long-term, this may lead to the extirpation of locally significant species (Nature Conservancy of Canada, 2008). This suggests that multiple sequential burns over a single habitat may be necessary to reduce legacy effects of hardwoods and uncharacteristic alvar species in successive seasons.

Understanding the timing and extent of fire season is crucial to effective prescribed burns. The post-fire species composition and dominance of species will differ if they are burned early in the growing season from those burned in the summer or fall seasons. Differences in timing relate directly to individual species morphologies, seeding phenologies, or germination requirements, and the species' availability to reproduce through rhizomes. Cooler and damper conditions brought on by spring time conditions in March allow for greater control and manipulation over the burn area. However, fire in spring time poorly burns woody material due to the high moisture content of above ground vegetation and the present litter layer, along with the general moisture of mosses (Copeland et al., 2002; Limb et al., 2011; Nature Conservancy of Canada, 2008; Reschke et al., 1999). Fires prescribed in the early spring increase the infloresence of heat tolerant plants while supressing some ephemeral growth (MacDougall, 2004). Fires conducted in April are generally too early in the year and cause damage to critical foliage components and above ground structures of cool-season plants, leaving late-season plants untouched and more competitive (MacDougall, 2004). Burning woody species at the beginning of a drought event, during high intense stress events (July and August), during flowering, or when moisture levels are low are ideal as the plants are already stressed and this leads to significantly less re-sprouting than in other times (Reschke et al., 1999; Stubbendieck & Volesky, 2007). Thinning of shrubs, and adding cut material to the burn area to increase fuel loading prior to burning can increase the effectiveness at controlling these species of interest (Nature Conservancy of Canada, 2008). Grasses are usually non-selective on burning season, underground rhizomes and reproductive parts will sprout given correct post-fire conditions (Anderson, 2006). This may be delayed somewhat on alvars during drought periods brought on by mid-summer conditions.

Fire intensity is also a subject of concern and directly related to vegetation fuel loads, fuel moisture, prevailing weather, and the spatial arrangement of fuel in the area of interest. High intensity fires may adversely impact even fire tolerant or fire dependent species by impacting growing points below the soil surface or destroying reproductive mechanisms entirely (Knapp, 2009; Parminter & Bedford, 2006). Conversely, low-intensity fires may not have the desired impact to reduce woody and fire-intolerant species from a system and may be an ineffective use of conservation resources and capacity if not applied properly.

Ultimately, burn season and timing should correspond to the current vegetation, site in question, and management goals for the site. Further, it should be applied accordingly to attain the desired effect in recognition that systems dominated by a woody state for a long period of time may require a fire return interval of 2-3 years to attain a more open condition, to reduce woody dominance, and to reduce legacy effects when trying to restore fire to a degraded system.

2.9.3 Historical and Bounded Ranges of Variability

Establishing the range of appropriate fire return intervals over time and space is critical for ecosystem management in fire-prone ecosystems. Historical accounts and reference conditions, including physical scarification, have been the basis for evaluating the status of fire in modern systems. They have been used as benchmarks for setting fire management goals contextually appropriate to systems in question. One approach to quantify fire return intervals has been the historical range of variability (HRV) framework. HRV uses scarification evidence and environmental parameters to establish the range of variation in fire return and patterns that are influenced from a broad set of ecological patterns and processes in order to quantify the naturally occurring fire intervals. The premise of HRV is that retaining key ranges of variability in many ecosystem components can reinforce ecological resilience, and preserve function over a broad range of system components (Moritz et al., 2013). HRV identifies baseline conditions for past fire regimes, or past composition of vegetation mosaics, and aims to capture natural ecosystem composition before human perturbations (Moritz et al., 2013). For management, HRV would inform on how to restore or emulate natural fire processes in space and time. HRV would also help to avoid crossing ecological thresholds that would send the system into some different ecological state, perhaps even a stable state with no means of return (Hobbs et al., 2014; Moritz et al., 2013).

One notable issue with HRV, is the influence First Nations peoples had on some key ecological processes, shaping land, and employing the use of fire (Pyne, 1982). In addition, future environmental states and conditions will be vastly different than those informing HRV. Opponents to HRV have established a different framework. Rather than managing around and trying to attain historical conditions, the bounded ranges of variation (BRV) accounts for the full range of possible conditions (McKenzie, Miller, & Falk, 2011; Moritz et al., 2013). BRV goes further in recognising the importance of disturbance regime thresholds to restrict transforming invaders from a system, as well as the abundance of thresholds of the invader beyond which the probability of ecosystem transformation is greatly enhanced (McKenzie, Miller, & Falk, 2011). BRV critically acknowledges changing suites of parameters in response to climate change to inform dynamic fire policy. Because crossing a threshold can result in rapid and profound change in composition, structure, and function, one goal of fire management is to increase resilience within prescribed basins of attraction that themselves are stable (Scheffer & Carpenter, 2003). An example of this could represent a suite of disturbances (e.g., fire, herbivory, frost cycle, and precipitation) that cause relatively small tolerable shifts between vegetation states, but within desired windows (Keane, Hessburg, Landres, & Swanson, 2009), thereby maintaining certain compositions of vegetation in space and time by clearing away species intolerant of those disturbances. Thus, BRV is a more desirable approach in systems where invasive species are a transformational concern, and should include elements of both biotic (invader abundance or percent cover) and abiotic (fire regime parameters) boundaries within which the desired system functions are maintained (Gosper, Prober, & Yates, 2013; Moritz, Hessburg, & Povak, 2011; Moritz et al., 2013).

2.9.4 Experimental Approaches to Prescribed Burning

Small-scale fire approaches can be highly effective at facilitating experimental control over variables to allow for greater replication of fire effects. Experimental burning aids in the understanding of fire season, return interval, magnitude, heat dosage on plant communities, post-fire succession, on native plant communities and wildlife habitat. Most research fails to measure fire characteristics and burning conditions which may contribute to varying results when trying to quantify fire effects (Bailey & Anderson, 1980; Limb et al., 2011). To increase replicability and experimental control, fire ecology researchers have the ability to conduct fires on smaller scales (from several square meters to several hectares) compared to wildfires and prescribed burns. Currently in the literature, three other studies that have employed the use of a 'burn box' device to contain and test fire effects on small scales, these are elaborated upon below (see: Kral et al., 2015; Richards et al., 2014; Sharrow and Wright, 1997).

Kral et al. (2015) tested an in-situ burn box approach to compare time-temperature profiles of the experimental burn to prescribed burns. Kral et al.'s (2015) burn box consisted of four 2 m x 2 m aluminum sheets constructed around in-situ vegetation and augmented with timothy hay to adjust fuel loads to 3000 kg/ha and ignited using a propane torch. They used thermocouples to log time-temperature profiles within the box. It was found that heat dosages, burn times, and mean time-temperature curves were within range of other fires in mixed grass and shrubland situations, and performed well at combusting biomass.

In contrast Richards et al. (2014) used a wood variant of a burn box to measure the survivorship and fecundity of *Lygodium microphyllum*, an exotic-invasive plant in the Florida

Everglades, USA, to determine if fire is an effective management strategy against its encroachment. They inserted metal tags painted with heat-sensitive paints and thermocouples inserted into the soil to measure heat at the roots of the plants. They ignited back burns using a gas drip torch and propelled the fire using a large industrial fan. The authors were able to compare burnt *Lygodium* plant responses to unburnt control plots in order to inform and develop a more effective conservation management involving the use of prescribed burns strategy to cope with invasive-exotics in the Everglades.

Sharrow and Wright (1977) tested the effects of fire, ash, and litter on soil nitrate, temperature, and moisture to manage *Pleuraphis mutica* communities in Colorado, USA. The study aimed to understand in greater detail the effects of fire on soil nitrogen and how the various factors associated with post-fire conditions affected plant growth. Sharrow and Wright (1977) used a 2 x 2 x 1 m, 16-gauge steel burn box to contain the fires. They found that small-scale fires had significant effects on soil temperature (7.5-8 cm below soil surface), and on soil moisture availability. They also found that, even using a containment device, the effects of fire were replicable to natural, uncontained prescribed burns and significantly increased *Pleuraphis* grass yields.

2.91 Justification for Treatments Used

I chose to use three common management interventions that are principally aimed at vegetation control and providing a disturbance. The modes of application and other specifics are elaborated upon in the *Methods* section. This section serves to justify the particular choices and other areas of research where either practical or experimental applications have been employed.

2.91.1 Prescribed burning using the "burn box" technique

The choice to employ fire was driven under the NCC Property Management Plan developed for North Bear Alvar (Nature Conservancy of Canada, 2013). Multiple considerations factored into employing the burn box technique including spatial considerations, the temporal timeline for low-complexity licensing, burn site scoping and development of a burn plan, and budgetary limitations. Most of these barriers were overcome by utilising a mesocosm burn box device that contained the spread of the fire. Other studies (Kral et al., 2015; Richards et al., 2014; Sharrow & Wright, 1997), employed the use of a containment device when prescribing fires. The particular modes of application and outcomes are reviewed above under *Experimental Approaches to Prescribed Burning*.

2.91.2 Topical glyphosate application via backpack sprayer

Experimental and practical treatments using herbicide have been highly effective at killing individually treated problematic plants on terrestrial (Pitt, Thompson, Payne, & Kettela, 1993; Sullivan & Sullivan, 2003) and aquatic environments (Caffrey et al., 1999; Mozdzer, Hutto, Clarke, & Field, 2008; Riemer, 1976). However, the long-term effects of herbicide application and its effect on assembly factors remain uncertain in native alvar plant communities (Reschke et al., 1999). Glyphosate is typically only applied as a control to a particular species or group of species posing some conservation risk. Although other treatments are known for their utility as ecological disturbance mechanisms, glyphosate has little to no direct or indirect benefits to other ecosystem components (Kaiser-Bunbury, Mougal, Valentin, Gabriel, & Blüthgen, 2015; Mattrick, 2012) and has assembly-limiting legacy effects in the soil (Lancaster, Hollister, Senseman, & Gentry, 2010; Landry, Dousset, Fournier, & Andreux, 2005).

Management authorities, NCC included, typically only consider herbicide application to woody material after it has been cut on the exposed surface in order to reduce the likelihood of re-sprouting. This topical application is not broadcast to understorey herbaceous plants, and only species of concern are targeted. The additional use of stem-injections of individuals using glyphosate is an approach to selectively target problematic species given the presence of other sensitive ecological elements while effectively killing targeted individuals (Edwards, Clay, & Dixon, 2000). Application of herbicide in this study was targeted to plots that were heavily encroached by woody species, and where herbaceous layer did not contain sensitive or rare plant species. The particular methods used herein will be elaborated further in subsequent sections.

2.91.3 Manual removal through lopping

Manual removal of shrubs and trees, either mechanical, lopping or other means, is an important conservation management and restoration practice to facilitate openness and increase light penetration for light-dependent herbaceous vegetation, especially as shrub and tree cover encroach into sensitive ecological areas (Sundberg, 2012). Woody removal is aimed at ameliorating effects of increased competition, lower biodiversity, along with attenuation of light, nutrient, and water resources, and increasing herbaceous richness (Ratajczak et al., 2012). Best management practices of removal of woody species have involved cutting stems at or slightly above ground level and applying a topical herbicide to the cut material to limit the potential of re-sprouting (Lett & Knapp, 2005; Sundberg, 2012). Combinations of cutting and prescribed burning are also practiced, though this may have potentially negative impacts if fire severity and burn dosage increases to thresholds intolerable for native seedbanks and vegetation (Quinlan et al., 2003).

2.92 Existing Gaps

Few studies have looked at replicable experimental mesocosm studies to study the acute effects of a treatment on alvar community to determine responses. This gap is most evident when responses in a system are not known, when regime shifts may have catalysed new system characteristics that are unknown in their response to disturbance, or when a method is being tested against the efficacy of others.

While there is considerable debate on the mechanisms responsible for the maintenance of open alvar systems, discourse in literature cannot agree to the specific recipe of disturbance. The recipe includes the frequency, type, and magnitude responsible for maintenance. Questions and research on alvars centered around disturbance have shifted from 'is disturbance necessary' to 'what disturbance is naturally precedent in a local context'. This research aims to contribute to knowledge gaps in the Carden Alvar through a comparative treatment approach to reduce woody encroachment and promote characteristic open alvar species.

2.93 Purpose of this Research

The purpose of this experiment is to contribute to the existing knowledge bank and to fill in knowledge gaps with the Nature Conservancy of Canada in the field of restoration and conservation ecology in the maintenance of alvar habitat. Specifically, it is aimed at preserving and restoring the correct array of open alvar habitat for the conservation of rare and endemic species along with provisioning habitat for grassland breeding birds through the use of prescribed fire. Specifically, this study seeks to compare and contrast the effects of prescribed burning using the experimental burn box design against herbicidal application and manual removal techniques. This research contributes to the field of restoration ecology and alvar ecology by first understanding the ecology of alvars and then identifying objectives and treatments likely to restore alvar habitat. The results of this study will help guide future management actions for best practices for prescribed burning and can be tangibly built upon for future studies to look at fire effects in ecological systems in a controlled mesocosm study before applying it to the whole system in question. In doing so, there are three objectives to this thesis:

1. To use an experimental burn design to compare and contrast the effects of prescribed fire to herbicide and manual removal and how these facilitate native biodiversity and reduce woody cover and abundance.
2. To assess and quantify the composition and status of post-treatment vegetation among the treatment types to determine best modes to conservation management of North Bear Alvar.
3. To assess the utility of the experimental burn box design at making tangible inferences from small-scale mesocosm studies to inform wider conservation management activities.

Hypotheses:

H_0: There will be no significant differences in the cover or richness across the treatment types

H_1: There will be significant differences in the cover and richness of vegetation after treatment by fire compared to glyphosate application and manual removal

H_0: Soil nutrients do not vary significantly across the three treatment types

H_2: Soil nutrients will be significantly affected by prescribed burning, yielding higher abundances of nutrients in the alvar soil

H_0: Prescribed burning did not achieve complete burns, soil and vegetation were not significantly affected.

H_3: Prescribed burning using the burn box is representative of natural prescribed fires in burn severity, time, and dosage

3.0 Methods

3.1 Study Site

The research was located on the North Bear Alvar (44° 39’ 39 N; 79° 11’ 56 W), a 314 ha parcel of alvar owned and managed by NCC and found within the Carden Plain, 27 km NE of Lake Simcoe (Figure 1). The site has a history of some unorganised small-scale and low-impact logging activities which ceased over 80 years ago, and unlike other alvars in the area, North Bear has not been subject to prolonged grazing by cattle (NCC Property Management Plan, 2013). As far as records show, North Bear Alvar hasn’t had any fire event in over 100 years (cf. pers comm. Rick McNamee, 2016; cf. pers comm. MNRF, 2015), and no burn scar evidence was found on the site[1]. North Bear remains mostly undisturbed by human activity and in relatively good condition in the sense that it has a low ratio of non-native compared to native species on the northern part of the property, though there is a higher proportion of non-native species on the southern part (NCC Property Management Plan, 2013). The alvar is surrounded by almost continuous natural cover of alvars, forest, and wetlands - with the exception of minor unmaintained roads and trail allowances abutting the property on the South and East boundaries.

North Bear has 11 ecologically defined communities including: wetland, open alvar, shrubland, and treed habitats (NCC, 2013) (Figure 1). The open alvar communities contain dominant species of *Poa compressa*, *Danthonia spicata*, *Deschampsia cespitosa*, *Packera paupercula*, *J. communis*, *Rhus typhina*, and *Asteracae* and *Solidago* sp.. Forested alvar communities are mainly dominated by *Thuja occidentais*, *Tilia americana*, *Tsuga canadensis*, *Ulmus americana*. The surrounding landscape is used by moose, black bear, white-tailed deer, a variety of other mammals, along with over 230 species of grassland and forest specialist bird species (Reid, 2011).

[1] In contrast, the Cameron Ranch Alvar and Blue Bird Ranch Alvar, located 5 km South, have had evidence of natural fire and prescribed burns, along with other areas on the Carden Plain (cf. pers comm. Rick McNamee, 2016; cf. pers comm. MNRF, 2015).

3.2 Experimental Design

3.2.1 Synthesis

I focused my research on open and shrubland alvar habitats on North Bear because NCC wanted to determine appropriate treatment options to manage and reduce encroachment of woody shrub

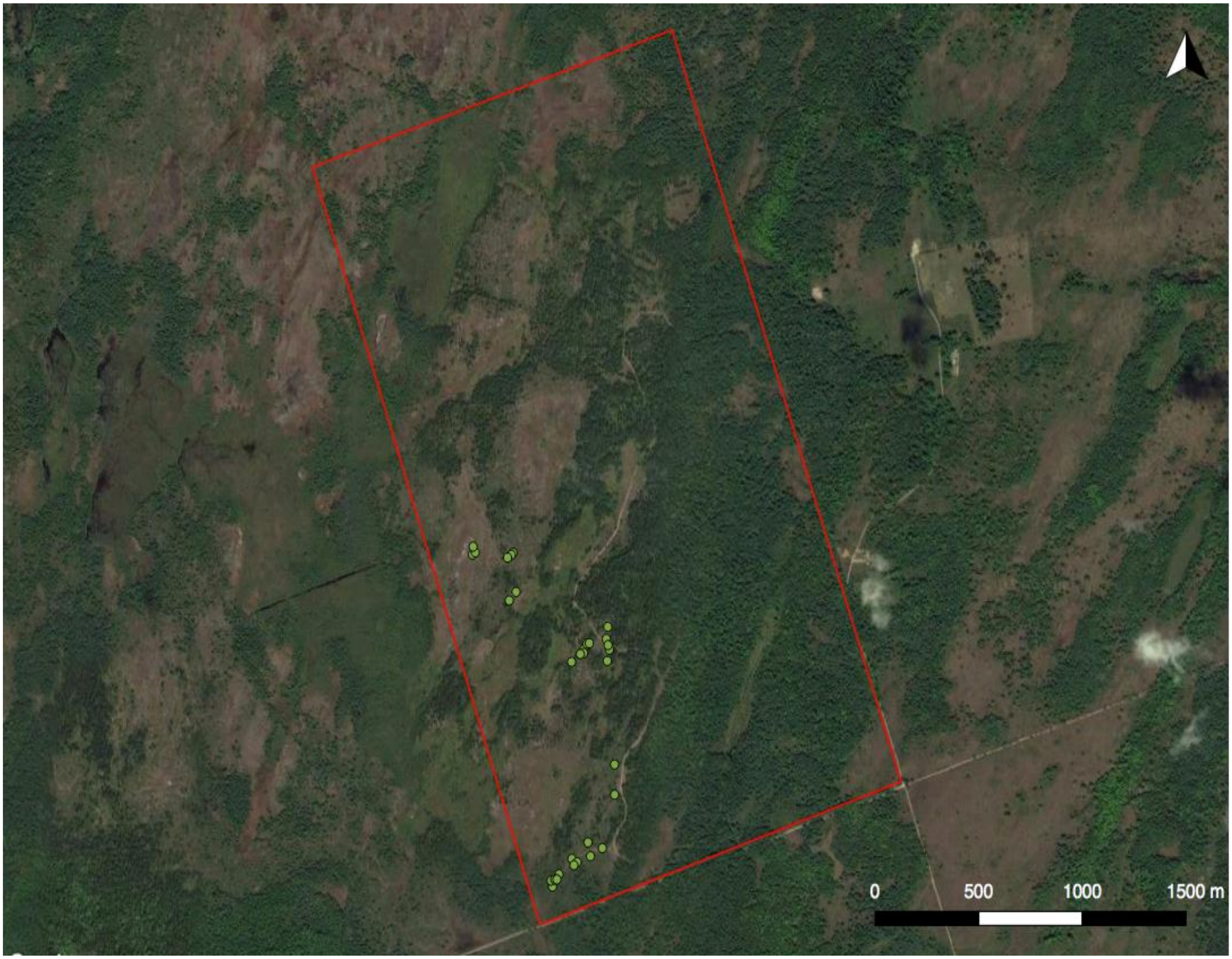


Figure 1. Landsat depicting property of interest - North Bear Alvar (red polygon) and research plots (green dots).

species into open alvars and increase native herbaceous species cover and richness. I used a before-after control-impact (BACI) (Smith, 2002) to test the effectiveness of each of three management treatments. The main treatment of interest was the use of 'contained' prescribed burns, i.e. a ‘burn box’, that does not allow for accidental spread of fire. For comparison, the other two treatments were glyphosate application and manual removal (by lopping). Glyphosate application was targeted at plots heavily encroached by *J. communis,* and manual lopping was targeted at both encroached and open plots, the latter of which to simulate a grazing disturbance. Percent cover and species richness of herbs and forbs were used to measure treatment success. I measured soil bulk density, nitrate-nitrogen (NO_3), phosphorus (P), and potassium (K) to determine how treatments affected alvar soils and, consequently, colonisation and regeneration of plant species. For the prescribed burn (burn box treatment), I also examined the duration, temperature, and dosage of the fires to determine if burn dynamics using the burn box approach align with natural prescribed burns.

3.2.2 Sample size calculation (number of plots)

To determine sample size (number of plots), I used a *priori* power analysis (G*Power Version 3.1.9.2) with a two-tailed test and a power of 90% and an effect size of 0.6; based on this, the sample size was 32 permanent plots (power = 0.911).

3.2.3 Plot locations

The location of these plots was based on use of stratified random samples, though it was further constrained by several factors. Locations were removed from selection if they exhibited any or all of (a) tree species over 3 m in height, (b) the largest percent cover measured was rock, (c) dominated by non-herbaceous species, (d) vulnerable to effects of having to use ATVs to carry equipment to locations (i.e. risk further damage to an environmentally sensitive and protected areas). The 32 plots were mainly located on the Southern portion of North Bear (Figure 1); this was one of the most degraded portions of the site with large patches of common juniper, and easily accessed with the ATVs and equipment without creating lasting damage to the alvar soils. While 1 m x 1 m plots are more common (e.g. Barker, 2001; Lamb, 2003), I used larger 2 m x 2 m plots. My rationale for this size was to allow my research to observe fire dynamics that would be more representative of the very patchy alvar and to allow more air circulation within the larger plot when using the burn box method (described in a subsequent section).

3.2.4 Plot marking and physical characterization

Once plots were selected, the boundaries were clearly flagged so that monitoring could continue over two seasons. A pink flag denoted the North corner of the plot, while three other orange flags oriented the plot against fixed cardinal directions. This was done to align four 1m x 1m quadrats within the plot to obtain four subplots within each plot for monitoring consistency. Depth to bedrock was measured using a metal depth probe. The average soil depth was an average of 7.4 cm (+/- 2.2 cm) to bedrock, validating the premise that all plots could be defined as alvars (see also Neufeld et al., 2012).

3.2.5 Allocation of treatments to plots

15 plots were designated as 'control'; the remaining 17 were allocated to one of the three management treatments (Figure 2). Prescribed burning by burn box treatment was considered the focus of the study and driven by NCC interest; I allocated nine plots to treatment by prescribed burning in order to test varying degrees of shrub encroachment on North Bear (open, moderately encroached, and encroached alvar situations) with n=3 replicates in each category. Constrained by logistics and needing to maximize samples for the primary treatment of interest to NCC – the burn boxes - I assigned four plots each to be treated by glyphosate and by manual lopping and removal.

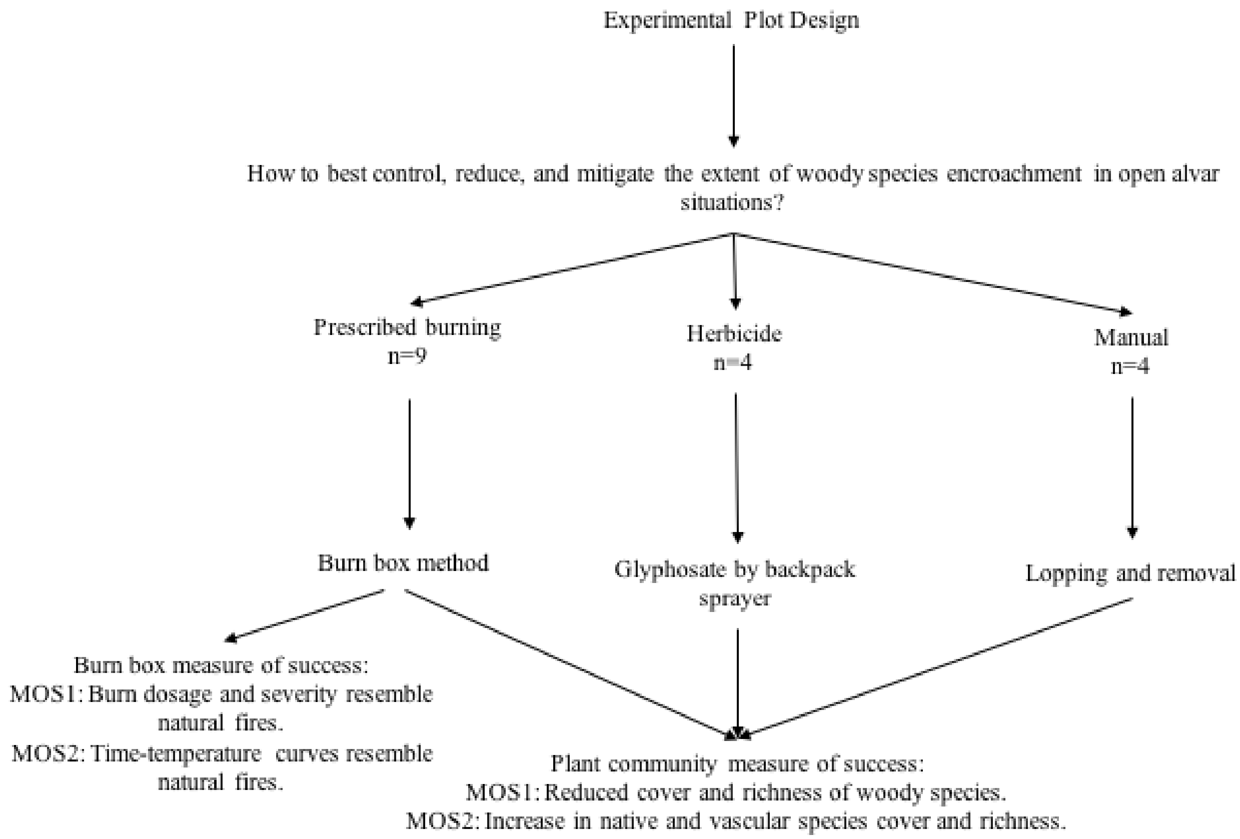


Figure 2 Experimental design with the three treatment types and the number (n) of plots assigned to that treatment. "MOS" = measure of success.

3.2.6 Implementing prescribed burning using burn boxes

I conducted nine replicate prescribed burns on June 9, 2016 using a "burn box" as adapted from Kral et al., (2015) and Sharrow & Wright (1977). The burn box in this particular study is a 2 m x 2 m x 1 m (LWH) square 11-gauge steel box comprised of eight individual 1 m x 1 m x 1 m panels with a total weight of 345 kg (Figure 3). The use of individual 1 m panels was for ease of transport on the back of an ATV, allowing these to be reassembled at spatially independent plots, and allowing the panels to be adjusted for the heterogeneity in plot topography. Each panel was framed with 2.5 cm aluminum square tubing on three sides to give the panels more structural integrity and a way to assemble and attach the panels in the field via

C-clamps (Figure 3). Each panel had five holes drilled to provide air flow for continued combustion. Panels were clamped and affixed together in-situ around vegetation.

Figure 3. An assembled burn box with two panels removed to show interior of the burn area. Perimeter of the box has a 1 m vegetation buffer to prevent ignition of non-target vegetation.

A Thermocouple Data Logger with a Type-K probe (Omega – Laval Quebec) with insulated wire to withstand temperatures up to 1000 °C. The probe was inserted at 8 cm above the soil surface to capture time-temperature profile curves. Fire and weather conditions were monitored during the duration of the burn with a Kestrel 4000 meter (Kestrel Meters – Minneapolis Minnesota). Cloud cover, air temperature, relative humidity, dewiness, and time of day were recorded.

Prescribed burns in the burn boxes were ignited using a gas drip torch with an 80:20 mixture of gas and diesel fuel. Average vegetation fuel loading was low (grass dominated)–medium (shrub dominated). Any woody material that exceeded the height of the burn box was lopped at 1 m to ensure vegetation outside the box were not ignited. No additions were made to the vegetation fuel loads; these were left naturally to conform to the management aims and goals established by NCC and the need to replicate a controlled burn with representative vegetation fuel loading. Burn times fluctuated between 2 m 55 s (grass dominated) and 7 m 54 s (shrub dominated). Once the panels had cooled post-fire, the panels were deconstructed within ten minutes of flame-out and moved to the next site for reassembly.

3.2.7 Implementing treatments using glyphosate or manual removal

I applied glyphosate [N-(phosphonomethyl)glycine)], under the brand name Roundup WeatherMAX® with Transorb 2 Technology (PCP# 27487, Class 4). Four plots were allocated for treatment by glyphosate, they all exhibited a high degree of woody percent cover with a mixed understorey of native and exotic herbs and forbs. I used a Monsanto backpack mounted sprayer at 15% concentration. During afternoon periods (no dew, low winds), glyphosate was applied generously along the stems and foliar areas of woody species and broadcast on herb layer species until all species on the plot had a coating of the herbicide, these plots were more dominated by woody species and contained an exotic-dominant herb layer. For comparison purposes with prescribed burning, where all biomass was consumed, all plants within glyphosate-treated plots were targeted for glyphosate.

For manual removal treatments, I used 1 m pruning loppers to cut down all above ground vegetation within the plot to 2 cm above ground. Lopping was conducted on the morning of June 10, 2016. Again, all vegetation within the plots was targeted, thus providing the removal mechanism for unwanted woody species and simulating a grazing disturbance on forbs. These were removed and discarded nearby in a pile to reduce unintentional seed transport throughout the alvar.

3.2.8 Sampling and measuring plant and soil variables to test impacts of all three treatments

For all treatments, percent cover, abundance and richness were collected. In each 1 m^2 quadrat, percent cover was determined by measuring of the spatial extent of the individual species (where 10 cm x 10 cm in each quadrat = 1% cover). Total species richness was measured by counting the number of different species present within each plot. A subset of species, ‘species of interest’, were analysed for abundance by counting the number of stems belonging to species within each quadrat, relative abundance of species of interest was calculated by dividing the number of individuals belonging to a species by the total number of individuals on the plot; relative abundance data for herbs and woody species is presented and described in the results. A total of eight problematic woody species and 10 native herbaceous species were chosen because they represent good indicators of open alvar habitat as determined by other monitoring projects on Carden Alvar (see Couchiching Conservancy and Ontario Parks: Cameron Ranch Monitoring Data 2015), and in some cases, have high confinement to open alvars (Brownwell & Riley, 2000; Reschke & Jones, 2005), additionally they were present in before and after conditions over the treatment categories.

Sampling was done for all four quadrats in each plot and composited together for each plot, with 32 total plots sampled. Species were identified in field and/or comparing non-plot specimens with herbarium specimens at the University of Waterloo. I categorized these plant data as belonging to one of three life forms (1) native herbaceous species, (2) exotic herbaceous species, or (3) woody species (native or exotic, combined). The first two categories are likely intuitive to readers; they were used because the goal was to increase native herbaceous species and decrease exotic herbaceous species. The ‘woody’ category combined native and exotic species because the NCC’s management goal for this alvar was to restore and maintain open alvar communities with less than 25% tree and shrub cover and was indiscriminate about types of woody material.

Soil samples for macronutrient analysis were collected and composited from two random locations in each of the 2 m x 2 m plots. Samples were collected using an 88.5 cm^3 hole saw; as this has a relatively small diameter and shallow penetration depth needed for the shallow and rocky nature of alvar substrates. The saw was hammered into the soil until the top of the hole saw was flush with the soil surface yet avoiding compression of the soil or soil migration to/from the ring. Samples were collected once in May 2016 and August 2016. I did not collect samples in 2017 because I anticipated there would be no exacerbation of any initial significant effects of treatments after one year. Weathering, frost heaving, and precipitation runoff of lithic udorthent alvar soils normally overwhelm all other factors in a short (two year) research time period (see Stark, Lundholm, & Larson, 2004).

I used a portion of the soil samples to test how treatments affected nitrate-nitrogen (NO_3), phosphorus (P), and potassium (K). The composited samples from the field were stored in sealed labelled Ziploc bags in a cooler while in the field for two days at ~10 °C. The samples were returned to the University of Waterloo labs and refrigerated for 14 days at 4 °C. Nutrients were extracted using an acid extracting solution and processed with the LaMotte Smart3 Colorimeter ® (LaMotte Company, 2001). The LaMotte Colorimeter uses Cadmium Reduction for nitrogen-nitrates, Ascorbic Acid Reduction for phosphates and Tetraphenyl Borate for potassium (LaMotte Company, 2001).

3.2.9 Statistical analysis for plant community and soil data

I used ANOVAs to test the effects of treatments on percent cover, species richness, and soil macronutrients. I used data from June 2016 (prior to treatment) and 2017 (1-year following) for plant community data, and June 2016 (prior to treatment) and July 2016 (1-month after treatment) for soil data. I tested these data for normality and homoscedasticity using an

Anderson-Darling test and Levene's method respectively. If data violated the normality assumption ($p > 0.05$), they were log-transformed and tested again for normality. If this did not yield normalized data, I still used univariate ANOVAs (Minitab Express 1.5.1) because tests with N ~ 30 are not affected by this violation (McDonald, 2014). If data were not homoscedastic ($p < 0.05$), a Welch's ANOVA was used (Minitab 18.1.1). Post-hoc Tukey Tests Honest Significant Difference (HSD) or Games-Howell Pairwise Comparisons were used to separate treatment effects to determine where the significant difference existed.

3.2.9.1 Measuring and testing impacts of variables specific to burn box treatments

Beyond the common variables and statistical analyses outlined for all three treatments above, I also evaluated burn box time-temperature profiles, burn dosage, and severity to test if this method was consistent with expectations from unconstrained fires. Fires were spatially mapped and dominant cover of vegetation was recorded to a resolution of 10 cm x 10 cm prior to burning on each 2 m^2 burn plot and fitted it to a grid matrix using Microsoft Excel (version 15.32 for Mac). This resulted in a matrix of 20 x 20 (400 unit cells total) grid units for each burn allocated plot where vegetation was assigned codes corresponding to the life form of the plant, where G=graminoids, H=herbaceous, S=shrub, T=tree, and 0=bare/exposed soil, such that each grid cell contained the dominant cover plant. Burn severity was qualitatively ranked using colour shading after the burns using a burn severity index by Ryan (2002) and overlaid on top of the cells containing the spatial arrangement of vegetation.

Green shading indicated low-severity burns (plant parts somewhat green or moderately scorched, surface litter, mosses and some herbs charred or consumed), yellow shading indicating medium-severity (all understorey plants charred or consumed, fine dead twigs on soil surface consumed, and tall shrubs or trees exhibit some canopy combustion), and red shading for high-severity (entire shrub consumed with deep charred woody material remaining, all herbaceous species consumed, surface litter of all sizes largely consumed, and a white ash deposition left behind). The burn severity index was overlaid on the existing fuel arrangement to yield a depiction of fuel loads/vegetation arrangement and the performance of the fire within the confines of the burn box.

Data from thermocouples were analysed for significant differences in temperature across the open, moderately encroached, and encroached alvar conditions tested to determine which alvar state produced more natural burn effects (i.e. uninhibited by burn box containment, reaching a desired temperature and for a desired duration). The time-temperature values were tested for equal variance and normality; they were found to be heteroscedastic so a Welch's ANOVA was performed. Temperature (°C) was used as the response variable and alvar condition (open alvar, moderately encroached, and encroached) were used as the fixed factor levels. When significant differences were found, a post-hoc Games-Howell Pairwise Comparison test was performed to determine where the differences existed.

4.0 Results

4.1 Effect of treatments on percent cover of plants

The percent cover of woody and native herbaceous species was significantly affected by treatment (Table 1). Relative to control plots, the percent cover of woody species was significantly reduced by glyphosate treatment (Tukey's HSD = 0.004), fire (Tukey's HSD = 0.02) and by manual lopping (Tukey's HSD = 0.02) (Table 2).

*Table 1 ANOVAs testing effects of treatment type on percent cover of each of the species grouped as native-herbaceous, exotic-herbaceous, and woody. Results are from one-way ANOVAs (native-herbaceous and woody) and Welch's ANOVA (exotic-herbaceous). A * indicates a significant difference.*

Category of Plant	MS	F	P
Native Herbaceous sp.	$MS_{Treatment\ (df\ =\ 3)}$: 0.907 $MS_{Error\ (df\ =\ 28)}$: 1.0	0.9	0.04*
Exotic Herbaceous sp. (Welch's)	- -	1.51	0.28
Woody sp.	$MS_{Treatment\ (df\ =\ 3)}$: 4.29 $MS_{Error\ (df\ =\ 28)}$: 0.648	6.63	0.002*

*Table 2 Outcome of the one-way ANOVA Tukey Test HSD performed on percent cover of woody species. A * denotes a significant difference.*

Comparisons of Treatment	Difference of Means	SE Difference	t	Adjusted P
Fire-Control	-1.051	0.339	-3.1	0.02*
Herbicide-Control	-1.725	0.453	-3.81	0.004*
Manual-Control	-0.992	0.453	-2.19	0.02*
Herbicide-Fire	-0.674	0.484	-1.39	0.514
Manual-Fire	0.059	0.484	0.12	0.999
Manual-Herbicide	0.733	0.569	1.29	0.578

Fire caused declines in percent cover of mainly targeted invasive woody species: *J. communis*, *R. aromatica*, and *P. virginiana*. All woody species declined to an average of 4% on plots treated by herbicide and manual removal, though some minor re-sprouting was evidenced by *Symphoricarpos albus*, *R. aromatica*, *P. virginiana*, and *Amelanchier spicata* and rose to 11% cover on plots by the end of the monitoring cycle.

Relative to control plots, prescribed burns significantly increased the percent cover of native herbaceous species by 33% (+/- 4.2%) (Tukey's HSD = (0.03) (Table 3 and Figure 4).

*Table 3 Outcome of the one-way ANOVA Tukey HSD performed on percent cover of native herbaceous species. A * denotes a significant difference.*

Comparisons of Treatment	Difference of Means	SE Difference	t	Adjusted P
Fire-Control	0.87	0.42	-1.13	0.03*
Herbicide-Control	-0.83	0.57	-1.46	0.48
Manual-Control	-0.31	0.57	-0.55	0.98
Herbicide-Fire	-0.35	0.6	-0.57	0.94
Manual-Fire	0.17	0.6	0.28	0.99
Manual-Herbicide	0.52	0.71	0.73	0.89

Prescribed burns increased the following native herbaceous species percent cover: *Danthoria spicata*, *Carex richardsonii*, *Deschampsia cespitosa*, *Solidago ptarmicoides*, *Packera paupercula*, various asters and other grass species. Other treatments did not produce significant

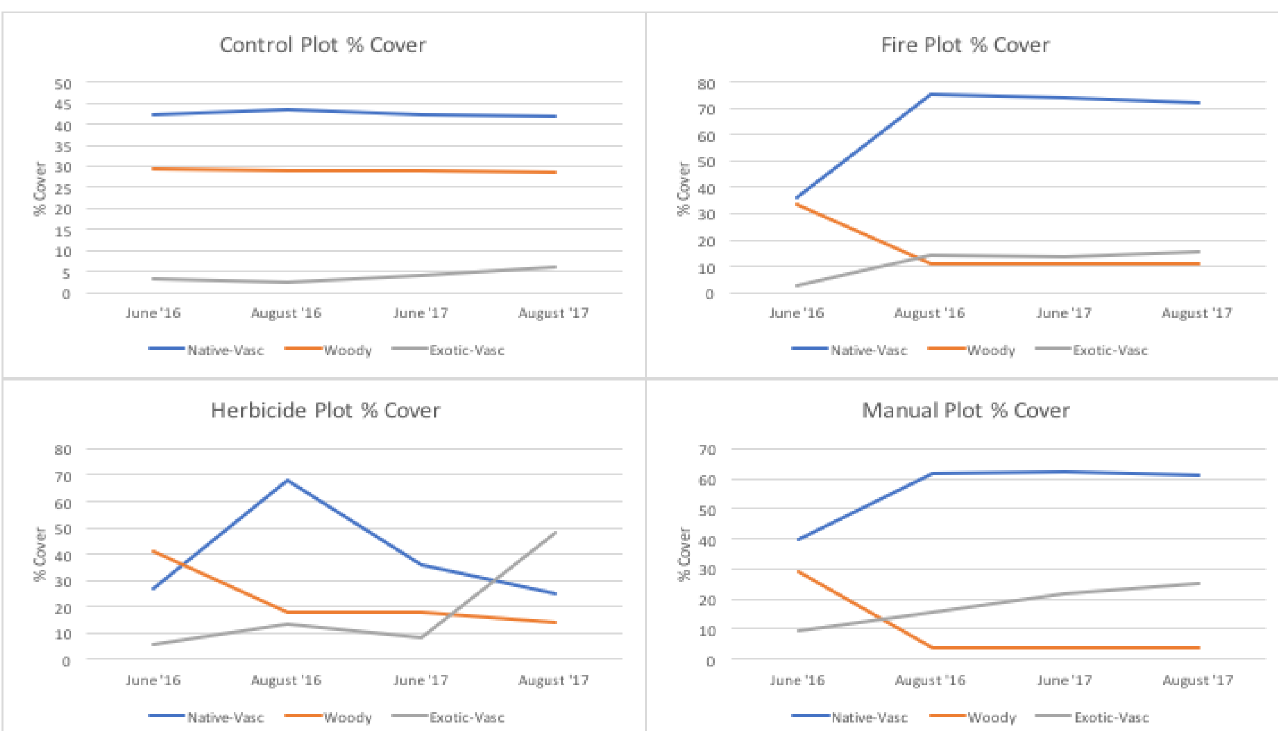


Figure 4 Plot-aggregated time-series graphs depicting pooled mean species cover per treatment type categorised by native herbaceous, exotic herbaceous, and woody species

differences in the percent cover of native herbaceous species compared to control plots, this is mainly attributed to the application of glyphosate to target woody species and not the herbaceous layer. The same can be seen in exotic herbaceous species which did not differ significantly (p=0.28) between treatment and control plots (Table 3 and Figure 4).

4.2 Effect of treatments on total species richness

A total of 96 species of plant were observed on the study plots after treatment, of those, 57 were native herbaceous, 21 exotic herbaceous, and 18 woody species (Table 13). There was a significant difference in the richness of native herbaceous and woody species (Table 4).

*Table 4 Outcome of the Analysis of Variance for species richness across the four treatment types, showing degrees of freedom, mean square, F and P values. A * denotes a mean with a significant difference in the analysis.*

Category of Plant	MS	F	P-Value
Native Herbaceous sp.	$MS_{Treatment\ (df = 3)}$: 3.64 $MS_{Error\ (df = 28)}$: 0.71	5.08	0.006*
Exotic Herbaceous sp.	$MS_{Treatment\ (df = 3)}$: 1.37 $MS_{Error\ (df = 28)}$: 0.96	1.43	0.26
Woody sp.	$MS_{Treatment\ (df = 3)}$: 3.08 $MS_{Error\ (df = 28)}$: 0.77	3.98	0.02*

*Table 5 Outcome of the one-way ANOVA Tukey Test HSD performed on native herbaceous species richness. A * denotes a significant difference.*

Comparisons of Treatment	Difference of Means	SE Difference	t	Adjusted P
Fire-Control	0.35	0.35	0.97	0.77
Herbicide-Control	-1.58	0.48	-3.31	0.02*
Manual-Control	0.16	0.48	0.33	0.99
Herbicide-Fire	-1.92	0.51	-3.78	0.004*
Manual-Fire	-0.19	0.51	-0.37	0.98
Manual-Herbicide	1.74	0.6	2.9	0.03*

Relative to control plots, native herbaceous richness was significantly decreased on plots treated with glyphosate by 46% (Tukey's HSD = 0.02) (Table 5). Plots treated with fire were similar in their native herbaceous species richness to control plots (Tukey's HSD = 0.77) (Table

5 and 7), though mean species richness on fire plots increased 67% from before fire conditions (Table 5 and 7). Relative to control plots, woody species richness was significantly reduced by 17% in plots treated with glyphosate (Tukey's HSD = 0.04) (Table 6 and Table 7), and woody

*Table 6 Outcome of the one-way ANOVA Tukey Test HSD performed on woody species richness. A * denotes a significant difference.*

Comparisons of Treatment	Difference of Means	SE Difference	t	Adjusted P
Fire-Control	0.35	0.372	0.94	0.77
Herbicide-Control	-1.403	0.496	-2.83	0.04*
Manual-Control	-0.464	0.496	-0.93	0.79
Herbicide-Fire	-1.754	0.529	-3.31	0.01*
Manual-Fire	-0.814	0.529	-1.54	0.43
Manual-Herbicide	0.948	0.623	1.51	0.45

Table 7 Mean species richness by vegetation classification before and after treatment. Standard Error (SE) of the mean is presented is parentheses.

	Native Herbaceous		**Exotic Herbaceous**		**Woody**	
Treatment	June 2016	June 2017	June 2016	June 2017	June 2016	June 2017
Control	9.9 (0.8)	10.1 (0.8)	2 (0.4)	1.6 (0.4)	1.8 (0.4)	1.7 (0.5)
Fire	6.8 (1.3)	11.4 (1.2)	1.4 (0.2)	2.2 (0.5)	2.2 (0.4)	2.5 (0.6)
Glyphosate	8.3 (1.4)	4.5 (1.5)	2.25 (0.5)	0.9 (0.4)	3.5 (0.7)	1.5 (0.5)
Manual	7.3 (0.7)	11 (1.5)	2 (0.6)	1.5 (0.8)	2.8 (0.7)	2 (1.1)

Weighted means were used for species across the four treatment types to correct for unequal sample sizes

species richness decreased 57% as a result of glyphosate within the glyphosate treatment category. Conversely, mean woody richness slightly increased by 12% in plots treated by fire using the burn box method (Tukey's HSD = 0.43) and decreased by 40% through manual removal (Tukey's HSD = 0.79). Increases in woody species richness were largely due to re-sprouting of individuals (particularly *Symphoricarpos albus* and *Rhus aromatica*).

There were no significant differences (p=0.26) in richness of exotic herbaceous species across the four treatment types. Manual removal by lopping and trimming all herbaceous plants down to the ground to simulate a grazing disturbance increased native herbaceous species richness by 34% and decreased richness across the other two plant groups (Table 7).

4.3 Effects of treatment on species of interest

Treatment by fire generally decreased woody species abundance, although, *A, spicata, P. virginiana*, and *Rhus sp.* sprouted and spread vegetatively in the year following prescribed burning (Table 8), though no significant effects were evidenced among woody species

Table 8 Outcome of ANOVA for herbaceous and woody species of interest

Plant Form	N	dF	Mean	F	P
Herbaceous[1]	32	3	40	12.51	<0.001
Woody[2]	32	3	11.79	1.05	0.408

1: Performed using a one-way ANOVA 2: Performed using a Welch's ANOVA due to non-equal variances for woody species

of interest following treatment [F(3,31)=1.05, p=0.408]. While woody species abundance was reduced by fire, herbaceous species of interest including *C. richardsonii, D. spicata*, D. *cespitosa*, and herbaceous species of *G. trifolium, P. paupercula* all significantly increased in post-fire conditions compared to before treatment [F(3,31)=12.51, p<0.001]. As an annual hemiparasitic plant, *C. ccoccinea* was widely noted on the landscape and in research plots in 2016, though conditions in 2017 were unfavourable for its development and was not evidenced on any plots, on North Bear Alvar, and rarely on surrounding parcels of alvar in the Carden Plain.

Treatment by glyphosate largely reduced all woody species of interest abundance (Table 9), *P. virginiana* was quick to send out shoots outside of study plots which influenced abundance on plots following treatment. *J. communis* was immediately impacted by glyphosate and while it was reduced in cover, some individuals did persist after treatment. *S. albus* took advantage of opened conditions and quickly regenerated and colonised plots treated by glyphosate. Any native herbaceous species present on plots were immediately impacted by treatment and glyphosate did not facilitate adequate growing conditions post treatment for specially adapted alvar species.

Treatment by manual removal and lopping stimulated some advantageous woody species to colonise sites quickly. Treatment by lopping either increased abundance or it remained the same for woody species, only *D. fruticosa*, *R. typhina*., and *T. occidentallis* were completely removed following lopping treatment (Table 9). Native herbaceous species generally decreased though were not inhibited by treatment as was the case with glyphosate; they re-sprouted in near-similar abundance from before treatment. *B. inermis*, *C. crawei*, and *D. cespitosa* all increased in abundance following treatment by lopping (Table 9).

Table 9 A comparison of relative abundance of species of interest with levels of confinement derived from Catling (1995) and Catling and Brownell (1999) Before treatment is represented by June 2016, and after treatment by 2017. Levels of confinement: Extreme (E) with 86-100% of occurrences in the Great Lakes region on alvars; High (H) with 71-85%; moderate (M) with 50-70%; and not ranked (NR) for taxa that are in high abundance on North Bear Alvar. Species obtained from Ontario Parks and CC inventory.

		Control		**Fire**		**Glyphosate**		**Manual Removal**	
	Scientific Name	June 2016	June 2017	June 2016	June 2017	June 2016	June 2017	June 2016	June 2017
Conf.	Shrubs/Trees								
M	*Amelanchier spicata*	0.002	0.002	0.003	0	0.02	0	0	0.004
M	*Juniperus communis*	0.002	0.003	0.02	0.001	0.008	0.008	0.001	0.004
NR	*Prunus virginiana*	0.0009	0.0009	0.001	0.002	0.01	0.02	0.004	0.004
M	*Rhus aromatica*	0.003	0.003	0.004	0.03	0.009	0.004	0.005	0.01
NR	*Rhus typhina*	0	0	0.0008	0.001	0	0	0.0007	0
M	*Dasiphora fruticosa*	0.0003	0.002	0.0002	0.002	0	0	0	0
NR	*Symphoricarpos albus*	0.002	0.002	0.004	0.004	0	0.004	0.001	0.02
NR	*Thuja occidentalis*	0.002	0.002	0.007	0	0.0009	0	0.002	0
	Herbs								
H	*Bromus inermis*	0.003	0.003	0.007	0.08	0	0	0	0.001
E	*Carex crawei*	0.04	0.07	0.07	0.09	0.01	0	0.006	0.03
E	*Carex richardsonii*	0.06	0.11	0.04	0.10	0.005	0	0	0.02
M	*Castilleja coccinea*	0.09	0	0.05	0.11	0	0	0.001	0
H	*Danthonia spicata*	0.11	0.12	0.14	0.17	0.05	0.02	0.19	0.14
H	*Deschampsia cespitosa*	0.02	0.006	0.01	0.13	0.26	0	0.002	0.01
E	*Geum trifolium*	0.007	0.013	0.02	0.14	0.008	0.01	0	0
NR	*Packera paupercula*	0.02	0.07	0.06	0.15	0.0006	0	0.08	0.07
H	*Scutellaria parvula*	0.001	0.001	0.001	0.001	0.01	0	0	0
E	*Solidago ptarmicoides*	0.02	0.05	0.02	0.17	0.02	0	0.03	0.02
E	*Sporobolus heteroplepis*	0	0.004	0.001	0.018	0.001	0	0.002	0.003

4.4 Effects of treatment on soil macronutrients

There were only significant differences found in nitrate-nitrogen (p=0.03) (Table 10) among the treatment types. Relative to control plots, there were significant increases in nitrate-nitrogen

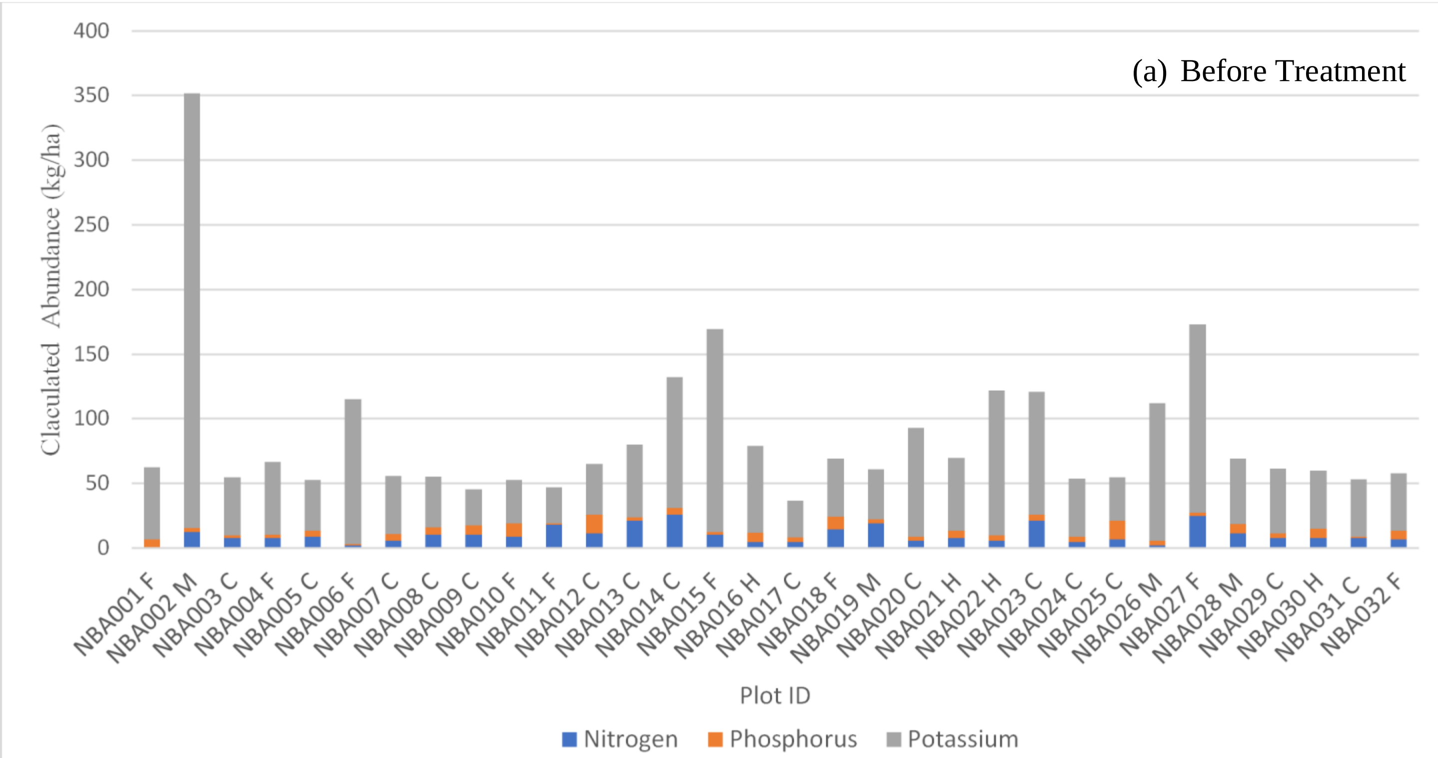
*Table 10 Outcome of the Analysis of Variance for soil macronutrients across the four treatment types, showing degrees of freedom, mean square, F and P * denotes a mean with a significant difference in the analysis.*

Nutrient	MS	F	P
Nitrate-Nitrogen	$MS_{Treatment\ (df\ =\ 3)}$: 3184.38 $MS_{Error\ (df\ =\ 28)}$: 929.37	3.43	0.03*
Phosphorus	$MS_{Treatment\ (df\ =\ 3)}$: 82.57 $MS_{Error\ (df\ =\ 28)}$: 82.26	1	0.41
Potassium	$MS_{Treatment\ (df\ =\ 3)}$: 1875.47 $MS_{Error\ (df\ =\ 28)}$: 930.78	2.01	0.14

*Table 11 Outcome of the one-way ANOVA Tukey Test HSD performed on nitrate-nitrogen. A * denotes a significant difference.*

Comparisons of Treatment	Difference of Means	SE Difference	t	Adjusted P
Fire-Control	1.04	0.38	2.73	0.04*
Herbicide-Control	0.84	0.51	1.66	0.37
Manual-Control	-0.22	0.51	-0.44	0.97
Herbicide-Fire	-0.20	0.54	-0.37	0.98
Manual-Fire	-1.26	0.54	-2.32	-0.12
Manual-Herbicide	-1.06	0.64	-1.66	0.36

content on plots treated with fire (Tukey's HSD =0.049) (Table 11 and Figure 5). The other comparisons of treatment to control did not produce any significant effects for nitrate-nitrogen analysis. Phosphorus and potassium nutrient content were not significantly affected by treatment type (Table 11 and Figure 5).



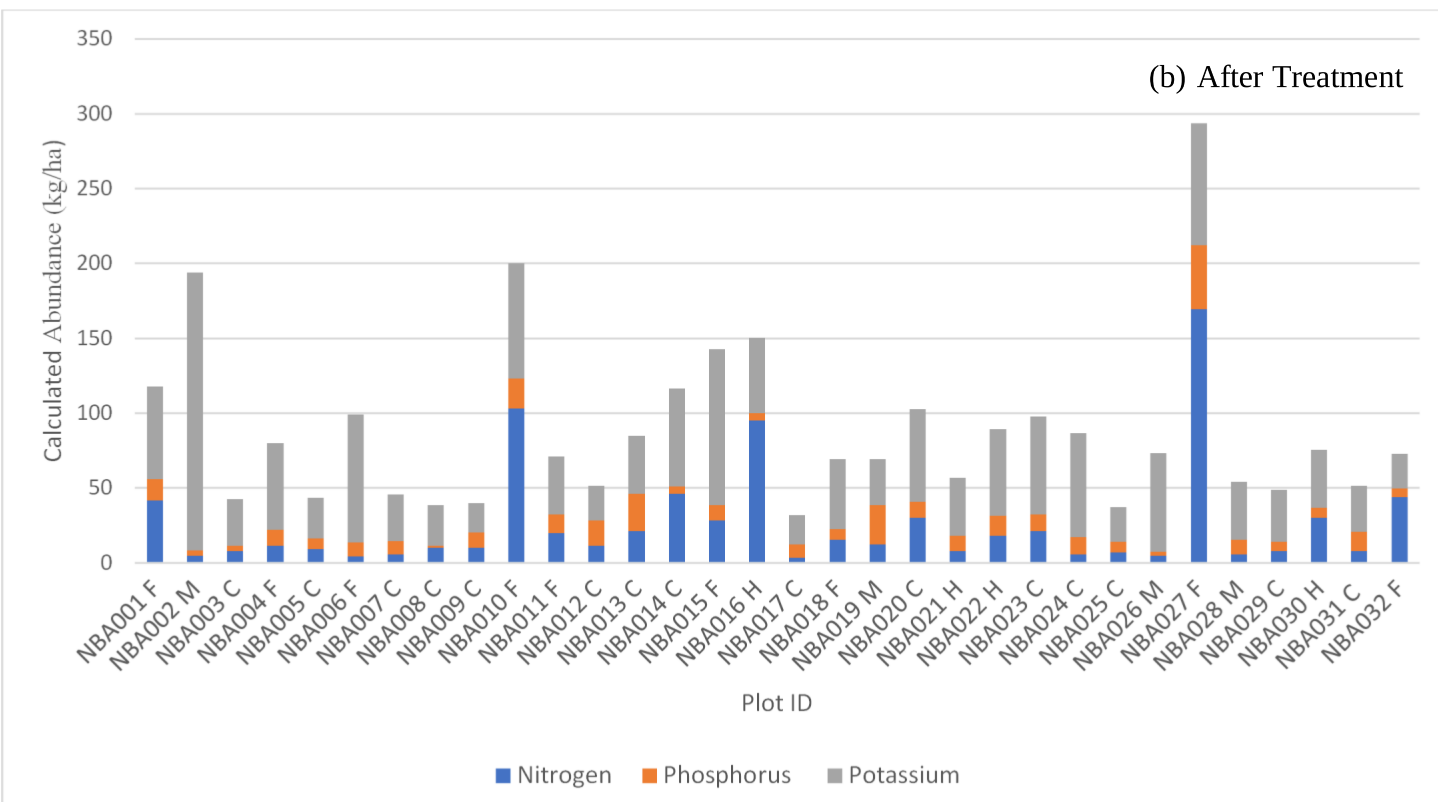


Figure 5 Outcome of macronutrient content analysis per plot (a) before and (b) after treatment. Letters beside the plot ID along the x-axis represent treatment assigned to that plot; where C=Control, F=Fire, H=Herbicide, M=Manual

4.5 Variables unique to the burn box treatments

There were significant differences on burn temperature and time [$F_{301.32}$=39.92, p<0.0001] between the vegetation categories. A post-hoc Games-Howell Multiple comparisons found that burn temperature was significantly lower and burn dosage shorter on open alvar conditions compared to moderately encroached and encroached (Table 12 and Figure 6). Encroached alvar conditions had the second longest burn dosage and highest mean burn temperature of 214.1 °C (Table 12 and Figure 6).

Table 12 The outcome of the univariate Welch's ANOVA and Games-Howell Pairwise Comparisons on burn temperature and the condition of alvar (open, moderately encroached, and encroached). Means that do not share a letter are significantly different.

Alvar Condition	N	Mean	StDev	95% CI	Games-Howell Pairwise Comparison
Encroached	47	214.1	152.6	(173.16, 206.32)	A
Moderately Encroached	53	189.74	112.08	(191.3, 236.9)	A
Open	25	120.92	57.91	(109.82, 132.01)	B

"N" represents the number of temperature readings over 5-second intervals

The burn box performed best when a mixture of herbaceous groundcover and around 50% woody cover was inside the box allowing for more complete combustion and consumption of plant material (Figure 8, 9, 10). Fire dynamics inside the box were variable across the three alvar conditions, with open alvar situations, dominated by herbaceous species, having the lowest mean temperature (Figure 6).

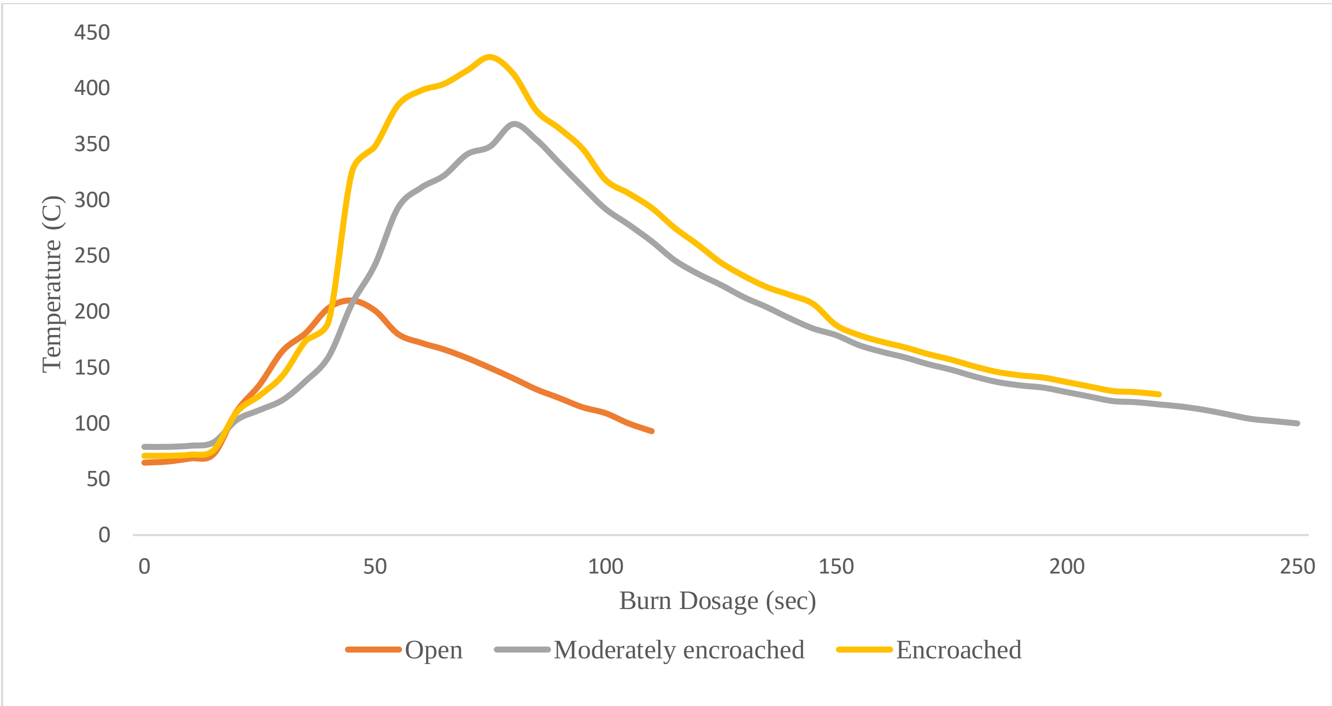


Figure 6 The mean time-temperature profile curves from the prescribed burns using the burn box. Each curve is an aggregation of three curves within the alvar condition that was tested.

The mapping of the burn severity within the box found that burns were variable across vegetation categories (Figure 6 & 7). In the matrices representing open alvar conditions (Figure 7.1, 7.2, 7.3) where there is a defined "less than 25% tree and shrub cover, dominated by forbs and herbaceous species" (Catling, 2016), the mean average temperature for open alvar burns was 135 °C with a peak maximum temperature of 210 °C. These burns did not meet the targeted time-temperature profiles characteristic of grasslands (see Archibald et al., 1998; Vermeire and Roth, 2013; and Strong et al, 2013). Plant mortality was between 9.75-56.5% due to the fuel moisture levels; burn times in the open alvar situations were on average 3 minutes (Figure 8).

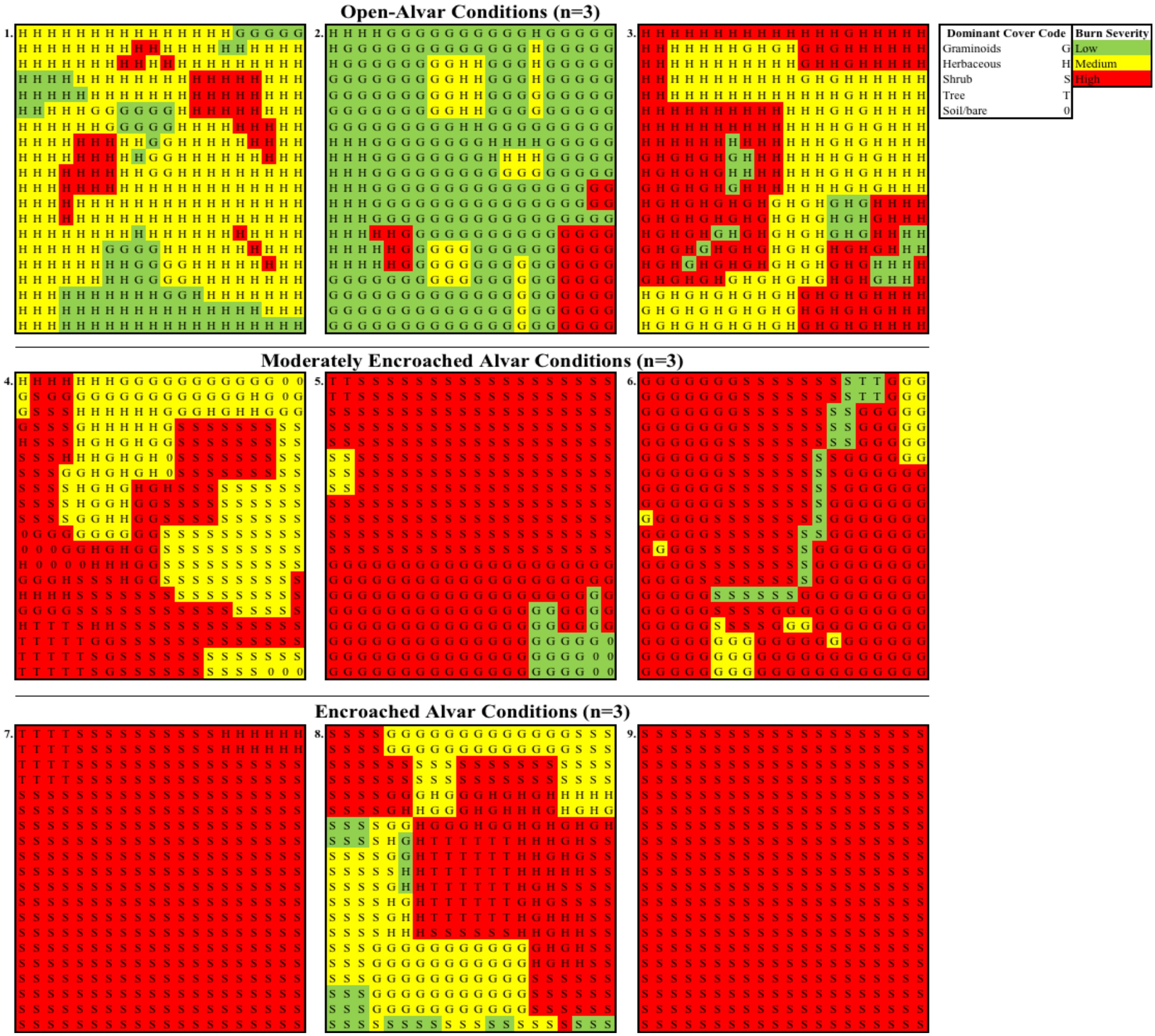


Figure 7 Fuel load arrangement and burn severity mapping across the nine-prescribed burn treated plots. Green shading indicates low severity burn (plant parts somewhat green or moderately scorched, surface litter, mosses and some herbs charred or consumed), yellow shading indicating medium severity (all understorey plants charred or consumed, fine dead twigs on soil surface consumed, and tall shrubs or trees exhibit some canopy combustion), and red shading for high severity (entire shrub consumed with deep charred woody material remaining, all herbaceous species consumed, surface litter of all sizes largely consumed, and a white ash deposition left behind). Severity index adapted from Ryan (2002).

In moderately encroached communities (Figure 7.4, 7.5, 7.6) with over 25% shrub and tree cover, the mean temperature was 186 °C with a maximum temperature of 268 °C. Generally, the fuels were drier and had greater carrying capacity to sustain fire and combust vegetation with plant mortality between 50-93% and a burn time on average of 4 minutes (Figure 9).

The shrub-dominated encroached plots (Figures 7.7, 7.8, 7.9) had the greatest mean of 224 °C and a maximum peak temperature of 428 °C and a burn duration of 7:28 sec. Plant mortality in these shrub-dominated situations were between 42.24-100% and left deep charring, a white ash deposition, and consumed most to all of the biomass (Figure 10).

5.0 Discussion

5.1 Prescribed Burning

As was hypothesized, prescribed burning resulted in more desirable conditions for plant regeneration and assembly, and was sufficient to reduce woody species encroachment in the first year. This was evidenced in the space clearing properties fire exhibited over treatment by herbicide, but also the change in nutrient content resulting from post-fire combustion of biomass which likely contributed to facilitating more desirable growing conditions for native herbaceous vegetation than those treated by herbicide or manual removal. Changes in nutrient content were consistent with those in similar studies by Sharrow and Wright (1977), where charred material from burning increased soil nitrate-nitrogen content in the upper soil layers (0-5 cm).

5.1.2 Effect on Plant Community

There were statistically significant changes in the percent cover of both woody (p=0.02) and native herbaceous (p=0.03) species. Woody plant species exhibited an overall decline in cover, they were mainly combusted and foliar areas that escaped fire died. Some regeneration was evident, particularly on *P. virginiana* and *R. aromatica* (Figure 9, 10). *P. virginiana* is well adapted to disturbance by fire, and most studies report an increase in stem numbers following (Meyer & Witmer, 1998; Johnson, 2000; Young, 1983). While the species is subject to top-kill by fire, fire stimulates prolific regrowth from underground rhizomes and surviving root structures. Post fire regeneration may also be attributed to seed drop or dispersal by birds (Volland & Dell, 1981). In my experiment, *P. virginiana* had the greatest percent cover post-fire (an average of 3% across treated plots). Similarly, while existing stems of *R. aromatica* were subject to top-kill by fire, disturbance by fire stimulated vigorous re-sprouting from rhizomes and roots in the year after treatment. Generally, an increase in *R. aromatica* is found in years following a fire due to increased light penetration and less light competition, the species relies on recruitment and establishment from seedlings following a fire (Taylor, 2004). Recruitment sources are prevalent on the site, the closest *P. virginiana* or *R. aromatica* were within one to three meters away on average and may influence the trajectory of the specific plot following growing seasons as a result of seed drop or by asexual regeneration in the case of *R. aromatica* (Taylor, 2004). It would be common then to assume that these species would be present in the early succession of fire disturbed alvar habitat and are not necessarily a cause for concern, but rather the natural succession of vegetation following a fire disturbance.

The main woody species of interest, *J. communis* was killed or seriously damaged by the fires (Figure 10). All basal and most crown areas were completely charred and did not exhibit any growth in follow-up monitoring. No sprouting was found after burning, this is common with most *J. communis* studies (e.g. Tirmenstein, 1999) following a fire disturbance. Some regrowth can take place if some basal branches remain alive, which may occur in early-season fires or where individual arrangement is patchy (Tirmenstein, 1999). Some basal areas in my experiment did escape fire when using the burn box device. This factor may be mitigated by prescribed burning without a containment device. Regeneration and recovery of *J. communis* in post-fire conditions is generally facilitated through seed dispersal by birds or mammals (Diotte & Bergeron, 1989; Tirmenstein, 1999). The nearest *J. communis* was on average 1 meter away as some plots were burned in thickets to mimic impacts of medium-old growth effects on plant community regeneration and fire dynamics. So these sites will likely be influenced by seed drop or vegetative spread in subsequent growing seasons (Collins & Calabrese, 2012). Some seeds may have persisted in the soil, though this was not determined by this study, albeit *J. communis* seeds are not stimulated by fire (Mallik & Gimingham, 1996), and prescribed burning of these patches should allow for other species to assemble and regenerate unhindered by competition of juniper scrub. Given the prolific extent of *J. communis* on the Carden landscape, it is likely that sites managed against *J. communis* encroachment will likely encounter the shrub in early years following disturbance events given some seed survival and dispersal mechanisms (Limb, Engle, Alford, & Hellgren, 2014; Romme et al., 2009). Post-treatment vegetation composition and structure trajectories can be highly variable following the removal or reduction of juniper, and postfire increases may be seen within 20 years, depending on seedling recruitment, and dominance of other species in the area (Ansley, Wiedemann, Castellano, & Slosser, 2006; Ansley & Rasmussen, 2005).

Woody species richness increased by 14% compared to pre-fire levels, though I emphasize that this was not statistically significant (p=0.708). These increases in richness were mainly due to the recruitment and colonisation of a few opportunistic species (*P. virginiana* and *R. aromatica*) which assembled on plots following treatment, noting the previously discussed mechanisms for regeneration. Whether these species increased from germination or re-sprouting from roots and rhizomes or from seedling contributions from nearby individuals was not evaluated in this study. *J. communis* did not contribute to any of the increases in richness or cover observed and fire had a 95% crown kill efficacy following fire, and was further reduced and killed by fire treatment. Overall, the growing season burn conducted in June did well in removing most living woody species; cover decreased from 34% down to 4% cover on average immediately following burn box treatment. While woody species cover did increase to 11% (p=0.02) by the end of the following growing season, it is still a significant reduction of living woody biomass and contributed to facilitating more light penetration for an increase in native species (discussed later in this section). It is worth noting when prescribed burning was conducted, there were still influences from the wet spring and morning dew on foliar areas and the ground in the plots and this could have contributed to the uneven burning of woody material on some of the designated burn plots. Woody fuel moisture was also higher than would be in the dormant season (Limb et al., 2011). It is likely that these same fire dynamics of spread, patchy distribution, and incomplete combustion of moist fuel sources would be exhibited in environments with no containment device (Brawn et al., 2001; Catling & Brownell, 1998; Marty, 2015). The containment device mitigated the spread of the fire outside of the intended burn area, and mitigated wind-fire dynamics limiting fire spread and scorching.

Native herbaceous species were significantly different and exhibited an increase in percent cover, abundance, and richness in the post-fire succession treatment. Native herbaceous species cover increased significantly from 36% to 57% in post fire conditions (p=0.03). The sharp decline of woody species, space clearing, and increase in soil nutrients are likely contributors to facilitating the increase in native herbaceous cover (Howe, 1999; Lett & Knapp, 2005). In post-fire conditions, native herbaceous species richness was not significantly different from control plots (p=0.77) in the season following fire. Within the fire treatment category native herbaceous richness increased by 67%. It is likely these will continue to increase in year two and subsequent years following fire (Ansley & Rasmussen, 2005); that awaits to be tested by the next phase of research. A comparison of before and after species richness indicates 13 new species that were previously not found on plots treated with fire (Table 14). Three of which are exotic (*Verbascum Thapsus, E. repens,* and *M. lupuina*) and one woody species was found on plots following prescribed burning (*R. alnifolia*).

Some of the native herbaceous species of interest (Table 9) increased significantly (p=0.001) in abundance following prescribed burning. One notable exception was *C. coccinea.* As mentioned earlier, it is not clear why *C. coccinea* did not appear on the entire Carden Alvar landscape in year two of the study. It was evident that prescribed burning stimulated vigorous recruitment and establishment of alvar-adapted native species in post-disturbance habitat. It wasn't determined whether these seeds were contributions from the soil seedbank or brought in through other vectors, but they have been found in post fire conditions. Of these species, *D. spicata* is adapted to fire by tillering and establishing from seed after top-kill (Sheiner & Samuel, 1988). Several studies in barrens, forest, prairie and flatwood ecosystems found a large increase in *D. spicata* in the growing-season following a fire (Shceiner, 1987; Sheiner & Samuel, 1988; Vankat & Snyder , 1991). In Shceiner & Samuel (1988), they found that fire stimulated *D. spicata* to produce 4.5 times more vegetative culms and 1.5 times more flowering culms than those in more successionally advanced communities. My experiment on alvar habitat showed a slight increase in abundance in the following year where more vegetative culms added to abundance. *D. spicata* does well as a native contributor to abundance and cover in years following fire as the plant allocates more resources to vegetative growth rather than reproductive effort. Competitive dominance and reproductive capabilities may not be sustained more than 30 years following a fire disturbance, highlighting the need for a disturbance in some interval to maintain native competitiveness within open alvar communities.

Similarly, *Deschampsia cespitosa* increased in cover and abundance significantly compared to pre-fire levels on plots previously encroached by *J. communis*. While *D. cespitosa* was subject to top kill, fire stimulated vigorous growth in the following year with new culms colonising areas opened to sunlight. Similar increases in abundance of species of interest was found with *P. paupercula*, *S. parvula*, *S. ptarmicoides*, and *S. heteroplepis* in a post-fire disturbance.

Exotic herbaceous species cover did not exhibit significant differences from BACI comparisons in any of the treatment types. This is likely due to the presence of the same suite of exotics before and after treatment with the exception of three new exotic herbaceous species observed after treatment (*Fallopia convolvulus*, *Turritis glabra*, and *V. thapus*). In post fire conditions, species with greatest cover were *Elymus repens* (~3%), *V. thapsus* (~2%), and

Medicago lupulina (~2%), though these were all in fairly low abundances and cover values. These are species that are strong competitors and generally increase in post fire conditions, taking advantage of new space and in some instances, being stimulated by fire. *E. repens* cover and flowering can increase following an early spring burn (Snyder, 1992), facilitated mainly by rhizomes which are stimulated by above-ground culm top kill, and to a lesser extent by seed dispersal and seedbank survival (only 25% are viable per plant per season) (Olson, 1976; Majek, Erickson, & Duke, 1984). It generally invades any newly-disturbed area. Of the new exotics found after treatment, *V. thapsus* was moderately present on the landscape and North Bear Alvar site, no individuals were found on plots prior to treatment, though the high abundance of seeds per plant, nearby individuals, and increases in light penetration and space clearing likely aided in the assembly on plots treated with fire (Gucker, 2008). Additionally, seeds of *V. thapsus* can be persistent and remain viable in the soil for up to 100 years and can disperse within 11 meters of the parent plant (Gross & Werner, 1978). There were individuals present nearby open and moderately encroached plots, but were absent from encroached plots. It is likely that contributions from persistent seed banks germinated in conditions following low-moderate severity fires (Korb, Johnson, & Covington, 2004), especially in areas with increased sun penetration to the ground level and less woody competition (Booth, Murphy, & Swanton, 2010). This was evidenced in 3/9 fire treated plots with *V. thapsus*. Though while it may be widespread and abundant in certain locales, the effects of the species in an ecological sense are fairly benign (Booth, Murphy, & Swanton, 2010). No information on species response to fire was found for *M. lupulina* on the Fire Effects Information System (see FEIS: https://www.feis-crs.org/feis for more information). This species is a strong competitor in disturbed habitats and outcompetes native grasses for space in many open or newly disturbed habitats (Booth, Murphy, & Swanton, 2010) and may inhibit regeneration and assembly of desirable herbs and grasses. None of these pose serious threats to the post-disturbance succession of alvars, they are found within the landscape and in insignificant abundances. Some (such as *E. repens*) may help contribute to post-succession structure regeneration by providing grazing opportunities for herbivores (Olson, 1976).

As evidenced in other treatments, exotic herbaceous richness did not change significantly using prescribed burning compared to prior levels. While some small increases were evidenced in the following year, it is likely that these will level out and decline as native and alvar-adapted species dominate in the later successional stages. Persistent exotic species were mainly *Elymus repens* and *Prunella vulgaris*. Both can be moderately invasive and may become locally dominant in optimal conditions (Urban Forest Associates, 2002); they are all common on the Carden Alvar and in non-alvar open grassland and prairie-meadow communities (Brownwell & Riley, 2000). They don't pose the same problems in terms of competitive dominance and structure changes as do woody species and *E. repens* further serves to add structure and grazing opportunities for herbivores, and cover for grassland breeding birds (Majek, Erickson, & Duke, 1984; Kirsch & Higgins, 1976).

5.1.3 Effects of prescribed burns on macronutrient abundance

As was hypothesized with prescribed burns or natural fires, there were increases in the availability of soil macronutrients, primarily evidenced in nitrate-nitrogen ($p=0.04$), following prescribed burning. Of the nine fire treated plots, four (NBA 001, NBA010, NBA027, NBA032) exhibited significantly large increases in nitrate-nitrogen abundance immediately following fire. Soil disturbance by fire is known to accelerate mineralisation of nitrogen (Parminter & Bedford, 2006) and contributed to post-fire recovery of plants stimulating germination and growth. The substantial increases in nitrogen availability in the soil contributed to increases in species richness across all three vegetative groupings, mainly evidenced in native herbaceous species by

67%. Plots which had a greater cover of shrubs and greater live fuel accumulations resulted in more nutrient release into the soil material following burning. These same plots had correspondingly higher species richness. Nutrient inputs don't last long in alvar soils, generally due to weathering, uptake, and other erosive environmental factors (Stark et al., 2004). The stimulation that the plant community received as a result of prescribed burn nutrient cycling substantially increases post-disturbance assembly in areas with higher nutrient concentrations as was evident in the effects noted in the section above (Dudley & Lajtha, 1993; Sherman & Brye, 2009). These effects were remarkably different across the three treatment types: changes in the abundance and presence of soil macronutrients were insignificant with glyphosate application and manual removal. These will be discussed in a subsequent section below. Fire interactions with soil nutrients influenced the increases in abundance and richness of native plant species and also facilitated the regeneration of exotic species taking advantage of local short-lived increases in nutrients.

5.2 Glyphosate Treatment

As expected with glyphosate treatment, species percent cover and richness were significantly decreased across some plant categories (Figure 11). This was mainly within the woody species category ($p=0.004$) as this was a selected target for plots allocated to glyphosate application.

5.2.1 Effects on plant community

Woody species percent cover was immediately affected by application, and immediately reduced cover from 41% to 14% with further losses to 11% in glyphosate treated plots by the next growing season. Woody richness declined by 57% following treatment within herbicide treated plots. *P. virginiana* persisted even while most of its foliar areas were sprayed. Some treated plots were re-colonised by *R. aromatica* from nearby individuals following defoliation. In some cases, contributing sources of colonisation were less than 1 meter away from the treated plot. *J. communis* was immediately affected by glyphosate though some foliar areas survived and persisted with new growth. Evidently application was not complete and even under stress, *J. communis* individuals persisted.

Native herbaceous cover was not impacted significantly (p=0.48), though richness did decline significantly by 46% (p=0.02) through mist drift. Three native species persisted after treatment. *Aquilegia canadensis* and *Monarda fistulosa* both contributed to native herbaceous cover following treatment from within the plot. *Geranium bicknellii* was not evidenced in baseline species surveys and colonised after treatment by glyphosate. All three of these were in relatively low (3% and less) cover. Characteristic native alvar herbaceous species like *D. spicata*, *G. trifolium*, and *D. cespitosa* were killed off and were not evidenced in the year after treatment. Legacy effects of glyphosate in the soil and environmental conditions may be responsible, these effects may persist for some time, or be weathered away given the multiple soil disturbances that occur on alvars. The interactions of glyphosate in soils is considered generally safe due to rapid sorption onto soil particles, and degradations by microbes where it becomes inactivated quickly (Chandler, Murphy, & Swanton, 2010). There is no current or past research on the legacy effects of glyphosate on alvar soils, though I believe this need not be conducted due to other more desirable management options. The exotic *Medicago lupulina* was incredibly invasive (~9% cover) in the first sere observed after treatment by glyphosate.

Unless other actions are performed vegetation dies in situ, which also, to a degree inhibits plant assembly and regeneration of the site until it decays through natural processes over time. Chemicals can strongly increase the efficacy of invasive woody control, though short-term detrimental effects on desired native characteristic alvar species are to be expected through drift or unintentional contact with the herbicide. Glyphosate application impacts may be temporary -

any reductions in herbs and shrubs may not persist beyond a season or two (Sullivan & Sullivan, 2003).

5.2.2 Effects of glyphosate on macronutrient content

Glyphosate application had no significant effects on macro-nutrients. Only two plots exhibited slight increases in nitrogen and potassium, though they were not statistically or ecologically significant as was the case with prescribed burning. These are likely the result of site specific differences, influences of surrounding vegetation, or soil movement due to frost heave and weathering processes. Those that did exhibit increases in nitrogen were located in dense thickets of vegetation and to a degree, litter decomposition may have influenced this more so than in plots that were more open. Since glyphosate is applied to foliar leaf areas, it has little or no herbicidal activity in the soil and is not found to significantly contribute to nutrient fluctuations (Duke et al., 2012). In comparison to fire, glyphosate has no clear accompanying physiochemical process which would accelerate the mineralisation of nutrients in the soil, which aids in explaining the static nutrient abundances from before and after treatment. This was within the range of the hypothesized effects of treatment and was as expected.

5.3 Manual removal

Lopping and removal of cut material resulted in significant differences in woody species percent cover as would be expected, surprisingly this did not affect the richness or cover of native species (Figure 12).

5.3.1 Effect on plant community

Lopping and removing slash from plots obviously resulted in large significant declines in woody species cover ($p=0.02$), reducing cover of woody plants on treated plots to 1% or less on average. *J. communis* exhibited some small shoots from lopped individual stumps which was to be expected as the species is stimulated to send up shoots when the crown is damaged (Tirmenstein, 1999). Generally, woody richness declined 29% within the treatment category through manual removal, though this was not statistically significant (p=0.79) when compared to control plots. Here again opportunistic *P. virginiana* and *R. aromatica* contributed to assembly on manually treated plots.

After removing woody biomass, the effects of lopping native herbaceous species invigorated growth to recovery equal to or greater than before treatment levels of abundance; cover was affected as would be expected. No significant differences were observed between manually treated plots and control plots in native herbaceous cover (p=0.98) or richness (p=0.99). Existing exotic species that were on the plots persisted following treatment. The lopping simulated a light point in time grazing disturbance and was not necessarily as persistent as grazing. Many of the exotics are ruderals and may persist for a short time before being out-competed by alvar species. Target Cyperaceae species, especially *Carex crawii* and *C. richardsonii,* increased in abundance following removal of shrubs and lopping, they also increased in percent cover on plots following lopping.

5.3.2 Effects of manual removal on nutrient content

No significant differences were observed in any of the macronutrient abundances on manually treated plots, as hypothesized. Some slight decreases are evidenced (Figure 5). These are likely attributed to nutrient uptake by plants after a disturbance to aid in regrowth. Again, site specific factors and soil disturbance by environmental processes may also be responsible for changes in nutrient availability.

5.4 Performance of the Burn Box

Prescribed burning using the burn box was found to be similar to the time-temperature profiles and combustion characteristics of prescribed burns in situations with no burn box (P Catling & Brownell, 1998; Kremens, Faulring, & Hardy, 2003; Marty, 2015), and in instances where a burn box has been used prior (Kral et al., 2015). The burns in this experiment were ecologically similar in their effects to prescribed burning (Stubbendieck & Volesky, 2007; Engle et al., 1989; Archibold et al., 1998; Ohrtman et al., 2015), including patch dynamics brought on by arrangement of vegetation, soil moisture, and ground cover. The time-temperature curves reached a short-lived apex temperatures of 207 °C, 368 °C, and 428 °C in open, moderately encroached, and encroached vegetative situations respectively before slowly cooling (Figure 6). The tails of the time-temperature profiles (Figure 6) in this experiment were influenced slightly from the heated temperature of the steel wall confines, which caused the lingering right-tail. Though this effect is negligible when looking at burn dynamics within the box as the cool down period would not have altered already combusted biomass. Burn times were well within the limits of natural fires, where heat dosage (in seconds) rises quickly through a short-lived apex, and then cools rapidly as there is no fuel to sustain long burns (typically less than 125 seconds between ignition and flameout) (Bailey & Anderson, 1980; Kral et al., 2015). Though heat dosages were slightly lower compared to Kral et al. (2015) as I did not adjust fuel loads with supplementary fuel. The results of this study indicate that prescribed burning using the burn box attains representative burn effects and inferences can be used to evaluate fire effects and species responses in post-fire environments.

Moderately encroached and encroached alvar conditions, those with the greater fuel loads and more compact fuel arrangement obtained longer and higher heat dosages than those on open alvar situations dominated mainly by herbaceous species (Figure 7 & 8). In these two alvar conditions, I found the box performed best at combusting and consuming the available fuel, reduced existing biomass and was associated with increased assembly and richness following burning. The burns within the box consumed between 50-93% of existing vegetation on moderately encroached plots, and between 42-100% on encroached plots. While there was considerable variability in vegetation clearing characteristics between the two groups, this may be attributed to a matrix of various fine and coarse fuels exhibited in moderately encroached plots which allowed a greater fire carrying capacity in these plots.

Open alvar situations may not represent the best testing grounds for the burn box approach when left naturally, they only combusted and removed between 9-56% of the existing vegetation. This suggests that they may need to be augmented with fuel prior to testing fire dynamics on open alvar communities, such as those done in Kral et al (2015). *Phleum* sp. or other local native grasses can increase fuel loading in order to observe more representative fire dynamics (Kral et al., 2015; Limb et al., 2011). Factors such as vegetation composition and arrangement, fuel moisture, and soil moisture led to patchy burns on open alvar plots during the growing season. Vegetation patch dynamics as a result of fire are exhibited on other prescribed burns sites on the Carden Alvar landscape in open alvar communities (cf pers comm. Robin Vernon, OMNRF, 2015), and aid in seed preservation and refuges for species during ecosystem recovery from fire events (Hobbs & Huenneke, 1992; Reschke et al., 1999a). Some of these patch dynamics may be the result of edaphic factors (e.g. soil moisture, soil composition), presence of surface lying bedrock, fuel moisture, or fuel arrangement (Jones & Reschke, 2005). The lower severity burns are to be expected on sparse vegetation as the ability to sustain the carrying capacity of fire is directly correlated with fuel load and arrangement.

5.4.1 Implications for the practice of conservation and restoration management

Biologically rich communities typically necessitate a fire return interval to maintain productivity and stability. With the advent of European settlement, landscape fragmentation, land-use, and fire suppression have reduced many biologically diverse communities to less productive and less diverse communities, leading to declines in plant species richness and impacts on animal richness and habitat utility. For such fire supressed communities, a major conservation goal has been restoration of these communities with the reintroduction of historic fire regimes (Varner et al., 2005). However, fire introduction in long-unburned systems may have undesired and novel effects given the current geological epoch. The most compelling way to test fire effects may be the prescription of fire over small areas using natural or physical buffers to control flame-spread. The ability to manipulate the environment within the burn box can be an iterative learning tool for conservation managers when the effects of fire in a particular alvar (or other) community are not known, or to test species-specific responses to fire. The utility of burning using the burn box can be further captured in testing responses of fire to: determine the availability of the soil seedbank to be stimulated by fire, assess acute fire effects on a particular species or community, determine fire effects on nutrient cycling and mineralisation, or any other small-scale experiment. The size and construct of the box lent itself to increased replicability over a single habitat or community type. This was evidenced in burn dosage and combustion characteristics depicted in the burn severity mapping in Figures 7.1-7.9 within the dominant vegetation categories (open, moderately encroached or encroached). Habitats exhibiting uniformity of vegetation exhibited similar burn dynamics, which aids in reinforcing the notion that these can be tested within the same type of community to observe replicable fire effects without much variation in burn characteristics within groups.

5.4.2 Scope of applications to apply the burn box for testing fire effects

The physical confines of the burn box did narrow the scope of dynamics that would be normally observed in prescribed or natural fires. Fire spread and severity influenced by wind dynamics and patterns were hindered using the burn box approach. This was as expected, the five holes drilled into the base of each of the eight panels still allowed some near-ground air flow to feed the fire. Fire spread and severity would alter flammability and combustion characteristics in some vegetation communities and affect dispersal over a particular site. In addition, wind may increase or decrease burn severity and influence temperature at very small or large scales, and contribute to the mosaic burn that alvars experience when ignited by prescribed burning or naturally (Belcher et al., 1992).

The burn box results can be hard to apply to wider fire management principles (e.g. fuel loading, fuel arrangement) because of safety precautions which were required when the burn box was deployed in a remote field setting and the care needed to avoid accidental spread outside of the burn box. In order to prevent ignition of surrounding woody vegetation which was at or greater than the height of the box walls, shrubs and trees exterior to the box were lopped and removed one meter on every side. Additionally, trees or shrubs within the box were lopped to 1 meter and added to the fuel load in the box (i.e. left quasi-naturally). This also allowed me to assemble the box, ignite, and monitor the fire safely. I acknowledge though this may have inhibited some combustion and dispersal characteristics of the burn. The effects these may present are negligible in this situation due to the small $4m^2$ burn area, but are still worth noting.

Larger burn patches can be tested by using a larger box of the same steel 11-gauge construct. Circumstances can arise where larger patch dynamics need to be observed in order to analyse statistically and ecologically significant effects in some communities. This may be particularly useful in shrub-dominated communities, or the presence of abundant saplings or young trees. Increasing the spatial area within the burn box allows environmental variables like wind to have more representative effects in moving fire around vegetation. Enlarging the burn area does come with increased weight of the panels, where in my experiment each 1 m^2 panel weighed 36 kg. This may be an issue in remote areas or in sensitive terrain.

The burn box does lend itself to answer tough or uncertain questions on ecosystem response to fire disturbance and post-disturbance succession in fire-excluded ecosystems. The ability for land managers to determine fire effects and determine succession trajectory, along with additional interventions that may be needed can be of great utility in novel or uncertain ecosystems. The ability to model ecosystem trajectory in post-fire habitats is already well-documented, though there are a variety of local instances where the particulars of fire effects are not certain, the burn box can be of great utility to answer these questions and determine larger management options to manage sensitive or novel systems.

6.0 Future Directions and Conclusions

6.1 Management of North Bear Alvar

Current practices by NCC for conserving alvar habitat are already well documented in their approaches to conserving alvars on the Bruce Peninsula and on Pelee Island alvars. Both of which are managed for an attainment of habitat heterogeneity and preservation of isolated, rare, and significant species (Nature Conservancy of Canada, 2008). An aerial comparison of North Bear Alvar to similar alvars on the Carden plain notes considerable infilling of open communities by woody shrubs and trees. While this is not unique to the North Bear Alvar parcel, as communities surrounding it are heavily dominated by woody species and forests, management interventions such as prescribed burning should be considered as soon as possible before encroachment crosses a threshold of no return. Planning for a burn may take upwards of two years, and stochastic environmental patterns may preclude conducting burns in the order of years. It can be expected that as woody percent cover increases, there will be losses in native and rare species richness and abundances in these phytogeographically unique communities.

It is recommended that NCC undertake a strategic review of North Bear Alvar to determine target priority areas and access points for equipment required in low or high complexity burn scenarios. Feasibility may be constrained by site access in certain areas due to the presence of wetlands or mesic soils which may become rutted as a result of frequent traffic and equipment (Catling, 2016). Permanent quadrat or transect monitoring will lead to inferences on processes of shrub encroachment and control and will aid in assessing post-fire succession over time, which can continue to inform conservation management of North Bear Alvar. Woody species encroachment should be continually monitored by comparison of land-satellite imagery to identify priority areas. Large monotypic patches of *J. communis* and mixed communities may represent the best habitats to target for prescribed burning. Burning should only be considered before breeding birds nest or after fledglings leave since open alvars in central Ontario represent Important Bird Areas and breeding grounds for grassland breeding birds. Target species should be chosen to represent their status and rarity within the system. Relatively small areas (~30%) should be burned at any one time in order to maintain a matrix of habitat refuge areas for species to persist while burned areas recover.

It is evident within this experiment that early results of fire in successionary habitat promotes increases in native biodiversity when woody species are combusted and removed. Additionally, the increase in nutrient availability within the soil as a result of burning creates desirable growing conditions for regeneration. Within alvar complexes of various habitat types, prescribed burning should be promoted for management and maintenance of biodiversity and open alvar structures, as positive effects are noted in post-fire early succession and factors associated with prescribed burning (i.e. space clearing, stimulation of seedbanks, and increases in soil nutrients). Burning within a set regime will also stimulate productivity and yield competitive advantages for alvar-adapted native herbaceous species that, without burning, may be outcompeted by advantageous woody species along a successionary gradient. That being said, some woody species (*P. virginiana*, *R. aromatica* and *typhina*) take advantage of post-fire conditions and may present short to medium term problems with increases in stem density or cover (Olson, 1976; Johnson, 2000; Taylor, 2004). Summer fires during severe drought conditions may result in less regeneration in dry sites due to depleted soil organic matter, high summer soil temperatures, and a lack of seed bank. Should these species cross a threshold of concern in post-fire regenerating habitat, land managers might consider a couple of possible pathways for control. The most costly option might be multiple successive burns, staged 3-5 years after the initial burn in order to stimulate and yield a competitive advantage for herbaceous species and continue to top-kill and stress fire-intolerant woody species (Edwards, Krawchuk, & Burton, 2015). A second approach might consider a blend of top-down and bottom-up approaches where initial fire disturbance deals with stimulation and removal of biomass, with augmented planting or targeted removal of problematic woody species in post-fire conditions. These species, in an ideal sense, would be removed before seed-drop to limit contributions to the soil seedbank.

Future studies employing the burn box in alvars should be directed at the regeneration of fire-stimulated woody species and the particular effects of successive fire events on these species to analyse thresholds and factors that might decrease sprouting and regeneration in post-fire conditions. Some ecological community types on North Bear Alvar are not suitable for burning using the burn box (i.e. treed communities), these patches are more influenced from the landscape around them than small patches of open alvar habitat, and may be incorporated into a larger high-complexity burn plan than a low-complexity open grassland burn. Underpinning all of this, it must still be acknowledged that woody species are part of a natural succession timeline. Their processes, factors, and outcomes are important; some matrix of woody cover is a necessary habitat attribute and adds to the mosaic of complex habitat that alvars exhibit and that alvar or open-specialist species necessitate.

6.2 Conclusions

What defines successful (alvar) grassland, prairie, and meadow conservation is the presence of large-scale intensive disturbances such as fire or grazing which reinforce resilience, integrity, productivity, and native biodiversity (Bowman & Murphy, 2010; Catling, 2009; Catling & Brownell, 1998; Limb et al., 2011). While grazing effects weren't tested per se in this experiment (I tested quasi grazing through manual lopping), the combination of the fire and grazing has historically shaped prairies and a combination, magnitude, and frequency is needed today in order to maintain them. Without fire, these systems lose integrity in several ways, the most obvious being the change in structure and composition as a result of encroachment of shrubs and trees that fire otherwise controls and supresses such processes. My early results indicate that the regeneration of habitat proceeds along desirable trajectories to reduce woody encroachment and thereby promote native herbaceous richness and characteristic open alvar species in the seral stage following a fire disturbance. Considerations for burning should be directed at the attainment of habitat heterogeneity. In some cases, it may be appropriate to burn roughly 30% of a sizeable community or site at a time (Ansley & Rasmussen, 2005; Nature Conservancy of Canada, 2008; Van Sleeuwen, 2006). Unburned patches aid as critical refuges for fire-susceptible arthropods, herpetafauna, and retain some habitat structure and utility for grassland breeding birds (Davis, 2004; Hartley et al., 2007). Multiple sequential burns are necessary in areas of old-age long-unburned woody growth to kill off seedbank contributions from unwanted species from becoming too competitive with herbaceous species in early succession. If burning is not feasible in areas, tree and woody removal by cutting may result in communities similar to successional alvar burns, but they may persist only temporarily due to vigorous re-sprouting and stimulation of seedbanks by space-clearing without fire effects to supress fire-intolerant species (Catling and Brownell, 1998). In order to fully understand the impacts of fire disturbance to long-unburned alvars, future research should analyse the long-term regeneration of burned patches to determine fire return interval in order to establish a contextualised fire regime based on bounded ranges of variation in local alvar communities. It is widely accepted that alvars necessitate fire at some temporal interval, though long-term studies to determine the regime are lacking. Permanent plots, using vegetation sampling protocol (see: Ontario Ministry of Natural Resources, 2011) to evaluate the change in species composition over time would allow for trends and succession to be studied over time in regards to reference sites and changing climatic norms.

The goals and objectives of employing prescribed burning were to reduce the cover and extant of woody species, and influence the increase in alvar-adapted native species assembly in terms of richness and cover. Using the burn box approach, it is evident that fire has a significant impact on the local plant community, and influences open alvar successional habitat in post-fire conditions. It is acknowledged that a patchiness of communities will be the product of burns, and that to an extent, some woody and invasive species may persist in small patches. This matrix of structural composition is the basis of desirable grassland conditions, provides necessary habitat to species, and is considered of high quality (Catling, 2016). Problematic individual species that have high probabilities of spreading and encroachment may be dealt with on a contextualised basis following burning if they persist.

Glyphosate does not necessarily represent the best disturbance mechanism with multiple tangential benefits, the application of glyphosate is limited in scope solely to exterminating vegetation. Secondary benefits may be seen as space opening, and changes in soil chemistry (Chandler et al., 2010). Though these benefits are minimal and are not as immediate or perhaps as desirable as the processes and effects of fire. Legacy effects from chemical contact with soil may be persistent for 5-10 years in some cases (Lancaster et al., 2010; Landry et al., 2005; Sullivan & Sullivan, 2003), which may inhibit seedling germination for species with thin husks, or intolerant species (Landry et al., 2005). Additionally, the impact to non-target species was evident as it decreased richness across native herbaceous species, even though this was unintentional as woody vegetation was the main target, glyphosate drift still occurred. Choice to employ glyphosate should only be employed around problem noxious weed control and in situations where burning is not feasible but vegetation management is necessary. Alvars should be considered to be excluded from this due to easy runoff or infiltration into the underlying bedrock.

While manual removal was highly effective at reducing cover and encroachment, factors such as time and manpower might overwhelm conservation managers in areas of heavy encroachment and may not be practical for monotypic patches of large woody-dominated sites. Heavy machinery should not be considered as an option due to the high propensity for lasting rut damage in the shallow soils and increases in bulk density which may inhibit germination of seedbanks in the soil. NCC currently employs manual removal on some properties where goals are established to reduce sporadic woody material in 1 hectare blocks (cf. pers comm. Laura Robson, 2017). These are more likely to be considered practical interventions in communities that are moderately encroached, or that are too wet for prescribed burning (e.g. McGee Creek and Cranberry Wetlands).

Overall, the one year results of this study in addition to discourse on alvars (e.g. Catling, 2009; Jones & Reschke, 2005; Reschke et al., 1999; Rosen & van der Maarel, 2000; Taylor & Catling, 2011; The Ottawa Field-Naturalists' Club, 2002), reinforces the role of fire in maintaining desirable open alvar communities and reducing woody shrub encroachment. Fire timing and return intervals are crucial to establish. A short fire return interval may be needed when first applied to North Bear due to the length of time since the last fire event (>100 years). Considerations for prescribing burns will need to account for climate change and range shifts. A matrix of habitats should be selected for prescribed burning where multiple vegetation communities abut each other to attain a mosaic of alvar habitat types and patch sizes. The disappearance of disturbance regimes undoubtedly will result in large changes to alvar communities and perhaps the extirpation of certain specialist species of flora and fauna reliant on open alvar situations, underlying the need for immediate conservation and restoration actions of globally rare alvar habitat.

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8.0 Appendix

Table 13 Complete species observations for North Bear Alvar during 2016-2017

Scientific Name	Common Name	Form	Open_alvar	Native	Cat_inv	CC	SRANK
Achillea millefolium Linnaeus	Common yarrow	Herb	Y	E	0	0	SE
Agastache foeniculum (Pursh) Kuntze	Blue giant hyssop	Herb	Y	N	0	2	S4
Amelanchier spicata (Lamarck) K. Koch	Running serviceberry	Wood	N	N	0	5	S5
Anemone multifida Poiret var. multifida	Cut-leaved anemone	Herb	Y	N	0	3	S5
Antennaria neglecta Greene	Field pussytoes	Herb	Y	N	0	3	S5
Anticlea elegans (Pursh) Rydberg	Mountain death camas	Herb	Y	N	0	10	S4
Aquilegia canadensis Linnaeus	Red columbine	Herb	Y	N	0	5	S5
Arabidopsis lyrata (Linnaeus) O'Kane & Al-Shehbaz subsp. lyrata	Lyre-leaved rockcress	Herb	Y	N	0	7	S4
Arabis hirsuta (Linnaeus) Scopoli	Hairy rockcress	Herb	Y	N	0	8	S5
Arenaria serpyllifolia Linnaeus var. serpyllifolia	Thyme-leaved sandwort	Herb	Y	E	0	0	SE5
Asclepias purpurascens Linnaeus	Purple milkweed	Herb	Y	N	0	10	S2
Aster sp.	Aster sp.	Herb	Y	N	0		
Bromus inermis Leysser	Awnless brome	Herb	Y	E	0	0	SE5
Bromus kalmii A. Gray	Wild chess	Herb	Y	N	0	8	S4
Calamagrostis canadensis (Michaux) Palisot de Beauvois var. canadensis	Bluejoint reedgrass	Herb	Y	N	0	4	5S
Calystegia spithamaea (Linnaeus) Pursh	Low false bindweed	Herb	Y	N	0	7	S4S5
Campanula rotundifolia Linnaeus	American harebell	Herb	Y	N	0	7	S5
Carex bebbii (L.H. Bailey) Olney ex Fernald	Bebb's sedge	Herb	Y	N	0	3	S5
Carex crawei Dewey	Crawe's sedge	Herb	Y	N	0	10	S4
Carex flava Linnaeus	Yellow sedge	Herb	Y	N	0	5	S5
Carex intumescens Rudge	Greater bladder sedge	Herb	Y	N	0	6	S5
Carex pensylvanica Lamarck	Pennsylvania sedge	Herb	Y	N	0	5	S5
Carex richardsonii R. Brown	Richardson's sedge	Herb	Y	N	0	9	S4?
Castilleja coccinea (Linnaeus) Sprengel	Scarlet paintbrush	Herb	Y	N	0	9	S5
Celastrus scandens Linnaeus	Climbing bittersweet	Herb	Y	N	0	3	S5
Cirsium arvense (Linnaeus) Scopoli	Canada thistle	Herb	Y	E	1	0	SE5
Cornus racemosa Lamarck	Grey dogwood	Wood	N	N	0	2	S5
Crataegus sp.	Hawthorn sp.	Wood	Y	N	0		
Cypripedium parviflorum Salisbury	Yellow ladies slipper	Herb	Y	N	0	7	S4S5
Danthonia spicata (Linnaeus) P. Beauvois ex Roemer & Schultes	Poverty oatgrass	Herb	Y	N	0	5	S5
Dasiphora fruticosa (Linnaeus) Rydberg	Shrubby cinquefoil	Wood	N	N	0	9	S5
Daucus carota Linnaeus	Wild carrot	Herb	Y	E	0	0	SE5
Deschampsia cespitosa (Linnaeus) Palisot de Beauvois *subsp. cespitosa*	Tufted hairgrass	Herb	Y	N	0	9	S4S5
Dianthus armeria Linnaeus subsp. armeria	Deptford pink	Herb	Y	E	0	0	SE5
Drymocallis arguta (Pursh) Rydberg	Tall cinquefoil	Herb	Y	N	0	7	S4

Scientific Name	Common Name	Form	Open_alvar	Native	Cat_inv	CC	SRANK
Echium vulgare Linnaeus	Common vipers bugloss	Herb	Y	E	0	0	SE5
Elymus canadensis Linnaeus var. canadensis	Canada wildrye	Herb	Y	N	0	8	S4S5
Elymus repens (Linnaeus) Gould	Quackgrass	Herb	Y	E	0	0	SE5
Equisetum arvense Linnaeus	Field horsetail	Herb	Y	N	0	0	S5
Eupatorium altissimum Linnaeus	Tall boneset	Herb	Y	N	0	3	S1
Eurybia divaricata (Linnaeus) G.L. Nesom	White wood aster	Herb	Y	N	0	10	S2
Eurybia macrophylla (Linnaeus) Cassini	Large-leaved aster	Herb	Y	N	0	5	S5
Euthamia graminifolia (Linnaeus) Nuttall	Grass-leaved goldenrod	Herb	Y	N	0	2	S5
Fallopia convolvulus (Linnaeus) Á. Löve	Black bindweed	Herb	Y	E	0	0	SE5
Fragaria vesca Linnaeus subsp. vesca	Woodland strawberry	Herb	Y	N	0	4	S5
Fragaria virginiana Miller *subsp. virginiana*	Virginia strawberry	Herb	Y	N	0	2	S5
Geranium bicknellii Britton	Bicknell's geranium	Herb	Y	N	0	5	S4
Geum fragarioides (Michaux) Smedmark	Barren strawberry	Herb	Y	N	0	5	S5
Geum trifolium Pursh	Prairie smoke	Herb	Y	N	0	9	S4
Hieracium lachenalii subsp. cruentifolium (Dahlstedt & Lübeck) Zahn	Common hawkweed	Herb	Y	E	3	0	SE5
Houstonia longifolia Gaertner	Long-leaved bluets	Herb	Y	N	0	8	S4?
Hypericum perforatum Linnaeus subsp. perforatum	Common St. John's-wort	Herb	Y	E	4	0	SE5
Iris versicolor Linnaeus	Blueflag iris	Herb	Y	N	0	5	S5
Juniperus communis Linnaeus	Common juniper	Wood	N	N	2	4	S5
Lactuca hirsuta Muhlenberg ex Nuttall	Hairy lettuce	Herb	Y	N	0	7	S4?
Leucanthemum vulgare Lamarck	Oxeye daisy	Herb	Y	E	0	0	SE5
Linum usitatissimum Linnaeus	Common flax	Herb	Y	E	0	0	SE3
Lonicera canadensis Bartram ex Marshall	Canada fly-honeysuckle	Wood	N	N	0	6	S5
Lonicera dioica Linnaeus var. dioica	Limber honeysuckle	Wood	N	N	0	5	S5
Lonicera hirsuta Eaton	Hairy honeysuckle	Wood	N	N	0	7	S5
Lonicera tatarica Linnaeus	Tartarian honeysuckle	Wood	N	E	1	0	SE5
Lotus corniculatus Linnaeus	Garden Bird's-foot-trefoil	Herb	Y	E	2	0	SE5
Lysimachia ciliata Linnaeus	Fringed yellow loosestrife	Herb	Y	N	0	4	S5
Medicago lupulina Linnaeus	Black medic	Herb	Y	E	4	0	SE5
Mentha arvensis Linnaeus	Field mint	Herb	Y	N	0	3	S5
Mentha arvensis Linnaeus	Wild mint	Herb	Y	N	0	3	S5
Micranthes virginiensis (Michaux) Small	Early saxifrage	Herb	Y	N	0	6	S5
Minuartia stricta (Sw.) Hiern	Rock sandwort	Herb	Y	N	0	0	S4
Monarda fistulosa Linnaeus var. fistulosa	Wild bergamot	Herb	Y	N	0	6	S5
Muhlenbergia schreberi J.F. Gmelin	Schreiber's muhly	Herb	Y	N	0	1	S4
Oenothera biennis Linnaeus	Common evening primrose	Herb	Y	N	0	0	S5
Oxalis montana Rafinesque	White wood-sorrel	Herb	Y	N	0	8	S5
Packera paupercula (Michaux) Á. Löve & D. Löve var. paupercula	Balsam ragwort	Herb	Y	N	0	7	S5
Panicum philadelphicum Bernhardi ex Trinius	Philadelphia panicgrass	Herb	Y	N	0	6	S4

Scientific Name	Common Name	Form	Open_alvar	Native	Cat_inv	CC	SRANK
Sporobolus heterolepis (A. Gray) A. Gray	Northern dropseed	Herb	Y	N	0	10	S3
Symphoricarpos albus (Linnaeus) S.F. Blake *var. albus*	Thin-leaved snowberry	Wood	N	N	0	7	S4S5
Symphyotrichum cordifolium (Linnaeus) G.L. Nesom	Heart-leaved aster	Herb	Y	N	0	5	S5
Symphyotrichum novae-angliae (Linnaeus) G.L. Nesom	New england aster	Herb	Y	N	0	2	S5
Symphyotrichum urophyllum (Lindley ex de Candolle) G.L. Nesom	Arrow-leaved aster	Herb	Y	N	0	6	S4
Taraxacum officinale F.H. Wiggers	Common dandelion	Herb	Y	E	0	0	SE5
Thuja occidentalis Linnaeus	White cedar	Wood	N	N	0	4	S5
Thuja occidentalis Linnaeus	Eastern white cedar	Wood	N	N	0	4	S5
Toxicodendron radicans (Linnaeus) Kuntze	Poison ivy	Herb	N	N	0	5	S5
Tragopogon dubius Scopoli	Yellow salsify	Herb	Y	E	0	0	SE4?
Trifolium aureum Pollich	Yellow clover	Herb	Y	E	2	0	SE5
Trifolium pratense Linnaeus	Red clover	Herb	Y	E	4	0	SE5
Turritis glabra Linnaeus	Tower mustard	Herb	Y	E	0	0	SE5
Verbascum thapsus Linnaeus	Common mullen	Herb	Y	E	0	0	SE5
Vicia americana Muhlenberg ex Willdenow var. americana	American vetch	Herb	Y	E	1	0	SE5
Viola sororia Willdenow	Common blue violet	Herb	Y	N	0	4	S5

*This table reflects current status as of July 14, 2018. Sources used are: data.canadensys.net/vascan AND explorer.natureserve.org/servlet/NatureServe?init=Species. Invasive rankings from SER (2014).

Table 14 New species observations on plots following fire treatment

Scientific Name	Common Name	Form	Open_alvar	Native/Exotic	Invasiveness	CC	SRANK
Anemone multifida Poiret var. multifida	Cut-leaved anemone	Herb	Y	N	0	3	S5
Campanula rotundifolia Linnaeus	American harebell	Herb	Y	N	0	7	S5
Cypripedium parviflorum Salisbury	Yellow ladies slipper	Herb	Y	N	0	7	S4S5
Rhamnus alnifolia (L'Héritier) Hauenschild	Alder-leaved buckthorn	Wood	Y	N	0	7	S5
Eupatorium altissimum Linnaeus	Tall boneset	Herb	Y	N	0	3	S1
Eurybia macrophylla (Linnaeus) Cassini	Large-leaved aster	Herb	Y	N	0	5	S5
Fallopia convolvulus (Linnaeus) Á. Löve	Black bindweed	Herb	Y	E	0	0	SE5
Geranium bicknellii Britton	Bicknell's geranium	Herb	Y	N	0	5	S4
Penstemon hirsutus (Linnaeus) Willdenow	Hairy beard-tongue	Herb	Y	N	0	7	S4
Phlox divaricata Linnaeus	Wild blue phlox	Herb	Y	N	0	7	S4
Platanthera aquilonis Sheviak	Tall northern green orchid	Herb	Y	N	0	5	S5
Turritis glabra Linnaeus	Tower mustard	Herb	Y	E	0	0	SE5
Verbascum thapsus Linnaeus	Common mullen	Herb	Y	E	0	0	SE5

Table 15 Native herbaceous species observed assembling on plots treated with fire which were not previously observed

Scientific Name	**Common Name**	**CC**	**SRANK**
Anticlea elegans (Pursh) Rydberg	Mountain death camas	10	S4
Calamagrostis canadensis (Michaux)	Blue-joint reedgrass	4	S5
Calystegia spithamaea (Linnaeus) Pursh	Low false bindweed	7	S4S5
Campanula rotundifolia Linnaeus	American harebell	7	S5
Carex crawei Dewey	Crawe's sedge	10	S4
Carex richardsonii R. Brown	Richardson's sedge	9	S4?
Cypripedium parviflorum Salisbury	Yellow ladies slipper	7	S4S5
Danthonia spicata (Linnaeus) P. Beauvois ex Roemer & Schultes	Poverty oatgrass	5	S5
Deschampsia cespitosa (Linnaeus) Palisot de Beauvois *subsp. cespitosa*	Tufted hairgrass	9	S4S5
Drymocallis arguta (Pursh) Rydberg	Tall cinquefoil	7	S4
Eurybia macrophylla (Linnaeus) Cassini	Large-leaved aster	5	S5
Fragaria vesca Linnaeus *subsp. vesca*	Woodland strawberry	4	S5
Geranium bicknellii Britton	Bicknell's geranium	5	S4
Geum trifolium Pursh	Prairie smoke	9	S4
Houstonia longifolia Gaertner	Long-leaved bluets	8	S4?
Lactuca hirsuta Muhlenberg ex Nuttall	Hairy lettuce	7	S4?
Minuartia stricta (Sw.) Hiern	Rock sandwort	0	S4
Monarda fistulosa Linnaeus *var. fistulosa*	Wild bergamot	6	S5
Packera paupercula (Michaux) Á. Löve & D. Löve *var. paupercula*	Balsam ragwort	7	S5
Penstemon hirsutus (Linnaeus) Willdenow	Hairy beard-tongue	7	S4
Potentilla anserina Linnaeus *subsp. anserina*	Silverweed cinquefoil	0	SE5
Potentilla simplex Michaux	Common cinquefoil	3	S5
Sisyrinchium montanum var. crebrum Fernald	Strict blue-eyed grass	4	S5
Solidago canadensis Linnaeus	Canada goldenrod	1	S5
Solidago juncea Aiton	Early goldenrod	3	S5
Solidago ptarmicoides (Torrey & A. Gray) B. Boivin	Upland white goldenrod	9	S5
Symphyotrichum novae-angliae (Linnaeus) G.L. Nesom	New England aster	2	S5
Symphyotrichum urophyllum (Lindley ex de Candolle) G.L. Nesom	Arrow-leaved aster	6	S4
Toxicodendron radicans (Linnaeus) Kuntze	Poison ivy	5	S5
Viola sororia Willdenow	Common blue violet	4	S5

Figure 8 Permanent Photo Point Analysis showing 1/3 open situation burn plots dominated by herbaceous species

Figure 9 Permanent Photo Point Analysis showing 1/3 moderately encroached situation burn plots with a mix of vegetation structure

Encroached Situations

Figure 10 Permanent Photo Point Analysis showing 1/3 encroached situation burn plots dominated by woody species

Glyphosate Treated

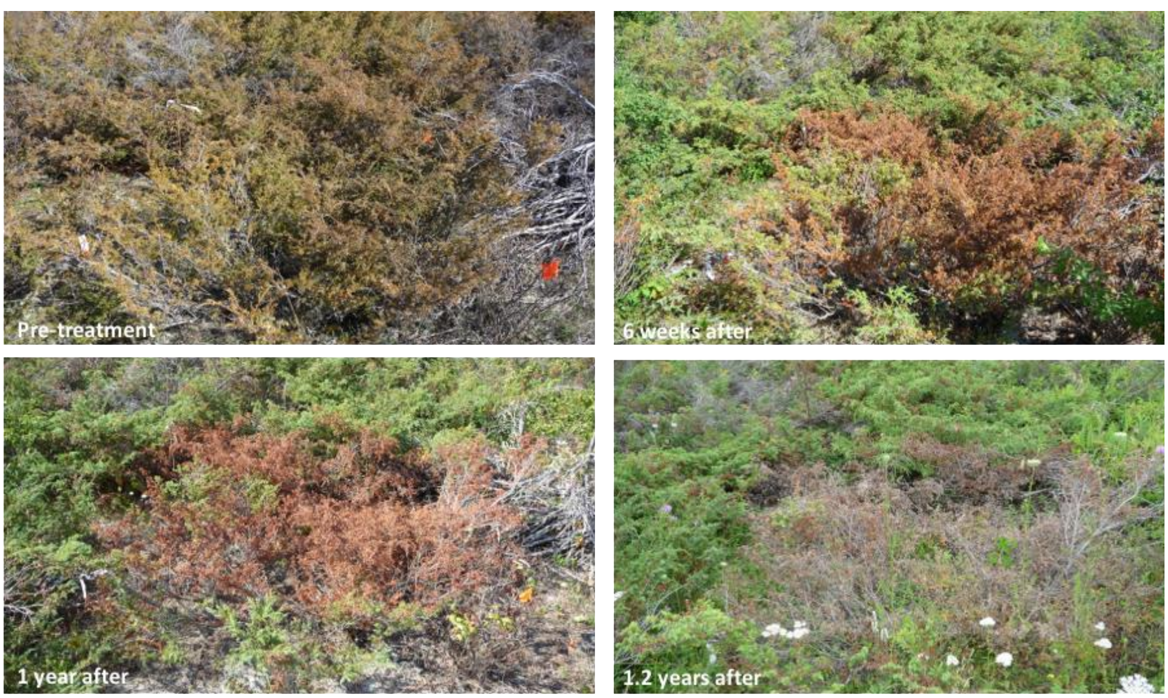

Figure 11 Permanent Photo Point Analysis showing 1/4 glyphosate treated plots

Manual Removal

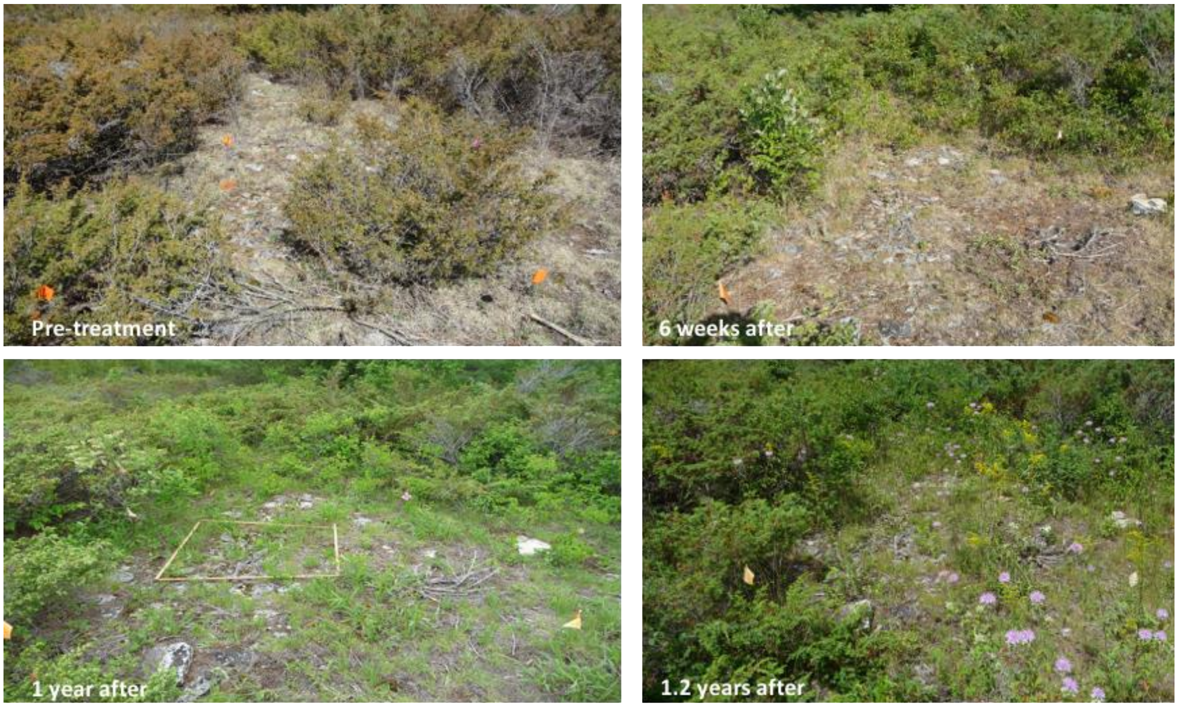

Figure 11 Permanent Photo Point Analysis showing 1/4 manually treated plots