HOARY ELFIN AND KINNIKINNICK RESPONSE TO FIRE ON PRAIRIES AT JOINT BASE LEWIS-McCHORD

by

Sarah Hieber

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Sarah Hieber

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The Evergreen State College

by

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Kevin Francis, Ph. D.
Member of the Faculty

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Date

ABSTRACT

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Sarah Hieber

The Fish & Wildlife program operating at Joint Base Lewis-McChord (JBLM), an active military base in Pierce County, WA, implements an extensive prescribed fire program to preserve the rare and unique prairie ecosystem found at JBLM. The prairies were maintained for millennia by the burning practices of indigenous civilizations and have suffered due to fire suppression, area loss, fragmentation, and invasive plant introduction, causing the decline of many endemic prairie-specialist species. While prescribed fire benefits prairie restoration efforts, populations declining due to habitat loss may struggle to cope with the added stress of fire, even if it improves their habitat in the long run. One such species is the Hoary Elfin butterfly (*Incisalia polia obscura*), which exclusively utilizes Kinnikinnick (*Arctostaphylos uva-ursi*) as a larval host plant. Hoary Elfin surveys are conducted during adult flight season from late April to late May. I analyzed spatial data on prescribed burn history and Hoary Elfin sightings at five prairies stewarded by JBLM. Additionally, I monitored Kinnikinnick regrowth at one site that burned in a wildfire in 2017 using 20 1m^2 plots that I observed for 21 months post-fire. The Hoary Elfin population in Johnson prairie appears to have been declining prior to the wildfire, but evidence suggesting the role of prescribed burning in this process is inconclusive. Hoary Elfin in Upper and Lower Weir prairies were concentrated in areas that had burned within the previous two summers. However, the Central Impact area, which almost never burns, contained a robust population. The Kinnikinnick that burned in Johnson prairie had reached an average of 38% cover within the meter plots 21 months post-fire. This research may serve to inform land management decisions that impact the habitat of a rare species. Furthermore, it is part of the much larger effort to restore South Puget Sound prairies and the unique biodiversity they harbor.

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INTRODUCTION

Lowland prairies in the Willamette Valley-Puget Trough-Georgia Basin (WPG) ecoregion are one of the most endangered landscapes in our region (The Nature Conservancy of Washington, 2011). This system has suffered area loss and fragmentation as a result of land use changes, namely urbanization and agriculture (Dunwiddie & Bakker, 2011). Prairies/oak savannas are characterized by their lack of tree cover, and dominant vegetation of grasses and forbs. Oaks are associated with these areas because they are generally found on the edges of the open grasslands, where they create a noncontiguous canopy over the same type of groundcover as the treeless prairies (Bachelet, Johnson, Bridgham, Dunn, Anderson, & Rogers, 2011). This landscape creates a unique habitat for a large array of species that need sunlight to thrive and are therefore excluded from the dense forests that characterize many other Pacific Northwest (PNW) landscapes. In addition to prairie-specific vegetation, prairies support wildlife that thrives in edge habitat or requires open areas to forage (Hamman, Dunwiddie, Nuckols, & McKinley, 2011). The overall effect is a landscape mosaic, rather than homogenous forest cover, and this diversity of habitat supports a greater diversity of species.

This ecosystem initially developed as vegetation reclaimed the areas exposed by the retreat of the Cordilleran ice sheet around 14,000 years ago (Crausbay, Higuera, Sprugel & Brubaker, 2017). These areas would have likely followed the trajectory of vegetative succession typical to the Pacific Northwest and become dense conifer forest were it not for indigenous civilizations who burned them regularly (Boyd, 1999), including but not limited to the Gitksan and Wet'suwet'en of British Columbia (Gottesfeld, 1994), Salish of the Puget Sound area (Boyd, 1999), and Kalapuya of the

Willamette Valley (Clark & Wilson, 2001). These civilizations maintained burning practices for millennia on WPG prairies to encourage the growth of certain edible plants and create open areas conducive to hunting game (Boyd, 1999). Evidence suggests that pre-colonial North American civilizations, even those classified as "hunter-gatherer," may have altered their environments to far greater extent and impact than once thought (Lightfoot, Cuthrell, Striplen, & Hylkema, 2013). Prairies have essentially always been a human-managed landscape, and without such intervention, they would not have persisted over as large an area as they did until European settlement (Boyd, 1999). The disruption of indigenous burning practices by colonization has contributed to decline of this ecosystem (Dunwiddie & Bakker, 2011).

Prior to European colonization, the WPG prairie ecosystem spanned from Oregon's Willamette Valley through Washington's Puget Sound to British Columbia's Georgia Basin (Figure 1), covering 845,000 ha. However, European settlers converted much of the best habitat to agricultural land, and urbanization has led to subsequent losses in area, leaving only 16,200 ha (Hamman et al., 2011), a reduction of 98% (Dunwiddie & Bakker, 2011). The remaining prairies are highly fragmented and shrinking due to fire suppression, which allows trees and woody shrubs to encroach upon them from the edges, and have experienced changes in their ecological structure due to the elimination of fire as an ecological process (Dunwiddie & Bakker, 2011; Hamman et al., 2011). The loss of WPG prairies as a result of these processes has led conservationists to classify them as an "endangered ecosystem" (Noss, LaRoe, & Scott, 1995).

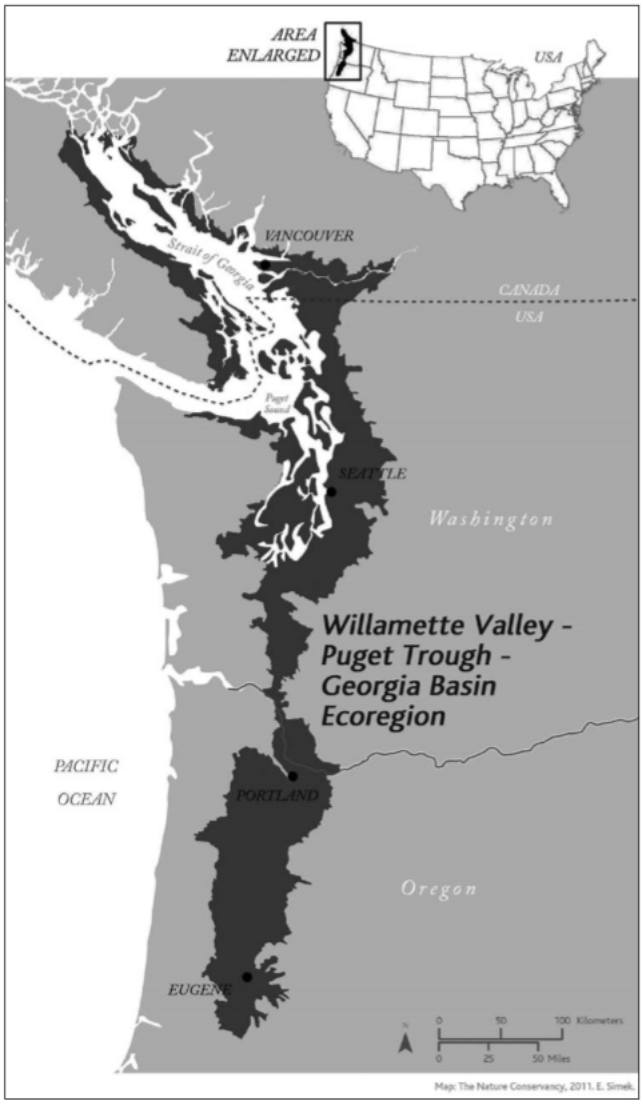


Figure 1. Historic range of WPG ecoregion.
Adapted from Hamman, S.T., Dunwiddie, P.W., Nuckols, J.L, & McKinley, M. (2011). Fire as a restoration tool in Pacific Northwest prairies and oak woodlands: Challenges, successes, and future directions. *Northwest Science*, 85(2):317-328. https://doi.org/10.3955/046.085.0218

The cumulative effects of area loss, fragmentation, cessation of frequent burning, and introduction of invasive plant species have caused the decline of a number of prairie-specialist species endemic to this ecoregion (Dunwiddie & Bakker, 2011). Hoary Elfin (*Incisalia polia obscura*) butterflies are one such species that have experienced habitat constraints due to these factors. This subspecies is endemic to the South Puget Sound (TNCW, 2011), and is currently considered a Washington state species of conservation

concern (Schultz et al., 2011). They exclusively utilize Kinnikinnick (*Arctostaphylos uva-ursi*), a low shrub that grows in mat-like patches in the prairies, as a larval host plant. Kinnikinnick possesses adaptations allowing it to rapidly recover from fire by re-sprouting from underground organs (del Barrio, Luis-Calabuig, & Tárrega, 1999), but little is known about Hoary Elfin's resilience to burning.

About 90% of remaining WPG prairie remnants in the South Puget Sound are contained by Joint Base Lewis-McChord (JBLM), an active military base in Pierce County, WA. They include some of the highest quality habitat left for threatened prairie species, and host significant populations of several of these species. JBLM has implemented an Integrated Natural Resources Management Plan (INRMP) in order to balance the objectives of conserving natural resources, including rare species and their habitat, on the lands it stewards while maintaining their use for military purposes. A branch of the U.S. Fish & Wildlife Service (USFWS) carries out the majority of projects implemented under the INRMP, which includes invasive species eradication as well as habitat monitoring and restoration to conserve threatened/endangered species. This includes an extensive ecological burning program; while prescribed burning affects entire landscapes, the primary purpose of the program at JBLM is endangered species conservation via habitat enhancement (Joint Base Lewis-McChord, 2018).

Prescribed burning has the potential to benefit prairie ecosystems and the native species which inhabit them, as they are adapted to habitat structured by frequent disturbance. However, the drastic environmental changes that have occurred since European colonization complicate the implementation of a successful prescribed burn program. Habitat loss and fragmentation have decimated the populations of many

species, including Hoary Elfin, by limiting both the size of population that available habitat can support, and the ability of the population to disperse and reproduce in a fragmented landscape (Dunwiddie & Bakker, 2011; Hamman et al., 2011; MacDougall & Turkington, 2005). Current prescribed burning activities show promising results for ecosystem restoration, but additional research into species-specific effects is necessary to determine whether sensitive species such as Hoary Elfin benefit directly or require additional attention (Hamman et al., 2011).

A plethora of research and monitoring projects are conducted on the same prairies, and evidence shows that burning, especially in conjunction with other restoration methods, is effective at promoting biodiversity, resilience, and ecosystem function (Hamman et al., 2011). These metrics are effective for evaluating restoration programs because they prioritize ecological health without imposing unrealistic expectations on landscapes heavily impacted by both human activities and environmental instability. Research and monitoring projects are essential because they generate understanding of potential impediments to restoration success and possible unintended consequences of misguided restoration efforts. The primary physical factors likely to hamper prairie restoration in general are the presence of invasive plant species, climate change, area loss and fragmentation, and habitat restructuring due to fire suppression (Dunwiddie & Bakker, 2011); restoration efforts may be derailed if knowledge of the impacts of these factors is inadequate.

Defining restoration goals can be challenging, especially because human perspectives, assumptions, and cultural ideologies implicitly shape land management decisions. Especially in the face of what many conservationists may view as

environmental destruction wrought by human activities, some may desire to return ecosystems to their natural state, restore equilibrium, or recreate conditions that predate European colonization. Based on the fluctuations of the past and major changes of the present, these are not the best characteristics by which to assess ecological wellbeing or restoration success. Not only are they unattainable due to irreversible environmental changes, but they are unrealistic considering the dynamism experienced by ecosystems throughout history. Reconstructions of ecological history over long timescales show that ecosystems do not necessarily exist in states of static equilibrium, and do not necessarily progress back to their original state after disturbances (Mori, 2011; Wu & Loucks, 1995). Indeed, disturbance has played a profound role as the catalyst of drastic environmental shifts in response to historical changes in climate (Crausbay et al., 2017; Mori, 2011).

Principles of disturbance ecology provide a basis for interpreting the effects that fire may have on a landscape. There is considerable evidence supporting the hypothesis that disturbance has the potential to increase biodiversity (Osman, 2008; Odion & Sarr, 2007; Roxburgh, Shea, & Wilson, 2004). Disturbance creates habitat for species that thrive in ecosystems at earlier stages of succession and makes resources available to species that are less able to compete for limiting resources in ecosystems at later successional stages. Additionally, a landscape containing patches of habitat at different stages of succession harbors greater diversity. However, characteristics of disturbance events, such as frequency (return interval), magnitude (severity), size (area affected), and seasonal timing influence outcomes. While 'optimal' disturbance may have ecological benefits, too much (e.g. too frequent, too severe, etc.) disturbance can reduce diversity by placing excessive stress on species (Odion & Sarr, 2015), especially those populations

already stressed by anthropogenic pressures or other environmental conditions. Species can only benefit from the potential habitat improvement of disturbance if their populations are resilient to the initial stress of a disturbance event.

The primary question I sought to address is: how does the burning of prairies at Joint Base Lewis-McChord affect the persistence of Hoary Elfin populations? This includes both prescribed fire (as a land management practice) and wildfire (as a stochastic natural disturbance), as the effects of both were considered at different sites. Very little is known about Hoary Elfin ecology, so investigating how these types of disturbances in Hoary Elfin habitat might affect the persistence of metapopulations, while broad, is an appropriate starting place. There is no succinct answer to this question, but this study aims to increase understanding of how disturbance in a fragmented landscape affects this rare species.

The study took place on several sites managed by JBLM with suitable Hoary Elfin habitat (prairies containing patches of Kinnikinnick). Most of these areas have been subject to restoration efforts including prescribed burning. Surveys were conducted for adult Hoary Elfin in these areas by observing patches of Kinnikinnick for butterfly activity. Spatial data on Kinnikinnick patch location and condition and Hoary Elfin sightings was collected as part of ongoing efforts to monitor Hoary Elfin populations. I assisted with the collection of some of the Kinnikinnick and Hoary Elfin data utilized in this study for USFWS at JBLM. This study also utilizes some pre-existing data on Hoary Elfin and Kinnikinnick. Additional data on burn history of prairies collected as part of prescribed burning operations was provided to me by those at JBLM. Spatial data was imported into ArcGIS software for representation and analysis.

Additionally, regrowth of Kinnikinnick in an area burned by wildfire was monitored for 21 months. Prior to the wildfire, the site hosted a population of Hoary Elfin, but all Kinnikinnick patches confirmed to be occupied were burned. This element of the study aims to track Kinnikinnick regrowth rate and pattern, namely by monitoring percent cover, to determine what timeframe is required and what patch characteristics are suitable for Hoary Elfin recolonization and occupation.

Research on the effects of prescribed fire has real-world implications for land management. Results have the potential to influence management decisions, which in turn can determine the success of conservation efforts. Restoration work requires direct action and physical intervention to promote conservation goals. Land managers are tasked with making decisions utilizing the best available science, but without the luxury of stalling until better science becomes available. Those at JBLM currently employ restoration tactics, including prescribed fire, in Hoary Elfin habitat, but information on the species and its response to these tactics is lacking. This project strives to provide useful insight that may influence restoration projects at JBLM by compiling and synthesizing available data, analyzing and reporting evident patterns, and serving as a resource to inform future research and management protocols.

LITERATURE REVIEW

A substantial amount of research has been published on WPG prairies, their history and decline, and plans to restore prairies as a key habitat component for PNW flora and fauna. Since prairies historically coexisted with frequent anthropogenic burning, almost all of the literature discusses fire to some degree, and a significant number of studies focus exclusively on burning. This work identifies the drawbacks and challenges of utilizing prescribed fire. Since fire has the potential to produce unintended consequences, especially in ecosystems such as prairies that are already vulnerable, the research and monitoring efforts represented by this literature are crucial to ensuring that restoration efforts are successful.

In order to determine whether prescribed burning is truly promoting ecological wellbeing within the current paradigm, it is essential to examine what constitutes "ecological wellbeing." Cultural notions promoting natural or equilibrium conditions, or conditions that existed at a specific point in history, were once prevalent in the natural sciences (Wu & Loucks, 1995) and may still persist in lay conservationist culture. However, literature on the historic variability of environmental conditions, including forested ecosystems of the PNW, shows that these concepts are not applicable because ecosystems are far more dynamic than these concepts assume. Essentially, ecosystems and the humans who desire to conserve them must be able to cope with change. Many of the publications urging land managers to consider environmental instability and non-equilibrium discuss biodiversity, resilience, and/or ecosystem function. In my review of the literature on prescribed burning of prairies, I utilize these concepts as a metric to assess the efficacy of fire as a restoration tool.

Most of the literature pertaining specifically to WPG prairies suggests that prescribed burning is effective for restoration. Most prevalently, prescribed fire is used to manage invasive species, reduce fuel accumulation (Hamman et al., 2011), and improve habitat for threatened/endangered species (JBLM, 2018). It appears to promote biodiversity, resilience, and ecosystem function. Not only do these attributes represent ecological health, they are essential to adapting to climate change (Mori, 2011). While the full effects of climate change are yet to be realized, research shows prairies may be more likely to persist under future climatic conditions than adjacent forests, therefore efforts such as prescribed burning that improve the quality of prairies are beneficial to climate change adaptation (Bachelet et al., 2011).

Principles of disturbance ecology provide a theoretical framework for interpreting the impacts of fire on ecosystems. In particular, the Intermediate Disturbance Hypothesis (IDH) suggests that disturbance has the potential to increase biodiversity. In addition to the effects of disturbance on competition and resource availability, the IDH addresses landscape heterogeneity and patch dynamics as outcomes of disturbance that promote biodiversity and increase landscapes' ability to adapt to changing conditions (Roxburgh et al., 2014). This context is instructive in understanding the importance of prairies in an otherwise forest-dominated landscape.

While literature on prairie ecology, threats, and restoration may not mention or pertain directly to Hoary Elfin, they provide insight into the greater context surrounding Hoary Elfin populations and their future. Publications on Hoary Elfin themselves are the least substantial component of the literature reviewed for this project. However, as all things in nature are inextricably linked, the impacts of invasive plants, climate change,

and other such challenges all impact Hoary Elfin habitat, and thus impact the butterflies themselves. Understanding these factors and how they influence conservation efforts is necessary. Additionally, the implementation of restoration practices, or any human activities, in Hoary Elfin habitat affects their populations regardless of whether or not such is intended, so analyzing other restoration projects underway in Hoary Elfin habitat is necessary to identify whether the effects of these projects might be factors in the persistence of Hoary Elfin.

Complicating Factors in the Reintroduction of Prescribed Fire

Reinstituting the fire regime sustained by the WPG ecoregion's original human inhabitants may appear to be an enticingly simple solution. However, several complicating factors make it virtually impossible to erase the dramatic changes that the region has undergone since this fire regime was disrupted. In addition to area loss, fragmentation, and impacts of fire suppression, non-native species introduction and climate change also threaten the persistence of native species, generating concern that the additional stress of reintroducing fire, if not done carefully, may do more harm than good to these vulnerable native populations.

The introduction of non-native species has altered the fuel characteristics of the prairies (Brooks et al., 2004; Martin & Hamman, 2016) and competition with these introduced species threatens native species' populations. It is common now for ecosystems to have native variety supplemented by species introduced by humans from far corners of the globe (McNeely, 2001). 'Non-native' species are those which have been transported intentionally or unintentionally by humans or human activity to areas

outside of their natural range and begun reproducing independently of human assistance in wild/unmanaged habitats. The consequences of this can either be relatively benign or can have devastating impacts on the native ecosystems affected. Non-native species causing significant disruption to the ecosystems in which they take up residence are termed 'invasive' (Dennehy et al., 2011; McNeely, 2001). The process of removing problematic invasive species tends to be tedious (Stanley, Dunwiddie, & Kaye, 2011) and expensive (McNeely, 2001). A concern for land managers regarding the re-implementation of prescribed fire is the potential for aggressive invasive species, many of which rapidly colonize disturbed areas, to establish in burned areas and reduce available resources for natives, thus causing the overall vegetative community composition to shift towards a greater abundance of invasive species (Keeley, 2006) and further marginalizing sensitive native populations.

Climate change is an increasingly prominent reality affecting all local ecosystems, including prairies, which makes it unrealistic to simply reinstitute an historic burn regime and expect it to produce historic conditions. Bridgham and Johnson (2013) experimented with the variables of increased heat and increased precipitation during the rainy season (congruent with climate projections for the PNW) at three WPG sites ranging from Puget Sound to Southern Oregon. They found that increased heat most detrimentally affected plant germination, and that the change in these variables to simulate anticipated climate change shifted the range in which species thrived northward, in congruence with general predictions that as global warming continues, species will shift their ranges north in pursuit of temperature and other climatic conditions to which they are adapted. They also found that the number of introduced species increased under the increased heat treatments

in their study sites, and overall special richness declined. This study does not investigate the effects of prescribed fire, but the findings further demonstrate the potential for fire to exacerbate the invasive species problem.

Climate change adds another aspect to the problem of fragmentation and limited dispersal ability: if species are to survive by shifting their ranges, habitat connectivity is necessary for them to do so. Populations surviving in isolated patches of habitat may be unable to colonize more climatically suitable areas if those areas are too far away for them to disperse to. As Crausbay et al. (2017) found, fire has the potential to catalyze shifts in vegetative composition during periods of climate instability by creating open habitat available to be colonized by species adapted to novel conditions. This may be considered an adaptive trait of plant communities, as it allows primary productivity to continue despite changes in climatic conditions (Tausch, Wigand, & Burkhardt, 1993). However, such shifts in species composition have the potential to result in extinctions of declining species, and are therefore still a conservation concern. Habitat loss and fragmentation have decimated native species populations available to recolonize burned areas, and disruption of a continuous range limits native species' ability to shift their ranges towards the poles as anticipated by conservationists and demonstrated by Bridgham and Johnson (2013). Invasive species further complicate this issue, since the opening of habitat niches through disturbance (i.e. fire) presents the opportunity, as Keeley (2006) found, to allow invasive species populations to dominate. This prevents vulnerable native populations from rebounding, especially when those native populations are limited in their reproductive and dispersal abilities. Fragmentation limits dispersal

ability, thus preventing species from recolonizing burned areas and reacting to climate change.

Habitat loss and fragmentation diminishes native species' ability to reproduce and disperse. MacDougall and Turkington (2005) argue that rather than driving ecosystem changes, invasive plant species are merely the "passengers" of changing environmental conditions. They tested this hypothesis by removing biomass of two dominant invasive grass species at WPG prairie sites with a history of fire suppression and some grazing, and found that this treatment favored native species occupying niches most distinct from those removed. This suggests that direct competition does not explain the decline in native species. The authors conclude that fragmentation and consequent dispersal limitation has a greater impact on reducing native plant populations, along with environmental changes such as fire suppression.

The disruption of the fire regime employed by indigenous cultures, to which the plants are adapted, has resulted in changes in vegetative structure and fuel characteristics. Thatch accumulation (largely due to invasive grasses) and shrub (e.g. Scot's broom (*Cytisus scoparius*)) and Douglas fir (*Pseudotsuga menziesii*) encroachment increase fuel load, which is likely to increase fire intensity and result in ecological outcomes more harmful to natives (Martin & Hamman, 2016). Plants are adapted to fire regimes, not fire *per se* (Keeley, Pausas, Rundel, Bond, & Bradstock, 2011). Fire regimes are characterized by attributes such as frequency, burn severity, and size; therefore, reintroducing fire in a way that results in different characteristics than the regime to which species are adapted will produce different outcomes. Two similar aspects of fire regimes, fire intensity and fire severity, particularly affect successional patterns. Fire

intensity specifically refers to the energy released during a fire, including the resulting surface temperature (Keeley, 2009). Several factors determine fire intensity, including weather, fuel moisture, and topography, as well as ignition pattern in prescribed fire (Hamman et al., 2011). Fire severity refers to the change in above- and below-ground biomass resulting from a fire (Keeley, 2009). Research indicates that frequent, low-severity fire has greater ecological benefits, especially in prairies and oak savannas, compared to high-severity fire (Scharenbroch, Nix, Jacobs, & Bowles, 2012). However, the presence of certain new plant species due to invasion, fuel characteristics due to fire suppression, and altered climatic conditions due to climate change make higher-severity fire more likely. Prairies are therefore not likely to react to fire in the same way they did prior to European colonization.

Claiming that a species or ecosystem is "fire-adapted" does not mean that it is directly dependent on fire or benefits from fire. It means that it has evolutionary adaptations enabling it to rebound from stress or mortality associated with fire (Keeley et al., 2011), and may experience habitat improvements in the long run due to fire (Hamman et al., 2011; MacDougall & Turkington, 2005; McMullen, Leenheer, Tonkin, & Lytle, 2017) (although many species who do not possess such adaptations still may thrive in burned due to the abundance of certain resources) (Keeley, 2006). Species and ecosystems in need of restoration work are already disadvantaged by the factors discussed above; therefore, simply reintroducing fire without careful planning, research, and additional restoration efforts is unlikely to produce overwhelmingly positive results.

These issues illustrate the need to consider the future rather than the past when enacting land management policies. Invasive species have altered fuel characteristics,

fragmentation has limited native species' ability to disperse, and climate change has begun altering the conditions under which species need to be able to survive. As environmental conditions shift, so do the concepts influencing management practices and the science to support them, and this serves to inform restoration efforts.

Defining Restoration Objectives and Management Goals

Restoration objectives must be clearly defined in order to be successful, and the process of defining such objectives may be guided by cultural perceptions as well as scientific knowledge. Science may help us measure whether restoration strategies are effective in producing certain outcomes, but how do we decide which outcomes are desirable? Preserving ecosystems in their 'natural' state, restoring equilibrium, or utilizing historical conditions as a baseline have prevailed in ecological theory in the past (Sprugel, 1991; Wu & Loucks, 1995), and may still hold appeal in lay conservationist culture as they seem to offer a way to right perceived human wrongs against the natural world. However, they have been shown to be misguided and unrealistic, as they misrepresent the true dynamism and variability of ecosystems (Sprugel, 1991; Wu & Loucks, 1995).

In conservation, an ecosystem in its 'natural' state generally means, or is interpreted as, free of human influence (Hunter, 1996). Not only have prairies always been shaped by human activities, but in the age of anthropogenic climate change, arguably nothing is free of human influence. Climate instability directly impacts disturbance regimes and ecosystem characteristics, especially vegetation (Mori, 2011). Naturalness by Hunter's (1996) definition is therefore unattainable if ecosystem

characteristics produced by indigenous civilizations are used to set targets or if current climatic instability is considered human influence.

Additionally, the concept of equilibrium rests on the assumption that healthy ecosystems exist in a stable state, and when disturbed progress to a “climax” state (Sprugel, 1991). Mori (2011) explains the role of both dynamic equilibrium and non-equilibrium in shaping ecosystems. Historically, prairies may have existed in dynamic equilibrium in that relatively small, regular disturbances maintained a landscape mosaic of relatively constant proportions of different vegetation types and successional stages. However, large and severe disturbances generate non-equilibrium in ecosystems, and these types of disturbances are associated with climatic instability (Mori, 2011). It may be unrealistic to expect ecosystems, even those subject to conservation measures, to maintain equilibrium under current and projected climate instability.

Furthermore, when considering long-term environmental history, it becomes apparent that environmental conditions have shifted countless times, even rapidly and drastically, altering vegetative composition of ecosystems (Crausbay et al., 2017; Tausch et al., 1993). Willis and Birks (2006) propose using paleoecological records to discern the full range of variability experienced by ecosystems over long timescales and using this perspective to conserve ecosystems more effectively. Swetnam, Allen, and Betancourt (1999) also advocate using this strategy, and discuss the drawbacks of using historical conditions as a reference, citing Sprugel’s (1991) argument that utilizing a fixed point in history to represent the correct state of an ecosystem is not an effective conservation strategy. Historical records cannot be used as a baseline to restore ecosystems to the way

they were at a fixed point in time, because the conditions that influenced them may have changed.

Drechshler, Lourival, and Possingham (2009) suggest the use of modelling algorithms using the spatial properties of disturbance events to improve site selection for conservation. Their strategy is designed to encompass a variety of successional models when choosing sites in order to represent variability. However, it fails to acknowledge that even in heterogenous landscapes, the representation of various successional stages and vegetative communities changes over time (Sprugel, 1991). The dynamic equilibrium described by Mori (2011), in which a patchwork of successional stages creates a diverse, heterogenous landscape, will be disrupted by large-scale disturbances, which Mori argues are a component of non-equilibrium with a fundamental role in shaping ecosystems.

Sprugel discusses several examples of landscapes that demonstrate non-equilibrium qualities, including forests of the Pacific Northwest. Over long historic timescales, catastrophic fires have influenced the proportion of landscapes occupied by various vegetative communities. Prior conceptions about successional processes assumed that even after these types of events, ecosystems would progress to their "climax" state (1991). However, Mori (2011) explains that alternate successional pathways are possible, even as a feature of resilience. Weisberg and Swanson (2003) support Sprugel's claim that PNW forests follow the non-equilibrium paradigm. Their analysis of the fire history of the western Cascades shows that major shifts in vegetative composition have happened historically, associated with large disturbance events (which Crausbay et al. (2017) show are associated with climatic shifts), resulting in different seral stages and vegetative communities being more dominant or relatively scarce at different points in time. Sprugel

points out that using conditions at a fixed point in history as a reference for what is "natural" vegetation is problematic because the point of history in question is arbitrary (e.g. old photographs, written documentation by ecologists or colonizers, or fossilized pollen only reveal vegetative composition at that point in time), neglecting temporal variation (1991). Furthermore, climate change and the persistence of invasive species make recreating past conditions virtually impossible (Dunwiddie & Bakker, 2011).

The caveat to this open-mindedness about climatic shifts and the ability of ecosystems to respond is that other anthropogenic environmental impacts have weakened this ability. For example, habitat fragmentation has limited species' ability to shift their ranges to areas with more suitable climatic conditions (Dunwiddie & Bakker, 2011), and invasive plant species may reduce diversity by forming monocultures (McNeeley, 2001). These compounding problems mean that ecosystems are less likely to be able to cope with change the way they have in the past. Therefore, a framework is needed to assess the health of ecosystems in the face of change. While 'natural,' 'in equilibrium', or 'historical' may not be the best options, biodiversity, resilience, and ecosystem function may serve as a lens through which to analyze and evaluate restoration activities, and adapt them to rapidly changing environmental conditions, scientific knowledge, and cultural values. Ideally, preserving these qualities of ecosystems will allow them to do what life on earth does best– adapt.

Biodiversity refers not only to number of distinct species (species richness), but to genetic diversity, functional diversity, and diversity at scales greater than species level, such as landscape and habitat diversity (Gaston, 2010). The more biodiverse an ecosystem, the more capable it is of adapting to change and rebounding from disturbance

(i.e. resilience) (Hisano, Searle, & Chen, 2018) because there is more genetic diversity upon which evolutionary forces can act, and more functionally diverse organisms capable of filling different niches created by the environment.

Mori (2011) defines resilience as "the key to conserving ecological integrity via the ability to cope with inevitable changes" (pg. 280) and "without interfering with inevitable change" (pg. 290). While the term may have entered the conservation dialogue as a way to describe an ecosystem's ability to return to its prior state following disturbance, it has expanded to include alternate stable states and the ability of an ecosystem to maintain functionality under a range of environmental conditions and the disturbances that come with them (Oliver et al., 2015). In fact, the concept of resilience in an ecological context was first developed in 1973 by Holling (Newton, 2016), who analyzed empirical models seeking to represent population dynamics and environmental shifts, and concluded that the concept of equilibrium did not accurately describe the dynamics of ecosystems (Holling, 1973).

Oliver et al. (2015) define ecosystem functions as "the biological underpinning of ecosystem services," where ecosystem services are "outputs of ecosystem processes that provide benefit to humans" (pg. 674). Ecosystem services can provide economic benefit, such as timber production, or simply survival needs, such as oxygen production via photosynthesis, but the term is generally applied specifically in the context of human needs. I argue that ecosystem function is a more appropriate focus in the context of conservation because it can be applied to processes which benefit non-human components of the environment and have intrinsic value regardless of human benefit.

Focusing on biodiversity, resilience, and ecological function is a flexible approach that allows the balance of combatting detrimental anthropogenic environmental impacts without attempting to recreate historical conditions despite irreversible changes. Additionally, even though misguided assumptions are often only revealed in hindsight, these concepts may offer an objective metric to guide management decisions that represent cultural conservation values without absorbing ambiguous cultural ideals.

Benefits of Reintroducing Prescribed Fire

Despite the challenges, prescribed fire has been reinstituted on South Sound Prairies, and a large body of research and monitoring programs have emerged to document threats to prairie persistence and the efficacy of restoration methods. A collaboration of several regional conservation groups provides the resources necessary to address the complexities of prairie restoration and implement effective strategies. Lands stewarded by JBLM are subject to extensive restoration activities considering their prominence as a major stronghold of intact prairie habitat in the region. The prescribed burning program began in 1983 as a component of the Integrated Natural Resource Management Plan. Initially, prescribed burning served the benefits of reducing fuel loads and the risk of wildfire, and maintaining open spaces for military training. However, the goals of this program have shifted to increasingly focus on ecological objectives, namely improving habitat for endangered species. In 2012, USFWS at JBLM took over the planning and implementation of the ecological burning program (JBLM, 2018). This spurred the formation of a partnership between JBLM and other regional groups, namely The Nature Conservancy and a local affiliate, The Center for Natural Lands Management,

in order to provide the skilled personnel necessary to carry out JBLM's ambitious ecological burning program (Puget Sound Ecological Fire Partnership, 2016), which includes the goal of burning 4,200 acres per year by 2020 (JBLM, 2018).

Research shows that prescribed fire is an effective prairie restoration method. One beneficial function is to prevent the shrinking of prairies due to the encroachment of trees onto the edges (Hamman et al., 2011). While forest succession is a natural process, in the absence of subsequent disturbance it reduces habitat heterogeneity and biodiversity as landscapes shift to homogenous forest cover rather than containing a variety/gradient of habitat types. Since this process would have happened thousands of years ago if not for indigenous burning (Boyd, 1999), the persistence of prairies depends on anthropogenic disturbance. While prescribed fire may not be effective in completely eradicating invasive species from WPG prairies, research shows that it has a positive impact on plant diversity (Rook et al., 2011). Additionally, while impacts on individual species may be uncertain, prairie landscapes may thrive under future climatic conditions while adjacent forests suffer, and prescribed fire may be useful in encouraging vegetative communities to transition to states better suited to future climatic conditions (Bachelet et al., 2011).

Based on available research into the effects of prescribed fire on WPG prairies, management of problematic invasive species appears to be one of the most prominent goals (Dennehy et al., 2011; Hamman et al., 2011; Rook et al., 2011). Prescribed fire has the potential to reduce the dominance enjoyed by aggressive invasive species and increase species richness (Dunwiddie, Alverson, Martin, & Gilbert, 2014; Rook et al., 2011; Trowbridge, Stanley, Kaye, Dunwiddie, & Williams, 2017) and habitat heterogeneity (Bachelet et al., 2011; Drechsler, et al., 2009). Despite evidence showing

that some ecosystems experience a takeover by invasive vegetation following fire (Keeley, 2006) several studies pertaining specifically to WPG prairies suggest that this is not the case, citing evidence that prescribed fire benefits native species, especially in the short term, and when combined with other methods such as herbicide treatment of invasive species and native seeding that mitigate the reproduction and dispersal limitations suffered by declining populations (Alba, Skálová, McGregor, D'Antonio, & Pyšek, 2015; Nuckols, Rudd, Alverson, & Voss, 2011; Rook et al., 2011; Trowbridge et al., 2017). Ultimately, the results of these studies do not indicate major success of prescribed burns in reducing the presence of invasive plants, and Dennehy et al. (2011) found mixed results when investigating species-specific responses of invasive plants to prescribed fire. It seems that burning has an impact in reducing the competitive dominance of these species temporarily and boosting native plant production.

Additionally, as MacDougall and Turkington discuss in their investigation of the "driver vs. passenger" explanation of invasive species dominance, much of the problem may be due to ecosystem restructuring rather than direct competition. While the authors do not explicitly advocate the reintroduction of prescribed fire in mitigating this situation, their conclusion supports the idea of using landscape level tactics that improve habitat for declining native populations and promote recruitment of these species (2005). Prescribed fire is certainly effective in changing ecosystem characteristics, but the effects themselves vary on short- and long-term timescales. Different aspects of fire regimes (e.g. intensity, return interval) lead to different outcomes and may have different effects on different species (Hamman et al., 2011).

In addition to evidence that low-severity fire has greater ecological benefits than high-severity fire in prairies and oak savannas (Scharenbrock et al., 2012), there is some evidence that wildfire favors invasive species establishment more than prescribed fire (Alba et al., 2015). Fire severity is strongly influenced by fire intensity. The factors that determine fire intensity (weather, fuel moisture, topography, and ignition pattern in prescribed burning) can be manipulated to produce certain outcomes in prescribed burn programs. By timing burns to target preferable weather conditions and utilizing topography and ignition pattern to determine spread pattern, for example, those administering prescribed burns have some ability to control for outcomes (Hamman et al., 2011). Because of the relative ability to control for certain variables that influence fire severity, prescribed fires are more likely to burn at lower severity than wildfires, if such is the desired effect, and those responsible for conducting the burn understand these variables (Martin & Hamman, 2016). Wildfire is a natural process, and over a long enough time frame it is a matter of 'when' an area will burn, not 'if.' In this sense, prescribed fire has the potential to mitigate the undesirable effects of wildfire by intentionally burning areas when conditions make low-severity burning likely, rather than risking a chance wildfire when high-severity results are likely. Especially considering predictions that climate change will increase the frequency of wildfires (Bachelet et al., 2011), prescribed fire may become more desirable as a mitigation tool and increase the ability of restoration projects to anticipate and achieve certain outcomes. While at least one study questions the efficacy of prescribed fire to reduce fuel loads without increasing the likelihood of invasive plant infiltration (Keeley, 2006), if the alternative is risking a

large wildfire likely to produce more detrimental impacts, prescribed fire may still be the best option.

The challenges that invasive plants present to ecosystem restoration-- and to biodiversity, resilience, and ecological function of ecosystems-- are well-realized, especially on South Sound prairies. The benefits and limitations of fire for invasive plant mitigation are relatively well documented. Additionally, the benefits of fire in reducing shrub encroachment and thatch/fuel accumulation are self-evident. However, another threat is gaining increasingly prominent concern and recognition for its drastic influence on virtually all aspects of environmental systems: climate change.

Climate projections for the PNW, presented by Bachelet et al. (2011), are based on data from the Intergovernmental Panel on Climate Change (IPCC) that has been used by two regional groups, the Climate Impacts Group (CIG) in Seattle, WA and the MAPSS team in Corvallis, OR, to produce regional climate models. Different projected outcomes vary based on interactions of temperature and moisture. In general, temperature is expected to increase on average. Precipitation is expected to increase as well, but due to increased temperature and therefore increased evapotranspiration, summer drought is still expected to worsen. In addition, the increase in temperature means that more precipitation may fall as rain instead of snow in the winter, thus decreasing summer moisture from snowmelt and causing more severe summer drought. The PNW already experiences climatic oscillations (such as El Niño and La Niña) and temperature variability, and these are expected to become more pronounced. This means that in addition to overall long-term climatic trends, weather patterns may vary more significantly from year to year as well (Bachelet et al., 2011). Increased understanding of

the inevitability of climate change and associated disturbance events has led to the emerging focus on resilience (Dunwiddie & Bakker, 2011; Mori, 2011).

Weisberg and Swanson (2003) demonstrate that over the course of long-term ecological history, the relationship between disturbance and environmental fluctuations resulted in changes in vegetative composition of landscapes. In the PNW, this resulted in changes in the relative abundance and sparseness of various seral stages of plant communities. Prairies are an early seral community, and historically were prevented from progressing to later seral stages by anthropogenic burning (Boyd, 1999). Old growth conifer forests are a late seral community on the far end of the spectrum of successional stages. Therefore, research demonstrating the historic fluctuations of plant community composition in relation to changing climate suggests that these ecosystems may shift in abundance as current climatic changes continue.

Literature on the potential effects of climate change on WPG prairies is sparse compared to the literature on invasive plants, especially in relation to prescribed burning (in fact, none of the research conducted specifically on WPG prairies explicitly investigates the relationship between prescribed burning and the effects of climate change). Concern over prescribed burning in regards to climate change generally reflects the concern that vegetative communities are less likely to return to their pre-burn characteristics due to environmental shifts. However, as the possibility of halting anthropogenic climate change becomes ever slimmer, strategies for adaptation become increasingly valuable options. If vegetative communities are bound to shift in response to climate change anyway, prescribed burning, especially as it is shown to benefit prairie ecosystems otherwise, may function as a component of adaptation strategies.

Literature on the history of climatic oscillations of the Quaternary period, and the climatic trajectory of the Holocene epoch, can help us further extrapolate a relationship between climate, fire, and vegetation that may be useful in understanding current and possible future paradigms. The last two million years of the Quaternary period have been characterized by glaciations intermediated by interglacial periods, the current Holocene epoch being the most recent (Tausch et al., 1993). The PNW began experiencing Holocene-like conditions with the retreat of the Cordilleran Ice Sheet about 14,000 years ago. The early Holocene was characterized by rapid and significant climatic changes. About 7,000 years ago the climate became relatively stable, and those conditions have persisted until the present (Crausbay et al., 2017).

Tausch et al. (1993) suggest that the climatic fluctuations of the Quaternary period have influenced the evolution of vegetative communities' response to climatic change. In this sense, one interpretation of past and current plant community changes is that perhaps plant communities as a whole have evolved the ability to change composition in response to changing climatic conditions. This ability may represent a type of resilience in itself, as it allows for primary production to continue across much of the Earth's land surface despite climatic fluctuations. This is relevant to present concerns over the possible effects of prescribed fire in that, while anthropogenic climate change is certainly problematic, concerns over the ecosystem changes that follow may not preclude the use of prescribed fire.

Crausbay et al. (2017) reconstructed the history of high-severity fire events and vegetative composition change by analyzing charcoal records and pollen records in western Washington. Their findings suggest that in times of relative climate stability,

vegetative communities are resilient to high-severity fire in that they tend to follow successional pathways towards their original pre-fire state. However, in times of directional climatic change, high-severity fire initiates vegetative succession towards alternate community composition in response to the changing climatic conditions.

While these findings may have dire implications for the forested ecosystems in close proximity to WPG prairies, the prairies themselves may not suffer under future climatic conditions in the same way. Bachelet et al. (2011) suggest that Douglas Fir-dominated mixed-conifer forests of this ecoregion may be more likely to suffer from severe droughts and extreme weather events than prairies, and they discuss the potential for prescribed fire to encourage the vegetative composition of ecosystems towards states that will be better adapted to future climatic conditions. Based on Crausbay et al.'s (2017) findings, this would essentially mimic the role of fire throughout the history of the Holocene epoch in catalyzing vegetative composition shifts in response to changing climatic conditions. Bachelet et al. (2011) advocate that prairies and the biodiversity that they harbor across the climatic gradient spanned by the WPG ecoregion especially hold conservation value in terms of climate change adaptation, as conserving prairies may present an opportunity to maintain a native ecosystem if global warming takes its predicted toll on PNW conifer forests. While the precise species composition of prairies may change, biodiverse, resilient, and ecologically functional early seral ecosystems are likely to increase in prominence. Prescribed fire plays a role in maintaining the ecological integrity of prairies, which will be of immense value to PNW landscapes if conifer forests decline.

Prescribed fire has the potential to enhance ecosystem resilience by increasing biodiversity, especially by benefitting populations of declining prairie-specialist species. By preventing landscape succession to homogenous forest cover, fire maintains greater diversity from the landscape to the species level and increases the availability of prairie habitat for species that depend on it (Hamman et al., 2011). In addition to maintaining prairies over time, disturbance via burning reduces the competitive dominance of aggressive invasive species and allows native species to reestablish (Rook et al., 2011). Conserving biodiversity is vital if ecosystems are to adapt to a changing climate (Mori, 2011); large disturbances such as wildfire are likely to increase under climate instability, and preserving biodiversity across the region allows the establishment of ecosystems that will thrive under new climatic conditions (Bachelet et al., 2011).

Perspectives from Disturbance Ecology

Disturbance ecology provides a framework for understanding the role of fire in ecosystems. The definition of ecological disturbance, provided in Sousa (1984) and cited in Odion and Sarr (2015), is "a discrete, punctuated killing, displacement, or damaging of one or more individuals (or colonies) that directly or indirectly creates an opportunity for new individuals (or colonies) to become established" (Sousa, 1984, pg. 356). In this context, disturbance does not necessarily disrupt equilibrium (Odion & Sarr, 2015), but disturbance is an integral part of the non-equilibrium paradigm described by Mori, as disturbance events are often the catalysts of changes in ecosystems (2011). The current environmental upheaval spurred by human activities raises legitimate concern for the future, despite evidence of past environmental shifts, as the extent of anthropogenic

impacts on all facets of the environment seems unprecedented. However, disturbance is shown to play a beneficial, or at least established and accommodated, role in natural systems.

The definition provided by Sousa (1984) relates environmental disturbance to the availability of resources in an ecosystem. The intermediate disturbance hypothesis (IDH) is one proposed explanation for positive outcomes of disturbance based on an increase in available resources. It postulates that disturbances of optimal size and frequency have a positive impact on species diversity, and that biodiversity is highest at intermediate stages of succession and significantly lower in older ecosystems. The speculation is that over time, more competitive species will monopolize resources and dominate, but disturbance may reduce their competitive advantage (Osman, 2008). Additionally, the IDH accounts for both diversity within patches and diversity between patches of habitat; not only do patches at intermediate successional stages (in theory) contain higher species diversity than older patches, but patches of different ages contain different assemblages of species, therefore increasing the diversity of the entire system (Roxburgh et al., 2004).

The IDH was first articulated by J. H. Connell in a 1978 paper that compared several explanations for the high diversity found in tropical forests and coral reefs and found the IDH to be most fitting (Osman, 2008). An alternative explanation, arguably the most common, for high diversity is niche diversification, where species diversity is achieved when species evolve to fill different ecological niches as a response to competitive pressures (Osman, 2008; Roxburgh et al., 2004). It is noteworthy that niche diversification rests on the idea that ecosystems reach a state of equilibrium, whereas the IDH rests on the idea that they do not.

While the IDH is appealing in its apparent simplicity and intuitiveness, it is complicated at best when applied to the real world. Some complexity is due to the fact that attributes of a disturbance, such as frequency, scale, and spatial extent are relative to the ecosystem or species in question (Osman, 2008). While Connell may have found evidence in rainforests and reefs, which both experience successional trajectories along similarly long timescales, disturbance levels that constitute frequent vs. infrequent timing, large vs. small magnitude, large vs. small spatial extent, etc. depend on the context. In cases where empirical evidence does not seem to support the IDH (meaning species diversity graphed against time since disturbance forms a unimodal distribution showing the highest diversity an intermediate amount of time since disturbance (Figure 2), it can be difficult to tell if the hypothesis has been falsified or if "intermediate disturbance" has been misinterpreted in context (Osman, 2008). It is therefore virtually

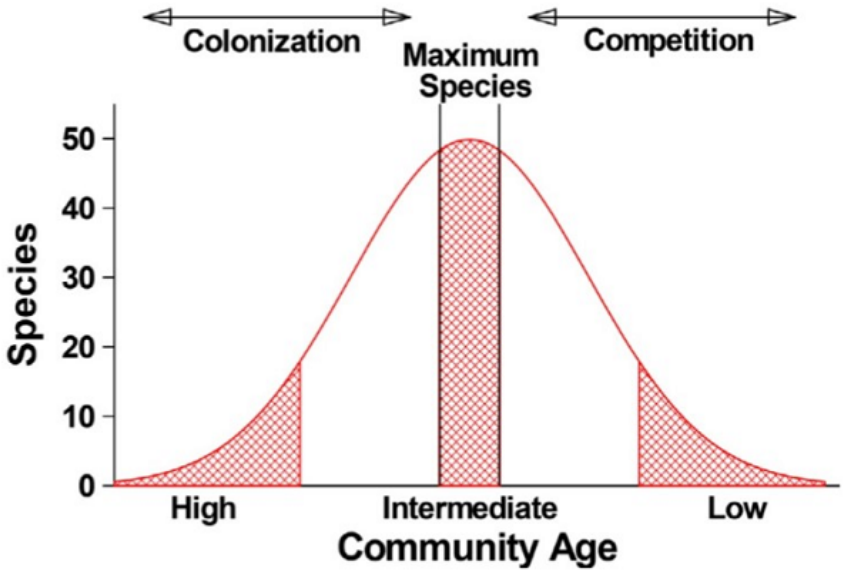


Figure 2. A simplistic representation of the relationship between time since disturbance (community age) and biodiversity (species) suggested by the Intermediate Disturbance Hypothesis.
Adapted from Osman, R. W. (2008). Intermediate Disturbance Hypothesis. *Encyclopedia of Ecology*, 1986-1994. https://doi-org.evergreen.edm.oclc.org/1016/B978-008045405-4.00510-3

impossible to completely prove or falsify the IDH. Roxburgh et al. conducted a number of empirical studies attempting to do so, and while their paper defends the utility of the IDH, it is hardly conclusive. The authors suggest the IDH may serve best to describe a collection of "mechanisms of coexistence" rather than as one universally applicable rule (2004).

A 2013 article by Fox criticizes the IDH on the grounds that it is empirically false and logically unsound. The claim against its empirical merit is essentially that species diversity of a disturbed ecosystem rarely forms the unimodal distribution representing the hypothesis that species diversity declines if enough time passes post-disturbance, shown in Figure 2. Fox's repudiation of the IDH's theoretical merit mainly critiques assumptions about interspecies competition that the IDH relies on, such as the ideas that disturbance weakens competition and reduces competitive exclusion (Fox, 2013). These criticisms may be warranted in the absence of definitive proof that species diversity declines in the absence of disturbance in all ecosystems. Additionally, literature discussing applications of the IDH relies on the assumption that species are either superior or inferior competitors and that this dynamic drives coexistence (Osman, 2008; Roxburgh et al., 2004). However, debate over the validity of the IDH shows that the definition itself remains unclear, with defenders claiming that the loopholes illuminated by critics are a misinterpretation of the IDH (Sheil & Burslem, 2013).

While the IDH may not be a proven theory, especially regarding the quantification of diversity levels represented by the unimodal graph in Figure 2, in the context of prairie restoration, the IDH may still be instructive in conceptualizing the relationship between disturbance, biodiversity, and non-equilibrium. Competitive

exclusion in prairies may be borne out by entire ecosystems, not individual species. The encroachment of Douglas fir-dominated forests creates habitat for many species that do not inhabit prairies; therefore, the competitive dominance displayed by Douglas Fir encroachment may not necessarily result in fewer species in a patch of habitat that has transitioned from prairie to forest. However, the mixed-conifer forest ecosystem itself can be seen as the exclusive competitor, forcing the prairie ecosystem out of the landscape in the absence of fire. This shows that disturbance which prevents forest encroachment has the potential to increase landscape level diversity by allowing both ecosystems to coexist, regardless of which habitat type boasts a higher tally of species present.

This supports the findings by Roxburgh et al. on the increase of between-patch diversity. These authors cite an older study in which Chesson and Huntley (1997) suggest the IDH be renamed the "successional mosaic hypothesis" (as cited in Roxburgh et al., 2004, p. 360). This interpretation of the IDH may be especially applicable to prairies, as patch dynamics are certainly at play as restoration practitioners apply fire. One mechanism of coexistence explaining between-patch diversity is described by Roxburgh et al. as the "storage effect." This concept refers to the potential for species to "store" their populations in patches of habitat at the optimal successional stage, thus maintaining a population robust enough to colonize other patches of habitat once they reach the optimal successional stage or (in theory) the original patch progresses to a less optimal successional stage (2004). Whether fire is applied to landscapes to stimulate edible plants, prevent forest encroachment, manage invasive species, or generally promote biodiversity, this offers an explanation for why fire might encourage these outcomes.

The storage effect explains a mechanism by which populations are resilient to disturbance. McMullen et al. (2017) develop a model to simulate population dynamics considering that disturbance may initially cause high mortality in a population but indirectly improve habitat and promote future population growth. This model conceptually supports the IDH in that it implies that disturbance reduces competitive pressure on species by increasing available habitat, which especially benefits early seral species.

Odion and Sarr (2015) utilize the IDH to examine PNW forested landscapes, and ultimately find evidence supporting the relationship between disturbance and diversity outlined by the IDH. Specifically, they find that landscapes dominated by even-aged conifer stands harbor lower levels of diversity across taxonomic groups, and that competition appears to be a major control on plant diversity in these landscapes. Landscapes containing early seral patches harbored higher levels of diversity. Osman's (2008) description of the IDH discusses disturbances of optimal size and frequency, and Odion and Sarr find that disturbances that result in excessive environmental stress can reduce diversity while disturbances that result in too minimal of an impact do not provide diversity-increasing benefits. However, they suggest that rather than a regime of one ideal type of disturbance, a varying range of disturbance characteristics (i.e. frequent and mild to infrequent and severe) contribute to maintaining diversity. Mori (2011) also claims that varying types of disturbance, including large, catastrophic ones, are instrumental in promoting biodiversity.

Excessive environmental stress via disturbances that are too frequent or too large to allow for biotic communities to establish and reproduce may reduce diversity and

result in biotic impoverishment. This concern is compounded by abiotic stressors such as fragmentation and climate change (Odion & Sarr, 2015), especially considering predictions that climate change will increase the frequency and magnitude of severe disturbances (Bachelet et al., 2011). It is therefore important to continue observe and assess ecosystem health while implementing restoration practices, to ensure that the balance of competitive exclusion vs. environmental stress is maintained. Different species and communities are likely to respond differently to various disturbances, and while this may be part of the reason that disturbance regimes promote diversity, it is important recognize when the compounding pressures of the modern age may be too much.

Hoary Elfin and Kinnikinnick

While a landscape-level perspective is essential for managing disturbance-prone ecosystems, impacts on individual species must be considered. As discussed, human activities have marginalized prairie habitat and many resident species, which impacts their ability to cope with landscape-level changes. Furthermore, legislation such as the Endangered Species Act (ESA) mandates the conservation of all species. While the requirements of the ESA apply to species listed as Endangered or Threatened, it still creates an imperative to enact species-level conservation efforts; pre-emptive action may prevent a species from being listed, but no action may result in a species declining to the point of being listed under the ESA.

Requirements of the ESA strongly influence the Integrated Natural Resource Management Plan at JBLM. 10 species hosted by JBLM are already listed under the ESA, three of which rely on prairie habitat (Streaked Horned Lark, Mazama Pocket

Gopher, and Taylor's Checkerspot Butterfly) (JBLM, 2018). Several other butterfly species unique to South Sound prairies are considered for conservation (TNCW, 2011).

Hoary Elfin (*Incisalia polia obscura*) butterflies inhabit some of the remaining South Sound prairies; they are classified as a Washington state species of conservation concern (Schultz et al., 2011). This classification is less severe than a listing as threatened, and does not come with management imperatives, but demonstrates that the population is vulnerable and in need of monitoring to detect further decline. The subspecies *I. polia obscura* is endemic to the South Sound (TNCW, 2011) (Figure 3).

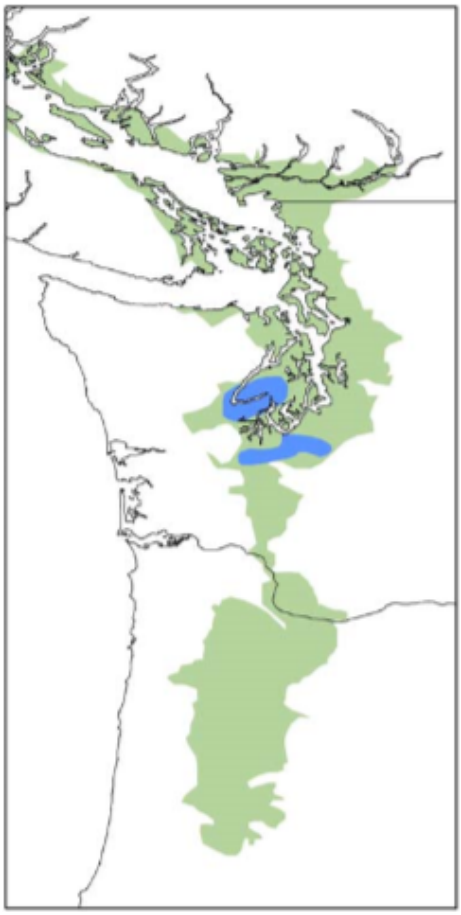


Figure 3. Historic range of WPG prairie system (green, larger shaded area) and known range of Hoary Elfin (*Callophrys polios obscurus*) as of 2011 (blue, smaller shaded area). Adapted from Schultz, C. B., Henry, E., Carleton, A., Hicks, T., Thomas, R., Potter, A., …&Anderson, H. E. (2011). Conservation of prairie-oak butterflies in Oregon, Washington, and British Columbia. *Northwest Science, 85*(2), 361-388. https://doi.org/10.3955/046.085.0211

Schultz et al. (2011) utilize the scientific name *Callophrys polios obscurus* when describing the same local subspecies, but mention uncertainty of whether the butterflies found in the Puget Sound area are actually a subspecies. Neither source provides an explanation of the differing taxonomy, nor how the populations occupying South Sound prairies relate to other Hoary Elfin populations in terms of geographic connectivity or taxonomic classification.

Hoary Elfin are a small, brown, relatively non-descript butterfly. Adults can be found on or near patches of Kinnikinnick from late April to late May, as they rarely stray far from their host plant. They spend their other life stages nestled within the patches of Kinnikinnick. Information on Hoary Elfin ecology is lacking; sources do not report the role they play in ecological processes such as pollination, herbivory, or as prey.

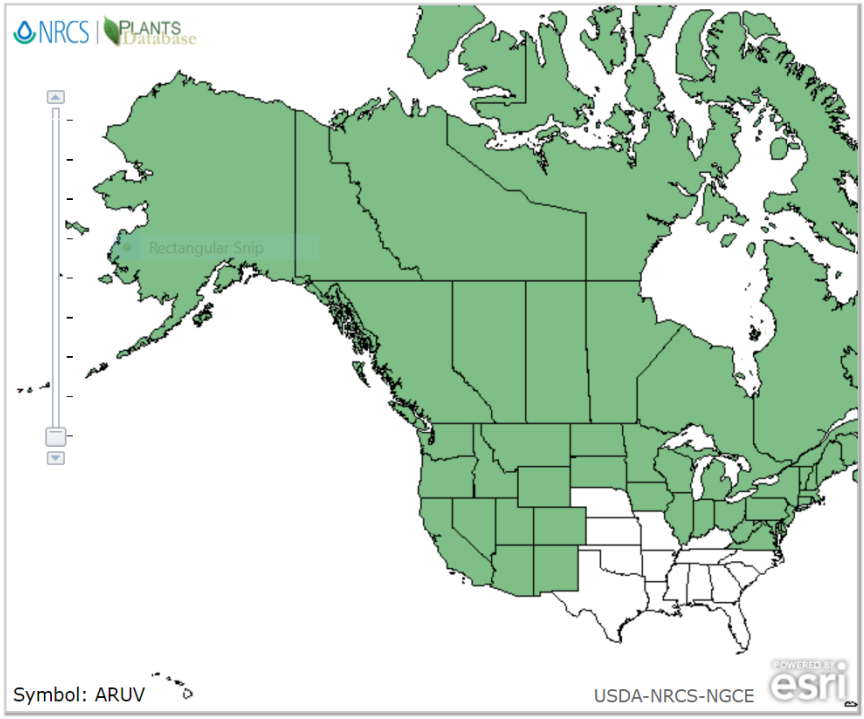


Figure 4. Distribution of Kinnikinnick (*Arctostaphylos uva-ursi*) in the United States and most of Canada.
Adapted from United States Department of Agriculture. (n.d.). *Arctostaphylos uva-ursi* (L.) Spreng. kinnikinnick. Retrieved from https://plants.usda.gov/core/profile?symbol=ARUV

Kinnikinnick (*Arctostaphylos uva-ursi*), the larval host plant of Hoary Elfin, occupies a broad distribution throughout North America (Figure 4) (United States Department of Agriculture, n.d.), and can be found in a very diverse range of habitats. It grows in early seral environments, in sunny areas, and grows well in sandy, rocky, or nutrient-poor soils. Its blooming coincides with Hoary Elfin adult flight season (likely providing a nectar resource), and produces red berries. It reproduces primarily asexually, growing in a creeping pattern with roots emerging from nodes on the stems. The berries serve as a food resource for wildlife, and may be significant for this purpose as they are very hearty and remain into the winter (Crane, 1991).

Kinnikinnick is known to be resilient to fire due to its ability to re-sprout from underground structures after above-ground biomass is burned. A study from del Barrio et al., conducted in Spain in 1999, compares Kinnikinnick's response to burning and cutting, and found no significant difference. This suggests that while Kinnikinnick may be adapted to cope with burning, stimulation of regrowth is not dependent on burning specifically versus other mechanisms of disturbance or biomass loss. Information on Kinnikinnick fire ecology synthesized for the US Forest Service by Crane (1991) states that it is best adapted to frequent, low-severity fire, and that it may not be able to regenerate if the root crown is killed by fire. Crane also reports that the seeds may survive low-severity fire in the seed bank, and germinate when heated by fire and when leaf litter and detritus build-up is removed from the soil surface (i.e. by burning).

Hoary Elfin persistence depends on the availability of quality habitat, which is essentially unshaded areas containing Kinnikinnick. Based on its broad geographic and ecological range and resilience to disturbance, Kinnikinnick does not appear to be facing

the same threats as Hoary Elfin in terms of continued existence as a species. However, Kinnikinnick's response to the specific fire regime implemented on South Sound prairies is relevant for study because its persistence in that particular habitat is essential to Hoary Elfin persistence. Additionally, prescribed fire is utilitarian in that it targets a myriad of species and their habitat, but this may create a complex tangle of outcomes because its effects are unlikely to be consistent among individual species. Ecological burning must strike a balance of creating the greatest benefit for the most species, but without irreparably harming populations that respond negatively. While this study seeks to inform management practices that best benefit Hoary Elfin, the way these practices affect other prairie species must be considered.

Conclusion

The most notable gap in knowledge in the literature is the absence of publications on Hoary Elfin butterflies. As recommended by Hamman et al. (2011), species-specific studies are important to fire effects research. These are necessary not only to conserve sensitive species themselves, but to find trends in the needs of different species to account for species conservation goals. Fire has the potential to be effective in that a landscape level tactic can benefit a wide variety of species, but the devil may be in the details in terms of effects on individual species, especially considering that strict legal conservation mandates, such as the Endangered Species Act, require that impacts on all individual species be taken into account. Climate change is emerging as an increasingly dire concern for land management and conservation, so increased understanding of its impacts will likely permeate most aspects of restoration. Additionally, as the need for

measures focused on adapting to the effects of climate change becomes more urgent, developing these strategies and testing their efficacy will warrant increased attention.

The primary area of disagreement appears to be on the risk of burned areas being susceptible to plant invasion by non-native species. However, virtually all literature pertaining specifically to South Sound prairies suggests that fire plays a beneficial role in restoration projects (Dennehy et al., 2011; Hamman et al., 2011; Nuckols et al., 2011; Rook et al., 2011; Stanley et al., 2011). Since prescribed fires are in targeted and relatively heavily managed areas where it is feasible to include herbicide and native planting, this increases its efficacy; in comparison, studies raising concern over the potential for fire to exacerbate invasive plant problems, such as Keeley (2006), seem to focus more on less managed areas more likely to burn in wildfires without mitigation strategies for invasive plants in effect.

There is disagreement on theoretical ideas, such as the IDH and the concept of resilience. This does not present major concern for South Sound prairie restoration because prairie restoration is very specific and targeted, allowing those dedicated to restoring these ecosystems to operate within a narrow scope and focus exclusively on what works for prairies. However, hypotheses about disturbance can serve as a basis for further research. For example, the storage effect may influence Hoary Elfin recolonization of Johnson prairie (or lack thereof, if the surrounding population is not able to rebound without assistance). When Hoary Elfin do reestablish a population in Johnson prairie, they might display a preference for burned or unburned Kinnikinnick, which will provide insight on the impact of fire on this particular species.

The most contemporary applicable research, such as McMullen et al. (2017) and Brown, York, Christie, and McCarthy (2017), shows that modelling technology is becoming prevalent in ecological and conservation research. These studies focus on the habitat effects of disturbance and the effects of fire on pollinators, respectively, and both are based on the development of digitized modelling to predict outcomes. Modelling requires defined variables in order to generate patterns and predict outcomes, therefore identifying these variables is essential to utilize this technology to its full potential, and ecological studies are the means for doing so. For example, the time it takes Hoary Elfin to recolonize a burned patch of Kinnikinnick relative to the distance of the nearest source population are aspects of species ecology that may aid in developing effective conservation plans.

Hamman et al. (2011) recommend the need for species-specific research on prairies subject to prescribed burning, in light of the complexities of reintroducing fire. I hope that my contribution to this project may add another small piece to the puzzle of effective conservation of a diverse, dynamic, and complex ecosystem. Beyond Hoary Elfin conservation, I hope my research may provide insight into the nature of ecological disturbance and how ecosystems adapt to changing environmental conditions.

METHODS

While Hoary Elfin occupy other sites in the South Sound area, all data used for this study were collected on sites stewarded by Joint Base Lewis-McChord, and are retained by USFWS. In spring 2018, I assisted with Hoary Elfin surveys as part of USFWS's ongoing monitoring project, along with other personnel (USFWS employees and interns). Additionally, I assisted in mapping Kinnikinnick patches in Training Area 6, along with other personnel. All other spatial data (Hoary Elfin observations from previous years, Kinnikinnick patches in all other locations, and burn history) was collected by other personnel. All spatial data was collected with USFWS GPS equipment, and provided to me by USFWS, including data that I assisted in collecting. Data on Kinnikinnick regrowth in Johnson prairie was collected independently with permission and support from JBLM USFWS.

The study sites are divided into three main areas of interest: Johnson prairie, Upper and Lower Weir prairies, and Training Area 6/Central Impact Area. These areas contain suitable Hoary Elfin habitat (patches of Kinnikinnick within prairies), and are all subject to prescribed burning. All study sites were surveyed, at least to some degree, in the 2018 adult flight season using the methods described. We deviated from ideal survey techniques in some instances due to time and personnel constraints, which I discuss subsequently. Some sites were surveyed in previous seasons, using similar methodology. Johnson prairie was also monitored for regrowth of the Kinnikinnick patches, many of which were previously occupied by Hoary Elfin, that burned in an August 2017 wildfire.

Study Sites

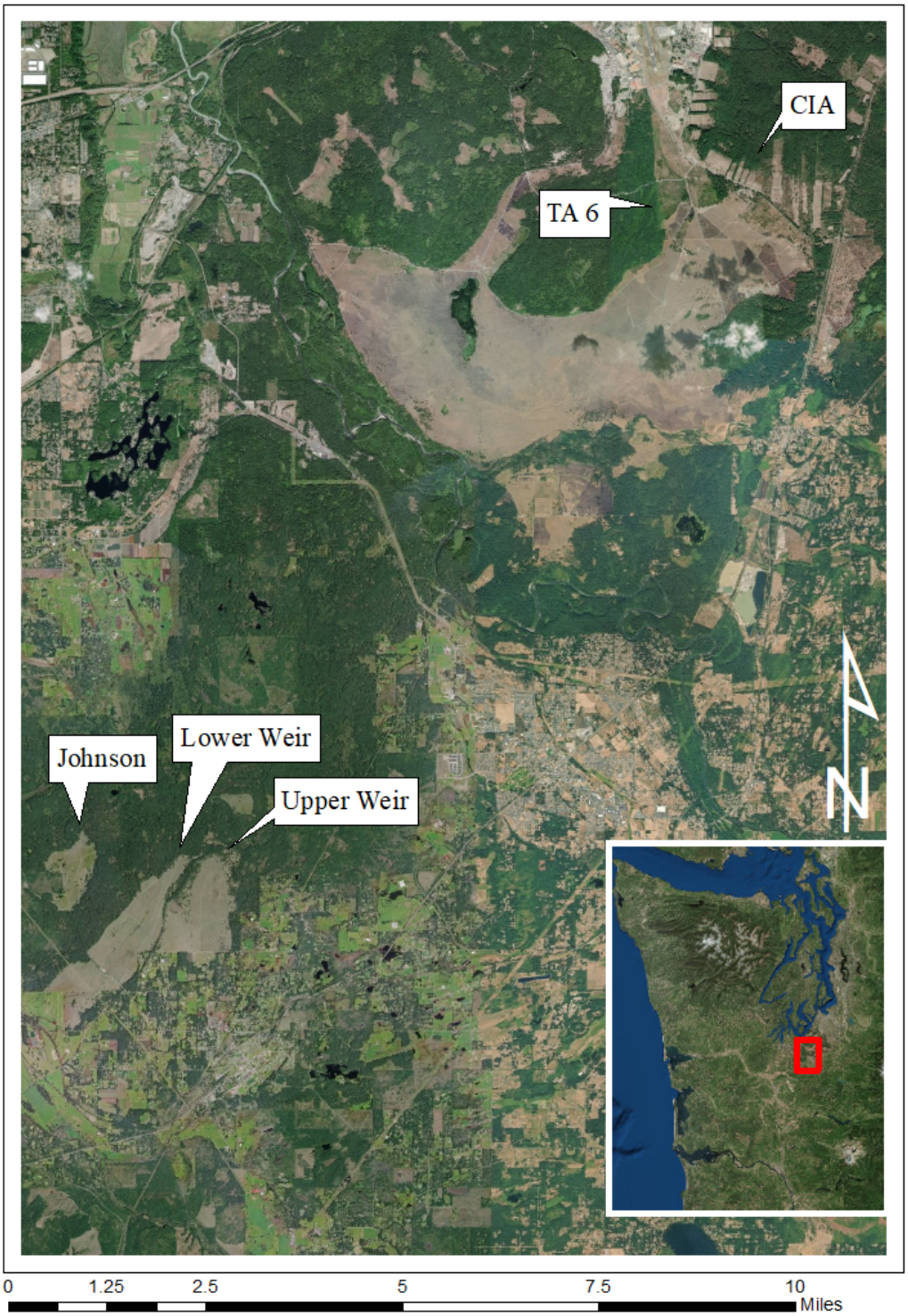


Figure 5. Data was collected at these five study sites.
(Service Layer Credits: Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community)

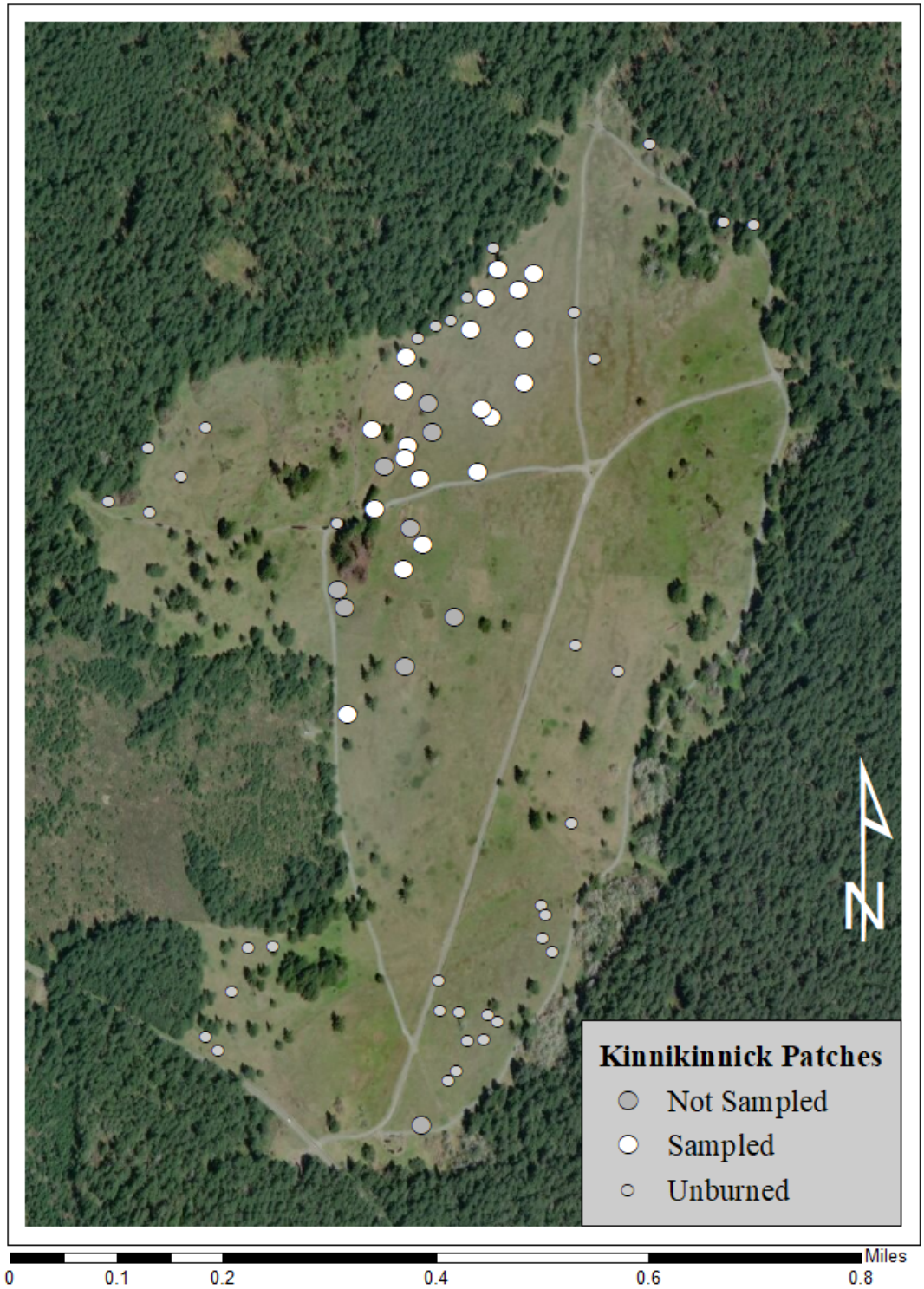
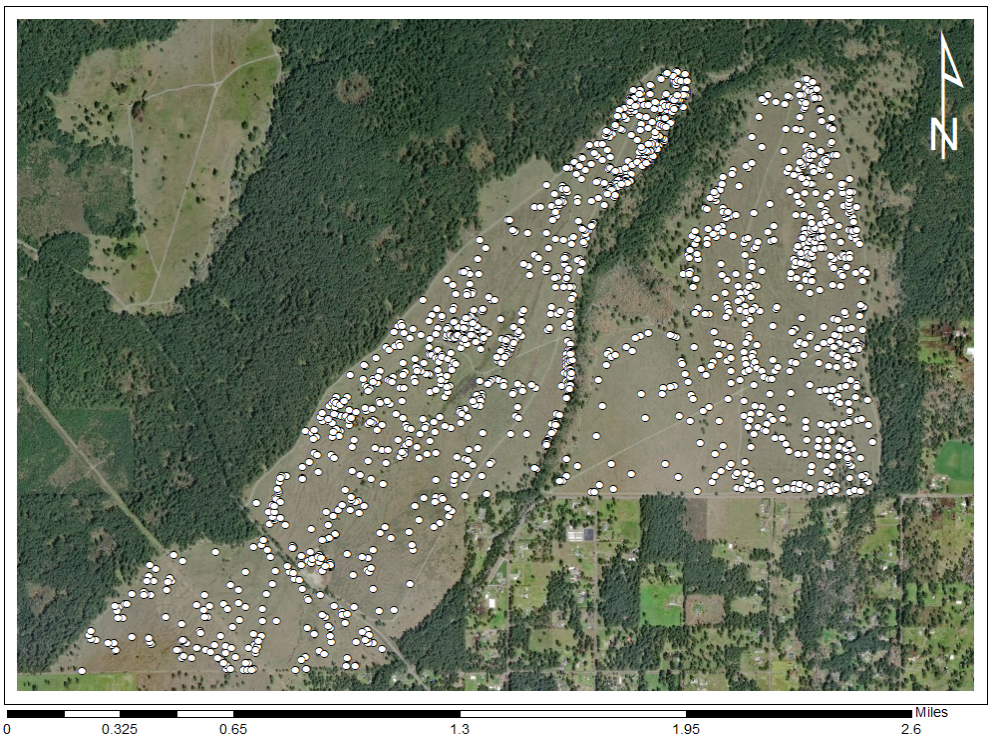


Figure 6. Johnson prairie with Kinnikinnick patches. "Unburned" refers to patches that did not burn in the August 2017 wildfire. "Not sampled" patches did burn, but were not selected for regrowth monitoring. The remainder, labeled "sampled," were included in the regrowth monitoring study.
(Service Layer Credits: Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community)



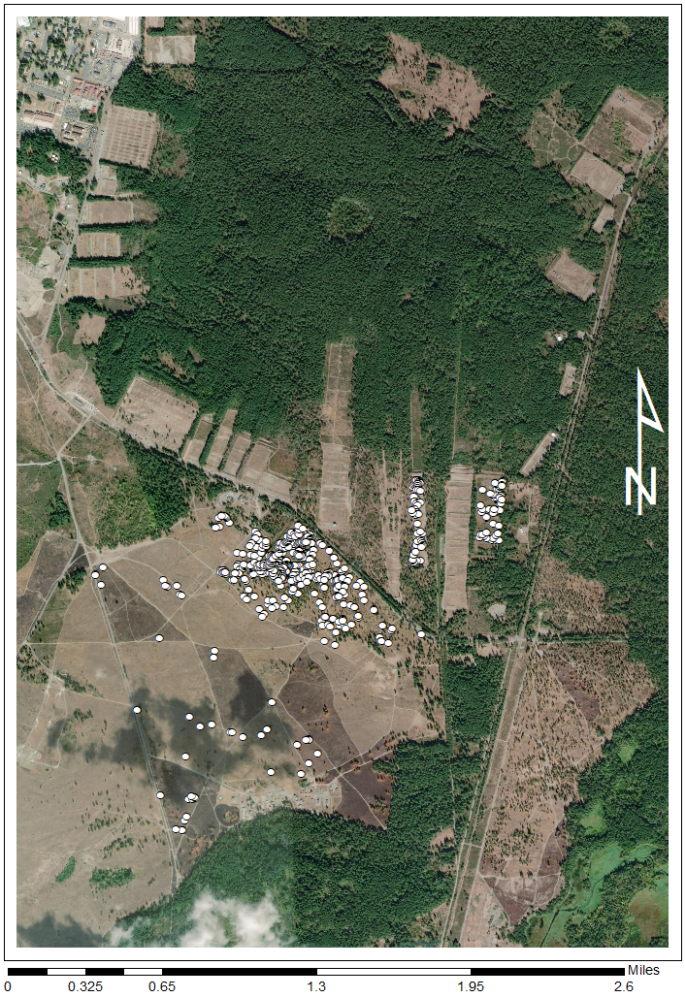


Figure 7. (Above) Upper and Lower Weir prairies with Kinnikinnick patches.

Figure 8. (Left) Training Area 6 and Central Impact Area with known Kinnikinnick patches (Kinnikinnick patches exist but have not been recorded in some areas of the CIA). The vast prairie adjacent to the west of TA6, as seen in Figure 5, is the Artillery Impact Area. Access is extremely restricted due to military activities. Hoary Elfin have been observed in this area, but I was not able to assist with surveys, and do not have comprehensive data on their distribution.

(Service Layer Credits: Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community)

Hoary Elfin Surveys

The methods used to survey for Hoary Elfin were established by the biologists at JBLM USFWS responsible for ongoing monitoring efforts for sensitive butterfly species. They are designed primarily to determine Hoary Elfin presence/absence in suitable areas of habitat. Surveys document the presence of adult Hoary Elfin, and are conducted during the adult stage flight season, from late April to late May. Sightings were documented using Trimble GPS devices, which record location via satellite. We aimed to conduct surveys during ideal weather conditions for butterfly activity. Ideal conditions are between 10:30am and 4:30pm, when the temperature is above 55 degrees F, winds are less than 10 mph, and there is enough sun to cast distinct shadows. Unpredictable weather and limited personnel availability impacted our ability to perform surveys in an ideal manner. Three different surveying techniques were used to obtain data on Hoary Elfin observations: surveys of individual Kinnikinnick patches (denoted as ARUV surveys), 100m North-South transect surveys, and wandering. Including wandering as a survey methodology mainly serves as a way to document butterfly observations that occur outside of formal surveys (i.e. a butterfly species of interest happens to fly past while a field observer is walking around, taking a lunch break, etc.), but was also used when time and personnel limitations interfered with our ability to conduct surveys using either of the other two methods. Transect surveys are generally used to cover areas that are so large and/or have such an abundance of Kinnikinnick that surveying individual Kinnikinnick patches is not feasible.

In Johnson prairie, Kinnikinnick patches were surveyed individually (Figure 9). Survey protocol is to observe each patch of Kinnikinnick for two minutes, standing in

one place. If the patch is too big to see all of it, the observer may move to another location for an additional two minutes (this rarely happened). If multiple adults are observed, the number of adults can be recorded, but the patch is considered occupied and does not need to be surveyed again. If no adults are observed, surveys continue periodically throughout the flight season. Johnson prairie has been surveyed more thoroughly than other areas because it is of special interest due to its history of hosting a population and experiencing a wildfire, as well as the feasibility of this survey method due to Johnson prairie's relatively small size and relatively small number of Kinnikinnick patches.

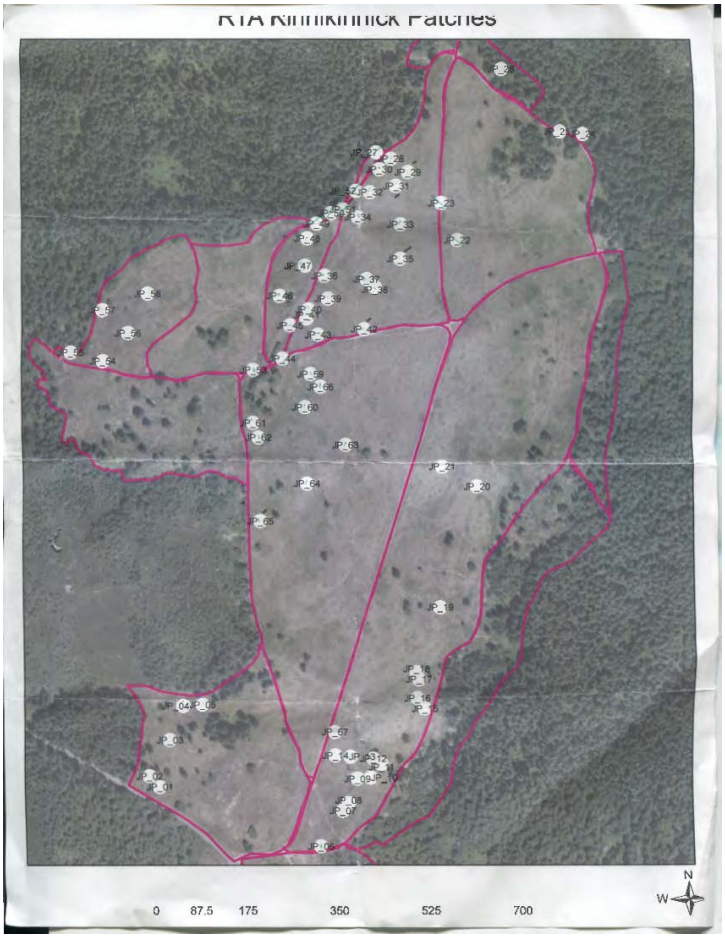


Figure 9. Scan of actual map of Johnson prairie used during surveys. A Trimble GPS or Avenza maps smarthphone app, which geolocate, were used to reference patch numbers by showing the user's location on the map in relation to the numbered patches.

Upper Weir prairie was surveyed using 100m North-South transects. Observers walked along the transects, looking for Hoary Elfin on patches of Kinnikinnick along the way. Observers could follow the transects using a Trimble GPS, which was also used to record Hoary Elfin sightings, along with a paper map. The transect lines appeared on the screen of the GPS without numbers, as well as the location of the observer georeferenced, and the paper map (Figure 10) could be cross-referenced for the transect number. Alternatively, observers using the Avenza smartphone app could download maps such as Figure 10 with their location georeferenced directly.

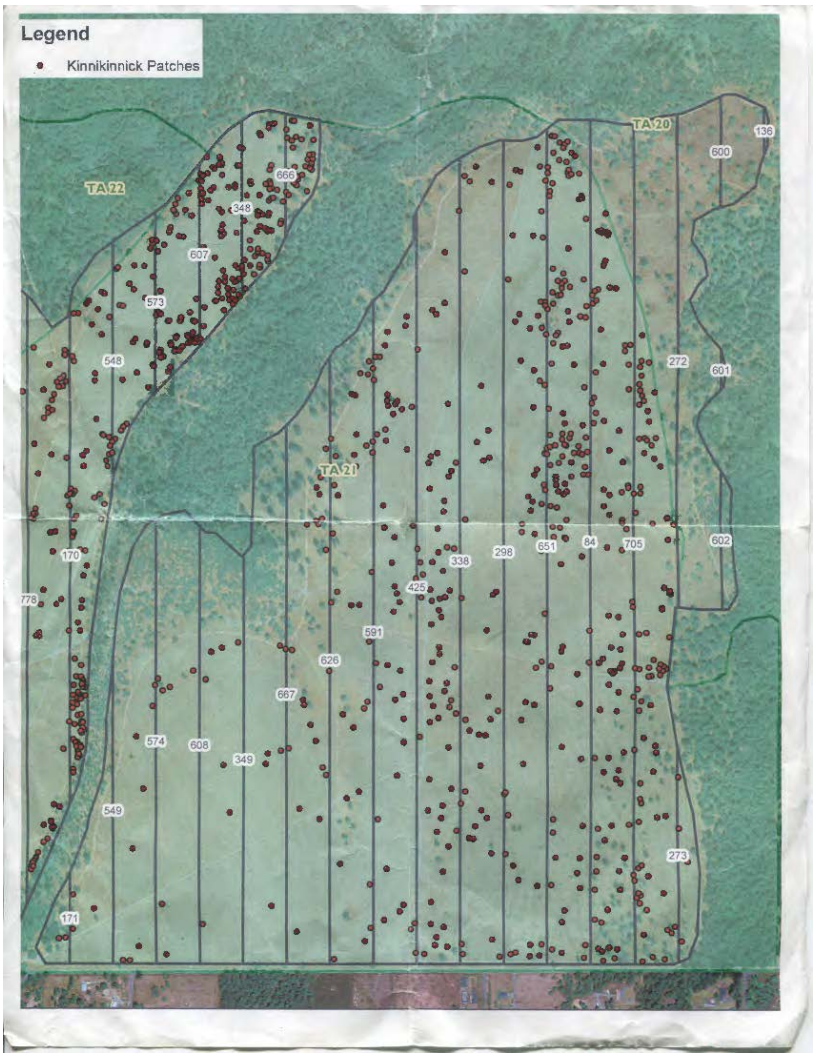


Figure 10. Scan of actual map of Upper Weir prairie used during surveys. Trimble GPS plus paper map or Avenza were used to locate transects and reference transect numbers.

Lower Weir prairie was surveyed by wandering in 2018. The goal was to cover as much ground containing Kinnikinnick as possible using available time and personnel. We loosely followed transects that intersected a relatively large amount of Kinnikinnick, but did not stick to 100m spacing.

The CIA was surveyed by transect, but rather than 100m spacing, observers spread out evenly across each range (which are small- 3-4 observers were generally less than 100m apart). Observers maintained even spacing as they walked the length of each range. Portions of TA6 with abundant Kinnikinnick were surveyed in a loose, wandering grid.

Kinnikinnick Regrowth

The second component of this study on Kinnikinnick regrowth after fire utilizes data collected at Johnson prairie. This area hosted a population of Hoary Elfin prior to a wildfire in August 2017, which burned a significant portion of the prairie, including all Kinnikinnick patches on which Hoary Elfin were observed during the previous spring. There are 67 patches of Kinnikinnick mapped in Johnson Prairie, and 30 burned in the wildfire, 11 of which were observed to be occupied by Hoary Elfin in spring 2017. The map used to survey Kinnikinnick patches in Johnson Prairie for Hoary Elfin (Figure 9) assigns numbers 1-67 to the patches, and this provides the patch ID number utilized as reference. I assigned each of the 30 burned patches an additional reference number 1-30 (since the original ID numbers of the patches burned are spread across 1-67). I used random.org to generate a random sequence of the numbers 1-30. I sampled the patches referenced by the first 20 numbers of this sequence, with the exception that I excluded

four patches because they fell within plots being utilized in other research experiments and I wanted to avoid potentially disrupting these experiments.

I began the regrowth monitoring project in May 2018, 9 months post-fire. In each selected patch, I chose a $1m^2$ plot to monitor regrowth. I delineated the plots using paracord, lawn staples, and pin flags. When I set up the plots in May 2018, I estimated percent cover of Kinnikinnick and counted how many sprigs or tufts were emerging from the ground within the meter plots. My criteria for a "sprig or tuft" was an area of plant regrowth separated from others by visible bare dirt. This ranged from a single stem to many stems covering a contiguous area of several inches. Counting every single stem was therefore impractical, but my goal was to measure how many sources for regrowth existed within each meter plot. I left the meter plots marked, and returned in November 2018, 15 months post-fire, to obtain a second set of measurements of percent cover and number of sprigs/tufts. I collected a third set of observations in May 2019, 21 months post-fire.

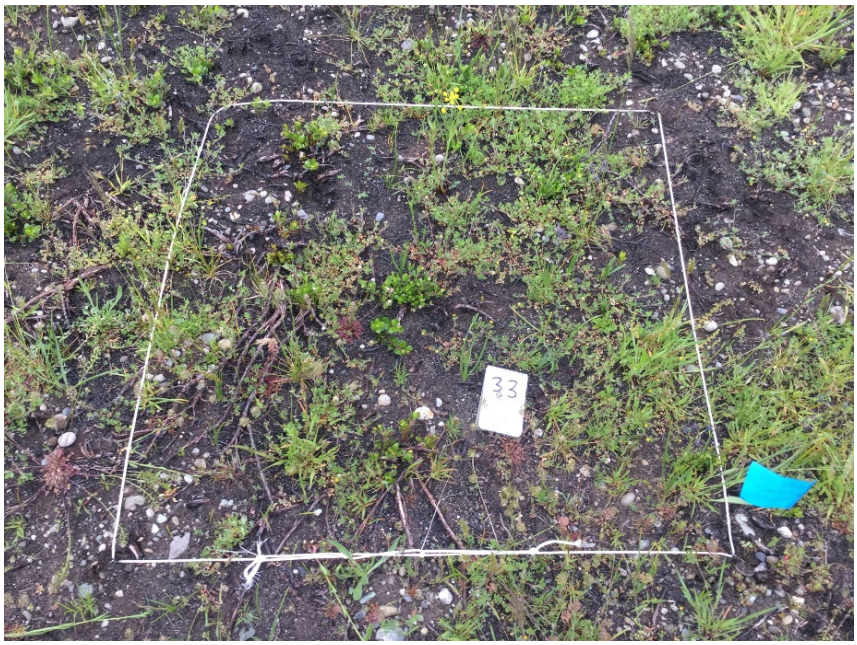


Figure 11. A meter plot for Kinnikinnick regrowth monitoring in patch #33, May 2018.

RESULTS

For each of the three areas of interest, available data on burn history is mapped to show which portions of each study site burned each year. This provides a visual on how much of each prairie burns each season and roughly how frequently each section burns. Subsequently, data on Hoary Elfin sightings are mapped for the same areas. This shows how the spatial distribution of Hoary Elfin within each prairie compares to the burn history of each area and can be used to identify temporal patterns in the relationship between burning and Hoary Elfin occupation. Regrowth monitoring of Kinnikinnick in Johnson prairie shows the rate and pattern of regrowth in response to elimination of aboveground biomass by burning.

Johnson Prairie

The August 2017 wildfire, which burned all patches of Kinnikinnick observed to be occupied by Hoary Elfin the previous spring, appears to have caused the drastic decline in the Hoary Elfin population seen in spring 2018. However, the population appears to have been declining prior to the wildfire as well. From 2015 to 2016, the number of individuals observed dropped from 38 to 23, a reduction of 39.5 percent, but those individuals were distributed over a similar geographic range across Johnson prairie. In 2017, 22 individuals were observed, comparable to 22 individuals in 2016, but these sightings were restricted to the northwest corner of the prairie. 2018 (post-wildfire) yielded the observation of only one individual. This sighting occurred in an area where no Hoary Elfin had been sighted in the immediate area since 2015, and in the general area since 2016.

The area occupied by the lone Hoary Elfin in 2018 last burned in 2016. The northwest corner that served as the Hoary Elfin stronghold in 2017 and was occupied in 2015 and 2016 burned most recently prior to wildfire in 2015, and prior to that in 2011. These two areas have the highest concentration of Kinnikinnick relative to the rest of the prairie. Areas of the prairie where no Hoary Elfin were sighted between 2015 and 2018 tend to have very sparse Kinnikinnick, and the patches are close to the forest edge.

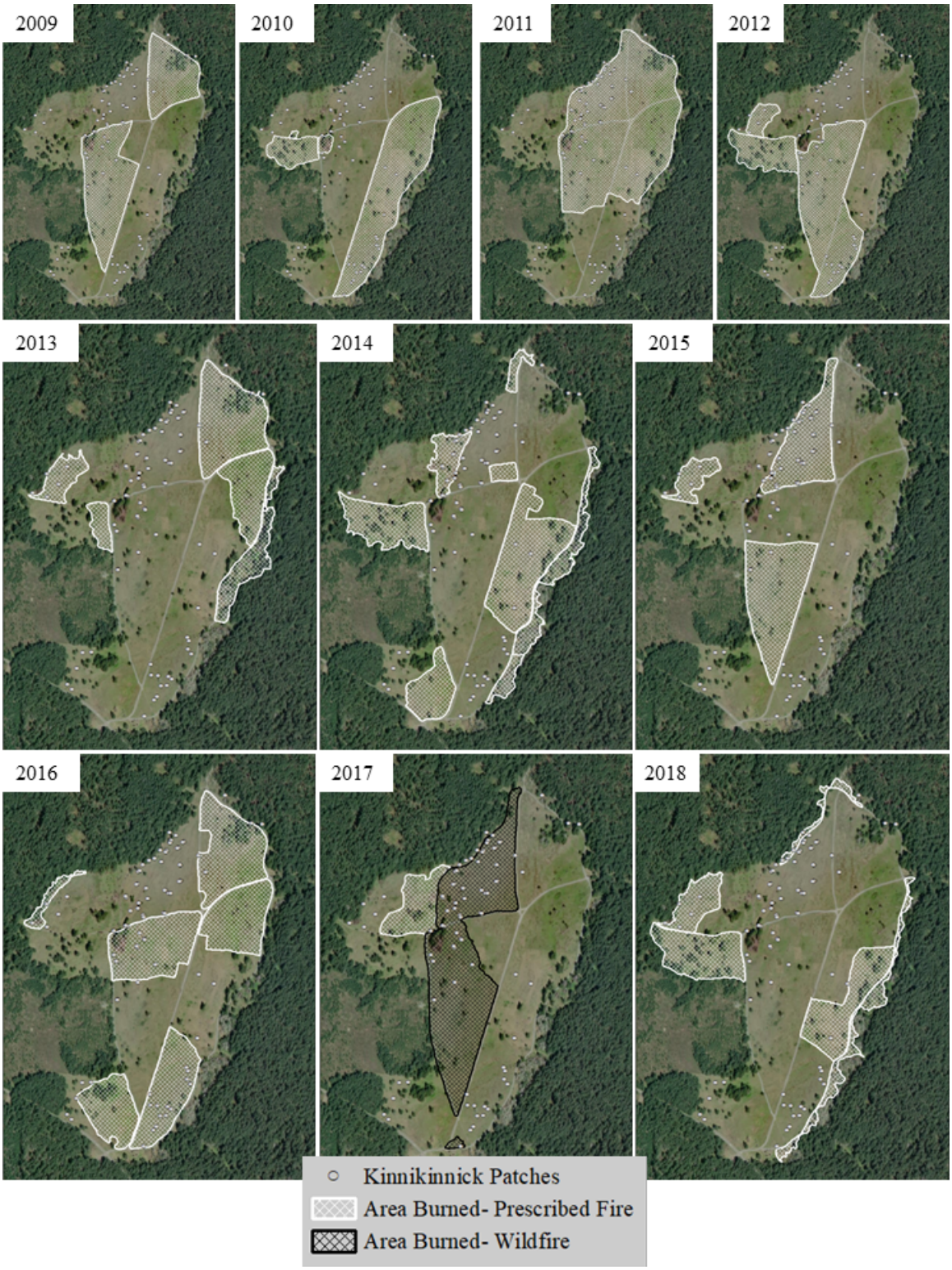


Figure 12. Johnson prairie burn history since 2009.
(Service Layer Credits: Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community)

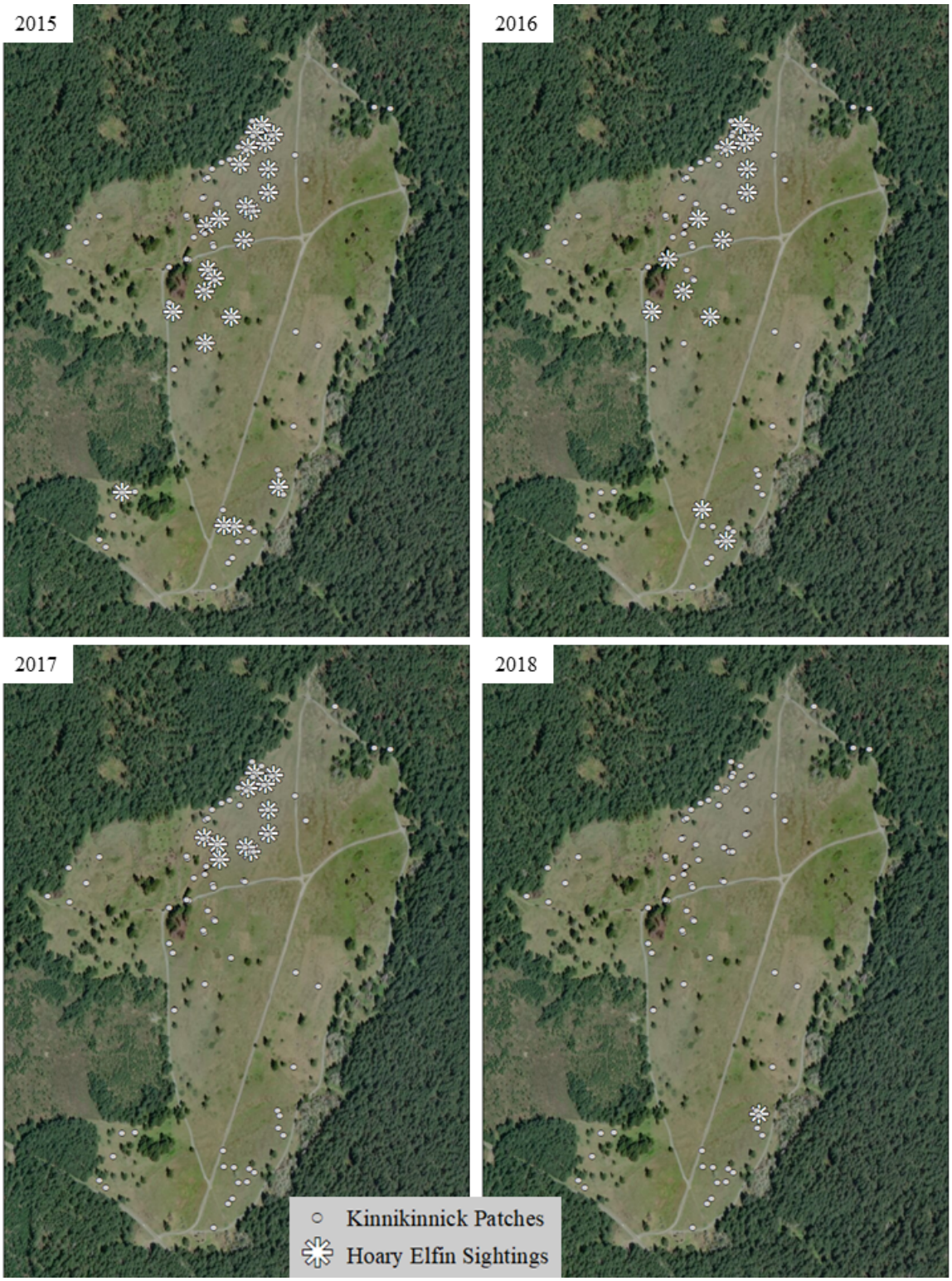


Figure 13. Johnson prairie Hoary Elfin sightings. Each symbol represents a location where Hoary Elfin were sighted, not individual butterflies. Multiple Hoary Elfin may have been observed at each location (see Table 1).

(Service Layer Credits: Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community)

Year	Total Observed
2015	38
2016	23
2017	22
2018	1

Date	Method	# Observed
2015-04-20	ARUV Survey	1
2015-04-20	ARUV Survey	1
2015-04-20	ARUV Survey	4
2015-04-20	ARUV Survey	1
2015-04-20	ARUV Survey	2
2015-04-20	ARUV Survey	1
2015-04-20	ARUV Survey	2
2015-04-20	ARUV Survey	2
2015-04-20	ARUV Survey	1
2015-04-20	ARUV Survey	1
2015-04-27	ARUV Survey	1
2015-04-27	ARUV Survey	2
2015-04-27	ARUV Survey	1
2015-04-27	ARUV Survey	1
2015-04-27	ARUV Survey	2
2015-04-27	ARUV Survey	3
2015-04-27	ARUV Survey	3
2015-04-27	ARUV Survey	4
2015-05-18	ARUV Survey	1
2015-05-18	ARUV Survey	1
2015-05-18	ARUV Survey	1
2015-05-18	ARUV Survey	1
2015-05-18	ARUV Survey	1

Date	Method	# Observed
2016-04-20	ARUV Survey	1
2016-04-20	ARUV Survey	2
2016-04-20	ARUV Survey	1
2016-04-20	ARUV Survey	1
2016-04-20	ARUV Survey	1
2016-04-20	ARUV Survey	1
2016-04-20	ARUV Survey	2
2016-04-20	ARUV Survey	4
2016-04-20	ARUV Survey	3
2016-04-20	ARUV Survey	1
2016-04-20	ARUV Survey	1
2016-05-03	ARUV Survey	2
2016-05-03	ARUV Survey	1
2016-05-10	ARUV Survey	2

Date	Method	# Observed
2017-05-03	ARUV Survey	3
2017-05-03	ARUV Survey	3
2017-05-03	ARUV Survey	1
2017-05-03	ARUV Survey	4
2017-05-03	ARUV Survey	2
2017-05-03	ARUV Survey	2
2017-05-03	ARUV Survey	1
2017-05-03	ARUV Survey	1
2017-05-18	ARUV Survey	2
2017-05-18	ARUV Survey	1
2017-05-18	ARUV Survey	2

Date	Method	# Observed
2018-05-22	Wandering	1

Table 1. Raw data on recorded Hoary Elfin sightings in Johnson prairie, exported from ArcMap.

Upper and Lower Weir

Far less data is available for Lower and Upper Weir prairies than for Johnson prairie. Only two Hoary Elfin sightings were documented in 2017, but I am unsure how thoroughly the rest of the prairie was surveyed. 39 individuals were observed in 2018. The heaviest concentrations appear to be on the east edge of Upper Weir and the north finger of Lower Weir, but sightings are distributed throughout the rest of both sites.

The southeast corner of Upper Weir, where sightings were concentrated in 2018, last burned in 2016, and in 2013 and 2009 prior. The northeast corner of Upper Weir, where less 2018 sightings took place, burned most recently in 2017. The north finger of Lower Weir, where sightings were also concentrated in 2018, also burned most recently in 2017.

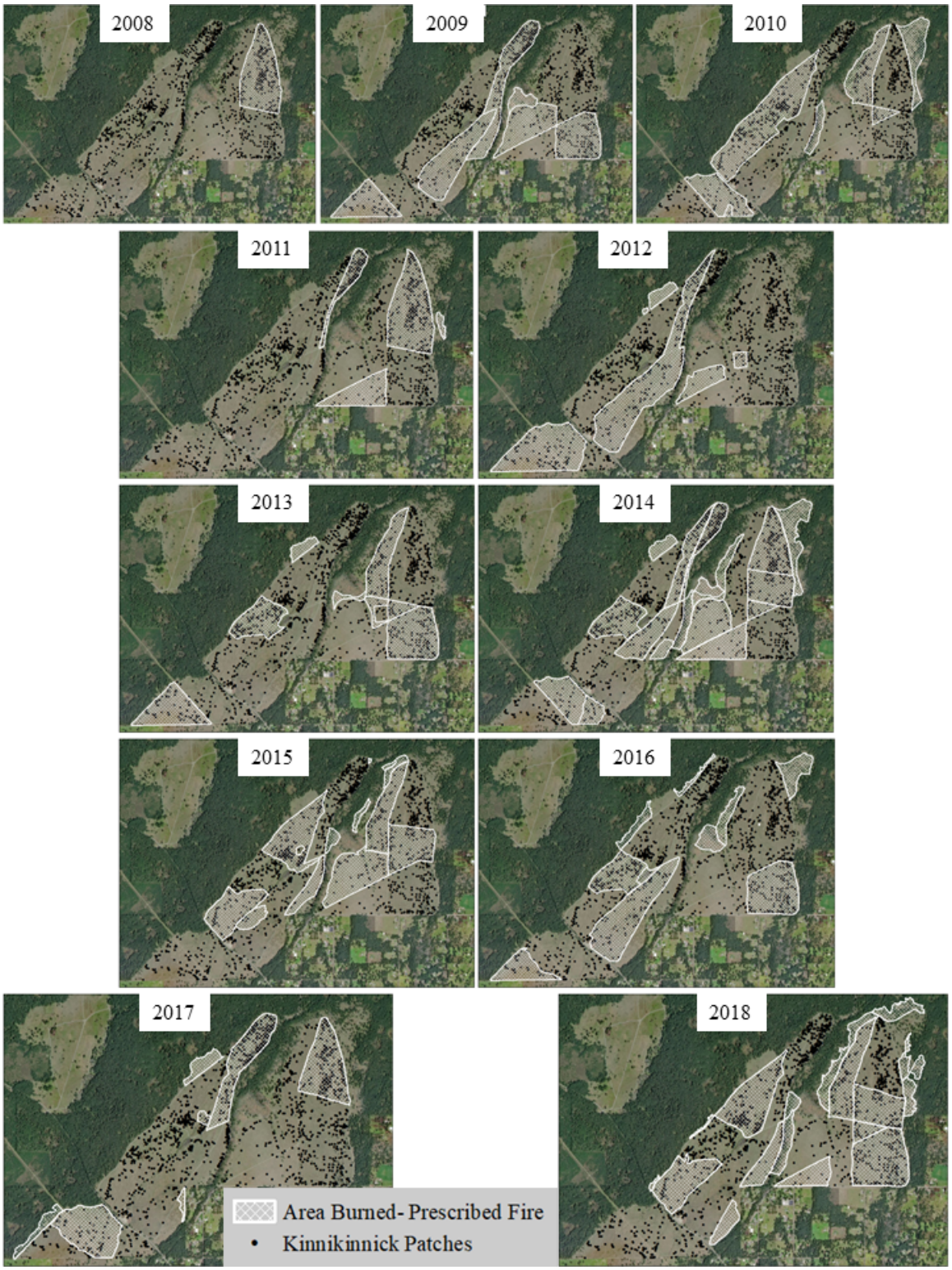


Figure 14. Upper and Lower Weir prairie burn history since 2008.
(Service Layer Credits: Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community)

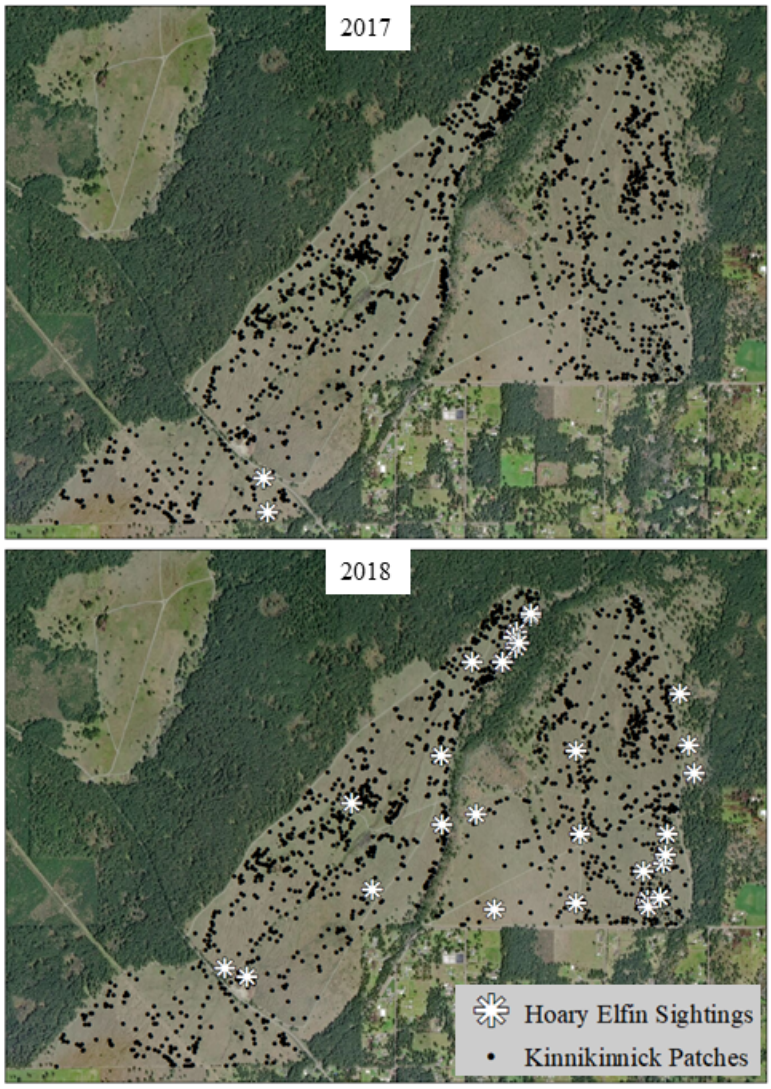


Figure 15. Upper and Lower Weir prairie Hoary Elfin sightings. Each symbol represents a location where Hoary Elfin were sighted, not individual butterflies. Multiple Hoary Elfin may have been observed at each location (see Table 2).
(Service Layer Credits: Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community)

Year	Total Observed
2017	2
2018	39

Date	Method	# Observed
2017-05-03	ARUV Survey	1
2017-05-18	ARUV Survey	1

Date	Method	# Observed
2018-05-03	100m NS Transect	2
2018-05-03	100m NS Transect	1
2018-05-03	100m NS Transect	2
2018-05-03	100m NS Transect	*1
2018-05-03	100m NS Transect	*1
2018-05-03	100m NS Transect	2
2018-05-03	100m NS Transect	*1
2018-05-03	100m NS Transect	*1
2018-05-11	100m NS Transect	*1
2018-05-11	100m NS Transect	*1
2018-05-17	100m NS Transect	*1
2018-05-17	100m NS Transect	*1
2018-05-17	100m NS Transect	*1
2018-05-17	100m NS Transect	*1
2018-05-17	100m NS Transect	*1
2018-05-17	100m NS Transect	*1
2018-05-17	100m NS Transect	*1
2018-05-17	100m NS Transect	*1
2018-05-21	100m NS Transect	*1
2018-05-21	100m NS Transect	*1

Date	Method	# Observed
2018-05-22	Wandering	*1
2018-05-22	Wandering	*1
2018-05-22	Wandering	*1
2018-05-22	Wandering	*1
2018-05-22	Wandering	*1
2018-05-22	Wandering	*1
2018-05-22	Wandering	*1
2018-05-22	Wandering	*1
2018-05-22	Wandering	*1
2018-05-22	Wandering	*1
2018-05-22	Wandering	*1
2018-05-22	Wandering	*1
2018-05-22	Wandering	*1
2018-05-22	Wandering	*1

Table 2. Raw data on recorded Hoary Elfin sightings in Upper and Lower Weir prairie, exported from ArcMap. * indicates sightings where no comments were recorded by the observer on how many individual butterflies were observed at that location, presumed to be one individual.

Training Area 6 / Central Impact Area

No Hoary Elfin sightings were documented prior to spring 2018 in Training Area 6 and the Central Impact Area. Three were observed in TA6 and a robust population, totaling 66 individuals, was observed in the CIA. TA6 is subject to a prescribed burn mosaic similar to other areas. The artillery ranges in the CIA do not receive prescription burning, likely due to extremely restrictive access due to military activities, but wildfires occur on occasion (Figure 16).

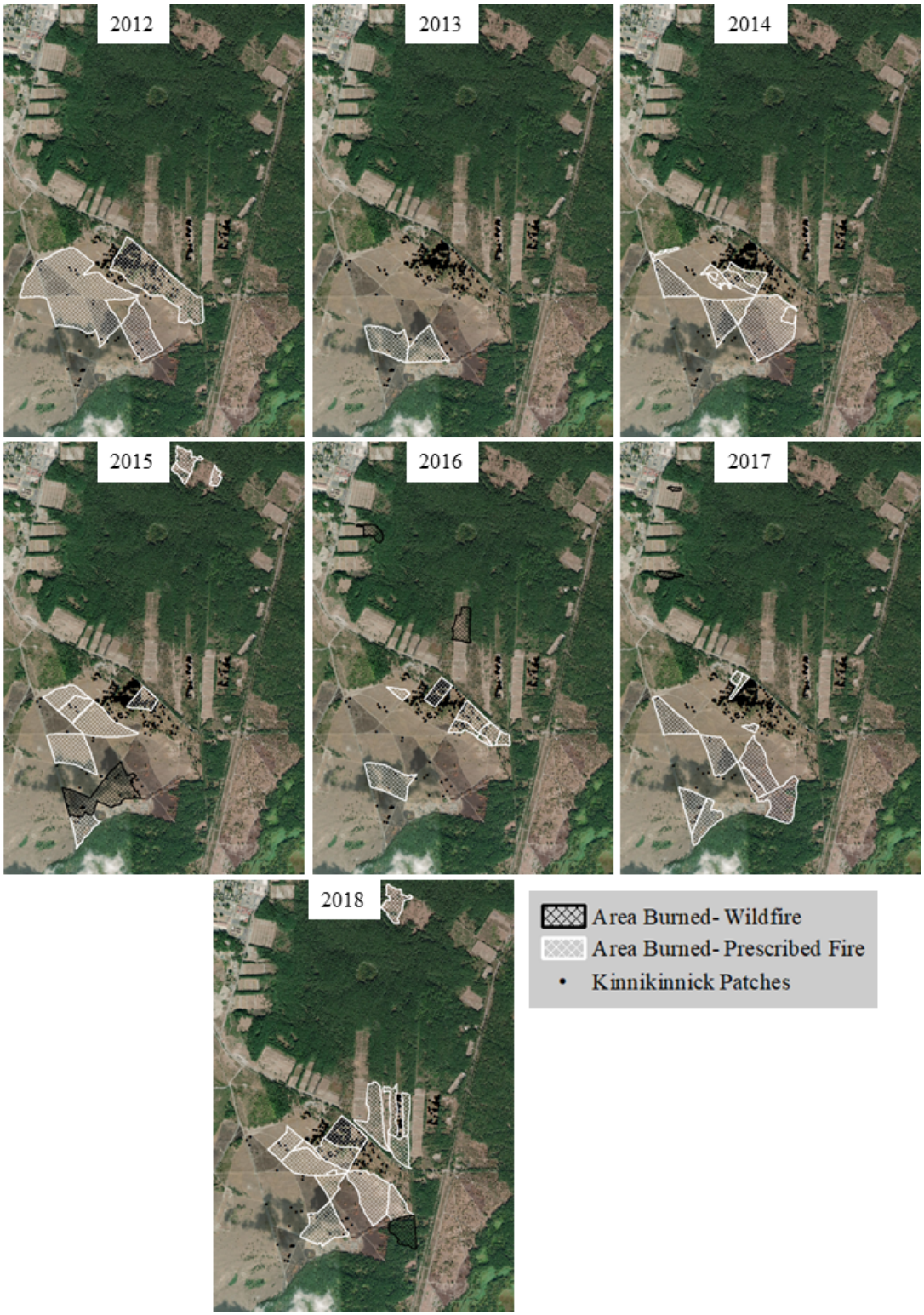


Figure 16. Training Area 6 and Central Impact Area burn history since 2012. (Service Layer Credits: Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community)

Figure 17. Training Area 6 and Central Impact Area Hoary Elfin sightings. 2018 is the only year in which surveys were conducted. Each symbol represents a location where Hoary Elfin were sighted, not individual butterflies. Multiple Hoary Elfin may have been observed at each location (see Table 3). Locations where Hoary Elfin sightings are documented but no Kinnikinnick is symbolized are areas where Kinnikinnick is present but has not been mapped and recorded via GPS.
(Service Layer Credits: Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community)

Year	Location	Total Observed
2018	TA 6	3
2018	CIA	66

Training Area 6		
Date	Method	# Observed
2018-05-01	Wandering	*1
2018-05-01	Wandering	*1
2018-05-01	Wandering	*1

Central Impact Area		
Date	Method	# Observed
2018-05-02	Wandering	2
2018-05-02	Wandering	1
2018-05-02	Wandering	2
2018-05-02	Wandering	2
2018-05-02	Wandering	3
2018-05-02	Wandering	1
2018-05-02	Wandering	1
2018-05-02	Wandering	1
2018-05-02	Wandering	1
2018-05-02	Wandering	1
2018-05-02	Wandering	1
2018-05-02	Wandering	*1
2018-05-02	Wandering	*1
2018-05-02	Wandering	*1
2018-05-02	Wandering	*1

Central Impact Area		
Date	Method	# Observed
2018-05-02	Wandering	*1
2018-05-02	Wandering	1
2018-05-02	Wandering	*1
2018-05-02	Wandering	*1
2018-05-02	Wandering	*1
2018-05-02	Wandering	*1
2018-05-02	Wandering	*1
2018-05-02	Wandering	4
2018-05-02	Wandering	*1
2018-05-02	Wandering	2
2018-05-02	Wandering	2
2018-05-02	Wandering	*1
2018-05-02	Wandering	*1
2018-05-02	Wandering	*1
2018-05-02	Wandering	2
2018-05-02	Wandering	2
2018-05-02	Wandering	*1
2018-05-02	Wandering	4
2018-05-02	Wandering	2
2018-05-02	Wandering	*1
2018-05-02	Wandering	*1
2018-05-02	Wandering	*1
2018-05-02	Wandering	*1
2018-05-02	Wandering	*1
2018-05-02	Wandering	*1
2018-05-02	Wandering	2
2018-05-02	Wandering	2
2018-05-02	Wandering	6

Table 3. Raw data on recorded Hoary Elfin sightings in Training Area 6 and the Central Impact Area, exported from ArcMap. * indicates sightings where no comments were recorded by the observer on how many individual butterflies were observed at that location, presumed to be one individual.

Kinnikinnick Regrowth

Immediately post-fire in August 2017, percent cover of Kinnikinnick was presumably at 0% in each of the areas that would later be contained by meter plots. In May 2018, the 20 meter plots were set up, each containing between 10% and 40% cover, with an average of 17% cover. In November 2018, each plot contained between 10% and 80% cover, with an average of 34.75% cover. This represents a gain of between 0% and 40% cover across all plots, with an average gain in percent cover of 17.75%. Percent increase in percent cover ranged from 0% to 400%, with a mean of 110% increase in percent cover. In May 2019, each plot contained between 10% and 90% cover, with an average of 38% cover. This represents a gain of between 0% cover and 10% cover, with an average gain in percent cover of 4.8%. Percent increase in percent cover ranged between 0% and 50%, with a mean percent increase in percent cover of 10.5%.

Data on number of sprigs is less conclusive and subject to interpretation. The goal was to study the growth pattern by which Kinnikinnick reclaimed burned areas. Overall, the number of sprigs decreased in all patches from May 2018 to November 2018 (meaning that the number of distinct areas of regrowth decreased, likely due to an increase in biomass causing the sprigs to grow together). The average number of sprigs decreased from 15.4 in May 2018 to 6.7 in November 2018, representing an average decrease of 5.8 sprigs and an average percent decrease of 44.7%. Data for November 2018 included 19 of the original 20 plots due to the boundary of one plot disappearing. In May 2019, I was only able to count sprigs in 13 of the original 20 plots due to the boundaries of plots disappearing. From November 2018 to May 2019, 3 of the 13 plots showed an increase in number of sprigs, while the other 10 decreased or remained the

same. The average number of sprigs decreased from 6.7 to 5.8, with a mean decrease of .5 and a mean percent decrease of 2.7%.

In the instances where the plot boundaries were disturbed or missing, I was able to locate the pin flags marking the corner of each plot. In cases where the twine delineating the boundary was disturbed but still visible, I was able to distinguish where the original was, and measured percent cover and number sprigs as usual. These instances are denoted with * in Table 4 and Table 5. In cases where the twine delineating the plot boundary was missing, estimates of percent cover could be deduced based on the percent cover of the area adjacent to the flag; either percent cover was relatively uniform on all sides, or certain areas had less percent cover than was measured 6 months previously and therefore could not have been contained by the original plot. However, number of sprigs could not be logically deduced in these instances. They are denoted with ** in Table 4 and Table 5.

Percent Cover			
	May 2018	Nov. 2018	May 2019
Mean % Cover	17	34.75	38
Mean Gain		17.75	4.8
Mean % Increase		110%	10.50%

Patch ID	% Cover May 2018	% Cover Nov. 2018	% Cover Gain	% Increase	% Cover May 2019	% Cover Gain	% Increase
29	40	80	40	100	90	10	12.5
30	10	*20	*10	*100	*30	*10	*50
31	20	60	40	200	60	0	0
32	30	70	40	133.3	70	0	0
33	10	50	40	400	**60	**10	**20
34	20	50	30	150	**50	**0	**0
35	10	15	5	50	20	5	33.3
37	10	20	10	100	**30	**10	**50
38	10	*20	*10	*100	*20	*0	*0
40	10	20	10	100	**20	**0	**0
41	10	20	10	100	20	0	0
42	20	50	30	150	60	10	20
43	30	50	20	66.6	**50	**0	**0
44	10	30	20	200	**30	**0	**0
46	20	20	0	0	20	0	0
47	20	40	20	100	50	10	25
48	20	*30	*10	*50	*30	**0	**0
60	10	20	10	100	*20	*0	*0
65	20	**20	**0	**0	**20	**0	**0
66	10	10	0	0	10	0	0

Table 4. Kinnikinnick regrowth in Johnson prairie, for patches burned by August 2017 wildfire. % Cover refers to the percent cover within each meter plot. % Cover Gained refers to the change in percent cover between the two previous observations of percent cover. % Cover Increase refers to the percent increase in percent cover between the two previous observations. * indicates plots where the twine delineating plot boundary was disturbed, but the original boundary was easily distinguished. ** indicates plots where the twine delineating the original plot boundary was missing, and the original edge was unknown.

Sprigs			
	May 2018	Nov. 2018	May 2019
Mean # Sprigs	15.4	6.7	5.8
Mean decrease		5.8	0.5
Mean % decrease		44.70%	2.70%

Patch ID	# Sprigs May 2018	# Sprigs Nov. 2018	Difference	% Change	# Sprigs May 2019	Difference	% Change
29	21	9	12	-57.1	1	8	-88.8
30	9	6	3	-33.3	5	1	-16.6
31	5	4	1	-20	4	0	0
32	16	5	11	-68.7	5	0	0
33	19	4	15	-78.9	**	**	**
34	13	4	9	-69.2	**	**	**
35	11	7	4	-36.3	6	1	-14.2
37	9	4	5	-55.5	**	**	**
38	5	2	3	-60	2	0	0
40	14	12	2	-14.2	**	**	**
41	3	2	1	-33.3	2	0	0
42	14	7	7	-50	4	3	-42.8
43	18	16	2	-11.1	**	**	**
44	10	6	4	-40	**	**	**
46	18	14	4	-22.2	13	1	-7.1
47	16	11	5	-31.2	13	-2	18.1
48	23	7	16	-69.5	11	-4	57.1
60	8	5	3	-37.5	4	1	-20
65	71	**	**	**	**	**	**
66	5	2	3	-60	5	-3	150

Table 5. Kinnikinnick regrowth in Johnson prairie, for patches burned by August 2017 wildfire. # Sprigs refers to distinct areas of re-sprouting within the meter plot delineated in each patch. Difference refers to the difference in number of sprigs between the previous two observations. % Change refers to the percent change in number of sprigs between the previous two observations. * indicates plots where the twine delineating plot boundary was disturbed, but the original boundary was easily distinguished. ** indicates plots where the twine delineating the original plot boundary was missing, and the original edge was unknown.

DISCUSSION

Johnson Prairie

The biologists at JBLM Fish & Wildlife and I were especially interested in Johnson prairie when we surveyed in 2018 to determine the impacts of the wildfire on the Hoary Elfin population. However, the analysis of burn data and Hoary Elfin observation data suggests that the population was declining prior to the wildfire. Between 2015 and 2016, the number of sightings decreased but were distributed across a similar area. The prescribed burns that took place between these observations, in the summer of 2015, included the area where the sightings remained concentrated through 2017. It appears, therefore, that these prescribed burning practices do not necessarily extirpate Hoary Elfin from the immediate area burned. The practice of burn exclusions, where patches of Kinnikinnick are exempt from burning during prescribed fire operations, have been utilized in some areas. However, in this instance of the burns conducted in summer 2015, there was a population of Hoary Elfin in an adjacent area of the prairie that was not burned. Regardless of whether the patches of Kinnikinnick that hosted Hoary Elfin in both 2015 and 2016 were excluded from the 2015 prescribed burns, it is unknown whether the Hoary Elfin observed in those patches in 2016 survived the 2015 burns or whether they dispersed to those patches from adjacent areas after pupating in 2016.

The data for Hoary Elfin sightings between 2016 and 2017, with the 2016 prescribed burn season in between, paint a different picture. The Northwest corner of Johnson prairie, where sightings remained concentrated from 2015 through 2017 despite being burned in 2015, was not burned in 2016, but the other two areas of the prairie where butterflies were observed in 2016 were burned. In 2017, no butterflies were

sighted in these areas. It therefore seems possible that the prescribed burns conducted in 2016 may have had a negative impact on the population. If so, this may suggest that Hoary Elfin surviving in the area that did not burn did not disperse to those recently burned areas. However, if this were the case, it may have been a result of the habitat characteristics of the burned area, and not necessarily a result of Hoary Elfin dispersal tendencies or abilities. Ultimately the answer may lie in whether there were significant differences in the circumstances of the prescribed burns between 2015 and 2016.

The area burned in the August 2017 wildfire included the Northwest corner of Johnson prairie, which had previously served as the Hoary Elfin stronghold for the previous three spring flight seasons. The wildfire virtually obliterated all patches of Kinnikinnick that were observed to host Hoary Elfin in 2017. The prairie was surveyed several times during the 2018 Hoary Elfin adult flight season. Only one individual was seen, towards the end of flight season (May 22), on a patch that did not burn in the wildfire. Where did this individual come from? Did it hatch and pupate on the Kinnikinnick patch on which it was sighted, or did it fly there from another area as an adult? Was it looking for a mate, or a place to lay its eggs? Hoary Elfin stay very attached to their host plant of Kinnikinnick, but it appears that very little to nothing is known about their dispersal capabilities. While the Kinnikinnick burned in Johnson prairie seems to be rebounding from the wildfire, the question remains of whether extant Hoary Elfin populations will be able to recolonize on their own. Johnson prairie was surveyed on May 22, 2019, and zero Hoary Elfin were observed.

Upper and Lower Weir Prairies

Surveys in Upper Weir prairie in the 2018 Hoary Elfin adult flight season revealed that a population appears to be thriving in that area. The majority of sightings occurred on the eastern side of the prairie. Unfortunately for this study, the most likely reason for this pattern is that we were unable to survey all transects in the same day and during similar weather due to time and personnel constraints. I worked with members of USFWS, CNLM, and interns with JBLM USFWS's internship program. To some degree, the time and personnel constraints we experienced due to Hoary Elfin flight season coinciding with the flight season of other butterfly species of greater conservation concern, such as the federally endangered Taylor's Checkerspot. Resources are often prioritized for these other species. Additionally, spring weather in the PNW can be finnicky and unpredictable; since we try as much as possible to conduct surveys during optimal weather for detecting butterflies, having to wait out periods of cold, wet weather during flight season further constrained the time available to conduct surveys. When we surveyed the eastern side of the prairie, where most of the butterflies were sighted, it was clear and sunny. We ended our survey on this day because the time of day was becoming unfavorable for butterflies and the end of the work day for the crew of interns was approaching. It was about two weeks before we were able to return and finish surveying the remaining transects. The day we finished surveying Upper Weir was significantly colder and cloudier than anticipated, and a precipitation event had occurred since the day we began the survey. Colder weather reduces the likelihood of detection, even if butterflies are present, as they are less active in colder temperatures. Additionally, it is unknown how Hoary Elfin abundance might fluctuate over the duration of their flight

season, and since our two survey days were two weeks apart, abundance of adults could have changed over this time.

Another variable that needs to be taken into account is the practice of burn exclusions. In some areas, Kinnikinnick has been excluded from burning during prescribed fire operations in order to avoid disturbing Hoary Elfin populations. This may make it difficult to discern a relationship between burn history and Hoary Elfin distribution even if surveys were to be conducted in a way that accounts for weather and time, because it's possible that Kinnikinnick within a burn unit may have a different burn history than the unit itself. Record of burn exclusions of Kinnikinnick does exist, but it still adds a layer of complexity to attempts to observe a relationship between burn history and Hoary Elfin distribution. However, surveying for Hoary Elfin distribution in areas such as Upper and Lower Weir prairie, accounting for weather and time, would be beneficial because establishing where, if anywhere, the Hoary Elfin population is concentrated in these areas would allow land managers to make better informed decisions about how to best conserve Hoary Elfin habitat. As previously discussed, it is possible for burning to have long-term benefits despite short-term costs in terms of localized mortality. However, even if this were proven true on principle, South Sound prairies still face the issues discussed previously (e.g. habitat loss) that stress native populations, so what might be an ecologically beneficial disturbance in the context of a thriving ecosystem could potentially be the final straw for species already battling extinction. Even if allowing Kinnikinnick to burn has the potential to create higher quality Hoary Elfin habitat, the extant population of Hoary Elfin may not necessarily be robust enough to enjoy such benefits given current ecological conditions.

Our survey of Lower Weir prairie was even further impacted by time and personnel constraints. Given the limited time available (one afternoon), it seemed more efficient to focus on areas that had large concentrations of Kinnikinnick and just wander and observe as many patches as possible. Our observations are sufficient to confirm that there is an extant population in this area.

Training Area 6 and the Central Impact Area

Training Area 6 involved a similarly informal survey. We walked in a grid, rather than following established transects, through some areas of the prairie, documenting Hoary Elfin sightings. This survey also took place very early in adult flight season, when the weather was still fairly cold; adults may have been just beginning to emerge, and cold temperatures reduced the likelihood of detection. Nonetheless, this survey at least confirmed the existence of a population in this area.

Surveys in the CIA always operate under severe time constraints since this area is almost always inaccessible due to military usage for artillery training. We had access for one week, and were able to survey the areas relatively thoroughly. We utilized a grid rather than transects, spreading survey personnel equidistantly within each range to cover ground. In most of the ranges, the area heavily used for training was mowed, with some Kinnikinnick in some areas. Towards the back of the ranges, there was generally an unkempt area being overtaken by vegetation. These areas had a greater abundance of Kinnikinnick and appeared to have thriving Hoary Elfin populations, despite the additional presence of Scot's Broom and other weeds. Areas used for artillery training are subject to unintentional ignition, but prescribed burns are not conducted. Based on burn

data (Figure 16), the CIA experiences the least burning of the areas surveyed, yet the undisturbed areas towards the back of the ranges seem to host the most abundant population, having the most individuals observed in 2018.

Strategies for monitoring and conserving Hoary Elfin populations are still developing. While these surveys may not have yielded significant scientific data, they are an important start for continuing to establish a conservation program for Hoary Elfin. Even confirming whether or not certain areas host an extant population is a valuable first step, and influences monitoring and land management decisions going into the future.

Kinnikinnick regrowth

The first time I visited Johnson Prairie was in January of 2018, approximately 5 months after the wildfire. Kinnikinnick was just beginning to grow back. The boundaries of the former patches were obvious because the burning of heavier fuels– shrubby Kinnikinnick– compared to the surrounding grass left round scars of heavier char in areas previously occupied by Kinnikinnick patches. Despite being in the middle of a cold, dreary PNW winter, the Kinnikinnick had already begun to re-sprout. Red buds were visible above the charred dirt– small, but abundant.

It's interesting that this process began well before "spring," when vigorous new growth usually begins. What factors initiate the regrowth? Is it time since the fire, with the simple process of burning spurring the beginning of regrowth? Is it water inundation from winter rains? Is it the anticipation of spring, causing the plant to utilize its stored reserves to push biomass aboveground to be ready for sunlight and warmth to allow photosynthesis to resume a few months later?

By the time I returned in May 2018 to set up the meter plots and collect my first round of regrowth data, I was amazed at the vigor with which the fresh, green tufts had begun to repopulate their previous domain. While establishing the meter plots earlier would have provided data on the full timespan over which Kinnikinnick regrowth occurred, I can infer with confidence that percent cover was at 0% immediately post-fire in August 2017, and can vouch that it was very close to 0% in January 2018, 5 months post-fire, based on casual observation.

Based on the vigorous regrowth that I observed in May 2018, I thought it plausible that the patches may return to close to 100% cover by November 2018, when I returned for my second set of formal observations. However, while percent cover did increase in all but three of the plots, I found that the patches looked fairly similar to the way they did in May, with tufts of Kinnikinnick re-growing amidst the grasses and forbs repopulating the still-visible charred ground. Specifically, I noticed that it appeared that most of the increase in Kinnikinnick biomass that appeared between May and November was a result of the tufts that had already re-sprouted in May growing longer, leggier branches that were spreading out across the ground, rather than a result of new tufts emerging from the ground.

The number of sprigs counted in each meter plot showed decrease as a general trend. Counting sprigs of Kinnikinnick proved difficult, as the pattern of regrowth often makes it difficult to discern distinct areas where it re-sprouts from underground structures. While I had to utilize discretion, I counted sprigs for all observations personally, therefore eliminating the bias of multiple observers. While the exact numbers produced by this methodology may not be completely precise, I think that the overall

trend that my results show– that the number of distinct areas of regrowth separated by bare earth decreased from May to November– represents an aspect of the pattern of regrowth. Based on my observations, I believe that the trend of decreasing number of springs/tufts with increasing percent cover was a result of the sprigs/tufts that already existed in May growing longer branches and spreading across the ground. (The alternative would be new sprouts emerging, but covering the bare earth to the point that they merged with other tufts and were not counted as individuals, hence the decrease in count of sprigs/tufts in November). Kinnikinnick sends its long runners across the ground as it spreads, which then send down new roots. Therefore, it does not need to rely exclusively on seed germination or new shoots to colonize bare ground as it spreads.

The third and final set of observations I collected in May, 2019, reveal significantly less gains in percent cover than I witnessed the previous winter. I hypothesize that during the first winter, Kinnikinnick rebounded from virtually zero percent cover by utilizing energy stored in the underground structures from which it re-sprouted. Without leaves, it would have been far less capable of photosynthesis anyway, and therefore less dependent on available sunlight to re-sprout. During the second winter, above-ground biomass capable of photosynthesis had reestablished, but the plant's capabilities were likely limited by reduced sunlight. In May 2019, as the growing season was beginning, new growth was emerging from the existing Kinnikinnick. Additionally, while some plots were observed to have a few new sprigs, the overall trend appears to be continued growth from existing above-ground stems rather that more re-sprouting.

This element of my study is similar to del Barrio et al. (1999) in that it aims to measure the rate and pattern at which Kinnikinnick regenerates after burning. The authors

compare response to burning and mowing by applying each treatment to 50cm diameter circular plots. They ultimately find negligible difference, though they report that initially it appeared that the burned plots experienced more regrowth in the center. The variables measured were cover and aboveground biomass, compared between the two treatments. Additionally, and perhaps more importantly, the authors observed whether the regeneration occurred primarily from adjacent unaffected stems or from underground re-sprouting, and compared this between the treatments.

Their results are difficult to interpret, however, because of inconsistency in the language used. When explaining their methodology, the authors state their intention to look at regeneration "from the aboveground, unaffected stems which gradually invade from close lying areas, or from subterranean organs…" (pg. 191, pg. 192). However, when discussing their results, they frame their observations in terms of regeneration from "vegetative re-sprouting" vs. "seedlings" (pg. 193, pg. 194). I infer that 'seedlings' refers to new shoots that have sprouted from seeds, rather than from underground organs. The authors explain their method for determining whether regrowth occurred from adjacent, unaffected stems or underground organs: the circular treatment plots were divided into two concentric circles, and they assumed that if regeneration occurred first on the edges and moved inward, it was due to regrowth of unaffected stems, but if it occurred uniformly across the circle it was due to underground re-sprouting (pg. 192). However, their results do not differentiate regeneration between the inner circle and outer circle of the treatment plots, although they do imply that they observed regeneration uniformly within the plots.

The initial question of whether regeneration occurs from adjacent stems or subterranean organs is comparable to my observations in November. In May, I could infer that all regrowth had occurred from subterranean re-sprouting; in January the patches I observed informally were gone. Moreover, in May the new spring growth was brighter green than older growth, making it easy to distinguish what had recently re-sprouted from the few remnants that survived the fire. Between May and November, it appeared that most of the increases in percent cover of the meter plots was due to growth of adjacent stems that had sprouted prior to May.

Figure 18. Patch #31 in May 2018 (left), November 2018 (middle), and May 2019 (right), as an example of the Kinnikinnick regrowth observed in Johnson prairie over one year.

The methods del Barrio et al. used to answer this question differ from mine, and the difference is notable in that their methods represent an unnatural disturbance pattern. They burned or cut 50cm diameter circular plots in areas where Kinnikinnick formed homogenous groundcover. It is unlikely that a natural disturbance would produce this kind of pattern. My methods utilized observations following an actual wildfire, and therefore may better represent the way Kinnikinnick responds to natural disturbance. The question of initial regrowth from adjacent stems in del Barrio et al.'s study is only relevant if adjacent stems are present. Since Kinnikinnick tends to grow in patches in the prairies, rather than in large, homogenous mats, it is less likely that there will be adjacent

stems present. This is supported by my observations in Johnson, where distinct patches of Kinnikinnick exposed to fire were for the most part fully consumed.

An additional aspect of this study informative to the context of South Sound prairies is that del Barrio et al. compare burning to an alternate treatment: cutting. Due to constraints regarding burning, such as air quality standards or other logistical issues (Hamman et al., 2011), or the potential for unintended consequences (Rook et al., 2011), the efficacy of surrogates for burning, such as mowing, is a pertinent question in South Sound prairie restoration (Nuckols et al., 2011; Rook et al., 2011; Stanley et al., 2011). This question seems to be either explicitly or implicitly framed in the context of whether surrogates such as mowing are as effective as burning in habitat restoration initiatives, such as mitigating invasive plant presence or tree encroachment, since mowing may be less controversial or logistically difficult. Stanley et al. (2011) conclude that mowing is not an effective substitute compared to burning for prairie restoration in general. However, Kinnikinnick is unique in that it is more difficult to discern the benefits of burning in the first place, especially as it pertains to Hoary Elfin; this is demonstrated by the implementation of burn exclusions for Kinnikinnick in order to preserve Hoary Elfin habitat (Hamman et al., 2011). This, of course, begs the question of whether allowing Kinnikinnick to burn improves habitat in the long run, even if it results in Hoary Elfin mortality in the short term (this is essentially the question that McMullen et al. (2017) investigate). Low-growing Kinnikinnick has to the potential to be overtaken by grasses and Scot's Broom, and it is unknown how this may impact Hoary Elfin's ability to utilize their host plant. Del Barrio et al. demonstrate that Kinnikinnick is capable of rebounding from cutting similarly to the way it rebounds from fire. This is significant because it is

plausible that Kinnikinnick could have responded more positively to fire, such as having regrowth stimulated by burning more so than cutting, but Del Barrio et al. show no differences in response. Cutting/mowing might prove to be an effective surrogate for Kinnikinnick if it shows the potential to provide the habitat benefits of disturbance while resulting in less butterfly mortality than burning.

To measure Kinnikinnick regrowth, Del Barrio et al. and I both used visual observation to estimate percent cover. They measured biomass by drying cut Kinnikinnick in a hot air chamber and measuring dry weight. Application of similar methods in experiments on South Sound prairies could potentially be instructive in generating scientific answers to the questions regarding habitat needs of Hoary Elfin. This includes how much Kinnikinnick regrowth is necessary for Hoary Elfin to reoccupy patches after burning, or if regrowth (measured in biomass) is the primary factor determining reoccupation at all (as opposed to time or distance from a source population). Continued monitoring of the burned patches for Hoary Elfin recolonization may provide insight.

CONCLUSION & RECOMMENDATIONS FOR FUTURE RESEARCH

This study serves primarily as a compilation of available data on Hoary Elfin populations and habitat on lands stewarded by Joint-Base Lewis McChord, and as much analysis as is appropriate given the limitations of the study design. Data on burn history of Hoary Elfin habitat at JBLM was compared to locations of Hoary Elfin sightings within the same area, with the goal of establishing a relationship between prescribed burning activities and Hoary Elfin occupation. Additionally, regrowth of Kinnikinnick in habitat previously occupied by Hoary Elfin but which burned in a wildfire in 2017 was monitored. I hope this study may serve to help shape the direction of future monitoring and restoration efforts.

Johnson prairie remains a primary area of interest for future monitoring efforts, as it contains Hoary Elfin habitat but the future of the population itself remains uncertain. Regular surveys of Johnson prairie during Hoary Elfin flight season should continue, despite the absence of adults observed in 2019. Further research on Hoary Elfin dispersal and colonization can benefit land management decisions regarding Hoary Elfin habitat. One knowledge gap is in regards to Hoary Elfin dispersal in adjacent patches of habitat (for example, into a burned area within the same prairie), such as how long it takes for disturbed (i.e. burned) areas to become inhabitable and subsequently when they are recolonized in the event of total mortality within the disturbed area. Additionally, knowledge is lacking on the ability of Hoary Elfin to travel between fragments of habitat separated by areas without suitable habitat.

Another missing piece of knowledge concerns the dispersing individuals themselves, such as whether females carrying fertilized eggs disperse to new (and

potentially unpopulated) sites or whether dispersing individuals must find mates at their destination. These questions relate back to principles of disturbance ecology, such as the storage effect and the relationship between disturbance and habitat structure. More importantly, they have implications for conservation because they provide insight into the landscape-level characteristics needed to sustain a population, or, conversely, what human intervention might be needed to sustain a population at the landscape level given current environmental conditions.

In addition to studying Hoary Elfin dispersal in terms of physical capability, site selection is another component in understanding habitat needs. Evaluating the variables that determine the quality of a patch of Kinnikinnick for Hoary Elfin use, including time since last burned, can provide insight into what constitutes high quality habitat. Further studies in Johnson prairie may provide an opportunity to answer such questions; whether Hoary Elfin recolonize naturally or are reintroduced via relocation, the areas of the prairie that they reestablish may reflect the habitat characteristics most conducive to their persistence.

Thorough surveys of Upper and Lower Weir prairies may be an opportunity to establish a relationship between fire and Hoary Elfin distribution, as well as track trends in Hoary Elfin distribution across this large area. Understanding the impacts of prescribed burning operations on this species would provide information immensely useful to habitat conservation and improvement efforts. It would be ideal to survey each prairie in one day by committing enough personnel to finish all the transects. This would remove weather conditions and time during flight season as variables potentially influencing spatial variance in Hoary Elfin detection. Repetition of this process multiple times during flight

season would be ideal to recognize patterns in spatial variance of Hoary Elfin detection. Discerning these patterns and eliminating as much as possible the influence of the variables of weather and time would bring us closer to examining burning as a variable influencing Hoary Elfin distribution. The primary question of this thesis is the response of Hoary Elfin populations to prescribed burning; the prairies are generally burned in a mosaic rotating 3-5-year intervals (as shown in Figure 12, Figure 14, Figure 16), so data on the distribution of Hoary Elfin in relationship to the burn history of different patches of prairie would provide insight into this question, granted other factors can be taken into consideration or controlled for.

Further documentation of burn exclusions and their impact on Hoary Elfin populations would be instructive in answering these questions. If burn exclusions are proven or even assumed to be beneficial for Hoary Elfin conservation, the vast quantity of Kinnikinnick present in Upper and Lower Weir prairies (as seen in Figure 7) causes me to question the practicality of preserving all Kinnikinnick in these areas from prescribed burning. Therefore, understanding Hoary Elfin distribution and identifying population hot-spots, if any, may allow land managers to target limited resources and efforts most effectively.

While the wildfire in Johnson prairie presented an opportunity to study the prairie's response, future studies on Kinnikinnick response to fire may benefit from including observation of patches before and after burning. Pre- and post-fire comparisons on characteristics such as total diameter and patchiness (essentially percent cover within the patch) can provide additional information on the reestablishment process. We know that Kinnikinnick can re-sprout even if all aboveground biomass is removed by burning,

but we don't know how burning influences its characteristics over time, especially in areas such as prairies that see frequent fire.

Prairies are dynamic landscapes dependent on frequent disturbances, which means that not only species but processes are essential to be considered in preserving ecosystems. Conservation of native biodiversity is a massive endeavor, one of the greatest environmental challenges we face. Why should we care about a small, brown butterfly that spends its life on one plant? Hoary Elfin conservation is one small but vital piece of the broader conservation effort to protect our rare prairie ecosystem in the South Puget Sound.

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