2005

Effects of cutting vegetation along trails in serpentine grasslands

Katie Beltrano
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EFFECTS OF CUTTING VEGETATION
ALONG TRAILS IN SERPENTINE GRASSLANDS

A Thesis

Presented to

The Faculty of the Department of Environmental Studies

San Jose State University

In Partial Fulfillment

Of the Requirements for the Degree

Master of Science

by

Katie Beltrano

December 2005

UMI Number: 1432450

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ABSTRACT

EFFECTS OF CUTTING VEGETATION ALONG TRAILS IN SERPENTINE GRASSLANDS

by Katie Beltrano

Trail maintenance, including mowing and brush-cutting, is of interest due to the presence of plant species' of concern in serpentine grasslands. This research assesses how timing of trailside brush removal affects native and non-native plant distribution along trails in serpentine grasslands in coastal California. Using a random block design, 32 blocked trail segments were assigned one of these four treatments: string-cut in May, string-cut in June, string-cut in July, or an uncut control. Plant community composition was recorded using quadrat (0.25 m) samples taken from each trail plot immediately before and one year after treatment. String-cutting along trails as compared to not cutting has no significant effect on abundance and species richness of native, serpentine plant species. To reduce non-native grasses and maintain native abundance and species richness, it is recommended that trail maintenance occur in May as compared to June or July.

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At Edgewood Park and Natural Preserve, I was able to witness spectacular sunsets, spring wildflower displays that could be postcards, and to meet a wonderful group of staff and volunteers, who are truly dedicated. Park Ranger Ricardo Trejo, with San Mateo County Parks, allowed me access to relevant planning documents, provided me with weather data, and was always available for conversation. Ken Himes, California Native Plant Society member, sparked a child-like curiosity in me to identify and to interpret everything in the natural world, through many long conversations literally out standing in the serpentine grasslands of Edgewood. To my parents, Pete and Judy Beltrano, thank you for your unconditional love and support, and for instilling in me at a young age a desire for endless learning, in which I will be forever grateful.

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INTRODUCTION

This study analyzes the effects of timed mowing using string-cutting of trailside vegetation on plant species composition (native vs. non-native) and distribution in serpentine grassland in Edgewood Park and Natural Preserve, San Mateo County, California.

The purpose of this research is to determine whether or not there are increases, decreases, or no change in the distribution of native and non-native plants alongside trails after string-cutting treatments. String-cutting, a similar method to mowing, is used in areas where mowers are unpractical, such as steep slopes or narrow trails. Trail maintenance in California parks typically consists of annual brush-cutting (vegetation removal via weed-wacking and clipping) within 1-3 feet of each side of recreational trails. The results of comparing cut and uncut trail segments can be used to improve trail maintenance practices, such that negative impacts to rare or sensitive native plants are minimized. Also, these results provide further insight into the potential spread of non-native invasive plants via disturbance activities, such as trail maintenance, so that increases in the spread of non-natives are also minimized.

Problem Statement

Anthropogenic demands and impacts. As the human population increases worldwide, the demand on the natural environment and natural resources also increases. Some human impacts that affect species are development, habitat fragmentation,

deforestation, fire suppression, desertification, sedimentation, agriculture, grazing, and hunting. The list of human activities that affect species diversity is extensive, and recreational impacts have recently been added. In parks, preserves, and open space areas, recreational use can cause impacts, such as trampling (Kobayashi et al., 2002; Monz, 2002), decreased species diversity (Radosevich, Stubbs, and Ghera, 2002), and the introduction of non-native species (Hall and Kuss, 1989; Walck et al., 1999).

The increasing popularity of outdoor recreation means more visitors in parks and open space areas using trails with the potential of causing more disturbances. Ecotourism locations, U.S. National Parks, and California State Parks have all experienced increases in visitors over the last 20 years. For example, the United Nations and World Tourism Organization designated 2002 as the International Year of Ecotourism to promote sustainable ecotourism while accommodating increasing demands (Hoad, 2002). In U.S. National Parks, the number of visits to parks has been steadily increasing from 282.5 million in 1979 to 436.3 million in 1999 (NPS, 2002). The demand for California State Parks has also increased in recent years to over 70 million visits (CA Parks, 2001). The impacts of this increased demand in parks, open space areas, and preserves are of concern.

Spread and impact of non-native species. A goal of many parks and natural areas is to protect and preserve native biodiversity. However, some park maintenance practices can negatively affect native species richness and diversity and aid in the introduction and spread of non-native species. For example, brush cutting and mowing may further the spread and distribution of invasive, exotic plants that are threatening California's native

species and the state's unique biodiversity by disturbing and removing environmental barriers allowing alien plants to be geographically transported. Disturbance may aid invasion by increasing the availability of light, water, minerals, and other nutrients (Grime, 1979; Hobbs and Huenneke, 1992; Williamson and Harrison, 2002).

Invasive non-native species are defined as those species that aggressively spread into areas where they are not native (Bossard, 2000). Invasive non-native plants displace native species and alter natural habitats. By out-competing native species and suppressing native species recruitment, non-native species change the structure of communities, which degrades and eliminates habitat for native plant, animal, and invertebrate species (USFWS, 1998). Non-native plant species alter ecosystem processes, such as nutrient cycling, erosion, hydrological cycles, plant succession, and wildfire frequency and intensity (Bossard, 2000). Invasive non-native species are becoming one of the leading causes of global extinctions (NRC, 1999). According to the USFWS, non-native species are the second most important factor in habitat loss, behind habitat destruction.

One highly invasive non-native species, yellow-star thistle (*Centaurea solstitialis*), can be seen along any major roadside in California and its range increased from 1.2 to 7.9 million acres between the 1950's and 1991 (Thomsen et al., 1996). By the early 1990s, at least 1,023 non-native plants were established in California (Hickman, 1993). The extent and spread of invasive plants are threatening our National Parks, preserves, and other protected areas where rare species that are highly vulnerable to invasion are susceptible.

Biological diversity. The Convention on Biological Diversity broadly defines biological diversity, biodiversity, as the “variability among living organisms from all sources (UNEP, 1992).” E.O. Wilson, one of the world’s experts on biodiversity, as well as other scientists in the fields of population biology, ecology, evolution, and conservation biology, widely agree that biodiversity is ‘the variety and variability of biological organisms’ (Wilson and Peters, 1988).

The state of California supports at least 875 genera and 5057 native vascular plant species (Munz, 1968). There are more species in California than in the entire central and northeastern United States and Canada (Barbour, Burk, and Pitts, 1987). Of these species, 1525 (30.2%) are endemic to California. Many of California’s serpentine endemic species are restricted near earthquake fault zones. This includes the South Coast Range, the San Francisco Bay Region, North Bay Counties (Marin, Sonoma, and Napa counties), North Coast Range (into Oregon), and the Sierra Nevada (Kruckeberg, 1984).

The Recovery Plan for Serpentine Soil Species of the San Francisco Bay Area (1998) states that California’s serpentine soils are considered unique and biologically diverse. Serpentine rock (California’s state rock) and the soils derived from it cover less than one percent of the state. The California Floristic Province, with serpentine soils representing unique, botanical richness, was recently proposed to be recognized as one of the Earth’s biological hotspots (Harrison and Inouye, 2002).

Background

Serpentine soil. Serpentine soils are characterized by their mineral toxicity to most plants, which results in a harsh environment for plant growth. Serpentine soil is high in magnesium and low in calcium minerals, which serves as a limiting factor to plant growth. Other restrictive components of serpentine soils are the lack of nitrogen, potassium, and phosphorous, all important nutrients for plant growth. Due to these factors, many of the plants that grow in serpentine soils are specifically adapted to these conditions. These rare and unique plants are referred to as serpentine endemics (USFWS, 1998).

There are at least five federally listed species (*Acanthomintha obovata* sp. *duttonii*, *Cirsium fontinale*, *Hesperolinon congestum*, *Pentachaeta bellidiflora*, and *Euphydryas editha bayensis*) and three species of concern (*Lessingia arachnoidea*, *Calicina minor*, and *Microcina edgewoodensis*) located in Edgewood County Park and Natural Preserve in San Mateo County, California (Figure 1), the study site for this research (USFWS, 1998).

Due to the harsh nature of serpentine environments, exotic plants spread at a slower rate as compared to the spread of exotics in fertile soils (Crawley, 1987; Hobbs, 1989; Mack, 1989; Rejmanek, 1989; Burke and Grime, 1996). Also, the species richness and abundance of exotics are lower in serpentine soils than in non-serpentine soils (Murphy and Ehrlich, 1989; Harrison, 1999).

Because serpentine soils support unique and endemic species, natural resource managers must include the inherent value of this biodiversity as they manage recreational

use in serpentine areas. For example, decisions about trail placement and maintenance must protect natural resources while allowing for public access. For this study, the plant distribution alongside trails and the potential impacts of trail maintenance in serpentine grasslands are of concern due to the presence of endangered and threatened species.

Policy framework. Trail maintenance and vegetation cutting is typically done to reduce hazards to allow for unimpaired recreation (8 ft. height and 6 ft. width for clearance) and for fire safety precautions. The San Mateo County Division of Parks and Recreation (1999) prepared an EIR for all trails including sections on design and maintenance guidelines.

The Edgewood County Park and Natural Preserve Master Plan (1997) outlines the policy framework for managing parks and the human impacts of recreation on natural resources. This document highlights the significance of adopting specific management strategies to protect serpentine soils and the rare endangered and threatened plant, animal, and invertebrate species, found in serpentine soil areas. From the results of this study, recommendations are provided for preferred trail maintenance procedures in serpentine environments.

In order to standardize trail maintenance procedures, the Santa Clara County Trails Master Plan Update (1995) and the work of Sternloff and Warren (1984) have also been reviewed focusing on the design and maintenance guidelines for trails. Few of the trail maintenance policies discussed focused on promoting native species and restricting non-natives, but rather the specifications were designed around human needs for height

and width clearance. There is a need for current research on the effects of trail maintenance in sensitive, natural areas.

Research design. To assess the effects of string-cutting on native and non-native plant species in Edgewood Park, data were collected on cut (treatment) and uncut (control) trail segments. Sites for both treated and control plots were selected to be as visually similar as possible in slope, topography, level of usage, and vegetation type. Along each trail segment, a random block design consisted of three cutting treatments (May, June, and July) and one control plot. Prior to string-cutting, I measured plant species richness and percent cover in sample quadrats (0.25 m). I measured the same parameters in 0.25 m quadrats one year after cutting. Inferential statistics (ANOVA tests) were applied to the data to determine whether or not there were significant changes in plant species composition and distribution alongside trails in serpentine grasslands.

Limitations. Due to time and financial constraints, this study did not extend beyond two years. With only two seasons of data, data interpretation is limited, whereas, with 3 to 5 seasons of data collection, data interpretation is more reliable and could provide a more accurate picture of gradual changes in plant species composition. Therefore, trends in plant species composition and changes that are not significant in this study may prove to be significant with more data in subsequent seasons.

The study was contained within one park. The results of this study are not intended to be generalized for all trails nor all grasslands, but should help guide management in other Northern California serpentine grasslands. Other common

vegetation removal techniques, such as herbicide spraying, prescribed burning, and grazing were not analyzed.

Some uncontrollable factors that could affect plant growth are weather and trampling. Recreational use in all treatment and control sections of trails was limited to hikers and equestrians. These trail users could have had an affect on vegetation growth. It would be difficult to control for unintended disturbances of human and equestrian trampling, erosion problems, or illegal activities such as off road vehicle activity. It is assumed that the probability of these unintended activities occurring would be equal for all treated and control sections of trail. A similar assumption of homogeneity was made with regards to fluctuations in weather.

RELATED RESEARCH

Research for this thesis topic is dependent on a number of ecological principles. In particular, the theory of plant succession and the role of disturbance in ecology are central. The impacts of exotic plant species are reviewed as are research studies conducted on vegetation removal, cutting, and mowing. Finally, relevant research on trail impact assessment is discussed.

Theories on Plant Succession and Disturbance

Clements (1916) laid out an organized interpretation of the phenomenon of succession. Succession accounts for the predictable and directional process that occurs after a disturbance. This process, referred to as the secondary or intermediate stage of succession, is observed in areas which supported communities prior to disturbance and exhibit survival of some residual species (Gutierrez and Fey, 1980). Soon after disturbance, nearby species, which are capable of rapid growth and spread, enter the disturbed area. Overtime, certain species, such as highly competitive or adaptable species, alter the environment until stabilization or a dynamic equilibrium is attained, known as the climax stage of succession.

Odum (1969) summarizes the changes in species diversity as it relates to secondary succession. During the pioneer stage, species diversity is low. Then, the trend in species diversity increases, peaking in the intermediate stage. Finally, species

diversity is high at the climax stage. Disturbance is an integral component of secondary succession as it initiates and maintains some ecosystems (Smith, 1986).

Contrary to the views of Clement (1916) and Odum (1969), Gleason (1926) argued that plant associations were unique and the product of random processes devoid of finite stages known as the individualistic concept. For example, in a given area, vegetation association, abundance, and diversity result from “the fluctuating and fortuitous immigration of plants and an equally fluctuating variable environment (Gleason, 1926).” Interaction between species and competition for nutrients, light, and moisture determine the structure of a plant community in which short lived colonizing species will eventually be displaced by species that are long lived and have adapted to a specific area (Gleason, 1926).

Succession describes the process of disturbance, dispersal, and establishment of natural communities. Grime (1979) presented his version of succession which was based on the adaptations of individual species reaction to stress and disturbance at different intensities. Disturbance critically affects the abundance and diversity of species in plant communities (Watt, 1973).

The intermediate disturbance hypothesis postulates that with the proper intensity of disruption high species diversity can be attained because competition is reduced and vigorous species are limited from monopolizing resources (Petraitis, Latham, and Niesenbaum, 1989; Wootton, 1998). If levels of disturbance are too low and/or infrequent, then communities will be dominated by a few, highly competitive species. However, if disturbances are too intense and/or too frequent, then species diversity will

also decline allowing a few, resistant species to prosper while most species will not be able to reproduce fast enough.

Species diversity along trails in serpentine grasslands is dependent upon the past uses and levels of disturbance. Serpentine environments are considered to be steady-states that include disturbance (associated with earthquake fault zones) as part of the ecosystem (Kruckeberg, 1984). All serpentine habitats have been severely disturbed by human impacts including land development, grazing, off-road vehicle usage, and nitrogen enrichment from vehicle emissions in metropolitan areas (Weiss, 1999). Some disturbances, such as cutting trail edges to facilitate recreational use, are directly under park management control. Given good information park managers can adjust this disturbance to support native plant diversity.

Effect of the Spread of Invasive Plants on Biodiversity

Baker (1989) identifies grasslands and transportation corridors (roads and trails) as susceptible ecosystems that are vulnerable to invasion. There is evidence that rare, native plants are being directly displaced in the grasslands of Kentucky, U.S.A. by invasive plant species (Walck, J. Baskin, and C. Baskin, 1998). They explain that rare plants are typically vulnerable to invasion due to poor competitiveness and/or small geographic ranges. Radosevich, Stubbs, and Ghera (2002) state that plant invasions can profoundly affect biodiversity and discuss the need for protection of biologically diverse grasslands. They also indicate that human disturbances can aid introduced species as the invasions increase in geographic range.

Bossard et al. (2000) conclude that roads and trails can serve as transportation corridors for the spread of invasive species. They state that encouraging competition from natives, manual removal, prescribed fire, herbicides, biological control, flooding, grazing, and/or mechanical methods, such as mowing and string-cutting, can be effective land management treatments used to control exotic plants. They especially note the importance of the timing of treatments based on the plant's biology, in order to prevent the unintentional consequence of eliciting growth and furthering the spread of invasive species.

Frankel (1977) studied plant vegetation along roadsides in northern and central California. His findings show that habitat areas that lack canopy cover, such as grasslands, had a high percentage of introduced plant species. He also found that the number of introduced plant species decreased as distance away from roadsides increased. Also, this study found that annual plants are typically the dominate plant species established in disturbed sites.

The concept of dispersal patterns of invasive plants aligning with human transportation corridors is supported by Tyser and Worley (1992). They found that alien species are successfully invading grasslands from roadsides in Glacier National Park. Buckley et al. (2003) add to the corridor theory and state that areas surrounding old forest roads and skid trails in Michigan are displaying an increase in non-native plant species richness. Their methods included sampling with transects perpendicular to forest roads and skid trails. An increase in exotic plant species might be indicative of changes in native species diversity (Kharin, 2002).

Change from Perennial to Annual Grasslands

The introduction of European annual grasses has vastly changed California's grassland ecosystem by replacing native perennial grasses. In California, native grasslands are highly threatened, and serpentine grasslands represent a substantial portion of the remaining, native-dominated grasslands (Murphy and Ehrlich, 1989). In particular, perennial bunch grasses, especially those found in serpentine environments, are being displaced by annual grasses and forbs (Heady, 1977; Mack, 1989).

According to Brown and Rice (2000), California's native, perennial grasses, such as purple-needle grass (*Nassella pulchra*), are being out-competed by non-native, annual grasses. One explanation is that annual plants emerge faster than perennials, thus competing earlier for water and mineral resources (Bartolome and Gemmill, 1981). Another explanation is that annual plants have higher growth rates than perennial plants (Muller and Garnier, 1990).

Chiariello (1989) analyzed the relationships between climate, growth, development, and the controls on flowering date. In a Mediterranean climate, annual precipitation is followed by at least two months of summer drought. Plants have adapted to this environmental pressure by flowering in the spring before summer droughts. However, annual and perennial plants show variance in their responses to this climate, such that annual plants typically emerge earlier than perennial plants. According to this study, more research is needed that compares native and introduced grasses and forbs.

Vegetation Removal/Mowing Studies

Research for this study focuses on the impacts of string-cutting also known as weed-whacking alongside trail borders. No published research describes the impact of string-cutting on vegetation distribution and diversity alongside trails. However, studies that focus on mowing treatments are very relevant as mowing and string-cutting are similar management methods. String-cutting is a coarser method of vegetation cutting leaving stems at various heights. String-cutting is preferred to mowing in rocky areas due to fire danger associated with metal blade mowers.

Previous research on the impact of string-cutting, grazing, and fire for butterfly habitat restoration has been conducted at Edgewood Park by Stuart Weiss (2002). His results identified string-cutting (instead of mowing) as the preferred method over grazing due to the non-selective nature of grazers with the presence of endangered species in Edgewood Park and as preferred to prescribed fire due to close proximity of residential areas to the park. He recommended cutting vegetation during the spring as compared to cutting in the fall. In string cut plots, native plant species richness increased from 7.8 to 11.3 species per quadrat. This finding was attributed to a reduction in thatch which allowed for higher forb densities (Weiss, 2002).

In northern Finland, Huhta (2001) tested the effects of restorative mowing and grazing on semi-natural grasslands at the community and species levels. As compared to grazing, mowing favored late flowering non-rosettes and was conducted with more intensity, but less frequently than grazing. This study identified the timing of mowing treatments as important. Mowing earlier delayed and/or prevented flowering in late-

successional species, which had effects on production and species composition. Huhta (2001) also found that mowing later maintained existing vegetation allowing plants to set seed. The late mowing treatment also removed litter accumulation and preserved grasslands such that there were no major short-term changes in plant species composition.

A study conducted by Howe (1999) tested the timing of mowing treatments in a tall-grass prairie in Wisconsin. He found that mowing in May had no effect on the population of *Zizia aurea*, a native plant. Mowing in August resulted in double the population. He attributed these results to allowing plants to set seed and that late mowing reduced accumulated litter and removed the canopy cover of dominant species.

Thomsen et al. (1996) also tested the timing of mowing analyzing its effect on yellow-star thistle (*Centaurea solstitialis*) in the Sacramento Valley, California. They found that mowing at the time of defoliation (usually in July) is an important factor in reducing yellow-star thistle seed production and dispersal.

In Japan, Kobayashi, Hori, and Nomoto (1997) researched the effect of mowing to remove vegetation on plant species diversity. Their initial observations showed a rapid recovery and increases in native plant diversity after vegetation removal. They attributed this recovery to increased sunlight and decreased competition. The study's results concluded that in areas where vegetation was removed, there was an increase in plant species diversity.

In a related study in Japan, Kitazawa and Ohsawa (2002) studied the impact of management types, including mowing, on plant species diversity in rural herbaceous

communities. They found higher species richness and abundance of unique species in the mown sites, as compared to the cultivated, trampled, and abandoned study sites.

Overall, most researchers agree that mowing results in an increase in the number of species present, but not necessarily an increase of native species present (During and Willems, 1984; Gibson, Seastedt, and Briggs, 1993). Therefore, further research is needed to analyze the effect of mowing on native and non-native plant species abundance and species richness.

Trail Impact Assessment Studies

Trail impact assessment studies provide further insight into the effects of disturbance on vegetation alongside trails. Researchers conducting trail impact assessment studies often cite the work of Cole and Bayfield (1993), who provide a reliable source for methods, sampling techniques, and data interpretation. Another potential source of methods for trail-side disturbance studies is Monz (2002), who measured plant species cover, percent bare ground, and vegetation height in arctic tundra plant communities. Monz (2002) discussed the importance of gaining site-specific information on the response of plant communities to human disturbance, especially with regard to management decisions.

Leung and Marion (1999) used different computer models to examine sampling strategies for trail impact studies in Great Smoky Mountains National Park. Their research showed that trail impact assessment can be accomplished accurately through a re-sampling simulation method. Their findings revealed an excellent level of accuracy in

trail sample intervals that are less than 100 m. They also recognized that one of the most established ecological methods for vegetation sampling is to use line transects and/or quadrats.

Further evidence of human impacts to plant communities along trails was found by Adkison and Jackson (1996) in Midwestern forests of the U.S. They found that the trampling of vegetation near trails produced plant composition changes, organic matter decreases, and soil compaction increases. Hall and Kuss (1989) also studied vegetation alteration along trails in forested areas and reported that plants along the trail border bloomed earlier and seemed to display low growth.

Previous researchers analyzing trails and vegetation distribution have been primarily concerned with the impacts of human trampling. No published literature reports the effect of cutting vegetation alongside trails in biologically diverse grasslands.

OBJECTIVES

Natural resource managers and field staff are charged with the dual missions of providing recreation to visitors and protecting the natural environment. Unfortunately, these two mandates are often in conflict, especially with regard to maintenance activities in sensitive habitats. This study concentrates on analyzing plant species changes, namely the spread of invasive, non-native plant species, within serpentine grasslands.

Specifically, the activity of string-cutting is analyzed along recreational trails in Northern California serpentine grasslands. The primary question addressed is whether string-cutting, as a trail maintenance practice, and the timing of cutting affect the diversity and abundance of native versus non-native plant species. Results of this study can be used to improve trail maintenance procedures that better balance the demands of humans and the protection of public lands, while limiting impacts to the natural environment.

The hypotheses tested in this study are:

H₀: There is no significant difference in native and non-native plant cover and species richness between trail edges that were not string-cut versus trail edges string-cut in May, June, and July.

H₀₁: There is no significant difference overall between string-cut and uncut trail edges.

H₀₂: There is no significant difference when trail edges are string-cut at different times.

H₀₃: There is no significant difference among the three blocked trail sites.

METHODS

Study Site

Research for this thesis was conducted at Edgewood Park in Redwood City and in the unincorporated lands of San Mateo County, California, U.S.A. (Figure 1).

Redwood City is located within the Mediterranean climate of Northern California that usually consists of distinct wet and dry seasons with moderate temperatures. For Redwood City, the average, annual precipitation is 50.8 cm, with about 43.2 cm usually falling in the months of November through May. Average temperatures range from 1° to 18° C in the winter months and from 10° to 32° C in the summer months.

Edgewood Park offers approximately 16 kilometers of hiking trails in 189 hectares (Figure 2). It contains a variety of vegetation types, including serpentine grassland, chaparral, oak woodland, and wetland communities. The park varies in elevation from 73 m to 266 m above sea level, and it lies in the rain shadow of the Santa Cruz Mountain Range. Annually, Edgewood Park receives 30,000 visitors with usage groups including hikers, joggers, equestrians, bird watchers, wildflower viewers, picnickers, school groups, and volunteers (Master Plan, 1997). The park is bordered on the east and the south by residential areas; to the north by Pulgas Ridge Open Space of Mid-Peninsula Regional Open Space District; and, to the west by Interstate 280 and the San Francisco Watershed District.

Edgewood Park was established in 1982 after many years of off road vehicle activity. It was designated a Natural Preserve in 1992 to protect the Bay Checkerspot

Butterfly (*Euphydryas editha bayensis*) as a Threatened species under the Federal Endangered Species Act (Master Plan, 1997). Edgewood Park contains known occurrences of the Federally Endangered San Mateo Thornmint (*Acanthomintha duttonii*), Marin dwarf flax (*Hesperolinon congestum*), white-rayed pentachaeta (*Pentachaeta bellidiflora*), fountain thistle (*Cirsium fontinale*), and over 20 special status species listed by the California Native Plant Society (USFWS, 1998). Areas with known occurrences of these species were avoided during this research.

Edgewood Park contains approximately 58.3 hectares of serpentine soil, and it supports the following endemic plants: serpentine linanthus (*Linanthus ambiguous*), fragrant fritillary (*Fritillaria liliacea*), woolly-headed lessingia (*Lessingia hololeuca*), California gilia (*Gilia capitata*), fringed checkerbloom (*Sidalcea diploscypha*), and large-fruited lomatium (*Lomatium macrocarpum*) (Master Plan, 1997).

Edgewood's serpentine grassland supports a high diversity of native wildflowers. Some common species include the California poppy (*Eschscholzia californica*), tidy tips (*Layia platyglossa*), purple owl's clover (*Castilleja exserta*), gold fields (*Lasthenia californica*), bicolor lupine (*Lupinus bicolor*), Ithuriel's spear (*Triteleia laxa*), and royal larkspur (*Delphinium variegatum*) (Master Plan, 1997). A large number of invasive non-native plant species are threatening Edgewood's grasslands including yellow-star thistle (*Centaurea solstitialis*), Italian thistle (*Carduus pycnocephalus*), Italian ryegrass (*Lolium multiflorum*), teasel (*Dipsacus fullonum*), false purple brome (*Brachypodium distachyon*), wild oat grass (*Avena fatua* and *Avena barbata*), soft chess (*Bromus hordeaceus*), and ripgut (*Bromus diandrus*) (Rana Creek, 2002).

Research Study Design

This research was conducted during two seasons with quadrat samples being measured in late April of 2004 and in early May of 2005. Three treatments of string-cutting were conducted to test the effect of timing on the growth of native and non-native plant species. Before the treatments, plant parameters were measured in 2004 to serve as baseline data for comparison. The first, string-cutting treatment occurred on Thursday, May 6, 2004; the mid-season treatment was on Tuesday, June 1, 2004; and, the late-season treatment occurred on Thursday, July 1, 2004. Treatments were not conducted on weekend days in order to minimize impact to park visitors.

A total of 8 trail segments were selected and blocked in 3 serpentine sites: Clarkia Trail (Trl.), Serpentine Trl., and Service Rd. Trail segments included both sides of the trail, and treatments and control plots were paired following a random block design (Figure 3). Each of the trail segments contained these 4 plots: treatment in May, treatment in June, treatment in July, and a control plot. Each plot spanned 10 meters in length and 0.5 meters in width on each side of the trail border forming parallel line transects.

Two sample quadrats (0.25 m) were taken from each of the trail plots following the methods used by Weiss (1999). Samples were spaced at 3.5 m – 3.75 m and 6.25 m – 6.5 m along each 10 m transect in order to be accurately relocated after treatment. Quadrat samples were offset approximately 10 cm away from the dirt/trail and vegetation edge to account for unevenness along trail edges within the transects (Figure 4). A total

of 64 samples were collected prior to string-cutting, and 64 samples were collected one year after string-cutting.

Trail segments selected in serpentine grasslands had similar slopes (less than 10% grade), topography (less than 200 m elevation change), and level of usage (less than 5,000 users per year). The assignment of the treatments and control to sample plots within the trail segments followed a random block design. Sample plot assignments were interspersed in adjacent trail segments.

Data Collection

Based on weather fluctuations and other seasonal factors, the window of data collection was from April through May in order to capture a majority of blooming species. Data were collected from April 29, 2004 to May 8, 2004 in the first year prior to the cutting treatments. In the second year, data were collected from May 1, 2005 to May 7, 2005. A timeline of photographs was compiled portraying growth changes from May 2004 to April 2005 for the Clarkia Trail (Figure 5), for the Serpentine Trail (Figure 6), and for the Service Road (Figure 7).

Prior to data collection, vegetation maps, topographic maps, and USGS soil maps were examined to select serpentine grasslands and trail segments. In addition, each site was visited by me and Dr. Stuart Weiss to ground truth map information and field observations. Each plot along the trail segment was assigned an identification number and plotted on graph paper. The treatment and control plots were randomly assigned within each trail block and marked with colored, wooden stakes.

Before data collection in 2004 (first year), I identified and roughly surveyed plants found in the study area with the aid of Ken Himes, California Native Plant Society member and leader of Edgewood Park's yellow-star thistle restoration program. This information aided in assembling a data sheet (Figure 8) and increased confidence in accurately identifying plants at the species level.

Primary data were collected on all plots in trail segments in April 2004 before string-cutting commenced. Total percent cover was estimated using sample quadrats (0.25 m) that were divided into 25 squares by transparent fishing line. Thus, a plant that occupied one square in the quadrat represented 4% of the total plant cover. For most of the quadrat samples, total percentage of plant cover exceeded 100 percent. A category was also included to record bare ground.

Two quadrat samples of 0.25 m were taken from each treatment and each control plot and then averaged (Weiss, 1999). The parameters measured were the total number of plant species and the relative percent cover for each plant species for all trail segments (Walck et al., 1999; Kitazawa and Ohsawa, 2002; Buckley et al. 2003).

Data Analysis

Plant species abundance and richness for native, serpentine, and non-native plant species was compared using repeated measures ANOVA (Analysis of Variance) for treatment and control segments, for the trail block sites, and for time with 2 years of data (Walck et al., 2002). Mean percent cover for each plant species was calculated by averaging percent cover of the two quadrat samples for each plot. This model tested the

effects and interactions on native, serpentine, and non-native plant abundance and species richness for time (2004 data and 2005 data), for the treatments (May, June, July, and control), and for the blocks (Clarkia Trl., Serpentine Trl., and Service Rd.). To summarize, this model fits into these equations: Time + Error → Native; Time + Error → Serpentine; Time + Error → Non-Native; Time*Treatment + Error → Native; Time*Treatment + Error → Serpentine; Time*Treatment + Error → Non-Native; Time*Block + Error → Native; Time*Block + Error → Serpentine; and, Time*Block + Error → Non-Native.

A block effect was also tested using Random Block ANOVA tests for the three, trail blocks. This model determined whether or not there were inherent differences between the trail sites independent of the treatments.

Serpentine plant species included in this analysis are not necessarily restricted to pure serpentine soil, but are frequently associated and found in serpentine habitats as listed by Toni Corelli (2002) and Hickman (1993). These species include harvest brodiaea (*Brodiaea coronaria*), morning glory (*Calystegia subacaulis*), farewell-to-spring (*Clarkia rubicunda*), squirrel-tail grass (*Elymus multisetus*), goldfields (*Lasthenia californica*), tidy-tips (*Layia platyglossa*), lomatium (*Lomatium macrocarpum*), California plantain (*Plantago erecta*), and fringed checkerbloom (*Sidalcea diploscypha*).

Abundance of indicator species was tested to determine whether they were affected by the treatments. Indicator species were singled out for one-way ANOVA tests. The native indicator species include tidy-tips (*Layia platyglossa*), California plantain (*Plantago erecta*), owl's-clover (*Castilleja densiflora* ssp. *densiflora*), purple needlegrass

(*Nassella pulchra*), slender tarweed (*Madia gracilis*), and dandelion (*Agoseris grandiflora*). Tidy-tips (*Layia platyglossa*), California plantain (*Plantago erecta*), and owl's-clover (*Castilleja densiflora* ssp. *densiflora*) are host plants for the federally listed the Bay Checkerspot Butterfly (*Euphydryas editha bayensis*). Purple needlegrass (*Nassella pulchra*) was chosen because it is the state grass of California. Slender tarweed (*Madia gracilis*) and dandelion (*Agoseris grandiflora*) are frequently found in disturbed trail sites at Edgewood Park and are examples of weedy, native plant species.

Five non-native plant species were chosen as indicators primarily because they were the most abundant and commonly found along trail plots. Non-native plant species chosen as indicators include Italian ryegrass (*Lolium multiflorum*), soft chess (*Bromus hordeaceus*), burclover (*Medicago polymorpha*), slender wild oat (*Avena barbata*), and six week fescue (*Vulpia bromoides*).

RESULTS

Percent cover of all plant species showed variability and patchiness (Table 1). The critical value of $p \leq 0.05$ determined significance (Table 2-19). All of the ANOVA tests performed for indicator species satisfied the assumption of similar variances. Mean percent cover was calculated by averaging percent cover for each plant species in the 32 treatment and control plots for each year.

Abundance

Native plant species. Between year 1 and year 2, mean native plant cover for all plots including the control tended to increase from 33% to 38.5%, which was significant with $p = 0.002$ (Figure 9). After comparing the May, June, July, and the control treatments, there was no significant difference in native abundance from year 1 to year 2, but an increasing trend was observed in plots string-cut in May (Figure 10). Within the trail blocks, there was no significant difference between year 1 and year 2 in mean percent cover of native plant species (Table 2).

Serpentine plant species. From year 1 to year 2, mean percent cover of serpentine plant species did not significantly change (Figure 9). No significant difference was detected when tested by treatment (Figure 11). Also, there was no significant difference found within the trail blocks (Table 3).

Non-native plant species. Mean non-native plant cover did not change significantly from year 1 to year 2 (Figure 9). For the treatments, non-native plant cover

tended to increase in year 2 for the June and the control plots and to decrease in the May and July plots, but neither of these trends were significantly different with $p = 0.107$. However, in the plots string-cut in May, mean percent cover showed a strong decreasing trend in year 2 (Figure 12). There were no significant differences within the trail blocks from year 1 to year 2 using repeated measures ANOVA where $p = 0.078$ (Table 4).

Species Richness

Native plant species. A total of 32 native plant species was identified across all plots. Mean native species richness significantly increased from year 1 to year 2 with $p < 0.001$, (Figure 13). When tested by treatment, there was a significant difference in the number of native plant species with $p = 0.005$ (Table 5). Specifically, in plots cut in May, native species richness significantly increased (Figure 14). From year 1 to year 2, significant differences were found within the trail blocks with $p = 0.005$, which was most notable in the Service Rd. plots (Figure 15).

Serpentine plant species. Of the 32 native plant species sampled, 8 are classified as serpentine species. An increasing trend was noticed for these plants from year 1 to year 2, with $p = 0.072$ (Figure 13). For the treatments, serpentine plant species richness also tended to increase with $p = 0.084$, but there were no significant changes detected from year 1 to year 2 within treatment plots (Figure 16). Within the trail blocks, no significant changes were noted from year 1 to year 2 (Table 6).

Non-native plant species. A total of 24 non-native plant species was identified. Between year 1 and year 2, mean non-native species richness increased significantly with

$p < 0.001$ (Figure 13). Amongst the treatments, all displayed increasing trends in the mean number of non-native species with $p = 0.096$ (Table 7). However, the plots string-cut in June showed a significant increase in non-native plant species richness in year 2 (Figure 17). There were no significant differences between year 1 and year 2 within the trail blocked sites.

Native Indicator Species

Six native species were selected as indicator species primarily because they were commonly observed in most plots, and three are host species for the Bay checkerspot butterfly. The native indicator species include tidy-tips (*Layia platyglossa*), California plantain (*Plantago erecta*), owl's-clover (*Castilleja densiflora* ssp. *densiflora*), purple needlegrass (*Nassella pulchra*), slender tarweed (*Madia gracilis*), and dandelion (*Agoseris grandiflora*) (Figure 18).

Tidy-tips (*Layia platyglossa*). Mean percent cover of tidy-tips dropped slightly in year 2, but this decrease was not significant (Table 8). Trends found in the treatments were not statistically significant with $p = 0.171$ (Figure 19). Within the trail segments, there was a significant change in mean percent cover of tidy-tips with $p = 0.045$, especially in the Service Rd. plots (Figure 20).

California plantain (*Plantago erecta*). From year 1 to year 2, mean percent cover of *Plantago* remained at 5% (Figure 18). The effect of treatment was not significant (Figure 21). Also, there were no significant differences detected between year 1 and year 2 within the trail blocks (Table 9).

Owl's-clover (*Castilleja densiflora* ssp. *densiflora*). Mean percent cover of owl's-clover showed an increasing trend, but this was not significant with $p = 0.072$ (Figure 18). No significant differences in treatment were detected (Figure 22). The differences between year 1 and year 2 within the trail blocks were not significant with $p = 0.066$ (Table 10).

Purple needlegrass (*Nassella pulchra*). Mean percent cover of *Nassella* increased from 3.5% in year 1 to 6% in year 2 (Figure 18), and this result was significant with $p = 0.023$ (Table 11). No significant differences in mean percent cover of purple needlegrass were found between the treatment and the control plots (Figure 23). Differences amongst the trail blocks were not due to changes between year 1 and year 2.

Slender tarweed (*Madia gracilis*). Mean percent cover of tarweed increased from 0.5% in year 1 to 1.5% in year 2 (Figure 18). This increase was significant with $p = 0.003$ (Table 12). There was no significant difference between any of the treatment and control plots (Figure 24). There was a significant increase in mean percent cover of tarweed between year 1 and year 2 within the trail blocks with $p = 0.007$, especially on the Clarkia Trl. (Figure 25).

Dandelion (*Agoseris grandiflora*). From year 1 to year 2, mean percent cover of dandelion increased significantly from 0.5% to almost 2% with $p = 0.001$ (Figure 18). The treatments all increased in year 2, but no significant difference was detected between the treatment and control plots (Table 13). However, a significant increase between year 1 and year 2 was detected in plots string-cut in June (Figure 26). Differences from year 1

to year 2 between the trail block sites were not significant, but within the Clarkia Trl. plots, a significant increase in mean percent cover of dandelion was found (Figure 27).

Non-Native Indicator Species

Five non-native plant species were chosen as indicators primarily because they were the most abundant and commonly found along trail plots (Figure 28). They included Italian ryegrass (*Lolium multiflorum*), soft chess (*Bromus hordeaceus*), burclover (*Medicago polymorpha*), slender wild oat (*Avena barbata*), and six week fescue (*Vulpia bromoides*).

Italian ryegrass (*Lolium multiflorum*). From year 1 to year 2, mean percent cover of *Lolium* tended to decrease from 13% to 11%, but not significantly with $p = 0.095$ (Figure 28). For the treatments, mean percent cover decreased in the cut plots (May, June, and July) and increased in control plots, which was significant with $p = 0.037$ (Table 14). The trail plots string-cut in May displayed a significant decrease in mean percent cover of Italian ryegrass in the second year (Figure 29). Changes from year 1 to year 2 within the trail blocks were not significant.

Soft chess (*Bromus hordeaceus*). There was no significant change in mean percent cover from year 1 to year 2 with $p = 0.138$ (Figure 28). For the treatment and control plots, no significant effect was found (Figure 30). There was a significant difference found within the trail blocks from year 1 to year 2 with $p = 0.001$ (Table 15). Notably, the Serpentine Trl. plots displayed a significant decrease in mean percent cover of soft chess in year 2 (Figure 31).

Burclover (*Medicago polymorpha*). Mean percent cover remained at 8% for burclover (Figure 28), and no significant change was found (Table 16). The varied means of the treatment and control plots were not significantly different (Figure 32). No significant differences were found from year 1 to year 2 within the trail blocks.

Slender wild oat (*Avena barbata*). From year 1 to year 2, mean percent cover increased from 4% to 5% which was significant with $p = 0.031$ (Figure 28). Even though increasing densities of wild oat were found in all plots (Figure 33), there was no significant differences detected (Table 17). From year 1 to year 2, differences within the blocks were not significantly different, but just barely, with $p = 0.058$.

Six week fescue (*Vulpia bromoides*). Mean percent cover of fescue increased from 1% in year 1 to 2% in year 2 which was significant with $p = 0.025$ (Figure 28). The control plots tended to display more of an increase than the treatments (Figure 34), but no significant difference was found (Table 18). Differences within the trail blocks were not due to string-cutting between year 1 and year 2 (Figure 35).

Block Effect

The Clarkia Trail, Serpentine Trail, and Service Road displayed similar, serpentine soil characteristics based on plant growth patterns; however, a block effect among trail segments was important in the analysis to explain sources of variability. A random block effect was detected in a majority of the statistical tests in the aforementioned results (Table 19). Significant differences between the trail blocks were found independent of the effects of time and treatment.

DISCUSSION

Abundance

Native plant species abundance increased significantly while non-native plant species abundance remained relatively the same. It should be noted that mean percent cover of bare ground also tended to decrease in all plots (Table 1). These trends may be indicative of differences in rainfall between the two years which affects plant growth and abundance. In year 1, the total rainfall was 64.7 cm, but in year 2, the total was 100.1 cm which is above the average rainfall for this area (Figure 36). The length of the rainy season was longer for year 2 as compared to year 1. Also, the amount of rain received during the spring was higher in year 2 than in year 1 with 30.3 cm of rain falling between March and June while only 4.1 cm was recorded in year 1 during the same months.

Plant growth is dependent on numerous factors including moisture, sunlight energy, and mineral nutrients. The plant species at Edgewood Park are adapted to summer droughts similar to the Mediterranean climate of Northern California. The amount of rainfall does vary from year to year as does the length of the rainy season. Generally, heavy and early rainfall benefits annual non-native plant species (Bartolome and Gemmill, 1981). Rainfall later in the spring season triggers perennial plant growth, which are mostly natives (Chiariello, 1989). The amount of sunlight energy received by plant species in the study site was relatively homogenous throughout the trail plots. It should be noted that trails plots have been cut annually and/or biannually for approximately the last 20 years.

In plots that were string-cut in May, an increasing trend for native abundance and a decreasing trend for non-native abundance were of interest based on the objectives of this study. Because non-native, annual grasses typically emerge earlier than native perennials, the early-season treatment in May is probably stunting and decreasing the amounts of seeds produced by these plants. Another explanation is that cutting early removes the thatch layer allowing late emerging native species to thrive.

Mineral nutrients in serpentine soil do limit plant growth due to low levels of nitrogen, calcium, and phosphorous. At Edgewood, there is evidence of nitrogen deposition into serpentine grasslands caused by vehicle emissions from the adjacent Interstate 280 (Weiss, 1996). As serpentine soils absorb more nitrogen, grasslands at Edgewood become more capable of supporting plant growth by non-native species and potentially at a faster rate of dispersal. Even though non-native abundance did not significantly increase in year 2, the relative abundance of non-natives was much higher than natives suggesting that nitrogen deposition has already allowed non-natives to dominate.

Species Richness

The average number of native plant species counted in all plots increased significantly from year 1 to year 2, but the total number of native species found in all plots only decreased by one species in the second year. This finding could be attributed to the fact that trail sites have become adapted to annual mowing and that string-cutting as a disturbance is maintaining the status quo of native species richness. Huhta (2001)

also found that mowing, like string-cutting, preserves plant species composition in grasslands. Similar to the reasoning for native abundance increasing, string-cutting in May positively affects native species richness.

Non-native plant species richness significantly increased in the second year. This could be due to low competitiveness of annual, non-native species and removal of thatch layer. According to Frankel (1977), the removal of litter accumulation decreases competition allowing annual non-native plant species to establish in disturbed sites. Buckley et al. (2003) note that transportation routes can act as corridors for the spread of non-native species. The trails in Edgewood Park may also function this way, but in order to test this hypothesis, this study would have needed to include data collection from non-trail sites.

Increases in non-native species richness especially in plots string-cut in June can also be attributed to early growth by annual plant species, thus giving annuals an advantage over native perennials in setting seeds earlier (Gibson, Seastedt, & Briggs, 1993). Typically, annuals emerge before perennials, because perennials have adapted to grow later in the spring due to drought summers (Chiariello, 1989).

Native Indicator Species

No significant changes were noted from year 1 to year 2 for the following native plant indicator species: Tidy-tips (*Layia platyglossa*), California plantain (*Plantago erecta*), and owl's-clover (*Castilleja densiflora* ssp. *densiflora*). Of these species, all

germinate annually. These native species are not being negatively affected by string-cutting when cut between May and July.

According to the Recovery Plan for Serpentine Soil Species of the San Francisco Bay Area, the survival of the endangered bay checkerspot butterfly (*Euphydryas editha bayensis*) depends on sufficient serpentine grassland habitat for its host plants. Plantain (*Plantago erecta*) is the primary larval host plant. Secondary host species for larvae are owl's clovers (*Castilleja densiflora* ssp. *densiflora* and *Castilleja exserta*). Finally, tidy-tips (*Layia platyglossa*) is one of the plant species commonly visited by adult butterflies for nectar. Results of this study indicate that string-cutting does not significantly affect these bay checkerspot butterfly host plant species; however, with 3 to 5 years worth of data, this may change and prove to be significant.

Significant increases from year 1 to year 2 were detected for purple needlegrass (*Nassella pulchra*), slender tarweed (*Madia gracilis*), and dandelion (*Agoseris grandiflora*). Even though tarweed and dandelion are native, these species are considered weedy, poor competitors, and commonly found in disturbed sites. These increases in abundance may be due to differences in rainfall from year 1 to year 2 (Figure 36).

Non-Native Indicator Species

All of the non-native indicator species are annual plants. Italian ryegrass (*Lolium multiflorum*), soft chess (*Bromus hordeaceus*), and burclover (*Medicago polymorpha*) showed decreasing trends in abundance from Year 1 to Year 2. Slender wild oat (*Avena*

barbata) and six week fescue (*Vulpia bromoides*) increased in abundance for the second year.

For Italian ryegrass (*Lolium multiflorum*), decreases in abundance were found in plots string-cut in May. In the control plots, abundance tended to increase in the second year. This finding could be attributed to the same theory underlying restorative mowing in grasslands that mimics a similar disturbance to grazing, which has been found to be beneficial in maintaining native grasslands and controlling invasive non-native grasses (Weiss, 2002). Burclover (*Medicago polymorpha*) is commonly found along trails and other open, disturbed areas. Results for burclover for the treatments and control were mixed with subtle increases and decreases with no apparent patterns nor trends.

Increases in abundance of wild oat (*Avena barbata*) and fescue (*Vulpia bromoides*) were found across all the treatment and control plots. This result suggests that both of these annual grasses are spreading, but not due to string-cutting. In particular, fescue grass was found to be significantly higher in abundance in the second year, even though mean percent cover around 1% to 2% for this species was low relative to the other non-native indicator species. A possible explanation for these increases could be associated with increases in rainfall and length of the rainy season in year 2 (Figure 36).

Block Effect

A majority of the variation found in the data and results is attributed to inherent differences between trail sites. There were significant random block effects in a majority

of statistical tests performed. Subtle differences between the Clarkia Trl., the Serpentine Trl., and the Service Rd. in soil depth, amount of serpentine material, soil nutrients, and other unmeasured attributes could account for the differences. Following the viewpoint of Gleason (1926) and his individualistic concept regarding plant association, the differences between trail sites is explained by the fact that “no two areas of vegetation are exactly similar, either in species, the relative number of individuals of each and their spatial arrangement, and the smaller the areas to be compared the greater proportionately are the differences between them (McIntosh, 1975).”

The Clarkia Trl. is located on a south-facing slope, whereas the Serpentine Trl. and the Service Rd. sites are located on north-facing slopes. South-facing slopes are warmer and usually drier than north-facing slopes which could affect abundance and species richness. Another explanation could be the proximity to areas that are highly invaded or extremely disturbed. For example, the Serpentine Trl. site is located within 100 feet of a P.G. & E. main tower. In general, no clear patterns were detected for any of the trail sites.

Limitations

A potential source of bias might have occurred during data collection, such as double-counting plants or estimation errors in percent cover. However, all data were collected by one researcher, and this type of random error did not skew results.

This study was limited to 2 seasons of data collection due to time and financial constraints; however, it is recommended that at least 3 to 5 year's of data be collected to

provide more insight into any changes or trends occurring for native and non-native plant species and their abundance and richness. Also, taking samples perpendicular to trail edges could shed a more accurate perspective with regards to the change and/or potential spread laterally of invasive plants due to disturbance.

To reduce some variability, a future study design should include more repetitions in each block and possibly 3 quadrat samples per plot as compared to 2 per plot. If the study were to be repeated, one should assign treatments after collecting baseline data in order to reduce variability.

CONCLUSIONS AND RECOMMENDATIONS

When vegetation is cut, removed, or severely disturbed by humans, a natural, successional process is initiated with the potential for invasion by non-native plants that are highly competitive, pioneer species. However, the timing of the cut or disturbance can be manipulated to match the growth and reproduction of annual plants, such that seed spread is minimized. Based on the results and conclusions of this study, string-cutting early in spring, approximately May, did positively affect native plant species abundance and species richness and did reduce non-native plant species abundance.

Future Management

String-cutting trails in early to mid-spring maintains the relative abundance of non-native plant species and is the preferred management practice as compared to cutting in the fall which has little to no effect on non-native plant species abundance (Weiss, 2002). I also recommend that trails in serpentine grasslands be cut in a strategic manner according to the approximate timing of the grasses turning brown, which varies by location based on slope, amount of rainfall, and other biotic factors. By definition, this approach to trail maintenance follows the adaptive management philosophy. When string-cutting trails, it is imperative that the equipment is properly cleaned after each usage so that invasive plant seeds are not spread from trail to trail.

It should be noted that regardless of refining the timing of cutting techniques, it is nearly impossible to match cutting with seed setting for all native and non-native plant

species. The target time to cut non-native plant species before setting seed is variable. For example, to reduce cover of Italian ryegrass (*Lolium multiflorum*), string-cutting should occur earlier in the Spring around April or May (Weiss, 2002). On the other hand, to reduce cover of yellow-star thistle (*Centaurea solstitialis*), string-cutting and/or mowing should be conducted later in the Spring usually in July (Thomsen, 1996). Hence, one should identify the non-native species that are highly invasive, their respective growth rates, and when is the most effective time to cut to reduce abundance. Most likely, the timing and growth rates will not match precisely and tradeoffs need to be considered.

For the trails located in serpentine grasslands of Edgewood Park, I recommend that string-cutting begin early in spring during May, in which the exact date will vary based on adaptive management that factors in annual precipitation. Invasive non-native grasses, such as Italian ryegrass (*Lolium multiflorum*), should be cut as soon as the plant begins to turn brown to minimize seed dispersal and to reduce cover, which is in May. In addition to reducing non-native invasive plant species abundance, string-cutting in May favors native species richness.

Future Studies

To increase reliability and validity of results and conclusions, this study should be continued for at least another 3 to 5 years. In addition to increasing the length of this study, data should also be collected in transects perpendicular to the trails to further understand the spread of non-native plants into less frequently disturbed areas.

Based on previous studies (Weiss, 1999; Weiss 2002), it is prudent to explore further research and consultation about vegetation management specific to issues surrounding the effect of nitrogen deposition in serpentine grasslands at Edgewood Park.

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APPENDIX A: Tables

Table 1. List of plant species. Species are sorted in descending order of abundance with mean and standard deviation for Year 1, before treatment, and for Year 2, after treatment. Total number of native species = 32. Total number of serpentine species = 8 in bold. Total number of non-native species = 24.

Species List	Mean Year 1	Standard Deviation	Mean Year 2	Standard Deviation
<i>Lolium multiflorum</i>	13.36	9.59	11.39	10.324
Bare soil	13.06	10.672	11.58	10.768
<i>Bromus hordeaceus</i>	10.83	5.481	10.14	5.264
<i>Layia platyglossa</i>	8.97	7.999	7.50	9.323
<i>Medicago polymorpha</i>	8.33	6.79	7.75	6.599
<i>Plantago erecta</i>	4.66	4.946	4.80	5.534
<i>Avena barbata</i>	4.09	5.108	5.11	5.952
<i>Hordeum murinum</i> ssp. <i>leporinum</i>	4.00	4.866	3.14	3.666
<i>Nassella pulchra</i>	3.55	4.343	5.78	5.439
<i>Hemizonia congesta</i> ssp. <i>luzulifolia</i>	2.63	2.59	2.02	2.026
<i>Castilleja densiflora</i> ssp. <i>densiflora</i>	2.47	2.609	3.13	3.15
<i>Trifolium campestre</i>	1.91	2.977	2.53	3.485
<i>Brachypodium distachyon</i>	1.80	3.656	2.02	2.291
<i>Bromus madritensis</i> ssp. <i>rubens</i>	1.72	3.925	0.84	2.026
<i>Bromus diandrus</i>	1.50	3.665	1.47	2.331
<i>Achillea millefolium</i>	1.36	3.261	1.39	3.2
<i>Lasthenia californica</i>	1.27	2.189	2.66	3.899
<i>Trifolium hirtum</i>	1.19	2.782	1.95	2.315
<i>Castilleja exserta</i> ssp. <i>exserta</i>	1.17	2.33	1.59	3.135
<i>Geranium molle</i>	1.14	2.38	0.89	1.354
<i>Lotus corniculatus</i>	1.09	1.907	0.78	1.177
<i>Lotus purshianus</i> ssp. <i>purshianus</i>	1.03	2.508	0.59	1.16
<i>Centaurea solstitialis</i>	0.98	2.883	1.41	3.107
<i>Vulpia bromoides</i>	0.84	1.181	1.78	2.144
<i>Chlorogalum pomeridianum</i> ssp. <i>pomeridianum</i>	0.81	1.655	0.39	1.014
<i>Eschscholzia californica</i>	0.64	1.471	0.38	0.813
<i>Clarkia rubicunda</i>	0.55	1.138	0.64	1.152
<i>Elymus multisetus</i>	0.53	1.344	1.39	2.271
<i>Madia gracilis</i>	0.41	0.946	1.52	2.51
<i>Bromus madritensis</i> ssp. <i>madritensis</i>	0.41	1.26	0.14	0.386
<i>Lotus humistratus</i>	0.38	1.27	0.47	1.436
<i>Lomatium macrocarpum</i>	0.38	2.121	0.45	2.153
<i>Agoseris grandiflora</i>	0.36	1.018	1.67	2.002

Table 1. List of plant species continued.

Species List	Mean Year 1	Standard Deviation	Mean Year 2	Standard Deviation
Sidalcea diploscypha	0.36	0.854	0.28	0.608
<i>Calochortus luteus</i>	0.31	1.083	0.16	0.369
<i>Hordeum marinum</i> ssp. <i>gussoneanum</i>	0.31	1.141	0.13	0.359
<i>Erodium botrys</i>	0.28	0.842	0.16	0.515
<i>Clarkia purpurea</i> ssp. <i>quadrivulnera</i>	0.27	0.568	0.64	1.159
<i>Danthonia californica</i>	0.22	0.74	0.13	0.336
<i>Micropus californicus</i> var. <i>californicus</i>	0.19	0.727	0.25	0.458
<i>Triteleia laxa</i>	0.14	0.462	0.19	0.749
<i>Linum bienne</i>	0.14	0.557	0.17	0.643
<i>Bellardia trixago</i>	0.14	0.795	0.09	0.53
<i>Lactuca saligna</i>	0.13	0.508	0.17	0.643
<i>Brodiaea elegans</i> ssp. <i>elegans</i>	0.11	0.535	0.09	0.39
<i>Vicia sativa</i> ssp. <i>sativa</i>	0.09	0.322	0.28	0.803
<i>Epilobium ciliatum</i> ssp. <i>ciliatum</i>	0.09	0.369	0.09	0.322
<i>Brodiaea coronaria</i>	0.08	0.314	0	0
<i>Achyraea mollis</i>	0.06	0.246	0.33	1.767
<i>Avena fatua</i>	0.05	0.265	0.03	0.177
<i>Silene gallica</i>	0.03	0.177	0.02	0.088
<i>Calystegia subacaulis</i>	0.03	0.177	0	0
<i>Gastroidium ventricosum</i>	0.03	0.177	0	0
<i>Dichelostemma capitatum</i> ssp. <i>capitatum</i>	0.02	0.088	0.05	0.148
<i>Lupinus bicolor</i>	0.02	0.088	0.03	0.177
<i>Microseris douglasii</i> ssp. <i>douglasii</i>	0.02	0.088	0	0
<i>Delphinium variegatum</i> ssp. <i>variegatum</i>	0	0	0.02	0.088

Table 2. List of Analysis of Variance results for native abundance. Bold p-values are significant using $p \leq 0.05$. N = 64.

Native Abundance

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	12.28	1,20	0.002
Time * Treatment	1.69	3,20	0.202
May	2.94	1,14	0.079
Time * Block	0.05	2,20	0.948

Table 3. List of Analysis of Variance results for serpentine species abundance. Bold p-values are significant using $p \leq 0.05$. N = 64.

Serpentine Abundance

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	0.47	1,20	0.500
Time * Treatment	0.39	3,20	0.756
Time * Block	0.82	2,20	0.456

Table 4. List of Analysis of Variance results for non-native abundance. Bold p-values are significant using $p \leq 0.05$. N = 64.

Non-Native Abundance

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	1.14	1,20	0.299
Time * Treatment	2.31	3,20	0.107
May	3.57	1,14	0.052
Time * Block	2.90	2,20	0.078

Table 5. List of Analysis of Variance results for native species richness. Bold p-values are significant using $p \leq 0.05$. N = 64.

Native Species Richness

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	17.66	1,20	0.000
Time * Treatment	5.78	3,20	0.005
May	5.70	1,14	0.032
Time * Block	5.17	2,20	0.005
Service Rd.	11.03	1,22	0.003

Table 6. List of Analysis of Variance results for serpentine species richness. Bold p-values are significant using $p \leq 0.05$. N = 64.

Serpentine Species Richness

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	3.61	1,20	0.072
Time * Treatment	2.56	3,20	0.084
May	2.02	1,14	0.177
Time * Block	1.82	2,20	0.187

Table 7. List of Analysis of Variance results for non-native species richness. Bold p-values are significant using $p \leq 0.05$. N = 64.

Non-Native Species Richness

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	34.53	1,20	0.000
Time * Treatment	2.42	3,20	0.096
June	5.52	1,14	0.034
Time * Block	0.08	2,20	0.924

Table 8. List of Analysis of Variance results for tidy-tips. Bold p-values are significant using $p \leq 0.05$. N = 64.

Tidy-tips (*Layia platyglossa*)

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	1.52	1,20	0.232
Time * Treatment	1.85	3,20	0.171
June	2.42	1,14	0.142
Time * Block	3.65	2,20	0.045
Service Rd.	7.78	1,22	0.011

Table 9. List of Analysis of Variance results for plantain. Bold p-values are significant using $p \leq 0.05$. N = 64.

CA plantain (*Plantago erecta*)

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	0.01	1,20	0.941
Time * Treatment	0.10	3,20	0.961
Time * Block	0.17	2,20	0.847

Table 10. List of Analysis of Variance results for owl's-clover. Bold p-values are significant using $p \leq 0.05$. N = 64.

Owl's-clover (*Castilleja densiflora*)

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	3.62	1,20	0.072
Time * Treatment	0.88	3,20	0.468
Control	1.04	1,14	0.325
Time * Block	3.13	2,20	0.066
Serpentine Trl.	1.51	1,14	0.240

Table 11. List of Analysis of Variance results for purple needlegrass. Bold p-values are significant using $p \leq 0.05$. N = 64.

Purple needlegrass (*Nassella pulchra*)

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	6.03	1,20	0.023
Time * Treatment	0.76	3,20	0.529
May	2.59	1,14	0.130
July	2.08	1,14	0.171
Time * Block	0.61	2,20	0.551
Clarkia Trl.	1.91	1,22	0.181

Table 12. List of Analysis of Variance results for tarweed. Bold p-values are significant using $p \leq 0.05$. N = 64.

Slender tarweed (*Madia gracilis*)

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	11.61	1,20	0.003
Time * Treatment	0.07	3,20	0.975
Control	1.82	1,14	0.199
Time * Block	6.54	2,20	0.007
Clarkia Trl.	6.42	1,22	0.019

Table 13. List of Analysis of Variance results for dandelion. Bold p-values are significant using $p \leq 0.05$. N = 64.

Dandelion (*Agoseris grandiflora*)

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	14.55	1,20	0.001
Time * Treatment	0.67	3,20	0.580
June	5.72	1,14	0.031
May	3.85	1,14	0.070
Time * Block	1.41	2,20	0.269
Clarkia Trl.	6.69	1,22	0.017
Service Rd.	3.93	1,22	0.060
Serpentine Trl.	3.58	1,14	0.079

Table 14. List of Analysis of Variance results for Italian ryegrass. Bold p-values are significant using $p \leq 0.05$. N = 64.

Italian ryegrass (*Lolium multiflorum*)

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	3.07	1,20	0.095
Time * Treatment	3.44	3,20	0.037
May	4.87	1,14	0.045
June	2.02	1,14	0.174
Control	1.96	1,14	0.183
Time * Block	0.48	2,20	0.627

Table 15. List of Analysis of Variance results for soft chess. Bold p-values are significant using $p \leq 0.05$. N = 64.

Soft chess (*Bromus hordeaceus*)

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	2.39	1,20	0.138
Time * Treatment	1.17	3,20	0.348
Time * Block	9.72	2,20	0.001
Serpentine Trl.	9.01	1,14	0.010
Clarkia Trl.	3.00	1,22	0.097

Table 16. List of Analysis of Variance results for burclover. Bold p-values are significant using $p \leq 0.05$. N = 64.

Burclover (*Medicago polymorpha*)

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	0.42	1,20	0.525
Time * Treatment	1.02	3,20	0.405
Control	1.54	1,14	0.236
Time * Block	0.13	2,20	0.876

Table 17. List of Analysis of Variance results for wild oat. Bold p-values are significant using $p \leq 0.05$. N = 64.

Slender wild oat (*Avena barbata*)

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	5.36	1,20	0.031
Time * Treatment	0.47	3,20	0.708
Time * Block	3.29	2,20	0.058
Serpentine Trl.	1.46	1,14	0.247

Table 18. List of Analysis of Variance results for fescue. Bold p-values are significant using $p \leq 0.05$. N = 64.

Six week fescue (*Vulpia bromoides*)

<u>Statistical Model</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Time	5.85	1,20	0.025
Time * Treatment	0.41	3,20	0.752
Control	3.19	1,14	0.096
Time * Block	0.65	2,20	0.533
Service Rd.	4.11	1,22	0.096
Clarkia Trl.	2.92	1,22	0.103

Table 19. List of Random Block Analysis of Variance results for all tests. Bold p-values are significant using $p \leq 0.05$. N = 64.

<u>Statistical Test Models</u>	<u>F Statistic</u>	<u>d.f.</u>	<u>p-value</u>
Native Abundance	1.81	2,58	0.173
Non-Native Abundance	7.24	2,58	0.002
Native Species Richness	9.10	2,58	0.000
Non-Native Species Richness	11.46	2,58	0.000
Tidy-tips (<i>Layia platyglossa</i>)	6.50	2,58	0.003
CA plantain (<i>Plantago erecta</i>)	30.72	2,58	0.000
Owl's-clover (<i>Castilleja densiflora</i>)	5.15	2,58	0.009
Purple needlegrass (<i>Nassella pulchra</i>)	8.47	2,58	0.001
Slender tarweed (<i>Madia gracilis</i>)	9.82	2,58	0.000
Dandelion (<i>Agoseris grandiflora</i>)	6.51	2,58	0.003
Italian ryegrass (<i>Lolium multiflorum</i>)	11.59	2,58	0.000
Soft chess (<i>Bromus hordeaceus</i>)	3.49	2,58	0.037
Burclover (<i>Medicago polymorpha</i>)	5.94	2,58	0.005
Slender wild oat (<i>Avena barbata</i>)	27.15	2,58	0.000
Six-week fescue (<i>Vulpia bromoides</i>)	0.45	2,58	0.533

APPENDIX B: Figures

Figure 2. Edgewood County Park and Natural Preserve Trail Map.

Figure 3. Trailside Sample Diagram.

Figure 4. Example of quadrat placement offset from trail edge.

Figure 5. Clarkia Trail located in Edgewood County Park and Natural Preserve.

MAY 2004

OCTOBER 2004

JANUARY 2005

APRIL 2005

Figure 6. Serpentine Trail located in Edgewood County Park and Natural Preserve.

MAY 2004

OCTOBER 2004

JANUARY 2005

APRIL 2005

Figure 7. Service Road located in Edgewood County Park and Natural Preserve.

MAY 2004

OCTOBER 2004

JANUARY 2005

APRIL 2005

Collector: _____

Plot I.D. and Sample #: _____

Date: _____

		1	2	3	4	5
Plant Species	Raw %	1-5%	6-15%	16-25%	26-50%	51-100%
Bare Soil						

Plant Species	Raw %	1 1-5%	2 6-15%	3 16-25%	4 26-51%	5 51-100%
Bare Soil						

Figure 9. Effect of time on mean percent cover of plant species. Gray shaded bars depict the mean for the first year sample, before treatment. White bars represent the mean for the second year sample, after treatment. Error bars depict $\pm$ standard deviation. N = 64.

Figure 10. Effect of treatment on mean percent cover of native plant species. Error bars depict $\pm$ standard deviation. N = 64.

Figure 11. Effect of treatment on mean percent cover of serpentine plant species. Error bars depict $\pm$ standard deviation. N = 64.

Figure 12. Effect of treatment on mean percent cover of non-native plant species. Error bars depict $\pm$ standard deviation. N = 64.

Figure 13. Effect of time on the mean number of plant species. Gray shaded bars depict the mean for the first year sample, before treatment. White bars represent the mean for the second year sample, after treatment. Error bars depict $\pm$ standard deviation. $N = 64$.

Figure 14. Effect of treatment on the mean number of native plant species. Error bars depict $\pm$ standard deviation. N = 64.

Figure 15. Effect of block by trail segment on the mean number of native plant species. Error bars depict $\pm$ standard deviation. N = 64.

Figure 16. Effect of treatment on the mean number of serpentine plant species. Error bars depict $\pm$ standard deviation. N = 64.

Figure 17. Effect of treatment on the mean number of non-native plant species. Error bars depict $\pm$ standard deviation. N = 64.

Figure 18. Mean percent cover of native indicator species for Year 1 and Year 2. Error bars depict $\pm$ standard deviation. N = 64.

Figure 19. Effect of treatment on mean percent cover of *Layia platyglossa*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 20. Effect of block by trail segment on mean percent cover of *Layia platyglossa*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 21. Effect of treatment on mean percent cover of *Plantago erecta*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 22. Effect of treatment on mean percent cover of *Castilleja densiflora*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 23. Effect of treatment on mean percent cover of *Nassella pulchra*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 24. Effect of treatment on mean percent cover of *Madia gracilis*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 25. Effect of block by trail segment on mean percent cover of *Madia gracilis*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 26. Effect of treatment on mean percent cover of *Agoseris grandiflora*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 27. Effect of block by trail segment on mean percent cover of *Agoseris grandiflora*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 28. Mean percent cover of non-native indicator species for Year 1 and Year 2. Error bars depict $\pm$ standard deviation. N = 64.

Figure 29. Effect of treatment on mean percent cover of *Lolium multiflorum*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 30. Effect of treatment on mean percent cover of *Bromus hordeaceus*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 31. Effect of block by trail segment on mean percent cover of *Bromus hordeaceus*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 32. Effect of treatment on mean percent cover of *Medicago polymorpha*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 33. Effect of treatment on mean percent cover of *Avena barbata*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 34. Effect of treatment on mean percent cover of *Vulpia bromoides*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 35. Effect of block by trail segment on mean percent cover of *Vulpia bromoides*. Error bars depict $\pm$ standard deviation. N = 64.

Figure 36. Total rainfall collected at Edgewood Park 2003 - 2005. The blue line with diamonds represents the total amount of rainfall (64.7 cm) from July 2003 to June 2004. The pink line with squares depicts the total rainfall (100.1 cm) from July 2004 to June 2005.