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Jona L. Ensley for the degree of Master of Science in Sustainable Forest Management presented on March 10, 2015.

Title: Comparing Himalayan blackberry (*Rubus armeniacus*) Management Techniques in Upland Prairie Communities of the W.L. Finley National Wildlife Refuge.

Abstract approved:

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John D. Bailey

Dedicated to the preservation and promotion of many of the nation's most threatened and endangered species, the W.L. Finley National Wildlife Refuge is highly invested in the management of some of the last remaining upland prairies once prevalent throughout Oregon's Willamette Valley. More than a century of land fragmentation, fire suppression, and cultivation has shifted species composition and physical structure of the native prairies toward woodland ecosystems, reducing habitat quality and quantity. This transition is greatly driven by the encroachment and introduction of the non-native, woody invader Himalayan blackberry (*Rubus armeniacus*).

This study focused on the three most common Himalayan blackberry management techniques used within the refuge's upland prairies: late summer mowing, late summer burning, and a combination of both. The goal was to analyze the efficacy of each technique in meeting the refuge objectives of controlling Himalayan blackberry populations and promoting native prairie physiognomy after one application using a

complete random block design. Findings were presented to refuge managers to facilitate improved management of these fragile communities.

Mowing was found to have the greatest reduction in blackberry stem and plant density and resulted in the lowest resprout density post-treatment. This treatment also met the refuge objectives of reducing woody cover and increasing graminoid cover, but did little to increase herbaceous cover when compared to the control group. Burning produced no significant increase in herbaceous cover or reduction in woody cover and provided favorable conditions for Himalayan blackberry seedling germination, contributing to a larger blackberry problem in years to come. In promoting herbaceous habitat for upland-dependent species, mowing with subsequent burning was the most successful technique. Though not as effective in reducing blackberry vigor as mowing alone, this treatment showed the most potential of the three in managing for all objectives.

Comparing Himalayan blackberry (*Rubus armeniacus*) Management Techniques in Upland Prairie Communities of the W.L. Finley National Wildlife Refuge

by
Jona L. Ensley

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I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

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Jona L. Ensley, Author

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1. INTRODUCTION

1.1 Willamette Valley Upland Prairies

The upland prairies of Oregon's Willamette Valley are currently among the rarest and most threatened ecosystems in North America. Prairies in this region provided 2/3 of the total habitat prior to Euro-American settlement, but are now limited to less than 1% of their historic area (Christy and Alverson 2011, Wilson 1998, Towle 1982, Johannessen *et al*. 1971, Habeck 1961). Home to a number of the state's most threatened and endangered species listed under the ESA, (golden paintbrush (*Castilleja levisecta*), Kincaid's lupine (*Lupinus sulphureus* var. *kincaidii*), and Fender's blue butterfly (*Icaricia icarioides fenderi*)), upland prairies are focal habitats in the fields of conservation biology and restoration ecology. Absence of the natural disturbance regime that shaped these ecosystems has made them vulnerable to new biological threats and continues to take a toll on their native community structure and species composition. The result is further reduction of historic upland prairie area and quality and diminishing habitat for critically threatened biota.

Before the 1830s, natural and indigenous fires held the Willamette Valley upland prairies in arrested succession, occurring at a return interval of 1-3 years (Storm and Shebitz 2006). Intentional ignitions for land clearing, food crops, and hunting removed the woody species that would otherwise convert the prairies to forests, woodlands, and savannas and prevented the establishment of fire-sensitive exotic species (Boyd 1986). While the high frequency, low severity burns allowed native perennial grasses and forbs to release dormant seeds, woody and non-native seedlings were burned before reaching reproductive and competitive age (Wilson 1998). This disturbance regime created year-round open grasslands with a wide variety of native grasses such as Idaho fescue (*Festuca idahoensis*) and forbs like Virginia strawberry (*Fragaria virginiana*). Such sites were essential for wildlife like the Fender's blue butterfly (*Icaricia icarioides fenderi*) and checkered skipper (*Pyrgus ruralis*). Today, many of the same species continue to depend on upland prairie habitats, preferring the sun-exposed bunchgrass structure of grasslands and to escape from the seasonal flooding and hydric soils of neighboring lowland and wetland prairies (Clark 2000). However, these habitat characteristics are lost if a form and frequency of disturbance similar to the historic regime is not present to reduce woody species cover and prevent intrusion of non-native species.

By the late 1800s, this historic disturbance regime was altered by the introduction of vast land fragmentation, cultivation, fire exclusion, and exotic species by Euro-American settlers to the Willamette Valley. These land use practices led to a decline in upland prairie size, number, and quality (defined as the proportion of total vegetation contributed by native species) (Johannessen *et al*. 1971, Habeck 1961, Towle 1982, Wilson 1996). Many prairies were converted to agricultural land for food crops or other economic profit while others were neglected and transitioned to dense woodlands and forests of Oregon white oak (*Quercus garryana*) and Douglas fir (*Pseudotsuga menziesii*) that prove inhospitable for many upland prairie species. With the new and infrequent disturbances, introduced exotic species easily outcompeted native inhabitants for vital resources. This was the case for the aggressive non-native tall oatgrass (*Arrhenatherum* *elatius*) and soft brome (*Bromus mollis*), which usurp water and nutrients needed for native plant growth in the upland prairies (Wilson 1998). Without the continuous disturbance under which these species evolved, native plant and animal populations declined along with prairie area (Wilson 1996).

Today, land surveys have shown only 400 hectares of native upland prairie remaining (USFWS 2000). Continued land conversion, rural development, and neglect continue to threaten the remaining historic prairies. Urbanization of the Willamette Valley is expected to increase these activities with a projected population over 4 million people by the year 2050 (ODFW 2006, Hulse *et al.* 2007). Further reduction in upland prairie area induced by this increase in population will likely lead to the extinction of currently threatened and endangered species under the ESA and federal listing of many more.

1.2 Invasive Species

Among the greatest threats to upland prairies is the introduction and spread of invasive species, especially those of non-native origin. While weeds are plants that grow where they are unwanted, McDowell (2002) defines an invasive as a weed that is first introduced, becomes established, and begins to reproduce and persist in a new habitat without human assistance. With time, an invasive is able to compete with other species for vital resources and can often become dominant within the plant community. Often, an invasive species will possess characteristics that enable it to inhabit unoccupied niches within a habitat (Radosevich *et al.* 2003). Under this definition, native species can also be considered invasive, however, invasions by non-native species are considered more aggressive and detrimental to communities (Randall 1996).

The invasion process is comprised of three stages: introduction, colonization, and naturalization (Cousens and Mortimer 1995). Introduction occurs when a species arrives at a location beyond that of its original geographic range and establishes itself as a population with reproductive capabilities. After successful introduction, invasive spread typically occurs as either a single front progression from the source population or by dispersal and establishment of multiple satellite populations, with the latter being more common (Radosevich *et al.* 2003, Baker 1986, Moody and Mack 1988). Colonization occurs as the initial and satellite populations become self-perpetuating and the number of individuals per area or in a population exceeds deaths (Radosevich *et al*. 2003). If a species successfully colonizes a site, occupies all possible niches, and both area and density of populations are unchanging, naturalization has been reached (Cousens and Mortimer 1995).

Though it is difficult to scientifically specify characteristics of a site's invasibility (vulnerability to invasion), certain habitats and the plant communities within them are hypothesized to be more easily invaded than others; grasslands, riparian areas, areas in close proximity to other invasions, and locations of high human perturbation and disturbance being among the more susceptible (Baker 1986). Radosevich *et al.* (2003) suggest that an area's evolutionary history, community structure, propagule pressure, disturbance regime, and levels of stress can all be considered factors of invasibility. At a smaller spatial scale, a community's number and quality of safe sites can increase an invasive's ability to germinate, survive, and reproduce (Radosevich *et al.* 2003). As a system characterized by grass species of low structural complexity (Clark and Wilson 2005), upland prairies are considered to be at high risk to invasion.

The invasiveness of the species itself is also considered a factor of successful invasions. Some common traits of highly weedy plants include multifaceted reproduction, adaptation to environmental stress, and the ability to thrive in a wide range of environments (Baker 1965). As a subset of weeds, Williamson and Brown (1986) add that invasive species may possess only a few or all of these traits. Some of the invasive species within upland prairies also possess more efficient growth characteristics than those of the native species within their biological niche (McDowell 2002). One example of this is Himalayan blackberry (*Rubus armeniacus*) growth in communities where its native counterpart, trailing blackberry (*Rubus ursinus*), historically grew.

1.3 Himalayan blackberry (*Rubus armeniacus*)

1.3.1 Introduction

Himalayan blackberry was intentionally introduced to the United States in the 1880s. It is assumed that an east coast settler, Luther Burbank, brought the species from Western Europe or acquired seeds from an unidentified Indian catalog and cultivated it for its sweet fruit, advertising it as the "Himalayan giant" (Francis 2014). Though traced back to Armenian origins, the plant is referred to by its misnomer, Himalayan blackberry.

Due to its advanced growth capabilities, the species escaped cultivation and expanded its distribution to the west coast of the U.S. and into British Columbia by 1945 (Soll 2004). Many other geographic regions like Hawaii, Europe, Australia, New

Zealand, and South Africa also currently have issues with Himalayan blackberry invasion (USDA 2008).

1.3.2 Morphology

Easily identifiable by sharp and stiff spines, individual Himalayan blackberry shoots (canes) can reach six to twelve meters horizontally and three meters vertically (Francis 2014). These biennial, partially woody canes are able to produce dense thickets of up to 525 canes per square meter and can live two to three years before senescing (Soll 2004). First year canes (primocanes) are infertile, consisting of leaves only, while second year canes (floricanes) develop from inside primocane axils and produce fruit, flowers, and leaves (Tirmenstein 1989).

Leaves are sharply toothed and typically compounded with five leaflets, atypically with three leaflets. The white or rose-colored flowers of floricanes generate from the lower axils in large terminal clusters (Starr *et al*. 2003) and typically have five petals transversely arranged in groups of five to twenty per flat-topped panicle (Hoshovsky 2000). Flowering season takes place between June and August with fruit remaining viable until September (Soll 2004). These fruit are large, aggregate, shiny, black drupelets that range in size from 12-20mm, with each drupe containing a single seed (Francis 2014).

Belowground, root crowns can reach 20 cm in diameter with rhizomes growing directly from each crown at depths of up to 1.5 meters into the soil. Roots growing laterally from the root crown can attain outward growth of 30-60 cm (Francis 2014).

Adventitious shoots, also called suckers, are occasionally formed on the roots and are able to emerge from depths up to 45 cm (USDA, NRCS 2008).

1.3.3 Ecology

Considered both a semi-woody and invasive species, Himalayan blackberry is one of the biggest threats to native upland prairies in the Willamette Valley of Oregon. The structural support provided by the partially lignified stems allows this species to occupy aboveground space that few of the native herbaceous and graminoid species are able to utilize. Further fueled by its invasive nature, Himalayan blackberry has become naturalized in the Pacific Northwest, ranging from California to British Columbia.

Himalayan blackberry can tolerate a wide range of soil conditions and types, growing even in infertile soils up to 1800 meters above sea level. The most limiting factor of its growth is hypothesized to be soil moisture (Francis 2014); it is present in both acidic and alkaline soils with more than 76 cm of annual rainfall, but cannot persist in true wetland areas, though temporary flooding is tolerated (Bennet 2006). It also does not grow as vigorous under dense canopies with limited sunlight than in fully exposed areas (Francis 2014, Jones 2004).

Himalayan blackberry also has a higher photosynthetic capacity and water-use efficiency than the two native *Rubus* species, black raspberry (*Rubus leucodermis*) and trailing blackberry (*Rubus ursinus*), allowing it to fix higher amounts of carbon than the two natives (McDowell 2002). A rapid growth rate and ability to outcompete native species after disturbances puts Himalayan blackberry within Grime's (1988) stress-tolerant competitor (C) category of the C-S-R model.

1.3.4 Growth and Reproduction

The maximum relative growth rate of the Himalayan blackberry primocanes is 0.104 g/g/day, which gives it a type of growth more similar to an herbaceous than woody plant (Amor 1974). The invasive ability of this species is well explained by the floricanes' ability to reproduce by taking root via the tip, creating a "daughter plant". This method of reproduction can result in a dispersal distance of three meters from the parent plant (Soll 2004).

Another major issue with the spread of this species is that root fragments and cane cuttings are able to establish new plants. In one case, a single cut has been able to produce a thicket five meters in diameter in just two years (Bennet 2006). In these cases, dispersal distances are dependent upon how far the cuttings are transported, often by mechanical cutting equipment.

While the majority of Himalayan blackberry growth is via asexual methods, it also spreads sexually with rapid production rates of up to 720 fruits per cane (Caplan and Yeakley 2006). Fruits from a single m^2 can contain 7,000-13,000 seeds with 7-10% viable to germinate. Of the viable seeds, 5% are able to germinate in the year after production and 10-17% germinate after two years (Hay 2012, Amor 1974, Northcroft 1927). However, the number of seeds and seedling recruitment are reduced when thickets are subjected to large amounts of shading (Bennet 2006, Jones 2004). Seedling recruitment is also lower in dense, mature thickets (Bennet 2006).

Seeds can be viable for several years as the seed coat is impermeable and the embryo is dormant until the coat breaks, causing slow germination that has been shown to increase by 17-30% after ingestion by birds and mammals (Francis 2014, Bruzzese 1998, Northcroft 1927). As a result, germination via seed dispersal is increased by birds, small mammals, and water (Soll 2004) as well as through human contact (Alaska Natural Heritage Program 2005). In these cases, dispersal distance is dependent upon the range of the animal.

1.3.5 Economic Impact

Due to the biological traits that allow it to outcompete most native North American species, Himalayan blackberry has degraded many natural ecosystems and interfered with agriculture, silviculture, and many restoration projects. This has led to crop damage and, in most cases, loss of money. Thickets of Himalayan blackberry can be vectors for diseases in many commercial plants (Hoshovsky 2000). As an example, the bacterial pathogen, *Xylella fastidiosa*, that causes Pierce's disease in grapevines is hosted and spread by Himalayan blackberry (Hill 2003).

There are a few cases, however, where the growth and spread of Himalayan blackberry can be beneficial. The berries produced by the canes are a food source for many birds and small mammals such as fox, squirrels, and beaver, and larger mammals such as deer, elk, and bear (Alaska Natural Heritage Program 2005). Honeybees are also known to use Himalayan blackberry as a nectar source for the production of honey and the fruit is often harvested in Oregon and Washington for preserves (Francis 2014). Though there is not much of an industry specifically for Himalayan blackberry fruit, this species often grows in association with evergreen blackberry (*Rubus laciniatus*), which is an important commercial, but also invasive plant in Oregon.

1.3.6 Listing

Himalayan blackberry is currently considered a naturalized invasive in the state of Oregon and listed as a species of concern in both Oregon and California (Caplan and Yeakley 2006). Assessment of Himalayan blackberry using the "criteria for categorizing invasive non-native plants that threaten wildlands" by the California Exotic Pest Plant Council and the SW Vegetation Management Association has resulted in an invasive score of "High" (Warner 2004), indicating its "severe" threat to native species and communities, high invasiveness, and vast distribution. The increasing abundance in Oregon has led to its inclusion in estimates of the state's primary productivity (Law and Waring 1994).

Under the Oregon Department of Agriculture's (ODA) Noxious Weed Policy and Classification System, Himalayan blackberry has been deemed a Class "B" weed, meaning it is a regionally abundant weed of economic and ecological importance that deserves intensive control at a site specific, case-by-case basis. ODA considers this plant beyond the feasibility of a statewide management plan given the difficulty of control (ODA 2014).

1.4 Management Methods

There are numerous methods currently in use to control the spread of Himalayan blackberry. Some methods have been more successful than others and a few have proved to do more damage than good, given the plant's complex biology. Methods include mowing, herbicide spraying, burning, browsing, and use of foliage-killing fungi. Often, a combination of these treatments is implemented to provide multifaceted disturbance. In choosing a treatment, the level of invasion is a determining factor and, more often than not, only control of Himalayan blackberry spread can truly be achieved with persistent and repeated treatments.

1.4.1 Mechanical Treatments

The use of mowing to cut the base of Himalayan blackberry canes in highly dense thickets has been effective in keeping spread low, but only when done multiple times throughout the year. Mowing is typically done between 8-20cm from the ground, depending on density of canes and the capability of the machine. This method is most effective in the summer growing season when resources are allocated to the aboveground biomass (Thorpe *et al.* 2008). Mowing treatments are focused on the reduction of the canes and do little to mitigate growth and spread belowground. Mowing is also a method that has been said to increase the presence of Himalayan blackberry plants due to asexual reproduction from cut plant fragments (Bennet 2006). Often, the mowing implement or the wheels will produce bare ground that creates additional space for these fragments to establish, though supporting data for this phenomenon is scarce. Another mechanical management option is to cut canes with hand tools and/or hand-pull the crowns during the growing season. Subsequent disposal of the blackberry biomass by burning is recommended (Soll 2004).

While very feasible options, these methods are risky endeavors as they all have high potential of leaving fragments on site for new plant establishment. Also, these methods are highly laborious and require much time to implement effectively. In general, 300-1000 hours of labor would be required to remove approximately one acre of densely infested Himalayan blackberry (Soll 2004). This, in turn, requires a good source of funding for personnel and equipment.

1.4.2 Herbicide Treatments

While often a controversial method, herbicide spraying is one of the most effective ways to reduce the cover of Himalayan blackberry from invaded sites (Bennet 2006). The compounds within herbicides are able to chemically alter the physiological processes within the plants, thus reducing growth ability and promoting mortality.

Particular herbicide brands are more effective than others in attaining control objectives of different Himalayan blackberry organs. Garlon® 3A and 4 (triclopyr) and Roundup® (glyphosate) are more successful in killing mature and new canes when applied in the fall season (Soll 2004). Herbicides containing picloram are more efficient in repressing the regrowth of the canes in newer patches, but also stimulate adventitious shoot growth from ground roots (Hoshovsky 2000). Application of herbicides is most effective if done in late summer to fall when the potential for translocation of chemical from foliage to roots is highest (Bennet 2006, Soll 2004).

An issue with herbicide spraying for mitigation of Himalayan blackberry is the high potential of the herbicide chemicals to leach into soil and water tables, which could have detrimental effects on neighboring vegetation and wildlife. For example, it has been shown that herbicides have adverse effects specifically on the larvae of the endangered Fender's blue butterfly (*Icaricia icarioides fenderi*) (Wilson and Clark 1997). Also, certain herbicides target broad groups of plants such as broadleaf weeds or woody plants. Consequently, targeting the invasive species itself is difficult without impacting surrounding plants in the same group (Rinella *et al.* 2009) If application is done incorrectly, the potential of these effects increases, thus, this method should be used with great caution, especially in areas with sensitive species.

1.4.3 Burn Treatments

Burning is likely the most cost effective and least labor-intensive method for Himalayan blackberry control. Used alone, however, this method does little for long term management (Bennet 2006). Burning can be very efficient in reducing the majority of the aboveground biomass, but, similar to the mowing treatments, results in areas of bare soil and therefore leaves high potential for Himalayan blackberry re-invasion and cover increase (Soll 2004). Like most other *Rubus* species, Himalayan blackberry resprouts vigorously after fire, especially from underground rhizomes, often creating more of a blackberry issue than initially present (Willoughby and Davilla 1984).

1.4.4 Browsing and Grazing Treatments

Another method used for Himalayan blackberry control is browsing and grazing by goats, sheep, cattle, and in a few cases, horses (Holshovsky 2000). Given the inhospitable thorns of blackberry, utilization of this method is often reserved for sites with higher cane density. Goats are best suited for this kind of blackberry management, as they prefer fibrous woody plants (Hodgson 1990). Most animals will consume the available fruit and foliage of Himalayan blackberry, but goats will also consume young canes, reducing stem and node density. However, an increase in primocane density and seedling establishment is also possible, indicating continued population vigor (Ingham

2014). Another issue with this method is that the consumption of Himalayan blackberry seeds by the animals could potentially increase blackberry's reproductive success (Soll 2004). This could lead to increased blackberry cover within the invaded site or new sites if the animals are moved.

1.4.5 Biological Treatments

A biological method that is currently in the experimental stage in western Oregon is the use of a fungus called blackberry leaf rust (*Phragmidium violaceum*) that affects the leaves of Himalayan blackberry and causes the plant to almost completely defoliate. This induces plant mortality and is also shown to reduce the occurrence of tip-rooting (Bennet 2006). However, the rust needs a great deal of time to spread before enough is present to overtake as much of the foliage as needed to induce mortality. Some Himalayan blackberry plants have also proved resistant to the rust for reasons yet to be discovered (Peters 2012) and most managers are hesitant to introduce this biocontrol to areas without knowledge of impact on other species. It is also illegal to introduce biocontrols such as this without permission and supervision from the Oregon Department of Agriculture.

1.4.6 Combined Treatments

Many strategies aimed at controlling Himalayan blackberry involve combinations of the above treatments. Each individual treatment focuses on a particular method of spread; allowing combinations of these strategies to increase treatment efficacy and likelihood of attaining management objectives. There is a multitude of treatment combination possibilities involving different seasonal timing, intensity of treatment, and sequence of treatment combinations. However, it must be noted that all treatments, combinations included, are typically ineffective unless repeated seasonally or annually (Bennet 2006, Soll 2004).

1.5 William L. Finley National Wildlife Refuge

As a site of some of the last remaining upland prairies in the Willamette Valley, the William L. Finley National Wildlife Refuge is a focal conservation location. Each year, efforts are made to clear encroaching woody and invasive species from the refuge's historic upland prairies and to protect the areas that have been successfully reclaimed. According to its Comprehensive Conservation Plan (CCP), a primary goal of the refuge is to "protect, maintain, enhance, and restore the native upland prairie/oak savannah habitats characteristic of the historic Willamette Valley" with attributes of these habitats including "<5% cover of non-natives of particular concern, a mosaic of low growing native grasses, native forbs, and bare ground with an absence of dense canopy vegetation, <10% canopy cover of shrubs, and <20% cover of invasive shrubs (e.g., Himalayan blackberry)" (USFWS 2011, Chapter 2, page 29, 30, 31). Strategies developed to achieve these objectives include controlling invasive/non-native species using mechanical treatments, such as mowing, and use of prescribed fire on a one to three year rotation (USFWS 2011).

1.6 Research Needs and Study Goals

With Himalayan blackberry control and upland prairie management being the focus of many conservation efforts, the U.S. Fish and Wildlife Service and other agencies throughout the Willamette Valley such as the Oregon Department of Fish and Wildlife, The Nature Conservancy, Bureau of Land Management, Institute for Applied Ecology, and others are greatly invested in research about the efficacy of Himalayan blackberry control treatments in reducing population vigor as well as promoting native prairie physiognomy (community structure and composition). As the methods most often used on the W.L. Finley National Wildlife Refuge to achieve these objectives, late summer mowing, late summer burning, and a combination of the two are the focus of this thesis.

Previous studies involving these techniques within the upland prairies of the Willamette Valley include an analysis of their efficacy in reducing the percent cover of Himalayan blackberry (Thorpe *et al.* 2008). While this metric is a good indicator of improved community structure, it provides little detail in how the blackberry population has been impacted. Though overall cover of Himalayan blackberry may decrease, it is highly likely that new seedlings and resprouting occurred, indicating a younger, but just as vigorous invasive population (Dennehy *et al.* 2011). However, if one technique results in a reduction of stem and plant density as well as fewer seedlings than other techniques, this technique could be suggested as a better tool in managing the blackberry-invaded upland prairies on the refuge. Given this, it is within the interest of the refuge and other land managers to understand the relationship between these techniques and Himalayan blackberry populations along with their surrounding plant communities. This study is intended to supply these land managers with scientific conclusions pertaining to Himalayan blackberry population dynamics and plant community functional groups response to their commonly used management techniques.

In the interest of analyzing the life history characteristics of Himalayan blackberry, Chapter 2 includes a comparison of three treatments (mow only, burn only, and mow then burn) against a control (no treatment) to show efficacy of the techniques. A comparison of every treatment against each other is also included. The questions of interest include the following: How are Himalayan blackberry population dynamics (change in stem density and plant density, average seedling density and resprout density) impacted by each treatment? Which method is best suited for reducing Himalayan blackberry vigor?

To investigate the community functional group response, Chapter 3 focuses on the changes in graminoid, herbaceous, and woody species cover. Questions of interest in Chapter 3 include: How is the percent cover of the functional groups impacted by each treatment? Which treatment provides the greatest change toward a more native prairie structure? Combined with Chapter 2, Chapter 3 will provide the components for a comprehensive discussion in Chapter 4 of the suggested management strategies to be used in the upland prairies as they pertain to the refuge's conservation objectives as outlined in their CCP.

2. HIMALAYAN BLACKBERRY (*RUBUS ARMENIACUS*) POPULATION RESPONSE TO MOW AND BURN TREATMENTS IN THE UPLAND PRAIRIES OF THE W.L. FINLEY NATIONAL WILDLIFE REFUGE

2.1 Abstract

Himalayan blackberry (*Rubus armeniacus*) is a non-native invasive shrub contributing to woody species encroachment into the fragile upland prairies in the Willamette Valley of Oregon. Controlling this species within the upland prairies of the W.L. Finley National Wildlife Refuge has proven difficult for refuge managers given the perennial shrub's aggressive invasion characteristics including various modes of reproduction, a wide range of environmental tolerance, and a positive response to disturbance. Refuge managers are responsible for the revitalization of federally listed species and conservation of the habitats that support them and are thus interested in understanding the response of Himalayan blackberry populations to a few of the refuge's most commonly used treatments: late summer mowing, late summer burning, and a combination of the two.

Measurements of blackberry stem density and plant density (number per m^2) were taken in spring of 2013, prior to treatment application in the same year. Post-treatment measurements were taken in spring of 2014 for comparison; including the same density metrics as well as other population structure variables like blackberry seedling and resprout density. The averages of these response variables were then compared among the three treatments and to a control with no manipulation to analyze treatment efficacy.

The mow only treatment was the most beneficial of the three in meeting refuge objectives pertaining to Himalayan blackberry (i.e. significantly reducing stem density).

This treatment also stimulated significantly fewer post-treatment resprouts compared to both alternatives. The mow then burn treatment showed an insignificant number of seedlings in the following growing season compared to the mow only treatment. The burn only treatment significantly increased seedling presence, indicating its negative influence in meeting refuge objectives and showing high potential for future spread and invasion.

Based on these results, the use of late summer mowing as means of controlling Himalayan blackberry populations is highly recommended on the W.L. Finley National Wildlife Refuge and in similar upland prairies within the Willamette Valley. Mowing with subsequent burning has the potential to provide greater control of seedling establishment and prevent a future increase in plant and stem density than the mow only treatment, but would require further investigation to warrant recommendation. Late summer burning is not advisable by itself, given the potential to create larger blackberry populations in the future. All results will assist land managers in deciding the best management options for controlling Himalayan blackberry spread in the fragile upland prairies.

2.2 Introduction

Himalayan blackberry (*Rubus armeniacus*) is one of the most aggressive invasive species in the Pacific Northwest (Caplan and Yeakley 2006) and poses a serious threat specifically to the vulnerable upland prairies of the Willamette Valley of Oregon. This semi-woody, non-native blackberry is currently invading the once predominant habitat type and altering the natural community structure upon which many endangered and threatened species including Kincaid's lupine (*Lupinus sulphureus* var. *kincaidii*) and

Fender's blue butterfly (*Icaricia icarioides fenderi*)) depend. With only 400 hectares of these habitats remaining within the pre-settlement range (Johannessen et al. 1971, Habeck 1961, Towle 1982, Wilson 1998), it is essential to analyze the efficacy of the management strategies used to control the spread of the species that threaten them.

Himalayan blackberry was intentionally introduced to North America in the 1880s for cultivation, but escaped human control and invaded the West Coast from Northern B.C. to California by 1945. It is now present in a large range of environments from semi-hydric to mesic soils, sea level up to 1830m in elevation, forest to prairie ecosystems, and in disturbed habitats where it quickly establishes before native competitors (Kearney *et al.* 1960). Himalayan blackberry invasion in the United States has greatly affected the economies of agriculture and forestry, as well as hindered the efforts of conservation and ecological restoration; contributing to the annual cost of invasive species management in the billions of dollars (NISC 2006).

Himalayan blackberry spreads both sexually and asexually once established. Clonal reproduction methods include taking root via tip of a mature cane (tip-rooting), as well as sprouting from rhizomes and fragments of cut plants (Soll 2004). Sexually, seeds are dispersed via gravity, wind, and wildlife that utilize the tall, dense thickets of canes for shelter, nesting, and foraging. After digestion, seeds become up to 30% more viable than if dispersed by other methods (Francis 2014, Bruzzese 1998). Once a population is established, spread can occur either linearly with growth from a single locus, or via satellite populations, with the latter being more common and resulting in a larger and more vigorous invasion (Radosevich *et al.* 2003, Baker 1986, Moody and Mack 1988).

Listing of Himalayan blackberry as a Class “B” Noxious Weed by the Oregon Department of Agriculture (ODA 2014) indicates its advanced invasive biology, difficulty of control, and the need for intensive control. Methods of controlling Himalayan blackberry throughout the Willamette Valley have included the use of herbicide, mowing, burning, animal browsing, hand pulling, and accidental introduction of defoliating fungi. While each method provides specific forms of control such as reduction of aboveground biomass or physiologic alteration of growth processes, no single method is supported for use in all invaded environments. The Oregon Department of Agriculture (2014) suggests that control of Himalayan blackberry be analyzed on a site-by-site basis as methods of invasion can vary by location.

Regardless of scale, Himalayan blackberry control is a poorly understood and seldom effective endeavor. Control methods typically focus on a single form of the plant’s reproduction, impacting part of the plant’s morphology but increasing vigor in others (e.g. seedlings and resprouts). Very little is known about the resulting population structure produced by commonly used treatments (Baker 1986), though it provides the opportunity to observe the potential for post-treatment spread and future invasion. Ingham (2014) contributed information about blackberry demographics after treatments of goat browsing, mowing, and a combination of the two, but it is worth investigation to evaluate if similar results occur after other forms of disturbance.

As a site of some of the last remaining upland prairies in the region, the W.L. Finley National Wildlife Refuge in Corvallis, OR, is a focal location for attempts to control the spread of Himalayan blackberry. Refuge objectives outlined in the

Comprehensive Conservation Plan specifically indicate the need to keep percent cover of invasive shrubs like Himalayan blackberry to less than 20% in the upland prairies (USFWS 2010). The primary management methods used by the refuge currently include late summer mowing, late summer burning, and a combination of both.

To date, evaluation of the refuge's Himalayan blackberry control techniques has been solely focused on aboveground response (e.g. change in percent cover, density, etc.). This study attempts to evaluate the impact of these methods on the post-treatment population dynamics of Himalayan blackberry that will allude to the potential for future spread and control success. More specifically, the questions of interest include: How are stem and plant densities altered by each treatment? What is the resulting population structure (seedling and resprout density) after each treatment? How do the treatments compare in their ability to meet refuge objectives? Analysis of these questions will provide refuge managers with scientific support for the continued use of certain treatments on the refuge as well as an understanding of which treatment contributes most to achieving specific management objectives.

2.3 Materials and Methods

2.3.1 Study Area and Design

The W.L. Finley National Wildlife Refuge is located 20 miles south of Corvallis (Figure 2.1) in the Willamette Valley of Western Oregon. The climate is wet and cool in the winter, and warm and dry in the summer with an annual high of 17.4 ºC, low of 5.5ºC, and rainfall of 1086mm (US Climate Data 2015). As loamy, well-drained soils are main indicators of upland prairies in the Willamette Valley (Wilson 1998), four sites with these soil types were selected within the refuge (Figure 2.1). Jory silty clay loam, Willamette silt loam, Dupee silt loam, and Hazelair complex soils were chosen based on upland prairie soil lists and GIS layers provided by the Natural Resource Conservation Service (2003). Dominant plants within the communities were fairly similar, all incorporating low-structured grasses and forbs such as tall fescue (*Schedonorus arundinaceus* (Schreb.) Dumort), silver hairgrass (*Aira caryophyllea*), bull thistle (*Cirsium vulgare*), and common sheep sorrel (*Rumex acetosella*). Sites ranged in elevation (92-132 meters above sea level), aspect, and slope (3-12 degrees). All sites had been managed with mowing and burning on a three-year rotation as per the CCP since the 1990s, but varied in time since management from one to three years and in method of management.

Three blocks per site were placed in areas with 20-80% blackberry cover, representing the range of blackberry invasion levels that the refuge sees fit to combat with the treatments of interest. Individual blocks were separated by at least 5 meters to maximize variation and had dimensions of 50x10m. Each block included four treatment plots of 10x8m separated by three 10x6m buffers (Figure 2.2) to reduce edge effects. Vegetation was sampled from these treatment plots with randomly located 1x10m belt transects delineated by two wooden stakes at each end (Grant *et al.* 2004). Belt transects were divided into ten 1m^2 quadrats and response variables were measured within each, then averaged at the plot level. Total area surveyed within each plot accounted for 12.5% of the treatment plot. Pre-treatment measurements were taken in June of 2013 during the beginning of the growing season.

2.3.2 Treatments

Each block received a complete set of all treatments: a control with no manipulation, a mow only, a burn only, and a mow then burn treatment. Treatments were randomly assigned to the four plots to allow for a complete random block design with twelve replicates per treatment. Mowing was completed on August 19th and 20th of 2013 when the majority of blackberry growth resources were allocated aboveground for flowering, fruit, and leaf production. A Bobcat® skid steer with a brush mower was used to cut plants 30cm from ground level. Mow only and mow then burn treatment plots were treated along with a 2m-perimeter buffer around the blocks to provide control lines for the subsequent burn treatments. Mowed debris was left undisturbed within all plots, as per common refuge procedure. Plots were burned on August 27th and 28th of 2013 after all debris had cured in the mow then burn plots, and when standing vegetation in the burn only plots had low moisture contents (Table 2.1). Backing fire ignition techniques were utilized to simulate the typical fire behavior and firing operations of prescribed fires on the refuge. Post-treatment measurements were taken in June of 2014, one year after the initial measurements.

2.3.3 Response Variables

Himalayan blackberry is known to respond to disturbance in multiple ways including: crown resprouting, tip-rooting, establishment of new plants from seed or cut plant fragments, rhizomatous sprouting, cane branching, and cane/crown subsistence (Amor 1974). As all treatments in this study focus on aboveground removal or high levels of physical alteration of blackberry canes, tip-rooting and cane branching were not observed or measured, as these require mature and intact canes. It is assumed that all remaining disturbance responses were a result of regrowth from belowground resources or establishment of new plants from seed.

Initial measurements taken within each quadrat in the 2013 field season included blackberry stem and plant density (number per m^2). Stem density encompassed all observed facets of blackberry growth (e.g. seedlings, crown sprouts, rhizomatous sprouts, and canes). Plant density grouped stems into clusters of two or more, seemingly growing from a single plant crown, as individual blackberry plants typically produce two new stems (primocanes) per growing season and retain the prior growing season's stems (floricanes) for sexual reproduction (Amor 1974). Some floricanes can persist for longer than two years, leading to observations of single plants with more than 18 stems. Given their ability to produce floricanes in the next growing season, crowns with canes, seedlings, and rhizomatous sprouts were considered individual plants. Together, stem and plant density represented the initial state of blackberry populations prior to treatment application. These same variables were measured again in June of 2014 to compare treatment efficacy.

Blackberry stem and plant density variables are valuable visual indicators of the efficacy of treatments in reducing population size and density; however, they suggest little about the potential for future population spread. Thus, in addition to the stem and plant densities, seedling and resprout density were recorded in each quadrat in the 2014 field season to bolster assessment of treatment efficacy. Seedlings were considered to be new plants that had sprouted from seed after treatment, while resprouts were individual stems growing from established plants as indicated by the presence of a crown. Resprout density indicated the potential for future invasion of surrounding areas via tip-rooting and seed production. Seedlings have the potential to become individual mature plants, adding to the plant and stem density of existing populations and serving as an undesirable response to treatments. Success of the treatments was judged based on low densities of seedlings and resprouts.

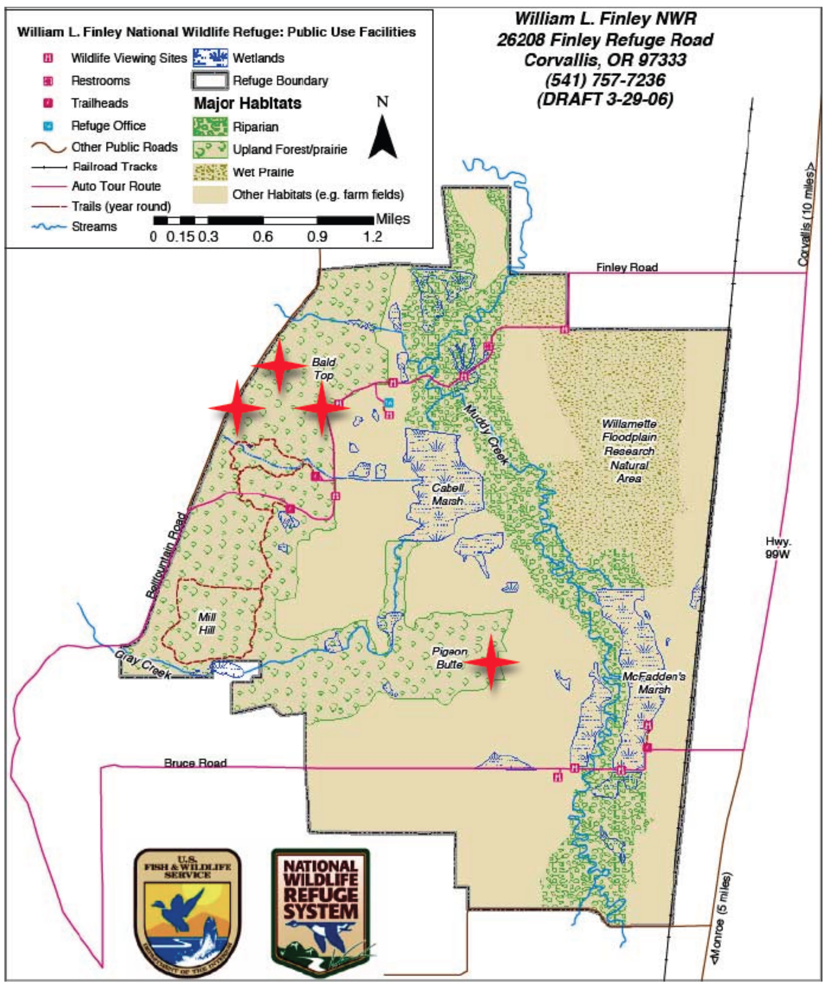

Figure 2.1 Map of location of W.L. Finley National Wildlife Refuge provided by the U.S. Fish and Wildlife Service. Red stars denote location of four study sites in four soil types.

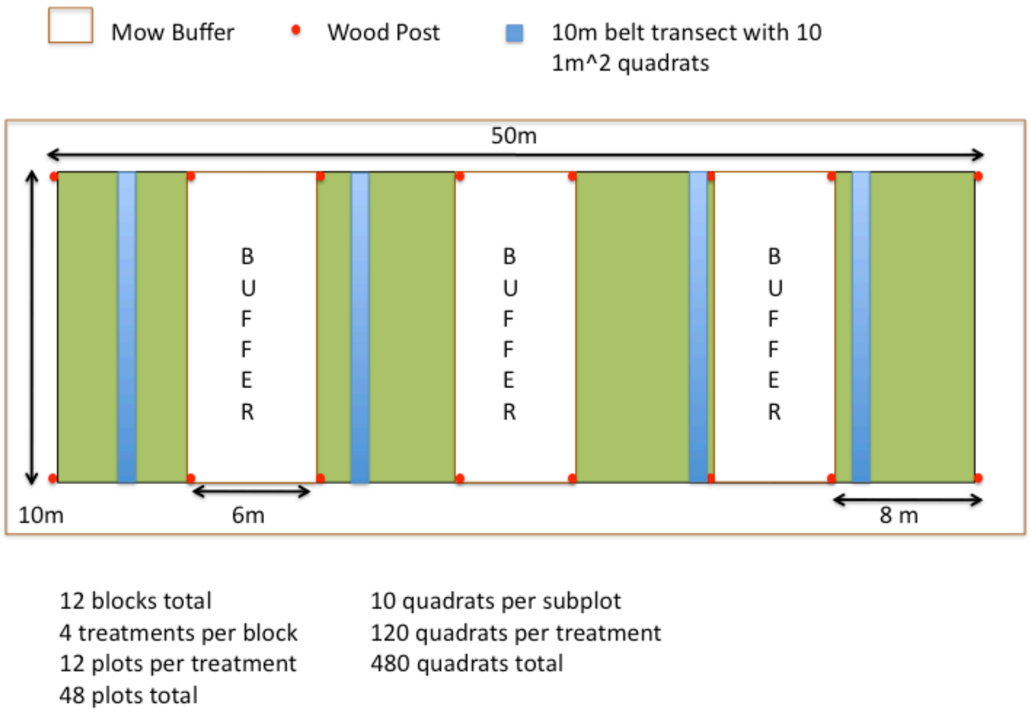

Figure 2.2 Example of an individual block used in the study design. Totals of each level of study are also provided.

Table 2.1 Weather and fuels conditions during burn operations. Weather data was provided by the Remote Area Weather Station (RAWS) located at W.L. Finley NWR. Fuel models were selected according to parameters outlined in Scott and Burgan (2005).

Treatment	Date	Fuel Model	Temp Range (C)	RH Range (%)	Wind Range (kph)	10 hr Fuel Moisture Range (%)
Burn	8/27/13	GS4	29-31	34-43	0-8	11-12
	8/28/13	GS4	27-32	38-44	0-6	10-21
Mow+Burn	8/27/13	GS3	29-31	34-43	0-8	11-12
	8/28/13	GS3	27-32	38-43	0-6	10-21

2.3.4 Statistical Model and Analysis

Treatments were compared using the following general linear model for each response variable:

$$Y_{ijk} = \beta_0 + \beta_1(x_b)_{ijk} + \beta_2(x_m)_{ijk} + \beta_3(x_{mb})_{ijk} + b_j + c_k + \varepsilon_{ijk}$$

where

Y_{ijk} is the plot average for the response variable (change in stem density, change in plant density, seeding density, and resprout density) in i^{th} treatment in j^{th} block in the k^{th} site, i=control, mow only, burn only, mow then burn, j=1,2,3, k=1,2,3,4,

β_0 is the mean average of the response variable for the control treatment,

β_1 is the incremental effect of the burn only treatment on the mean average of the response variable,

β_2 is the incremental effect of the mow only treatment on the mean average of the response variable,

β_3 is the incremental effect of the mow then burn treatment on the mean average of the response variable,

x_b is 1 when the i^{th} treatment is burn only and 0 otherwise,

x_m is 1 when the i^{th} treatment is mow only and 0 otherwise,

x_{mb} is 1 when the i^{th} treatment is mow then burn and 0 otherwise,

b_j is the random effect of the j^{th} block on the average of the response variable, $b_j \sim N(0, \sigma^2)$ and $Cov(b_k,b_{k'})=0$

c_k is the random effect of the k^{th} site on the average of the response variable, $c_k \sim N(0, \sigma^2)$ and $Cov(c_k,c_{k'})=0$

ε_{ijk} is the residual error term for the i^{th} treatment in the j^{th} block in the k^{th} site, $\varepsilon_{ijk} \sim N(0, \sigma2)$, and $Cov(\varepsilon_{ijk},\varepsilon_{ij'k'})=0$, and b_j, c_k, ε_{ijk} are all independent

To calculate the change in stem density and change in plant density, the counts of stems and plants of 2014 were subtracted from those of 2013. Average seedlings and resprout densities were calculated from only the 2014 counts and a factor accounting for the 2013 plant density was added as a covariate to the above model to compensate for the relationship between initial density and resprout and seedling density; it is expected that higher plant densities will influence resulting resprout and seedling densities. The statistical program RStudio version 0.98.490 (2013) was used to conduct each statistical analysis. All pairwise comparisons of treatments were made while ensuring control of the False Discovery Rate (FDR) as outlined in Verhoeven *et al.* (2005). After adjustment for pair-wise comparison, statistical significance was examined at the 95% confidence level.

To maintain the assumptions of the models, normality was checked with normal probability plots. Homogeneity of variances was checked using scatterplots of residuals and independence within and between sites and blocks was present during the design and data collection period. With the given elements of randomization, replication, and representation, the scope of inference of this study is limited to the refuge upland prairies, but could be extrapolated to Willamette Valley prairies of similar soil types and community composition.

2.4 Results and Discussion

2.4.1 Stem Density

Initial stem densities for all plots ranged from 0.4-37.2 stems/m^2 and averaged 10.52 stems/m^2. Average change in stem density from 2013 to 2014 for the control group was -0.99 stems/m^2 (n=12, SD=2.55). In comparing all treatments to the control, the mow only treatment produced the only statistically significant deviance with 2.8 fewer stems/m^2 (CI=0.7, 5.0, df=33, $p<0.01$). Burn only provided a non-significant difference of 0.78 fewer stems/m^2 and combined mowing and burning was also non-significant with 0.44 stems/m^2 fewer than the control (Figure 2.3).

With the significant difference in change in average stem density of 2.4 stems/m^2 (CI= 0.3, 4.5, df=33, p=0.02) between the mow only treatment and the mow then burn treatment, there is moderate evidence to suggest that burning after mowing promotes the growth of additional stems compared to mowing alone. Also, with the slightly non-significant difference of 2.1 stems/m^2 between the change in average stem density in mow only and burn only treatments, it is possible that burning promotes stem density through seedling germination or resprouting to a greater extent than mowing. This suggests that Himalayan blackberry responds favorably to fire, even after two disturbances; however, there is no statistical significance to support this claim. Analyses of seedling germination and resprouting are present in Sections 2.4.3 and 2.4.4.

An initial reduction in stem density is expected in all treatments (Table 2.2), as each altered the aboveground biomass of blackberry during a time when resources were allocated aboveground for growth and reproduction (Soll 2004, Tirmenstein 1989). However, mowing alone likely reduced stem density the greatest due to the almost complete removal of biomass above 30cm compared to the partial removal by burning alone and absence of the resprout stimulation often observed in burn treatments (Tirmenstein 1989). Thee observed decrease in stem density was unlike the findings of Ingham (2014), who found that density increased after one and two years of mowing. It is possible that this disagreement is due to the differences in initial population sizes and maturity, as Ingham (2014) began with greater blackberry densities in areas without recent management. Our findings were in agreement, however, with Giles-Johnson *et al.* (2010), who found that mowing at any time and frequency reduces percent cover of Himalayan blackberry within one year of treatment, though metrics were not similar to our study.

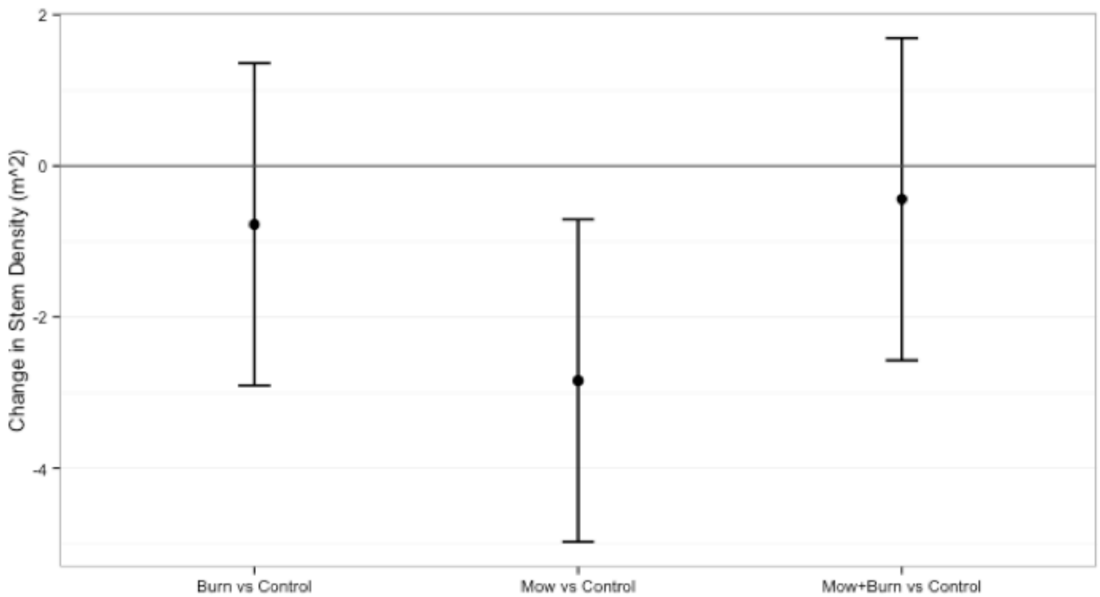

Figure 2.3 Plot of the estimated differences in mean change in average stem densities for each treatment compared to the control. Mean for the control is shown as a line set at 0. Bars represent FDR-adjusted 95% confidence intervals of the estimates.

Table 2.2 Estimates of the mean change in average stem density from 2013 to 2014 for each treatment and the control with their 95% confidence limits. * indicates significant values.

Treatment	**Estimate**	**Lower Limit**	**Upper Limit**
Control	-0.99	-3.09	1.11
Burn Only	-0.78	-2.91	1.36
Mow Only	-2.84*	-4.97*	-0.71*
Mow + Burn	-0.44	-2.57	1.69

2.4.2 Plant Density

Initial plant densities in the 2013 season averaged 4.27 plants/m^2 and ranged from 0.20-14.7 plants/m^2. Average change in plant density in the 2014 season for the control group was 0.18 plants/m^2 (n=12, SD=0.74). None of the treatments produced significant change in average plant density compared to the control at the 95% confidence level (Table 2.3). However, biological significance is implied in those differences detected with p-values less than 0.10, as these trends provide additional support for conclusions made within this study and therefore have potential management implications. It is essential to note that this method of data analysis has been justified and suggested for greater use in other biological studies (Nakagawa and Cuthill 2007).

The mow only treatment had a biologically significant reduction in plant density from 2013 to 2014 when compared to the control treatment (Figure 2.4). The mow only treatment resulted in an average difference of 0.68 plants/m^2 fewer than the control (CI=0.09, 1.28, df=33, p=0.06), while the mow then burn and burn only treatments had non-significant estimated averages compared to the control. Of note, the burn only treatment resulted in an increase in average plant density of 0.14 plants/m^2 more than the control, suggesting potential stimulation of rhizomatous sprouting and germination of seedlings. Comparison amongst treatments showed that the mow only treatment had a significant reduction in plant density compared to the burn only treatment, with a difference of 0.83 plants/m^2 (CI=0.08, 1.57, df=33, p=0.03). Comparison of both these treatments with the mow then burn treatment resulted in non-significant differences of 0.39 plants/m^2 greater in the burn only treatment, and 0.433 plants/m^2 fewer in the mow only treatment.

With all burn-associated treatments showing either greater plant density or less of a decrease than the mow only treatment, it is evident that fire stimulates Himalayan blackberry growth. This trend is in congruence with Bennett (2006) and Tirmenstein (1989), who suggest that burning in any manner increases blackberry resprouting and germination of new seedlings. It is suggested that fire also reduces competition from neighboring plants that are less fire tolerant (Pendergrass *et al.* 1998); however, without analysis of the belowground processes after disturbance and significant statistical support, there is not much weight in these conclusions without further analysis.

The differences observed between treatments utilizing mowing versus burning could likely be due to the varying effects each type of disturbance induces on the entire plant community. Mowing 30cm aboveground will likely reduce the vigor of most grass and few herbaceous species, but will not necessarily induce mortality, allowing for the persistence of most plants within the community. Burning, however, is more likely to induce mortality and even consume entire plants, as this type of disturbance is able to influence belowground as well as aboveground biomass. If our burn treatments were intense enough to consume neighboring plants, competition for resources needed for blackberry establishment and regrowth would be partially eliminated. To confirm this phenomenon would require additional research with utilization of various fire intensities and analysis of individual species percent cover, however.

Table 2.3 Estimates of the mean change in average plant density from 2013 to 2014 for each treatment and the control with their 95% confidence limits.

Treatment	**Estimate**	**Lower Limit**	**Upper Limit**
Control	0.18	-0.54	0.90
Burn Only	0.14	-0.61	0.89
Mow Only	-0.68	-1.43	0.07
Mow + Burn	-0.25	-0.10	0.50

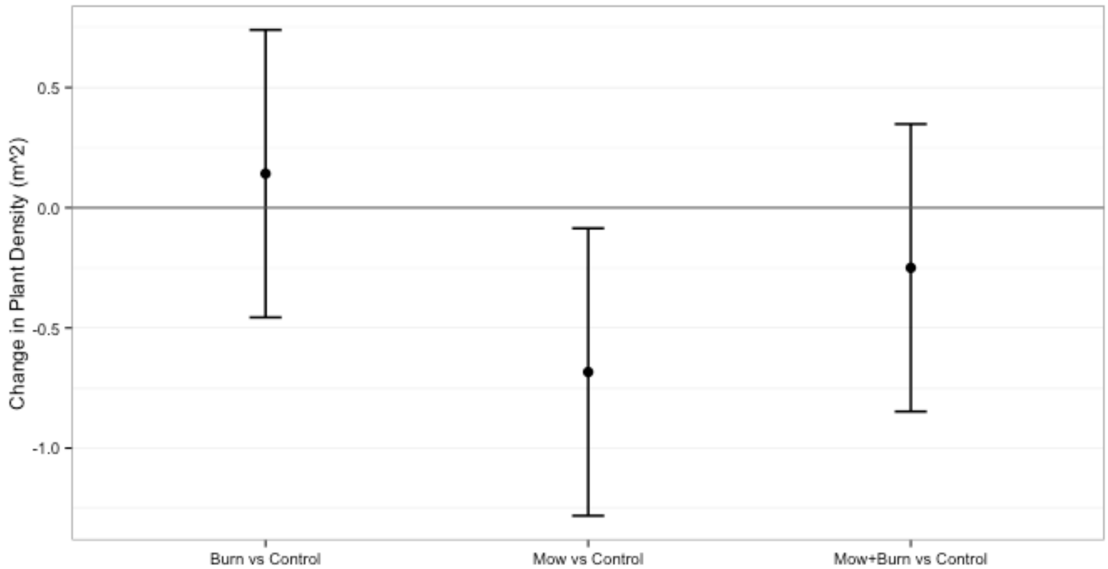

Figure 2.4 Plot of the estimated differences in mean change in average plant densities for each treatment compared to the control. Mean for the control is shown as a line set at 0. Bars represent FDR-adjusted 90% confidence intervals of the estimates.

2.4.3 Seedling Density

As all treatments reduced aboveground biomass, blackberry was expected to respond with higher seedling densities than in the control due to reallocation of resources from secondary growth to primary. Estimated average seedling density for the control group in the 2014 season was 0.48 seedlings/m^2 (n=12, SD=0.47). Of all comparisons made to the control (Table 2.4), the burn only treatment had the only significant difference with an estimated of 0.47 more seedlings (CI=0.13, 0.82, df=32, $p<0.01$)(Figure 2.5), indicating a positive response of blackberry to fire. There is biological support for this conclusion as the heat from burns has been shown to stimulate regrowth in *Rubus* species, and post-fire conditions facilitate establishment of woody species from refractory buried seeds (Keeley 1988; Kauffman & Martin 1990). This result also follows the general trend in our prior observations of burn treatments stimulating blackberry growth.

In comparing the treatments among each other, average seedling density in the burn only treatment was 0.45 seedlings/m^2 greater than the mow then burn treatment (CI=0.11, 0.79, df=32, $p<0.01$). The burn only treatment also had a statistically greater average seedling density than the mow only treatment by 0.40 seedlings/m^2 (CI=.0.06, 0.74, df=32, p=0.02). This indicates that blackberry responds most favorably to fire when no manipulation of aboveground resources has taken place. This result was expected as an increase in plant density was also observed in the burn only treatment and seedling production was the primary means by which new plants immerged.

Table 2.4 Estimates of the mean average seedling frequency in 2014 for each treatment and the control with their 95% confidence limits. * indicates significant values.

Treatment	**Estimate**	**Lower Limit**	**Upper Limit**
Control	0.02	-0.29	0.34
Burn Only	0.47*	0.13*	0.82*
Mow Only	0.07	-0.27	0.41
Mow + Burn	0.02	-0.32	0.36

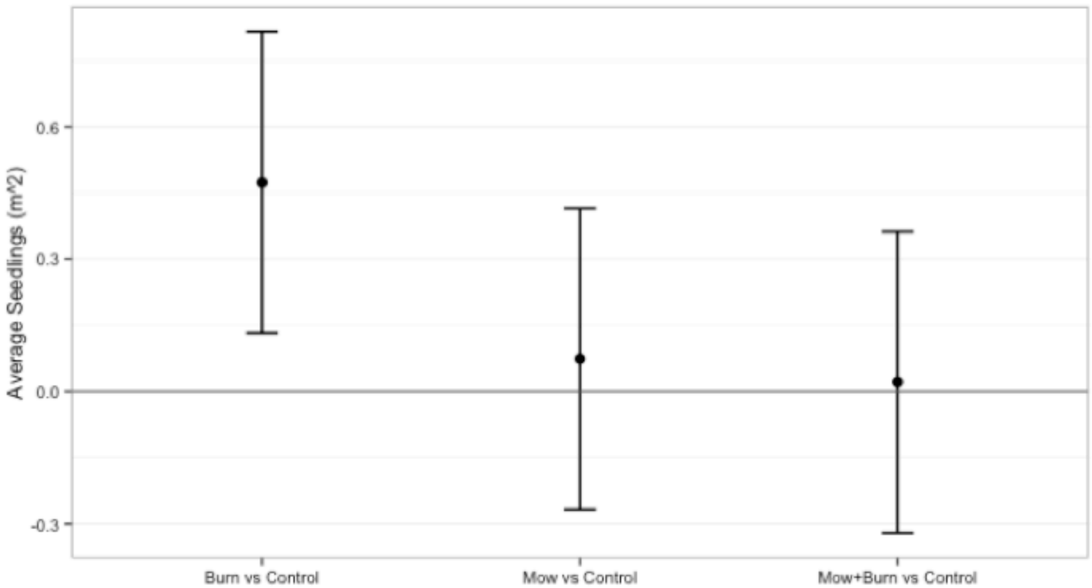

Figure 2.5 Plot of the estimated differences in mean average seedling frequency for each treatment compared to the control. Mean for the control is shown as a line set at 0. Bars represent FDR-adjusted 95% confidence limits of the estimates.

2.4.4 Resprout Density

In comparing the treatments to the control, which had an estimated average of 4.47 resprouts/m^2 (n=12, SD=3.90), all treatments had greater average resprouts densities (Table 2.5). The mow then burn treatment had the most significant difference of 3.3 resprouts/m^2 (CI=1.48, 5.05, df=32, $p<0.01$)(Figure 2.6). This suggests that adding multiple disturbance types or increasing disturbance intensity within one growth season stimulates the regrowth of existing blackberry plants compared to single-entry disturbance treatments.

The burn only treatment also significantly increased the average resprouts/m^2 compared to the control by 1.9 (CI=0.14, 3.71, df=32, p=0.04), suggesting that burning as a disturbance stimulates primary growth in blackberry crowns. Of note, the mow only treatment had a negligibly greater average frequency of resprouts (0.1 resprouts/m^2) when compared to the control.

In comparing the treatments among themselves, the mow only treatment had significantly fewer resprouts than the burn only treatment by1.8 resprouts/m^2 (CI=0.02, 3.59, df=32, p=0.05), and also fewer than the combined mow and burn treatment by 3.1 resprouts/m^2 (CI=1.36, 4.92, df=32, p<0.01).

Table 2.5 Estimates of the mean average resprout density in 2014 for each treatment and the control with their 95% confidence limits. * indicates significant values.

Treatment	**Estimate**	**Lower Limit**	**Upper Limit**
Control	-1.94*	-3.50*	-0.38*
Burn Only	2.00*	0.12*	3.88*
Mow Only	0.31	-1.59	2.20
Mow + Burn	3.35*	1.47*	5.22*

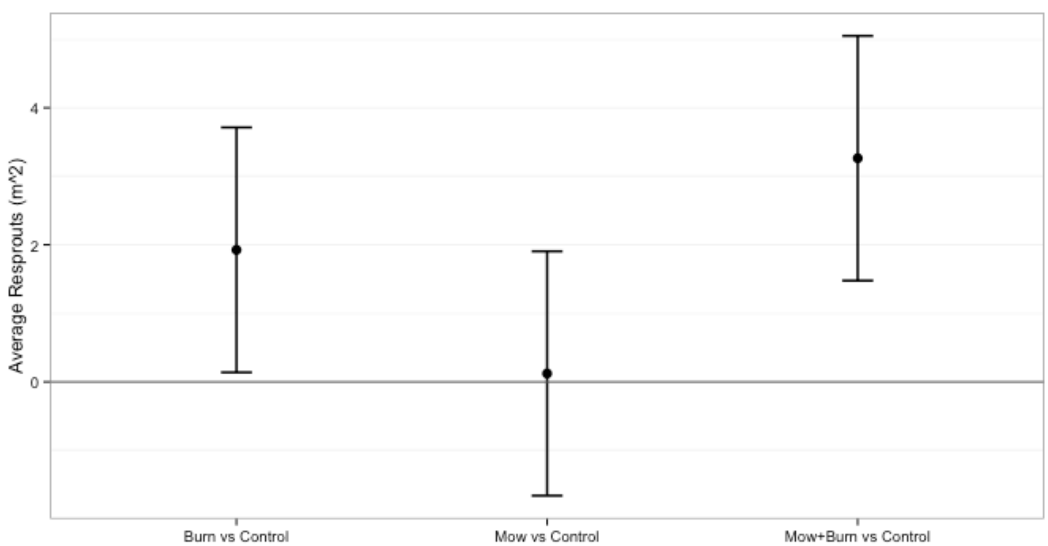

Figure 2.6 Plot of the estimated differences in mean average resprout frequency for each treatment compared to the control. Mean for the control is shown as a line set at 0. Bars represent FDR-adjusted 95% confidence limits of the estimates.

2.5 Management Implications

Late summer mowing has been shown to be a more effective means of controlling Himalayan blackberry populations compared to late summer burning or even a combination of the two techniques. Mowing reduces average blackberry stem density and plant density within existing populations and also stimulates low seedling density and resprouting post-treatment, suggesting less potential for future spread. Mowing is also the most logistically simple of the three treatments, requiring little equipment and labor.

Though capable of immediately reducing stem density, burning alone is not recommended for blackberry management as this treatment increases blackberry plant density in the following growing season. Seedling presence for this treatment was higher than either of the alternatives and resprouting was extensive, indicating the high potential for rapid re-establishment of the populations and future spread, ultimately contributing to a greater blackberry problem than initially observed. While this treatment is widely applied to refuge lands for native plant promotion, hazard fuels reduction, and wildlife food production (USFWS 2010), it is not advisable to utilize this treatment alone in areas where populations of Himalayan blackberry are established.

While the combined mowing and burning reduces stem and plant density, it is not as effective in controlling seedling promotion as mowing alone and produces more resprouts than both alternatives, indicating a higher potential for future spread. This treatment is also the most time-consuming and logistically complex of the three. While burning reduces the dead woody debris that mowing left behind, it shows little positive potential for controlling blackberry spread, likely due to an increase in nitrogen and carbon at the soil surface.

In meeting the refuge's management objectives pertaining to woody cover and invasive/non-native cover in the Comprehensive Conservation Plan (USFWS 2010) late summer mowing has the greatest potential for success. However, it is not feasible to completely eliminate burning and combined mowing and burning from the refuge's management arsenal, as these management methods have proven satisfactory in achieving other management objectives within the upland prairies such as increasing Fender's blue butterfly habitat and native species cover (USFWS 2010, Wilson 2004, Bartels and Wilson 2001). The preceding chapter discusses a few of these objectives in relation to the same treatment options.

While this study implemented the typical management techniques used on the refuge, it is advisable to conduct further research considering the seasonality of burning and mowing (spring vs. fall), frequency (twice per growing season vs. once), and intensity to test their potential influence on blackberry populations. It is possible that varying fire intensities, mowing heights, etc. have differing effects on blackberry than the narrow spectrums implemented within this study (Kauffman and Martin 1990). Therefore, additional research of these variables is recommended.

Though each treatment within this study provides some level of Himalayan blackberry control, it is unlikely that blackberry control will persist without subsequent management. For this reason, treatment application of any type, season, and intensity should continue as opposed to neglecting the prairies altogether.

3. UPLAND PRAIRIE PLANT COMMUNITY RESPONSE TO HIMALAYAN BLACKBERRY (*RUBUS ARMENIACUS*) CONTROL EFFORTS ON THE W.L. FINLEY NATIONAL WILDLIFE REFUGE

3.1 Abstract

Prior to Euro-American settlement, upland prairies of the Willamette Valley were characterized by mosaics of bunchgrass species and low-growing forbs with sparse clumps of woody species. Today, the encroachment of woody species into the few remaining prairies is due in large part to invasion by Himalayan blackberry (*Rubus armeniacus*), further degrading critical habitat for multiple threatened and endangered species. The possible management techniques used to control this non-native invasive are of great concern to managers of the W.L. Finley National Wildlife Refuge, especially in regard to their impact on prairie structure and functional group composition.

This study focused on the three most common techniques utilized by the refuge in managing blackberry-invaded upland prairies: late summer mowing, late summer burning, and a combination of the two. Plant community response to treatments was measured by change in graminoid, herbaceous, and woody percent cover after one season of regrowth. Comparing the individual treatments to the control, mowing with subsequent burning was the most successful in meeting refuge objectives, significantly reducing woody cover and increasing graminoid and herbaceous cover. Mowing alone significantly reduced woody cover and increased graminoid cover as well, but did not significantly increase herbaceous cover. Burning alone significantly increased graminoid cover, but did not meet the other management objectives. Understanding the success of

these treatments in meeting these individual refuge objectives will assist managers in determining the best management options for restoring the refuge's Himalayan blackberry-invaded upland prairies.

3.2 Introduction

Plant communities within the upland prairies of Oregon's Willamette Valley make up some of the most threatened ecosystems in existence (Christy and Alverson 2011, Wilson 1998, Towle 1982, Johannessen *et al*. 1971, Habeck 1961). Over a century of land fragmentation, cultivation, fire exclusion, and neglect has facilitated encroachment of woody species as well as non-native and/or invasive species in the once grass-dominated prairies. Consequently, this has altered the native species diversity and community structure upon which many federally listed species depend (Wilson 1998). As a semi-woody, non-native, and invasive species, Himalayan blackberry (*Rubus armeniacus*) has been one of the biggest contributors to this change in community structure and species composition, threatening to degrade the remaining 1% of native Willamette Valley prairies (Wilson 1998, Towle 1982, Johannessen et al. 1971, Habeck 1961).

Once introduced, Himalayan blackberry can quickly dominate plant communities given its ability to spread by seed and vegetatively via rhizomes, tip-rooting, and plant cuttings (Soll 2004, Amor 1974). The morphology and growth rate of weeds like Himalayan blackberry enable them to usurp resources more quickly and efficiently than nearby vegetation (Dekker 1997, Baruch and Goldstein 1999). Himalayan blackberry has higher photosynthetic capabilities, greater water, carbon, and nitrogen use efficiencies, and allocates more resources to reproduction than its native congeners (*Rubus ursinus*, *Rubus leucodermis*) (McDowell 2002). Additionally, the dense thickets created by the biennial canes allow little sunlight penetration and soil resources for understory plant growth (Fierky and Koffman 2006, Wilson 1998). Himalayan blackberry vigorously resprouts from crowns and roots after disturbance and is quick to take advantage of available resources compared to the less responsive natives (McDowell 2002).

The range and general appearance of native upland prairies can be gleaned from land survey records from the 1850s as well as analysis of the few areas that remain and serve as a guide to restoration efforts in the Valley (Johannessen *et al.* 1971, Habeck 1961). Wilson characterizes upland prairies as "small-stature communities dominated by perennial grasses and forbs" (1998, p.2), with the majority of the plant biomass present within 20cm of the soil and some grass flowering stalks reaching 150cm. Generally, vertical stratification is minimal in upland prairies and the presence of bunchgrasses leaves space in between for herbaceous species like wooly sunflower (*Eriophyllum lanatum*) and wild strawberry (*Fragaria virginiana*). If at all present, woody species were sparse and had low recruitment due to persistent burning by the local Kalapuya tribes and natural ignitions (Wilson 1998).

Some of the last remaining upland prairies of the Willamette Valley are located within the W.L. Finley National Wildlife Refuge, managed by the US Fish and Wildlife Service. Dedicated to providing habitat and promoting populations of some of the Willamette Valley's most threatened and endangered species under the ESA such as the golden paintbrush (*Castilleja levisecta*), Kincaid's lupine (*Lupinus sulphureus* var.

kincaidii), and Fender's blue butterfly (*Icaricia icarioides fenderi*)), refuge managers are greatly concerned with the influence of Himalayan blackberry on the plant communities within the refuge's upland prairies. According to the refuge's Comprehensive Conservation Plan (CCP), the management techniques used to restore these habitats to their native state and control the establishment and spread of species like Himalayan blackberry are still not well understood (USFWS 2010). The most commonly used of these techniques are mowing and burning, as well as a combination of the two methods.

Fire is known to stimulate growth and abundance of native upland prairie species such as the Willamette daisy (*Erigeron decumbens*) and golden paintbrush (*Catilleja levisecta*), and to increase seedling establishment in species like Roemer's fescue (*Festuca roemeri*) and blue wildrye (*Elymus glaucus*) (Clark and Wilson 2001). Mowing has also been shown to increase seedling establishment in other native species like California oatgrass (*Danthonia californica*) (Clark and Wilson 2001) as well as increase numbers of Fender's blue butterfly masses within upland prairies (Clark and Wilson 1997). However, these techniques have not been adequately analyzed in areas where Himalayan blackberry will compete with native vegetation post-treatment.

This study was designed to provide the refuge with an assessment of its efforts in utilizing mowing and burning for attaining a more native upland prairie physiognomy, defined by the community's vertical and horizontal structure as well as its functional group composition. In congruence with typical refuge practices in upland prairies, late summer mowing, late summer burning, and a combination of the two were chosen as techniques for analysis. One major question within the CCP that this study addresses is:

Which techniques are proving most fruitful in enhancing restoration? To further objectify this question, the impact of these techniques on the percent cover of graminoid, herbaceous, and woody species was analyzed.

3.3 Materials and Methods

3.3.1 Study Area, Design, and Treatments

For a description of the study area, experimental design, and treatments implemented in this study, please refer to Chapter 2, Section 3.1 and 3.2.

3.3.2 Response Variables

Percent cover of three functional groups was measured within individual quadrats along the belt transects in both the 2013 and 2014 field seasons. Estimates were made at 50cm above soil level with overlap counted between functional groups, but not within, allowing for greater than 100% total cover within an individual quadrat. For efficiency, any percent cover below 10 was rounded to the nearest 1%, the nearest 5% when between 10 and 25, and the nearest 10% up to 100. All estimates were averaged at the plot level for statistical analysis and treatment comparison.

This functional group approach to analyzing plant communities has been implemented in many ecological studies conducted within diverse prairie ecosystems. “Physiognomic aspects of vegetation play a greater role in affecting the environment than does the species composition in the habitat” (Brower and Zar 1998). With this, species were differentiated into the functional groups graminoids, herbaceous, and woody as defined by the USDA Natural Resource Conservation Service (2014)(Figure 3.1). Graminoids were considered those species in the grass (Poaceae), sedge (Cyperaceae), and rush (Juncaceae) families. The herbaceous group incorporated those vascular plants without significant woody tissue and that were not graminoids (forbs, herbs, ferns). The woody group (trees, shrubs, subshrubs) was defined by the presence of lignin in plant cell walls, allowing for greater vertical growth than graminoid and herbaceous groups (Table 3.1).

3.3.3 Statistical Model and Analysis

Treatments were compared using the following general linear model for each response variable:

$$Y_{ijk} = \beta_0 + \beta_1(x_b)_{ijk} + \beta_2(x_m)_{ijk} + \beta_3(x_{mb})_{ijk} + b_j + c_k + \varepsilon_{ijk}$$

where

Y_{ijk} is the plot average for the change in percent cover in i^{th} treatment in j^{th} block in the k^{th} site, i=control, mow only, burn only, mow then burn, j=1,2,3, k=1,2,3,4,

β_0 is the mean average of the change in percent cover for the control treatment,

β_1 is the incremental effect of the burn only treatment on the mean average change in percent cover,

β_2 is the incremental effect of the mow only treatment on the mean average change in percent cover,

β_3 is the incremental effect of the mow then burn treatment on the mean average change in percent cover,

x_b is 1 when the i^{th} treatment is burn only and 0 otherwise,

x_m is 1 when the i^{th} treatment is mow only and 0 otherwise,

x_{mb} is 1 when the i^{th} treatment is mow then burn and 0 otherwise,

b_j is the random effect of the j^{th} block on the average change in percent cover, $b_j \sim N(0, \sigma^2)$ and $Cov(b_k, b_{k'})=0$

c_k is the random effect of the k^{th} site on the average change in percent cover, $c_k \sim N(0, \sigma^2)$ and $Cov(c_k, c_{k'})=0$

ε_{ijk} is the residual error term for the i^{th} treatment in the j^{th} block in the k^{th} site, $\varepsilon_{ijk} \sim N(0, \sigma2)$, and $Cov(\varepsilon_{ijk}, \varepsilon_{ij'k'})=0$, and b_j, c_k, ε_{ijk} are all independent

To calculate the change in percent cover, the estimates of 2014 were subtracted from those of 2013 and covariates of 2013 pre-treatment functional group covers were included where significant. The statistical program RStudio version 0.98.490 (2013) was used to conduct each statistical analysis. All pairwise comparisons of treatments were made while ensuring control of the False Discovery Rate (FDR) as outlined in Verhoeven *et al.* (2005). After adjustment for pairwise comparisons, statistical significance was implied at the 95% confidence level.

To maintain the assumptions of the models, normality was checked with normal probability plots. Homogeneity of variances was checked using scatterplots of residuals and independence within and between sites and blocks was present during the design and data collection period. With the given elements of randomization, replication, and representation, the scope of inference of this study is limited to the refuge upland prairies, but could be extrapolated to Willamette Valley prairies of similar soil types and community composition.

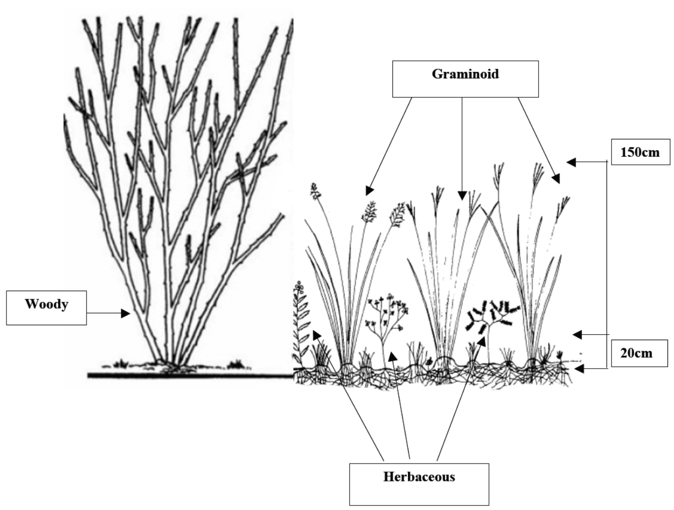

Figure 3.1 Diagram of functional groups as defined by NRCS (2014).

Graminoid	Herbaceous	Woody
Agrostis alba	*Brodiaea coronaria*	*Crataegus douglassii*
Aira caryophyllea	*Calyptridium embellatum*	*Rosa gymnocarpa*
Alopecurus pratensis	*Cirsium vulgare*	*Rosa nutkana*
Arrhenatherum elatius	*Crepis capillaris*	*Rubus armeniacus*
Cynosurus echinatus	*Daucus carota*	*Rubus ursinus*
Festuca arundinacea	*Fragaria virginiana*	*Symphoricarpos albus*
Holcus lanatus	*Galium divaricatum*	*Toxicodendron diversilobum*
Juncus effuses	*Geranium dissectum*	*Quercus garryana*
Lolium multiflorum	*Geranium lucidum*	
Poa pratensis	*Geranium molle*	
	Hypericum perforatum	
	Hypochaeris radicata	
	Lotus corniculatus	
	L. taraxacoides	
	Leontodon vulgare	
	Mentha pulegium	
	Physalis viscosa	
	Plantago lanceolata	
	Pteridium aquilinum	
	Rumex acetosella	
	Rumex crispus	
	Sidalcea virgata	
	Tanacetum vulgare	

Table 3.1 List of common species among blocks differentiated by functional group.

3.4 Results and Discussion

3.4.1 Woody Cover

Woody cover was dominated by Himalayan blackberry in all plots, but included a range of other species as well (Table 3.1). Initial average woody percent cover ranged from 2-72% in 2013. The average woody cover for the control group was estimated to be 3% less in 2014 than 2013 (n=12, SD=12.14), indicating a minute decrease in woody cover within 50cm to soil surface in the absence of disturbance. Compared to the control, the mow only and mow then burn treatments significantly reduced woody cover (Figure 3.2). The burn only treatment also reduced woody cover, but was not statistically significant.

The stimulation of resprouting observed in woody species after fire likely contributes to a lesser reduction in woody species cover in burn treatments (Chapter 2 Section 4, Wilson 1998). Pendergrass *et al.* (1998) observed an increase in the densities of 17 native and non-native woody species within similar Willamette Valley prairies one year after single and multiple burn seasons. Given the recent reintroduction of fire to these ecosystems and the change in species composition to non-fire adapted species (Pendergrass *et al.* 1998), it is likely that burning alone will not be sufficient in returning the prairies to their natural physiognomy.

Both treatments incorporating mowing showed greater reductions in mean average woody cover than the control and burn only treatment. The mow only treatment had a mean average change in woody cover of 8% less than that of the control (CI=2.67, 14.28, df=32, p<0.01) while the mow then burn treatment was 10% less (CI=3.90, 15.46, df=32, p<0.01).

Mowing eliminates most plant biomass above the set blade level, which typically accounts for the majority of woody cover. Combining mowing with burning has been shown to not only reduce woody cover, but also decrease woody species survival within other Willamette Valley prairies (Clark and Wilson 1996). While the mow only and mow then burn treatments showed significant reductions in woody cover, no treatment was shown to be statistically superior to another.

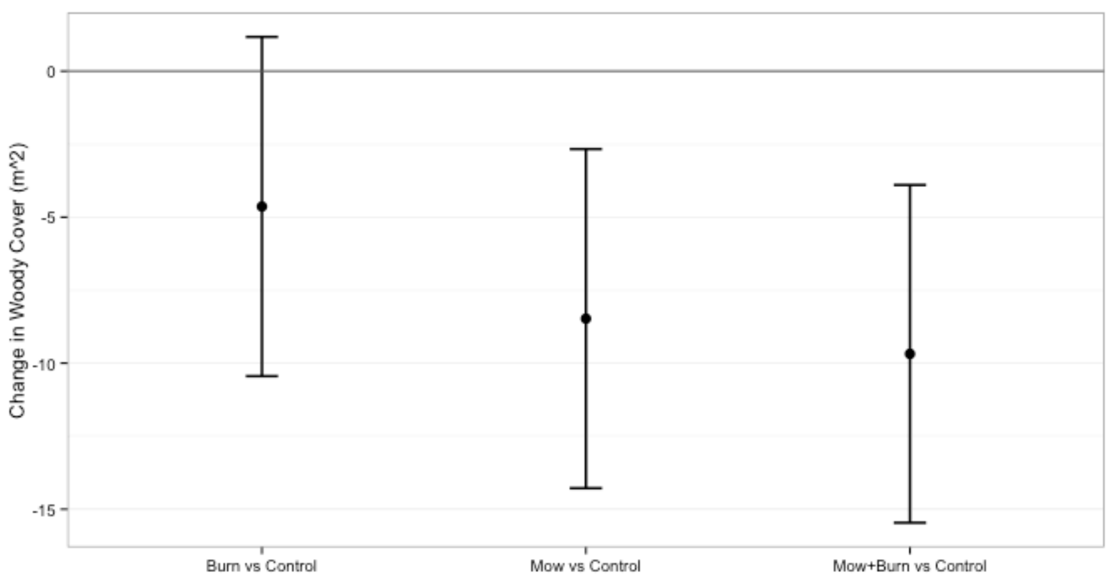

Figure 3.2 Plot of the estimated differences in mean change in woody cover for each treatment compared to the control. Mean for the control is shown as a line set at 0. Bars represent FDR-adjusted 95% confidence intervals of the estimates.

3.4.2 Graminoid Cover

Graminoid composition was predominantly non-native with residual agriculture species such as tall fescue (Table 3.1). Average graminoid cover ranged from 45-95% in the 2013 field season. Measurements in the 2014 field season showed a decrease in the average graminoid cover for each treatment as well as the control, indicating some unmeasured environmental driver within the plant communities. The control had a reduction in percent graminoid cover of 34% (n=12, SD=16.79). In comparison, the burn only treatment had a lesser reduction by 17% (CI=7.18, 26.88, df=33, $p<0.01$), the mow only was less by 17% (CI=6.98, 26.68, df=33, $p<0.01$), and the mow then burn treatment had the greatest difference with 18% greater cover than the control (CI=8.57, 28.27, df=33, $p<0.01$) (Figure 3.3).

All treatment estimates show a general improvement in graminoid cover compared to no disturbance; however, these averages are still less than the initially observed averages of 2013 and species composition was predominantly non-native. This trend is inconsistent with the findings of most studies in analyzing graminoid response to disturbance as most prairie grasses respond favorably to fire and mowing (Wilson and Clark 1997, Vogl 1974). Given the high abundance of non-native and residual agricultural perennial grasses within the prairies, it is unusual that there would be a reduction in percent graminoid cover within these prairies. Regardless, conclusions can be assessed based on comparison of treatments to the control group.

Fire is known to benefit bunchgrass species by releasing dormant seed (Wilson 1998), and mowing increases dispersal of seed, especially in non-natives with flowering components above the set blade level. Also, with the significant reduction in woody cover resulting from the mow treatments, ruderal species like exotic grasses are expected to establish as pioneers according to Grime's C-S-R model (1988). Thus, improvements in percent graminoid cover are expected in comparison to an absence of disturbance.

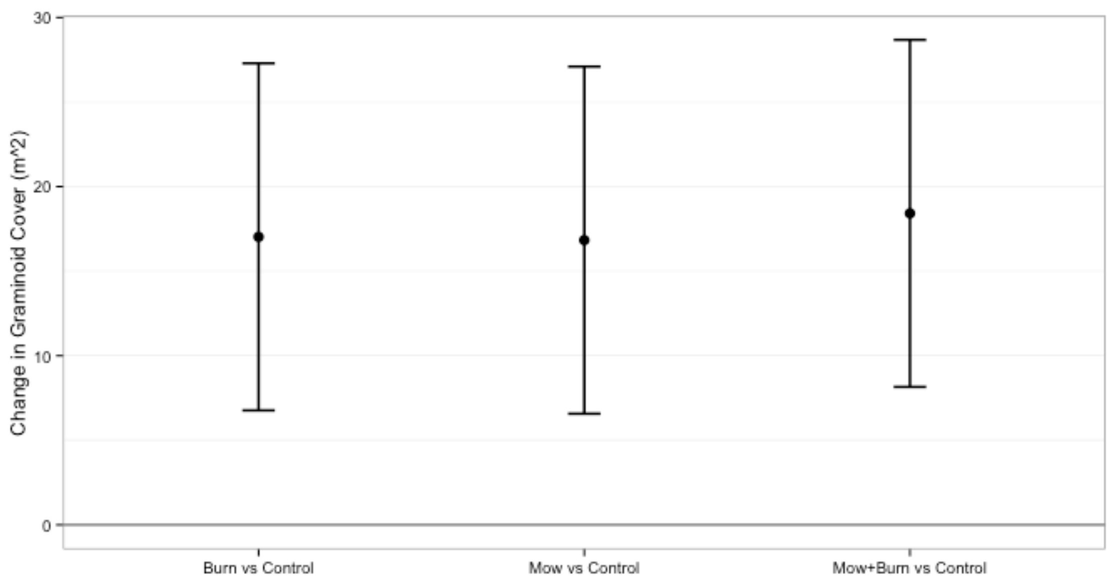

Figure 3.3 Plot of the estimated differences in mean change in graminoid cover for each treatment compared to the control. Mean for the control is shown as a line set at 0. Bars represent FDR-adjusted 95% confidence intervals of the estimates.

3.4.3 Herbaceous Cover

Herbaceous composition included native as well as non-native species (Table 3.1). Average herbaceous cover ranged from 1-38% in the 2013 field season. All treatments as well as the control group showed an increase in average herbaceous cover from 2013 to 2014; with the control showing a 14% increase (n=12, SD=15.90). Compared to the control, the mow then burn treatment was the only statistically significant difference with a 12% greater increase in mean average herbaceous cover (CI=2.82, 21.74, df=32, p=0.01) (Figure 3.4). When comparing the mow then burn treatment to the other treatments, there was a significant difference observed with the mow only treatment comparison of 12% greater herbaceous cover (CI=2.58, 21.61, df=32, p=0.01), but no significant difference in the burn only treatment.

It is likely that the mow then burn treatment showed the most significant increase in mean average herbaceous cover due to the horizontal growth patterns of many of the present herbaceous species. Most of the biomass for these species is within 5cm of the soil surface, allowing the majority of it to remain after mowing while grass and shrub biomass is removed (Wilson and Clark 1997). The addition of fire to this scenario would allow dormant herbaceous seeds within the soil to germinate soon after disturbance.

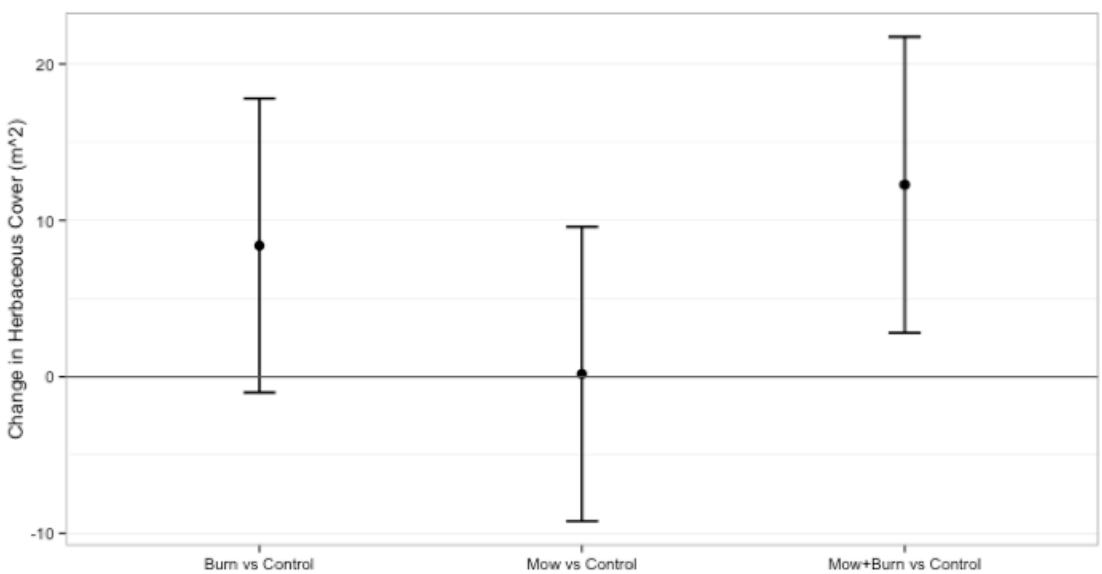

Figure 3.4 Plot of the estimated differences in mean change in herbaceous cover for each treatment compared to the control. Mean for the control is shown as a line set at 0. Bars represent FDR-adjusted 95% confidence intervals of the estimates.

3.5 Management Implications

While all treatments achieved at least one of the refuge objectives focused on promoting natural prairie physiognomy, combining late summer mowing with subsequent burning seemed to have the greatest benefit overall. Compared to mowing alone and burning alone, this technique had the greatest reduction in woody cover as well as increase in graminoid and herbaceous cover. This shows the greatest potential for success in redirecting prairie succession to more of the low-stature, non-woody communities historically prevalent in the Willamette Valley. This treatment is, however, the most time-intensive and logistically complex of all techniques analyzed, as it utilizes two forms of management. Regardless, the use of late summer mowing followed by burning in the management of upland prairies with Himalayan blackberry is highly recommended, as this treatment most adequately achieves the objectives outlined within the refuge's CCP.

To a lesser degree, late summer mowing alone proved to be successful in reducing woody cover as well as increasing graminoid cover in our prairies. However, this treatment showed little promise in increasing herbaceous cover, which is essential habitat to wildlife. Examples include multiple species of butterfly that rely on the broad leaves of forbs within prairies for egg laying and other reproductive efforts (Thomas *et al.* 1986, Williams 1981) and the streak-horned lark (*Eremophila alpestris strigata*) which prefers short, sparse prairie habitat (Pearson and Callaway 2003). However, it is likely that this technique is the only option in areas where applying fire could have detrimental effects such as in prairies with high levels of woody cover that would result in high fire intensity or around high-value structures. This technique is also a logistically simple and precise management tool requiring little time and effort. Therefore, the use of this technique as an alternative to combined mowing and burning is recommended when the above scenarios exist.

In achieving objectives related to graminoid and herbaceous cover increase, burning alone as a management technique proved successful. However, this technique was not significantly effective in reducing woody cover, especially in comparison to techniques utilizing mowing. In areas with minimal woody cover, burning alone could be a sufficient tool in managing upland prairies and, though more logistically complex than mowing, manages large areas in a short amount of time. Thus, we recommend the continued use of burning as a management option on the refuge.

It is clear that a single method cannot be relied upon in managing for native physiognomy in the upland prairies of the W.L. Finley National Wildlife Refuge. In meeting one CCP objective or combinations of two or more, there are a few successful management options, but others may prove more beneficial in certain circumstances. Management of upland prairies with Himalayan blackberry invasion needs to be evaluated based on prairie composition, values at risk, and prioritization of CCP objectives within the individual prairie. As the refuge prairies greatly vary in these aspects, the continued monitoring of prairies receiving the treatments analyzed within this study is needed to ensure continued effectiveness and applicability.

4. CONCLUSIONS

4.1 Management Summary

Successful upland prairie management on the W.L. Finley National Wildlife Refuge has the potential to restore and create better supportive habitats of ESA-listed threatened and endangered species populations throughout the Willamette Valley. In managing specifically for Himalayan blackberry control, success is likely to be even more achievable. This study's goal was to provide refuge managers with an analysis of the most common upland prairie management techniques used to achieve refuge objectives, including late summer mowing, late summer burning, and a combination of both. Specifically, we analyzed how these techniques influence blackberry population dynamics and surrounding functional group response. Refuge objectives for upland prairies were outlined in the CCP as a reduction of invasive species like Himalayan blackberry, increase in graminoid and herbaceous cover, and decrease in overall woody cover (USFWS, 2010).

Considering Himalayan blackberry populations alone, late summer mowing alone had the greatest success in meeting refuge objectives one year after treatment. This technique reduced stem density and showed low blackberry seedling presence, providing adequate control of blackberry spread in terms of future invasion potential and meeting the invasive species refuge objective. With only a single site entry and little equipment required, this technique also proves logistically simple for refuge managers to implement.

In sites where blackberry is the major management concern, we suggest this technique as the best option of the three tested within this study.

Managing blackberry-invaded upland prairies with plant community refuge objectives in mind required additional metrics in evaluating technique efficacy and suggested the addition of late summer burning to the mowing treatment. Though this combined technique was slightly less successful in controlling blackberry spread, it was the most beneficial of the three treatments in providing a more natural upland prairie physiognomy (Table 4.1). This included a reduction in woody cover and increase in graminoid and herbaceous cover, which encompasses the habitat structure that many threatened and endangered species prefer. Thus, in managing upland prairies where blackberry invasion is of less concern than maintaining or restoring desired physical structure and species composition, we advise the use of mowing with subsequent burning. Utilizing both management techniques requires the most intensive time, labor, and logistical requirements of the three, but shows the most potential in meeting all refuge objectives.

Of the three techniques, late summer burning alone resulted in the greatest blackberry seedling density and was the least successful in reducing woody cover. This counters the refuge's desired outcomes in both the categories of invasive control and improvement of upland prairie community structure. Other studies have stressed the importance of reintroducing fire to the prairies of the Willamette Valley (Pendergrass *et al.* 1998, Wilson and Clark 1997, Connelly and Kauffman 1991), but without a preceding mechanical treatment, detrimental effects are likely to occur, especially in blackberry-invaded upland prairies.

Table 4.1 Summary table ranking treatments from most effective (1) to least (3) in achieving refuge objectives according to each response variable. Ties were possible where differences between treatments were considered negligible. Statistically significant differences from the control group are also marked (*) as well as values that opposed refuge objectives (-).

	Burn	Mow	Mow+Burn
Blackberry control:			
Stem Density	2	1*	3
Plant Density	3-	1*	2
Resprout Density	2*	1	3*
Seedling Density	3*	2	1
Prairie Physiognomy:			
Woody Cover	3	1*	1*
Graminoid Cover	1*	1*	1*
Herbaceous Cover	2	3	1*

4.2 Future Research

While Himalayan blackberry control techniques have been previously analyzed in multiple environments, no silver bullet method emerges. It has become evident that with such a large geographic distribution and broad range of suitable site conditions, certain control techniques prove effective in some cases, but not in others. For example, this study found a minute reduction in woody (predominantly blackberry) cover after burning in upland prairies, while Thorpe *et al.* (2008) found a 70% blackberry reduction in wetland prairies. Jones (2004) also adds that the amount of shading after disturbance will greatly influence blackberry regrowth and this factor varies greatly across Himalayan blackberry's range. Consequently, the findings of this and other studies are severely limited in scope of inference. This necessitates the evaluation of all possible techniques on a site-to-site basis.

Just as treatment efficacy of blackberry control can fluctuate with varying environments, so can the response of the surrounding communities. Species composition prior to treatment can greatly influence which species return and replace blackberry (Dennehy *et al.* 2011). While this study was catered to resource objectives that were broadly categorized into functional groups, it would be beneficial to further tease these groups into native/non-native and individual species, as it is often desirable to restore to a native and/or diverse community. Evaluating these parameters is not possible if functional groups are the sole method of community response analysis. For this reason, future research of blackberry control methods should include analysis of finer-scale species composition.

Other treatments have proved to be more effective than those analyzed within this study. The use of glyphosate and triclopyr herbicides prior to burning and after mowing has been successful for other land managers in the Willamette Valley in controlling Himalayan blackberry aboveground (Soll 2004). While herbicide is not the refuge's first choice in control methods, it has recently been resorted to in areas of high blackberry density on the refuge. While refuge personnel are currently monitoring impacts of these herbicides, additional studies should commence in conjunction with these efforts to evaluate the treatment's efficacy and community response in a way that is similar to this study to facilitate comparison between this study's technique analyses.

This study served as a foundation for blackberry research on the refuge, providing refuge managers with feasible metrics and an example study design to continue analyzing the blackberry population response in the upland prairies beyond a single year of treatment. It is strongly suggested that additional evaluation of the populations within this study be conducted in subsequent years, as the stages of community succession and blackberry response are likely to vary temporally. It would also benefit the refuge immensely to repeat this study's treatment in a few of the blocks to assess the impact of repeated disturbance, as Dennehy *et al.* (2011) found a greater negative response of blackberry to annual disturbances than single entry treatments without follow-up.

To provide a larger scope of inference than just the refuge, blocks should be randomly placed throughout the upland prairies of the Willamette Valley. With similar soil types and community composition, it is possible that the technique efficacies observed within the refuge will be mirrored by those of the entire Willamette Valley. If this is the case, the total upland prairie area and quality (native structure and composition) will be improved and greater progress toward restoration of these historically prevalent systems will be made.

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