REESTABLISHING THE COMPETITIVE HIERARCHY IN AN INVADED CALIFORNIA GRASSLAND THROUGH THE PROCESS OF HABITAT RESTORATION FOLLOWING A PRESCRIBED BURN

A Thesis

Presented

to the Faculty of

California State University, Chico

In Partial Fulfillment

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Master of Science

in

Interdisciplinary Studies:

Biogeography

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by

Jason M. Mills

Fall 2015

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Land use practices over the past several hundred years have had a dramatic effect on many of California's ecosystems. As a result, native grasslands have become one of the state’s most threatened ecosystems. Close to 90% of the plants listed on California's Inventory of Rare and Endangered Species occur within grasslands. Native species can account for less than 1% of the composition of most California's grasslands today. Invasive species such as annual grasses (Poaceae) and *Centaurea solstitialis* ‘yellow star thistle’ (Asteraceae) from Europe were introduced to California and have rapidly overtaken exposed grasslands. Prescribed burns have demonstrated to be an effective management tool for reducing the density and seed bank of exotic grasses and forbs. Repeated burns have decreased *C. solstitialis* populations by as much as 99%; however, further research has shown that without continued management, *C. solstitialis* will reestablish in treated areas. I hypothesize that planting native grassland species will suppress the reestablishment of invasive species following a prescribed burn. Seeds of the perennial bunch grasses, *Stipa pulchra*, *Bromus carinatus*, and *Elymus glaucus*, along with *Grindelia camporum*, and *Madia elegans* were collected from the restoration area within the watershed of Big Chico Creek in the summer of 2012 and propagated in a greenhouse. Two separate fields, which were invaded with non-native annual grasses and *C. solstitialis* along Big Chico Creek, were burned in the fall of 2012. Three separate 4m x 7m blocks were established in each of the fields following the burn. Each native species treatment was planted into separated $1m^2$ plots and replicated four times within each of the blocks using systematic randomization. Planting of 1,152 native grass plugs, 216 *Grindelia* plugs, and 48 $1m^2$ direct seed treatments of mixed bunch grasses and *Madia elegans* was completed in the winter of 2012-13. The native species treatments were then monitored once a month over the course of fifteen months to determine their survival and composition. The establishment of native species through the process of restoration was found to be successful within each of the invaded grasslands. Out-planted perennial plug survival rates ranged from 38-97%, with percent cover values accounting for between 19-52% upon the completion of the experiment in May of 2014. In addition, the out-planted treatment plots were found to reduce the percent cover of *C. solstitialis* by as much as 28% as well as non-native annual grasses by 40%. Using the most effective perennial

species in grassland restoration efforts may aid in shifting the competitive advantage back to favor native taxa over the long term through mitigating the cycle of future invasion.

CHAPTER I

INTRODUCTION

Since the time of western colonization, humans have dramatically reshaped and altered the landscape to meet their needs. The environment of California is no exception and has been largely impacted by land use change over the past several hundred years. When the first settlers arrived with seeds to sow and livestock to feed, the fertile soils and robust grasses of California grasslands were ideal for both farming and ranching (Crampton 1974). Unfortunately, this came at the expense of the native grassland species, and resulted in ecological consequences. Grasslands are thought to have once made up 30% of California's landscape; today, few, if any, fully intact native grasslands remain (Lowry 2012), leaving grasslands as one of the state's most threatened ecosystems (Noss et al. 1995; Jantz et al. 2007). California grasslands have become fragmented through agriculture and development, and those which remain intact have become largely degraded (Stromberg & Griffin 1996; Jantz et al. 2007). Native species can account for as little as 1% of modern grasslands composition (Barry et al. 2012). Close to 90% of the plants listed on California's Inventory of Rare and Endangered Species occur within grasslands (Skinner & Pavlik 1994), and all but 4 of the 29 plants driven to extinction in California were found in grasslands (D'Antonio et al. 2007). Grassland habitats also currently support 50% of state listed vertebrate species and 48% of invertebrates (Jantz et al. 2007). It would be difficult to find an ecosystem in

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forbs. Repeated burns have decreased *C. solstitialis* populations by as much as 99%; however, further research has shown that without continued management, *C. solstitialis* will reestablish in treated areas. I hypothesize that planting native grassland species will suppress the reestablishment of invasive species following a prescribed burn. Seeds of the perennial bunch grasses, *Stipa pulchra*, *Bromus carinatus*, and *Elymus glaucus*, along with *Grindelia camporum*, and *Madia elegans* were collected from the restoration area within the watershed of Big Chico Creek in the summer of 2012 and propagated in a greenhouse. Two separate fields, which were invaded with non-native annual grasses and *C. solstitialis* along Big Chico Creek, were burned in the fall of 2012. Three separate 4m x 7m blocks were established in each of the fields following the burn. Each native species treatment was planted into separated 1m^2 plots and replicated four times within each of the blocks using systematic randomization. Planting of 1,152 native grass plugs, 216 *Grindelia* plugs, and 48 1m^2 direct seed treatments of mixed bunch grasses and *Madia elegans* was completed in the winter of 2012-13. The native species treatments were then monitored once a month over the course of fifteen months to determine their survival and composition. The establishment of native species through the process of restoration was found to be successful within each of the invaded grasslands. Out-planted perennial plug survival rates ranged from 38-97%, with percent cover values accounting for between 19-52% upon the completion of the experiment in May of 2014. In addition, the out-planted treatment plots were found to reduce the percent cover of *C. solstitialis* by as much as 28% as well as non-native annual grasses by 40%. Using the most effective perennial species in grassland restoration efforts may aid in shifting the competitive advantage back to favor native taxa over the long term through mitigating the cycle of future invasion.

California that has been more affected by disturbance than our grasslands (Corbin et al. 2007a), and they have in fact become the most altered ecosystem nationally (Jantz et al. 2007). Despite their economic, cultural, and ecological significance, grasslands have been largely overlooked in terms of conservation and policy throughout the west (Connor et al. 2001; Gelbard 2003; Jantz et al. 2007). Only in the past two decades have grasslands been considered for restoration (Keeler-Wolf et al. 2007) and only 4% of California grassland habitats are currently protected within designated reserves (Davis et al. 1998).

Land use practices such as agricultural and urban development account for the primary loss of grassland ecosystems, and have in turn led to their further decline by enabling introduced plant species to become established through disturbance (Randall 1996; Lonsdale 1999; Simberloff et al. 2005). When non-native species are introduced to a new environment, they are potentially freed from their natural reproductive constraints, competitors, and natural controls which allow some to become invasive (Schierenbeck 1995). Once non-native species become established, they can shift competitive hierarchies in their favor (Vitousek et al. 1996). Invasive species disrupt native ecosystem processes which alter their community structure and population dynamics (D'Antonio & Vitousek 1992). In order to address these issues, a deeper understanding of the biological mechanisms of competition and community invasibility must be obtained.

Here I focus on some of the most problematic invasive plants that are causing both economic and ecological impacts, not only in Butte County, but throughout California (DiTomaso 2000; DiTomaso et al. 2007; DiTomaso & Healy 2007; Schierenbeck et al. 2007). Non-native annual grasses and forbs such as *Centaurea solstitialis* (yellow star thistle) now dominate the fields once occupied by perennial native bunch grasses and wildflowers. *Centaurea solstitialis* is considered to be one of the most widespread and aggressive invasive plants of Western North America (LeJeune & Seastedt 2001; DiTomaso et al. 2007; Jackson & Bartolome 2007). According to state and county lists *Centaurea* spp. occupy more land area than any other invasive taxa in the Western United States (Skinner et al. 2000); *C. solstitialis* is the most widely distributed noxious weed in California having infested nine million hectares (DiTomaso et al. 1999; Dukes et al. 2007). Invasive species including *C. solstitialis* and non-native annual grasses have been shown to outcompete native taxa and alter hydrologic, fire, and nutrient cycles in grasslands (D'Antonio & Vitousek 1992; DiTomaso et al. 2007). Through these processes invasive species degrade wildlife habitat and decrease species diversity (Rikard & Cline 1980; Belcher & Wilson 1989; DeLoach 1991; Sheley & Larson 1994).

Once established, non-native, invasive species negatively affect ranching practices as well through the reduction of forage quality. The hard awns of many of the non-native annual grasses and spines of the thistles reduce livestock weights and degrade the quality of their meat, milk, wool, and hides (DiTomaso 2000; DiTomaso et al. 2007). *Centaurea solstitialis* also has been found to cause a lethal neurological disorder in horses known as "chewing disease" (Kingsbury 1964). The hard awns from introduced annual grasses such as several *Hordeum* spp. (foxtails) can be fatal for pets and wildlife if they become lodged in animals (Bussanich & Rootman 1981; Del Angel-Caraza et al. 2011). Invasive species burs, spines, and awns degrade recreational areas for human use as well and increase air borne allergens (DiTomaso et al. 2007).

The question now facing natural resource managers is how to preserve, protect, or enhance the remaining grasslands. Native grasslands not only support natural biodiversity, but also play an important role in nutrient cycling and carbon sequestration (Conant et al. 2001; Wigland 2007). Most native perennial grasses can live for hundreds of years (Crampton 1974; Corbin et al. 2007b) and have deep root systems that foster soil health and cycle nutrients through relationships with mycorrhizal fungi (Lowry 2012). These root systems help to filter runoff during storms by holding soils and preventing erosion, which directly benefits water quality and protects sensitive riparian species whose life cycles are affected by toxins and sedimentation (Crampton 1974).

Natural processes such as fire are a fundamental component of California's ecology and have historically played an important role in the maintenance of grassland ecosystems (Sauer 1950; Wells 1962; Vogl 1974; Hatch et al. 1991; Jacobs et al. 1999; Meyer 1999; Reiner et al. 2007). Prescribed burns provide land managers an effective means by which to control invasive populations without the use of chemicals. In addition the reduced frequency of burns in grasslands has led to the formation of thick layers of dried annual thatch (Biswell 1956; Heady 1958), which reduces germination and growth of native forbs (Heady 1956; DiTomaso et al. 1999; DiTomaso et al. 2007; Kyser et al. 2007). In fact, areas which have introduced fire have shown increased biodiversity while those which have reduced the roll of fire have exhibited lower diversity in plant communities (Kruger 1983; Hunter 1986; Parsons & Stohlgren 1989; Mitchelson 1993; Delmas 1999; DiTomaso et al. 1999; Harrison et al. 2003; Marty 2015).

Prescribed burning in grasslands has proven to be an essential management tool in controlling invasive species such as non-native annual grasses (D’Antonio &

Vitousek 1992; Delmas 1999; D'Antonio et al. 2006) and *Centaurea solstitialis* (DiTomaso et al. 2003; Keeley 2006; Reiner 2007). DiTomaso et al. 1999 reported that the repeated burning of *C. solstitialis* in two separate open grasslands in Sugarloaf Ridge State Park (Sonoma County, CA) over two years led to a 99% decrease in *C. solstitialis* seed bank and seedling density and an increase in the diversity of native forbs. The site was then monitored four years following the final burn and it was found that without continual management they reverted to their pre-burned state (Keyser & DiTomaso 2002).

The use of prescribed burns has shown to be effective in reducing invasive plant populations; however the long-term success of these control efforts has been limited (Keyser & DiTomaso 2002). After fire, early successional colonizers, such as non-native annual grasses, take advantage of the initial disturbance and quickly reestablish (D'Antonio & Vitousek 1992; D'Antonio et al. 2006). The shallow roots of annual grasses extend to depths of 30 cm (Seabloom et al. 2003) reducing topsoil moisture, while leaving the water in deeper soils untouched (Holmes and Rice 1996). This scenario has set the stage for *C. solstitialis* success by allowing it to take advantage of this untapped resource (Borman et al. 1992; Holmes & Rice 1986; Dukes 2001; Gerlach 2004). The phenomenon in which the initial invasion by a group of non-native species facilitates the introduction of another is known as "invasional meltdown" (Simberloff & Van Holle 1999; Morgan & Rice 2005).

Native plant revegetation following a burn gives native populations a chance to recover and potentially break the cycle of invasion. Once the disparity of resource allocation is addressed, native grasses can in fact compete with non-native annual species

(Rice 1989; Robinson et al. 1995; Brown & Bugg 2001; DiVittorio et al. 2007; Seabloom et al. 2003). Native perennial grasses can be effective competitors once established and have an increased resistance to invasion over time (White 1967; Jackson & Roy 1986; Brown et al. 1998; Hamilton et al. 1999; Corbin & D'Antonio 2004; Roche 1994; Callaway et al. 2001, 2003; Corbin et al. 2004; Reever Morghan & Rice 2005).

This study evaluates how the use of prescribed burns affects the grassland community composition, and assessed the ability of each native species to reestablish. I hypothesized that planting native grassland species following a prescribed burn will suppress the reestablishment of non-native annual grasses and *C. solstitialis* through the competition for light, nutrients, and water resources. Once established perennial native species root systems can compete for the deep-water resources which have enabled the wide spread of *C. solstitialis* infestations.

Species with broad ranges in California grasslands that occur within the local watershed were selected for the project. Seeds of *Elymus glaucus* (blue wild rye), *Stipa pulchra* (purple needle grass), *Bromus carinatus* (California brome), *Grindelia camporum* (great valley gumweed), and *Madia elegans* (tarweed) were collected from within the watershed of Big Chico Creek. Some of the seeds were stored for direct seeding and others were propagated into 15 cm long pots (plugs) at a local native plant nursery. The seeds and the nursery grown plugs were then spread or out-planted into the prescribed burn areas using a systematic randomization method. Species survival and composition were closely monitored to determine whether prescribed burns and native plant revegetation shift the competitive advantage toward native taxa. My goal for this research is to not only promote local native grassland diversity, but to develop a methodology for future grassland restoration in both parkland and agricultural lands throughout the west.

CHAPTER II

BACKGROUND

California Grasses and Grasslands

Grasslands ecosystems are an essential landscape of California and grasses themselves may be considered our states most abundant, widespread, and useful plants (Crampton 1974). The unique role and lifecycle of native perennial grasses are a quintessential component of the ecology of California grasslands. Perennial bunch grasses are vigorous and once supported the vast herds of deer, elk, and antelope which roamed California's grasslands (Crampton 1974; Bartolome et al. 2007). Perennial grasses sprout new growth 'tillers' from the previous season's dried vegetation which form their characteristic bunches over time that can persist for many years (Crampton 1974; Corbin et al. 2007b). Perennial grasses are instrumental to the dynamics of grassland ecosystems, the pockets of exposed earth between their bunches provide ideal conditions for the germination of annual forbs and perennial woody species.

The unique composition of grassland ecosystems makes their parameters hard to define and their borders difficult to map. The distribution of grasslands in California has been described as trying to put together a puzzle with half the pieces missing (Keeler-Wolf et al. 2007). For the purpose of this thesis, the general definition for a grassland ecosystem will be applied as that which consists of a majority of herbs, graminoids, and forbs with trees and shrubs accounting for less than 25% (Grossman et al. 1998). One fourth of the total area of California is covered with grasses (Keeler-Wolf et al. 2007), with grassland ecosystem themselves currently covering between ten to eleven percent (Davis et al. 1998; Jantz et al. 2007; Stromberg et al. 2007). This study will focus on the lower elevation grasslands of California from between 0-1,200 feet that extend along the Coastal Ranges east to the Central Valley and into the Sierra Nevada foothills (Corbin et al. 2007a).

History of California Grasslands

While there is some debate over their historic distribution before European contact (Hamilton 1997), perennial grasses are thought to have dominated California's grasslands in the nineteenth century (Heady 1977; Heady 1988; Holstein 2001). The grasslands of California were documented in accounts from early explorers including those of John Muir in the late 1800's (Muir 1974). The extent of California's grasslands has also been supported with the use of phytolith data (Bartolome et al. 1986) and paleovegetation analysis (Minnich 2007).

The structure and function of California's original grasslands were shaped by thousands of years of anthropological influences (Anderson 2007). Journals from 1769 describe how California's native people utilized fire for the management of grasslands (Aschmann 1977; Blackburn & Anderson 1993; Anderson 2005). Evidence suggests the human use of fire played a major role in shaping the grasslands of the Coast Ranges, Central Valley and Sierra Nevada foothills (Stromberg et al. 2007b). The indigenous use of fire increased fire frequency and is thought to have had a greater role in the structure of grasslands than lightening ignitions (Parsons 1981; Keeley 2002). Early records indicate that fire was implemented every few years (Margolin 1989) with summer and fall burns being the most frequently mentioned in the historical literature (Anderson 1993, 2005; Crespi 2001; Timbrook et al. 1993). Fire was used to convert areas of dense forest and brush to open grasslands (Vogl 1974). Burns were then regularly implemented in order to maintain the structure of these grasslands and to prevent brush from reestablishing (Aschmann 1977; Margolin 1989; Greenlee & Langenheim 1980, 1990). Nutrients released following fire increased yields through stimulating fresh growth in the leaves, flowers, and seeds of grasses (Lewis 1973). The expansion of grasslands increased quality of forage and in turn led to ideal hunting grounds for wild game (Anderson 2007; Hankins 2009).

Burns also promoted the abundance of certain desirable species, which were used for food, fiber, and medicine (Reynolds 1959; Dalrymple 2000; Hankins 2009). The legacy of these uses may have affected the distribution and populations of these species to this day. It has been documented that native people harvested and utilized many grassland species. Mounds of *Mulenbergia rigens* (deer grass) were specially treated with fire to produce the bundle foundations necessary for basketry (Crampton 1974; Anderson 1996). The seeds of native grasses such as *Bromus carinatus* (California brome), *Calamagrostis nutkaensis* (Pacific reed grass), *Danthonia californica* (California oatgrass), *Elymus condensatus* (giant wildrye), *E. glaucus* (blue wildrye), *Achnatherum hymenoides* (Indian ricegrass), *Stipa lepida* (foothill stipa), and *S. pulchra* (purple needlegrass) were reliable sources of food and grain (Lowry 2012). The flowering stalks were harvested, seed was separated from the chaff, parched in baskets, ground into flour, and roasted on rocks. In addition, a multitude of grassland geophytes were harvested and roasted throughout California including members of *Brodiaea*, *Camassia*, *Dichelostemma*, and *Triteleia* (Anderson 2007; Lowry 2012). In many cases, populations of native plant species were promoted by the spreading and replanting of their root fragments, cormlets, and bulblets for future harvests (Anderson 2007; Lowry 2012). The continued use and selection of these species likely increased their propagation and the composition of California grasslands over thousands of years (Picket 1976; White 1979).

The human influence over California's grassland landscape changed radically with European colonization (D'Antonio et al. 2007). As early as the 1600s, the implementation of agriculture and livestock practices by Spanish explorers facilitated the introduction and spread of invasive species (Bossard et al. 2000). Some of the earliest records from 1769 confirm the use of foreign livestock by Spanish soldiers and missionaries. The seeds of weeds were transported on ships in packaging material, and in the soil of fruit cuttings. Annual grasses such as *Avena barbata* (slender wild oats), *Avena fatua* (wild oats), *Bromus diandrus* (ripgut brome), *Bromus mollis* (soft chess) and *Lolium multiflorum* (rye grass) used for cereal, grains, and livestock forage were imported and grown in the surrounding areas of the first missions. Additional forbs were imported and used for grazing such as *Medicago polymorpha* (bur clover) and the filarees *Erodium botrys*, *E. obtusiplicatum, E. cicuatarium, and E. moschatum.* The transport of these species for forage also lead to the incidental spread of weedy species including *Aira caryophyllea* (silver hairgrass), *Brachypodium distachyon* (purple falsebrome), *Bromus madritensis* ssp. *rubens* (red brome), *Bromus tectorum* (downy chess, cheat grass), *Gastridium phleoides* (nit grass)*,* the 'fescues' *Festuca myuros*, *F. bromoides* and the 'barleys' *Hordeum murinum* ssp. *leporinum*, *H. murinum* ssp. *glaucum* (Crampton 1974).

In fact, evidence of some of the first introduced plants such as *E. cicutarium*, *L. multiflorum, Hordeum murinum* (foxtail barley), and *Poa annua* (annual bluegrass) seeds have been found within mission's original adobe bricks (Hendry 1931; Spira & Wagner 1983; Mensing & Byrne 1999; Bartolome et al. 2007). In the 1800s, livestock and non-native species quickly spread and advanced from the overgrazed grasslands surrounding the missions to ranches in the Central Valley (Crampton 1974; Bartolome et al. 2007). The widespread shift in composition to European annual grasses is thought to have occurred from 1860-1880 when agricultural growth and grazing practices began to dominate the grasslands of California (Burcham 1957; Heady 1977).

Modern Grassland Composition

Introduced non-native species perpetuate in grasslands today and some continue to be reseeded into pastures for optimal livestock forage (Crampton 1974). In grazing situations where fields have become depleted, additional non-native species have also been brought in to enrich the soils such as the 'clovers' *Trifolium hirtum*, *T. incarnatum*, *T. subterraneum* and hardy non-native perennial grasses like *Phalaris tuberosa* var. *stenoptera* (Harding grass) (Crampton 1974, personal conversation with ranchers in Chico, CA). Few, if any, of the native grassland species can be found in agricultural areas that were historically plowed or are currently tilled. Tilling is generally carried out once soils have become malleable for equipment (Crampton 1974). This coincides with the emergence of native grass seedlings, which over time has further exhausted remaining native resources (Stoll 1998). Considering that 80% of California's grasslands are currently owned and controlled by private landowners, their management and future largely rests in their hands (Jantz et al. 2007).

Where native plants do occur, they are found within a “matrix” of non-natives (D’Antonio et al. 2007). In 1974, 478 species of grasses were listed in California with 303 (63%) as native, and 175 (37%) as non-native (Crampton 1974). A later publication in 1998 accounted for 181 (40%) non-native grass species in California (Randall et al. 1998), with another survey in 2007 reported a total of 524 grass species, with 291 (55%) native, and 223 (45%) non-native (Peterson & Soreng 2007). Depicting a decrease in native grasses of 8% from 303 in to 291, and increase in non-natives by 8% from 175 to 223, in roughly a 30-year span from 1974 to 2007.

Of the 1,100 introduced species accounted for in California, 300-400 can be found in low elevation grasslands, and 65-90% of those are from Eurasia (Bartolome et al. 2007; D’Antonio et al. 2007). Among those found at lower elevations 60-70 are considered to be moderate to high concern invasive species (D’Antonio et al. 2007). Of the most common invasive plants found in grasslands 80% are divided between the Poaceae and Asteraceae, with the Brassicaceae, Geraniaceae, and Fabaceae accounting for majority of the rest (DiTomaso et al. 2007). The California Invasive Plant Council’s (IPC) invasive plant inventory listed forty-three of the non-native grasses as invasive (Peterson & Soreng 2007). Invasive members of the Poaceae listed as high concern include *Aegilops triuncialis* (barb goat grass), *B. madritensis* ssp. *rubens* (red brome), *B. tectorum* (downy brome), *Taeniantherum caput-medusa* (medusahead), and members of Asteraceae that are listed as such are *C. solstitialis* (yellow star thistle), *Cynaria carduncnlus* (artichoke thistle), and *Onopordum acanthium* (Scotch thistle) (DiTomaso et al. 2007). When comparing all of the invasive species present in California grasslands over 73% are winter annuals, 11% biennials, and 16% perennials (DiTomaso et al. 2007).

Competition in California Grasslands

Populations of introduced plant species have benefited from historical land use practices and several scenarios under which non-native species have been given competitive advantage over native species have developed. Competitive interactions between annual non-native grasses and native perennial grasses have been thoroughly researched (Biswell 1956; Bartolome & Gemmill 1981; Jackson & Roy 1986; Borman et al. 1991; Blumler 1995; Dyer et al. 1996, 2000; Holmes & Rice 1996; Dyer & Rice 1997, 1999; Brown et al. 1998; Hamilton et al. 1999; Jacobs et al. 1999; Brown & Rice 2000; Reynolds et al. 2001; Betts 2003; Seabloom et al. 2003; Corbin et al. 2004; Moyes et al. 2005; Seabloom et al. 2005; D'Antonio et al. 2007; DiVittorio et al. 2007). Comparison studies of competition between the native perennial, *Stipa pulchra* and non-native annual grasses have consistently favored the latter (Dyer & Rice 1997, 1999; Brown & Rice 2000). During the first fall rains, non-native annual grasses usually germinate several weeks before perennial native grasses, which has been demonstrated in the field and under controlled experimentation (Corbin et al. 2007b; Jackson & Roy 1986; Reynolds et al. 2001). The delay in native grass germination may hamper their growth from the start (D'Antonio et al. 2007). The seedling phase is the greatest period of interference between non-native annuals and native perennial grasses (Jackson & Roy 1986; Dyer et al. 1996), and seeds of *Stipa pulchra* were found to reach higher densities when sown alone than when mixed with seeds of non-native annuals (Bartolome & Gemmill 1981).

The physiological differences between native perennial grasses and non-native annual grasses causes a variation in the timing of their water and nutrient uptake (Holmes & Rice 1996; D'Antonio et al. 2007). During the early phases of development, annual grasses negatively affect both the germination and growth of *Stipa pulchra* through reducing light availability and soil moisture (Dyer & Rice 1997, 1999). The early onset and rapid vertical growth of annual species (Gulmon 1979) reduces available resources for smaller or slower growing species (Dyer & Rice 1999) before they have a chance to establish (Ross & Harper 1972; Bartolome & Gemmill 1981; Dyer & Rice 1999). Between March and April rising temperatures and increased daylight lead to a period of rapid biomass production (Pitt & Heady 1978; Gulmon 1979; Chiariello 1989) and light penetration can be as low as 5% among a field of non-native annual grasses (Dyer & Rice 1999).

The early development of non-native annual grasses has been shown to impede native perennial grasses' ability to reach below ground, deep-water resources they require for summer growth (Dyer & Rice 1999; Hamilton et al. 1999). Ninety percent of the roots of non-native annual grasses exist within the top 30 cm of the soil (Holmes & Rice 1996) and these shallow root systems have been found to greatly reduce topsoil water availability (Seabloom et al. 2003). However, during winter rains the roots of these annual grasses were found to be unable to uptake all of the available water (Holmes & Rice 1996; Gerlach 2004) leaving abundant soil moisture below them (Enloe et al. 2004). During the summer, annual grasses complete their life cycle and begin to senesce, which causes the remaining deep-water resources to be left untapped, in fact fields dominated by non-native annual grasses have been found to have higher levels of deep-water availability (Holmes & Rice 1996; Gerlach 2004). The accessibility of deep-water resources has led to the rapid invasion and success of *C. solstitialis* (Gerlach 2004), which has been found to alter water cycles in annual grasslands (Enloe et al. 2005). Once annual grasses begin to complete their lifecycle and set seed, *C. solstitialis* experiences rapid growth. Isolated populations of *C. solstitialis* have been shown to use more water than populations of non-native annual or perennial grasses (Enloe et al. 2004), and have been found to utilize most of the water resources in the early summer (D'Antonio et al. 2007).

Fire in California Grasslands

The control of invasive plant species is the greatest challenge facing the restoration of the remaining California grasslands (Stromberg et al. 2007a). A dramatic shift in the perpetual dynamics of invaded grasslands is needed and the reintroduction of fire may provide a partial solution. The altering of fire regimes, along with the widespread growth of agriculture has greatly reduced the roll of fire in grasslands (Reiner 2007). The elimination of indigenous burns can be dated to 1793 when the Spanish Governor José Joaquín de Arrillaga ordered that the "widespread damage" of setting fires to pasture lands would be met with the most severe punishment. Fire frequencies that were between 1-5 years are now few and far between (Margolin 1989; Greenlee & Langenheim 1990; Keeler 2002). Fire has historically played a predominant role in maintaining the structure of grasslands and its use carries important implications for their management (Meyer 1997; Bartolome et al. 2007; Reiner 2007). The use of fire has been shown to alter competitive interactions through effecting residual litter and duff, seed banks, germination, and standing vegetation (Corbin et al. 2007b).

The goal of using prescribed burns for invasive plant management is to eliminate invasive species before they have a chance to reproduce, reduce standing seeds, and diminish existing seed banks (DiTomaso et al. 2007a). Each prescribed burn area is site specific and the invasive species composition along with native species populations and seed banks that may be affected should be taken into consideration (Reiner 2007). Timing is paramount in all of these respects and the season of the burn, along with the duration, temperature, and fuel load can affect plant mortality (Miller 2000). The variation between the life stages of native and non-native plant species provide an opportunity to properly time burns to benefit native plants (Stromberg et al. 2007a). The early germination of annual grasses gives land managers the chance to burn in fall before natives have sprouted (personal conversation with Dr. Paul Maslin, Hankins 2009, Hankins 2013). Fall and winter burns are also beneficial because they do not take place during peak fire season when permits and fire crews are harder to attain (personal conversation with Dr. Don Hankins). Another optimal time to burn occurs later during spring once natives have set seed but invasive seeds have not been released (Moyes et al. 2005). Control burns during this time produce the highest temperatures above ground, which increases standing seeds mortality (Pollak & Kan 1998; DiTomaso & Johnson 2006). Prescribed burns that are carried out once invasive seeds have begun to form have been found to be most effective (Brooks 2001). Seeds from non-native species such as *Aegilops triuncialis, Bromus diandrus*, *Bromus madritensis* ssp. *rubens*, *Taeniantherum caput-medusae*, and *Centaurea solstitialis* have long awns and spines for animal dispersal and remain on the inflorescences allowing for optimal burn periods (DiTomaso & Johnson 2006; DiTomaso et al. 2007a). Prescribed burns carried out during this time have proven to be effective on invasive plants such as *A. triuncialis*, *B. diandrus*, *B. madritensis* ssp. *rubens*, *T. caput-medusae,* and *C. solstitialis* (Pollak & Kan 1998; DiTomaso et al. 1999; Hopkinson et al. 1999; Kyser & DiTomaso 2002; Betts 2003; DiTomaso et al. 2003; DiTomaso & Johnson 2006; Keeley 2006; Reiner 2007; Stromberg 2007a). In many cases, repeated burns may be required to control the seed banks of unwanted species (DiTomaso et al. 1999; Hopkinson et al. 1999; Kyser & DiTomaso 2002; Reiner 2007). Repeated burns have been found to be highly effective (Keeley 2006), but treatments must be conducted consecutively (Kyser & DiTomaso 2002).

Properly timed prescribed fires can not only control invasive species populations but reduce competition for species which persist in the seed bank (Fossum 1990; George et al. 1992; Menke 1992; Meyer & Schiffman 1999; Hankins 2013). Burns in annual grasslands remove dried thatch, which accrues over time from left over forage that has become hard and less palatable and inhibits the germination and development of native plant species (Biswell 1956; Heady 1958; Kyser et al. 2007). Burning exposes bare mineral soil while releasing essential nutrients such as nitrogen and phosphorus from the dead matter. In addition, fires increase light availability (Menke & Rice 1981; Whelan 1995; Foster & Gross 1998; Corbin et al. 2007b) which aides plant germination by increasing soil temperatures and enhance plant diversity (Stone 1951; Mutch 1970; Gill 1977; Baskin & Baskin 1998; Anderson 2007; DiTomaso et al. 2007). Fire can increase forb cover over time (Parsons & Stohlgren 1989; Delmas 1999; Meyer & Schiffman

1999; DiTomaso et al. 2001) and prescribed burns over three years for *Centaurea solstitialis* in Sonoma county increased forb cover from 17% to 67% particularly within the Fabaceae (*Acmispon wrangelianus, Lupinus nanus*, and *Trifolium gracilentum*) as well as for *Leptosiphon bicolor* (Polemoniaceae) and *Minueartia californica* (Caryophyllaceae) (DiTomaso et al. 1999). Similarly, native annuals such as *Castilleja attenuata* (Orobanchaceae) (Parsons & Stohlgren 1989), and native geophytes such as *Triteleia lugens* (Themidaceae) (Stone 1951), *Sidalcea calycosa* (Malvaceae) (Hunter 1986), and *Toxicoscordion fremontii* (Melanthiaceae) (Mitchelson 1993) increased following fire. The use of fire has proven to promote certain native species and could serve as an effective means to restore dwindling native plant populations when carefully applied with thorough monitoring and persistent follow-up.

Role of Restoration

Prescribed fires have proven to be an effective treatment method in reducing targeted non-native and invasive plant species populations, while increasing forb diversity; however, without continued maintenance invasive plant populations have reestablished (Keyser & DiTomaso 2002; DiTomaso et al. 2007a). Invasive plant management solely geared towards eradication may not be an effective means to restore ecosystems over the long term. The repeated use of grazing and herbicides has been effectively used to diminish invasive plant populations but neither of these treatments enhance native perennial grass abundance and in many cases may actually reduce existing native plant populations. The management of invaded ecosystems should include both the control of the invasive species and native species restoration. When native plant species resources are abundant they may recover on their own, however, if they have diminished the allocation of resources must be compensated and accounted for in order to restore ecosystem function. California grassland restoration efforts that have accounted for this disparity have proven to be most effective (Seabloom 2003; Corbin et al. 2004; Stromberg et al. 2007a).

Seed availability may be the primary cause hampering native grass population growth (Hamilton 1999; Seabloom 2003; DiVittorio et al. 2007). Hundreds of years of land use and grazing have greatly diminished native grass populations while non-native annual grass populations have rapidly increased with disturbance (Crampton 1974). Non-native annual grass species seed banks have become more abundant than native grasses and it is estimated that for every one native grass there may be hundreds to thousands of annual grasses within a 10 cm radius (Major & Pyott 1966; Bartolome & Gemmill 1981; Rice 1989; Young & Evans 1989). Although the seedling stage of non-native annual grasses has a negative affect on native perennial grasses (Biswell 1956; Heady 1956; MacDonald et al. 1988; Heady et al. 1992), seed limitation is the predominate factor limiting their establishment over direct competition (Seabloom et al. 2003). Experiments with an enhanced native grass seed supply led to increased density and cover of both native grasses and forbs (Corbin et al. 2007b). The addition of 5,000 *Stipa pulchra* seeds per m^2 increased their density by five times (Hamilton et al. 1999) and seed treatments of 500 per m^2 of *S. pulchra*, *Bromus carinatus*, *Elymus glaucus*, *Nassella cernua* and *Poa secunda* were all found to have increased percent cover in an invaded grassland (Seabloom 2003).

Native perennial grasses are adapted to survive in the unique conditions of the Mediterranean climate by using variations in rooting patterns and allocation of root to shoot tissue (Corbin et al. 2007b). The deep root systems of established native grasses can extend 1.5 meters or more into the soil (Brown et al. 1998; Dyer & Rice 1999) and can use water as deep as 50-100 cm below the surface (Holmes & Rice 1996; Enloe 2004). Native bunch grasses continue to gain biomass into the summer, which allows them to extend their periods of growth (Jackson & Roy 1986; Borman et al. 1992). Once native perennial grasses are have established they can live for decades to centuries (Corbin 2007b) and in the process resist further invasions (DiTomaso et al. 2007; Stromberg et al. 2007a). The root systems of native perennial grasses allow them more access to deep-water resources (Enloe et al. 2004) using 50% more soil moisture than annual grasses (Borman et al. 1992). Established native perennial grasses decrease the biomass of non-native annual grasses by as much as 80% (Corbin & D'Antonio 2004) and reduce the growth and reproduction of *Centaurea solstitialis* (Roche et al. 1994; Lym & Tober 1997; Dukes 2001; Reever Morgan & Rice 2005). The reintroduction of perennial grasses has also been found to increase forage production as well as promote plant diversity (Borman et al. 1991; Lym & Tober 1997).

Once thought of as an improbable task (Heady 1988) grassland restoration efforts are now becoming more common (Stromberg et al. 2007), and widespread restoration efforts are being carried out in California grasslands (Stromberg et al. 2007). Organizations such as The Nature Conservancy, the Trust for Public Land, and the California Rangeland Trust along with many county and local land trusts have invested hundreds of millions of dollars to acquire and protect grasslands (Jantz et al. 2007). The field of habitat restoration addresses the negative human impacts on a landscape (Packard & Mutel 1996), the Society for Ecological Restoration (SER) defines restoration as “the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed.” The grasslands of California fit this description and are in dire need of restoration. Successful grassland restoration activities should begin with the removal and control of invasive species followed by the revegetation of native species (Brown et al. 1994; Stromberg et al. 2007a). Before invasive control techniques are implemented past human activities should be considered as well as existing native species composition in order to protect remaining resources (Baker 1989). It is also important to take into account the adaptations of native plant populations and select species that are adapted to the local soils, elevation, and climate which will aid in their survival while preserving the genetic integrity of the ecosystem (Jacobs et al. 1999; McKay et al. 2005; Stromberg et al. 2007a).

Once the best candidates for restoration are identified, the most efficient reestablishment technique should be selected, which is determined through the scope, timeframe, and funding of the project. Seeds can be grown and propagated into pots (plugs) or direct seeded by hand or with the use of equipment. Both techniques have been applied to California grasslands with success (Stromberg et al. 2007a). While plugs may be more time consuming and require more resources to propagate, they may be the best choice for smaller scale projects with limited seed resources or sites that require more rapid establishment. Native perennial grassland species have been shown to propagate well in plugs and have proven to be highly effective in the field (Brown & Bugg 2001; Corbin & D’Antonio 2004; Huddleston & Young 2004). Direct seeding of native perennial grasses has also proven to be useful and can be the most efficient way to broadcast over a large area (Seabloom et al. 2003). The timing of the planting is quintessential to the success of revegetation efforts. The best time to plant plugs or sow seeds is during fall with the first rains in order to allow seeds enough time to absorb the moisture and warmth they require to germinate and give roots time to establish before the summer heat (Stromberg et al. 2007a). Following planting continued monitoring and maintenance for the first several years will ensure the best results (Stromberg et al. 2007a).

The goal of this research was to see if native species can effectively reestablish and by doing so curb the cycle of invasion through competing for similar available resources. My hypothesis is that the reimplementation of prescribed burns along with the compensation for the disparity in native species resources through restoration processes can shift competitive hierarchies within invaded grasslands. I monitored the survival and cover values of native species following the burn and observed whether the treatments then impeded the reestablishment and growth of non-native species. If so the information from these efforts can contribute to the long-term success of grassland restoration efforts by reducing the spread of invasion while increasing habitat resources in the process.

CHAPTER III

METHODS

Study Sites and Species

Two grasslands that fall with in the California Native Plant Societies extensive ‘California annual grassland series’ ranging from 0-1,200 feet (365m) in elevation that consists of varying amounts non-native species and Native annual species less than a meter high (Sawyer et al. 2009) were selected as research sites. Invaded grasslands with non-native annual grasses and *C. solstitialis* that had similar vegetation, slope and aspect were identified within the watershed of Big Chico Creek in Chico, California. The first location is part of Chico State University’s Big Chico Creek Ecological Reserve (BCCER) and the other site downstream, falls within the City of Chico’s Upper Bidwell Park (BIDWELL).

Native grassland species (*Bromus carinatus, Elymus glaucus*, *Stipa pulchra, Grindelia camporum,* and *Madia elegans*) whose growth period and traits overlapped with the targeted invasive species were selected for the restoration experiment. Seeds of each of these species were then collected from within the Big Chico Creek watershed during the spring and summer of 2012. Native species seeds were separated for direct seed treatments and for propagation in pots (plugs) for planting treatments. Seeds for the direct seed treatments were then weighed and divided into separate labeled zip lock™ bags containing 14,400 mixed bunch grass seeds (200 *Bromus carinatus,* 200 *Elymus* *glaucus* & 200 *Stipa pulchra*) [600 seeds per 1m^2 plot (Stromberg et. al 2007a] along with 6,000 *Madia elegans* seed [250 seeds per 1m^2 plot]). Seeds of the perennial native species *S. pulchra*, *E. glaucus*, *B. carinatus,* and *G. camporum* were distributed into four trays which each contained 98 3” inch pots ‘plugs’ to be germinated and grown in a local native plant nursery (Floral Native Nursery) greenhouse in October of 2012.

Site Preparation

Land managers in each of the site locations waited for optimal conditions for a prescribed burns (Rx). Fall burns in grasslands are conducted during the window of time when the leaf litter of surrounding vegetation has become saturated from the first fall rains while the layers of duff within the field are dry enough to properly carry the flame. Further, the Rx was timed to coincide with approximately three weeks following germination of non-native annual grasses. Prescribed burns (Rx) within invaded grasslands are most effectively done in areas with standing layers of duff to generate enough heat from fire to kill the targeted invasive seeds. Fires were set with wind conditions around 2-5 miles per hour; low enough to facilitate a ‘slow burn’, which allows longer residency time for the heat from the fire to pass and reach the invasive seeds. Linear backing fires were set using a drip torch to maximize the heat exposure to the designated area. I participated in the control burn of the research site on the Big Chico Creek Ecological Reserve using these techniques with the assistance of Dr. Paul Maslin. The control burn in Bidwell Park was administered by CalFire with city park officials. Prescribed burns (Rx) in both locations were carried out in October of 2012.

Following the Rx burn, six blocks (4m x 7m) were randomly placed within each burn location (three per site) and marked with latitude and longitude points for each corner of their location (SW, SE, NE, NW) and logged with a GPS Trimble unit (Figure 1 and Figure 2).

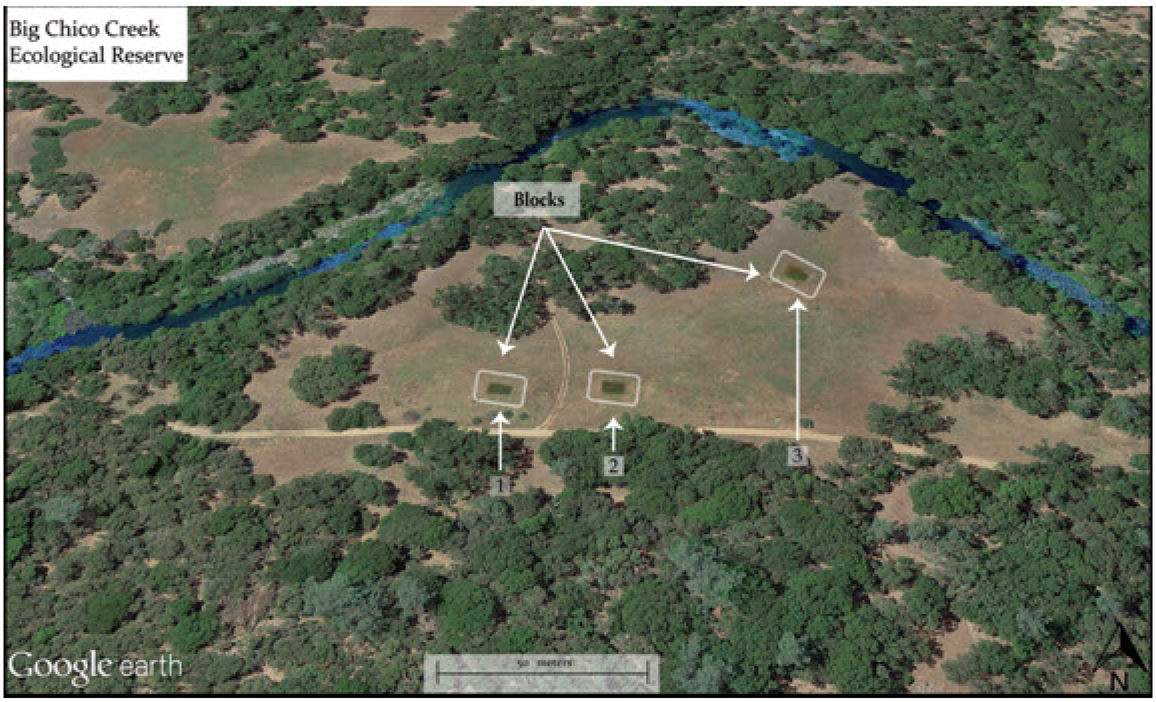

Figure 1. Map of the BCCER site location with each of the three 4m x 7m blocks.

Experimental Design

The corners of the blocks were established with rebar and labeled with florescent caps. Each of the blocks was then divided into 28 separate 1 m^2 plots marked with wooden stakes. One meter squared plots are an effective scale to record the plant diversity within grasslands (Stromberg et al. 2001; Harrison et al. 2003). The 1 m^2 plots were then assigned one of seven treatments (Table 1).

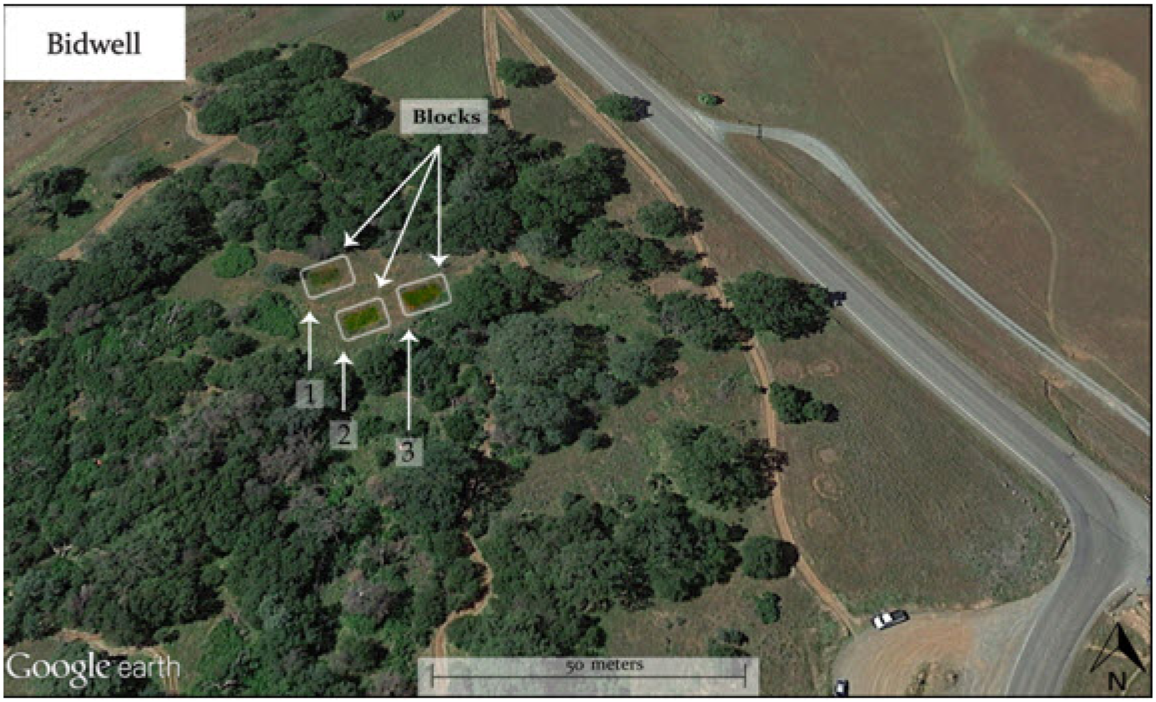

Figure 2. Map of the BIDWELL site location with each of the three 4m x 7m blocks.

Table 1. The seven treatment types within the blocks with the amount of plant materials used for each $1m^2$ plot treatment.

Treatment	*Amount per $1m^2$ plot*
#1: *Stipa pulchra*	(16 plugs)
#2: *Bromus carinatus*	(16 plugs)
#3: *Elymus glaucus*	(16 plugs)
#4: *Madia elegans*	(250 seeds)
#5: Control	X
#6: Native grass mix	(200 seeds of each[600 total])
#7: *Grindelia camporum*	(9 plugs)

The 1m² plots (four planted plugs, two direct seeding, and one control) were replicated four times within each block and distributed using a systematic randomization (Stromberg et al. 2007a) (Table 2).

Table 2. The systematic randomization experimental design for the blocks with each of the treatments receiving four replicates.

Treatments	*Random Values*			
#1	7	1	2	3
#2	6	7	1	2
#3	5	6	7	1
#4	4	5	6	7
#5	3	4	5	6
#6	2	3	4	5
#7	1	2	3	4

Treatments

The direct seed treatments of *Madia elegans* (#4) and mixed native perennial grasses *Bromus carinatus, Elymus glaucus,* and *Stipa pulchra* (#6) were raked in and spread evenly into the designated plots during December of 2012. The nursery grown plugs were ready for planting in January of 2013. Each perennial grass treatment (#1-3) was planted evenly into four rows of 4 plants (16 plugs per $1m^2$ plot) and the *G. camporum* plots (#7) received 3 rows of 3 (9 plugs per $1m^2$ plot) due to lower germination success. The planting of 1,152 native perennial grass plugs, 216 *G. camporum* plugs, 6,000 *Madia elegans* and 14,400 native perennial grass seeds was completed during the winter of 2012-13.

Monitoring

The experimental design was created with the goal of assessing the effectiveness of the different restoration techniques (plugs vs. seeding), and for tracking survival and composition of each treatment. Monitoring was carried out at the beginning of every month (1st-5th) over the duration of 15 months (March 2013- May 2014). Each individual $1m^2$ plot was recorded using a $1m^2$ square PVC pipe frame and data was entered using the following categories for vegetative cover and survival (Table 3.)

Table 3. The data collection format with plug survival and each of the five percent cover value categories recorded each month.

Plug Treatments (#1-3 & 7):	*All Treatments (#1-7):*
- # Total survived/planted	- % Cover of native species treatment - % Cover of *C. solstitialis* - % Cover of non-native grasses - % Cover of non-native forbs - % Cover of native forbs

Upon completion of the experiment the total survival and cover of all four $1m^2$ replicates were calculated and averaged for each of the blocks (4m x 7m) every month. The treatment averages for all 3 blocks were then summed and averaged again to find the combined average for each treatment in each site separately (BCCER & BIDWELL). The data analysis and results between sites were processed separately due to the fact that the BIDWELL research plots were inadvertently treated with herbicide in May and June of 2013. Tables with the results for each treatments average survival and cover values were created for both sites over 15 months from March 2013-May 2014.

Statistical Analysis

Statistical analysis was run using a one-way ANOVA for each site individually to assess the results of the treatment methods for survival and cover. Tukey pairwise comparisons were carried out for all analyses if a significance value of $\alpha = 0.05$ was found (Minitab 17, Minitab Inc.). Data from the last month of the monitoring (May 2014) for the out-planted plug treatments (#1-3 & #7) average survival was compared to show each of the plug treatments average survival success upon the completion of the experiment. The total survival numbers for each of the grass plug treatments (#1-3) were also compared to one another using data from the final month May of 2014.

The percent cover values collected to assess the competitive effect for each treatment on annual grasses, non-native forbs, and native forbs cover values were analyzed utilizing the same method with the data from April 2014. This month was chosen because that is when forbs and annual grasses generally reached their peak growth. However, treatment effects on *C. solstitialis* cover values were assessed during its period of rapid growth in May of 2014.

CHAPTER IV

RESULTS

The data from the experiment are conveyed using a combination of descriptive statistics to show the effects of each treatment over time (Figures 3, 4, 5, and 6; Appendices A and B) to show the final survival results. This is coupled with a series of one-way ANOVA comparisons in order to draw out the effects of each individual treatment survival as well as their effect on the five cover classes (Table 3) during the final months of monitoring (Figures 7-20; Tables 4-17). In order to maintain a distinction between locations (BCCER & BIDWELL) the data analysis and results were processed and analyzed separately for each site.

Survival of Out-planted Plugs from March 2013-May 2014

BCCER

The out-planted plug treatments in the BCCER average survival values remained quite high with over 97% in May of 2013 five months following the initial planting (Figure 3; Appendix A). Values receded during the summer months eventually lead to all of the plugs falling dormant by September with the exception of *Grindelia camporum*. Values begin to return with the first rains in October, and continue to resurge through the spring (Figure 3; Appendix A). Upon the completion of the experiment May 2014, 17 months following the initial planting in January of 2013 the average survival

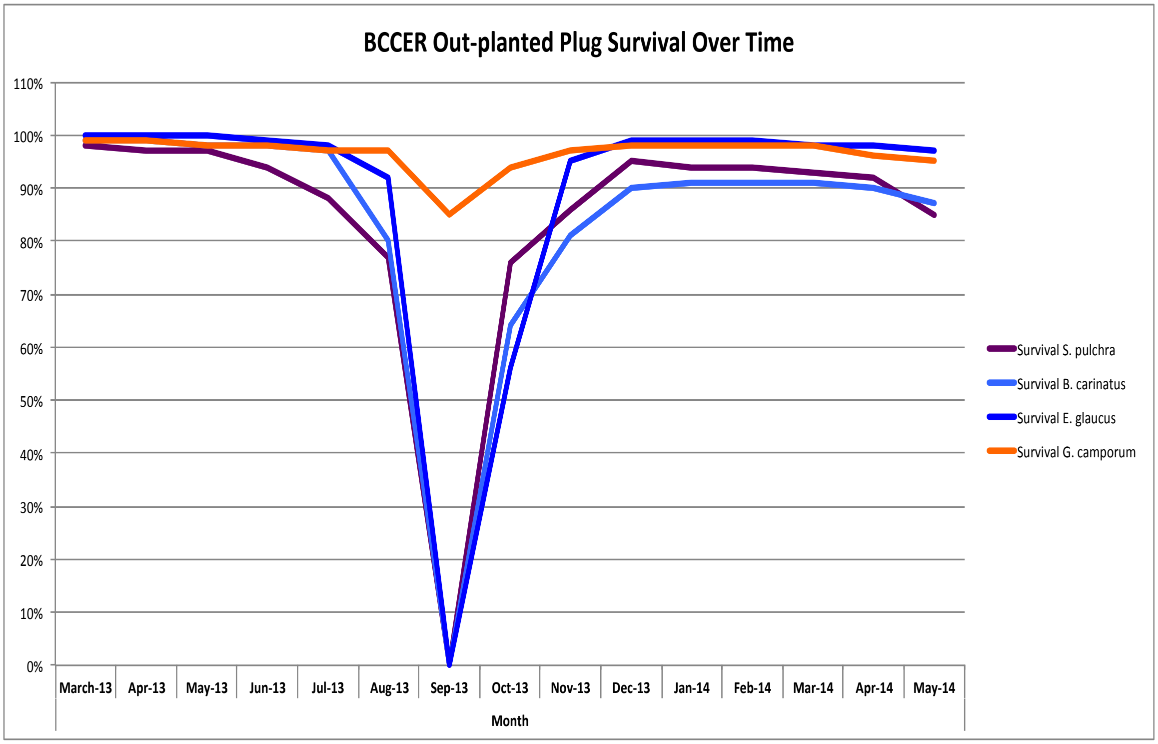

Figure 3. The survival of out-planted plug treatments shown over the course of 15 months of monitoring. Notice the period of senescence experienced by each of the perennial grass treatments during the driest time of year followed by new growth with the onset of the rainy season.

rates for all of the out-planted plug treatments was found to be highly successful ranging from (85-97%) (Figure 4).

BIDWELL

Out-planted plug treatments average survival values remained at over 90% in May of 2013 five months following the initial planting (Figure 5; Appendix A). Values receded during the summer months eventually leading to all of the plugs falling dormant by September with the exception of *Grindelia camporum*. *S. pulchra* begins to return with the first rains in October while the other grasses (*B. carinatus* and *E. glaucus*) do not emerge until December and then continue to resurge through the spring (Figure 5;

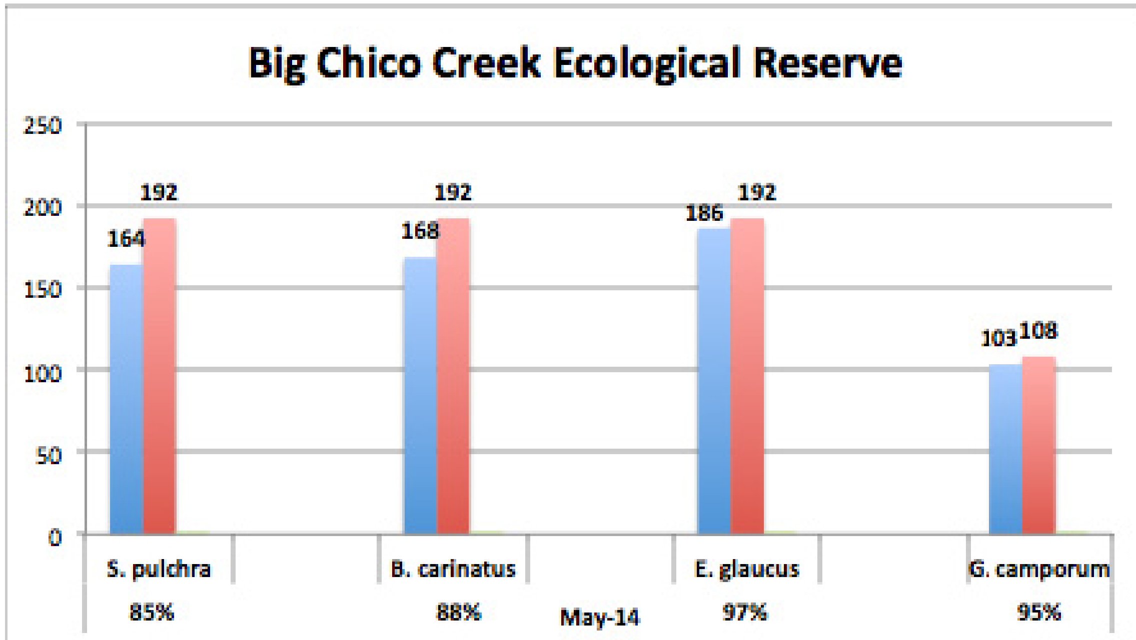

Figure 4. Percent survival of each of the out-planted perennial plugs for the BCCER site in May of 2014. The red column represents the total planted and the blue shows the total survived.

Appendix A). Upon the completion of the experiment in May of 2014 17 months following the initial planting in January of 2013 average survival values remained consistent among the out-planted plug treatments ranging from (54-79%) with the exception of *E. glaucus* (38%) (Figure 6).

Statistical Analysis of Out-planted Plug
Treatments Survival for May 2014

<u>BCCER</u>

During the final month of monitoring in May of 2014 the average survival remained high for all of the out-planted plug treatments in the BCCER location. *Elymus glaucus* (97%) and *G. camporum* (95%) showed the highest mean survival, followed by *B. carinatus* (87%) and *S. pulchra* (85%) (Figure 7). The one-way ANOVA and Tukey

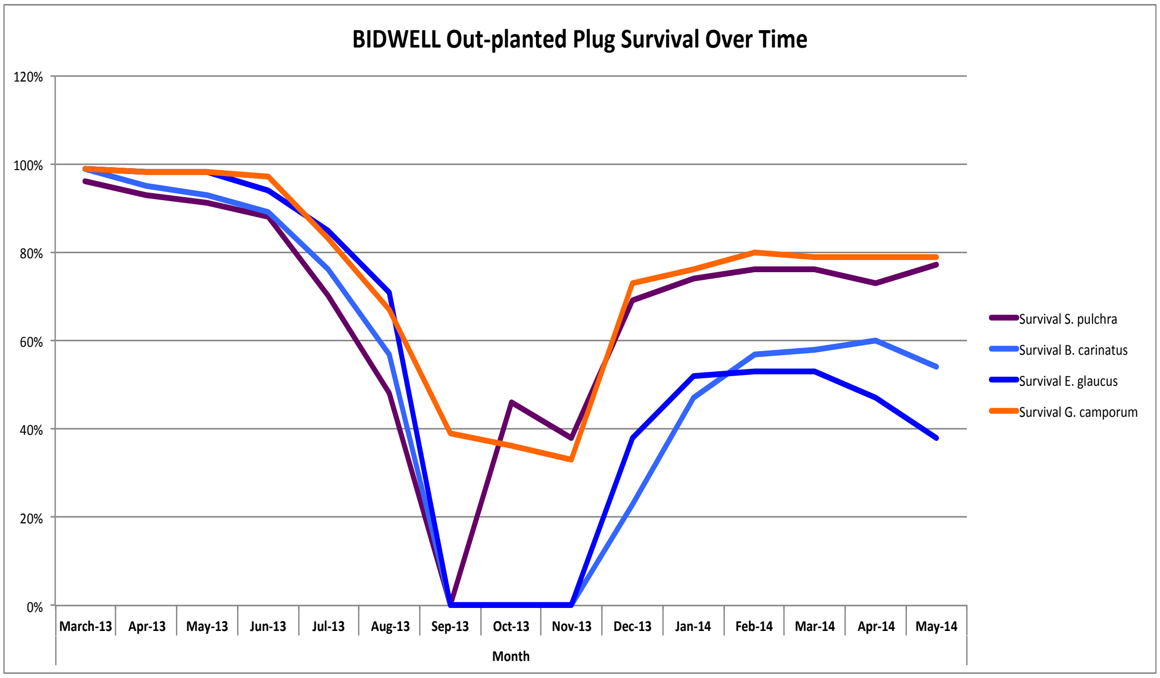

Figure 5. The survival of out-planted plug treatments shown over the course of 15 months of monitoring. The perennial grass treatments experience a period of senescence during the driest time of year followed by new growth with the onset of the rainy season. *S. pulchra* regrowth occurred several months before the other grass treatments as well as show an increase going into the summer of 2014 while the others decline.

pairwise comparisons (α = 0.05) showed that there were no significant differences among the mean percent survival for each of the plug treatments BCCER (F = 1.08, P = 0.412) (Table 4).

<u>BIDWELL</u>

In May of 2014 the BIDWELL site survival rates ranged from 38-79% where *G. camporum* (79%) and *S. pulchra* (77%) showed higher mean survival values than *B. carinatus* (54%) and *E. glaucus* (38%) which finished with the lowest mean survival overall (Figure 8). The one-way ANOVA and Tukey pairwise comparisons (α = 0.05) showed that there were no significant differences among the mean percent survival for the plug treatments BIDWELL (F = 3.48, P = 0.070) (Table 5).

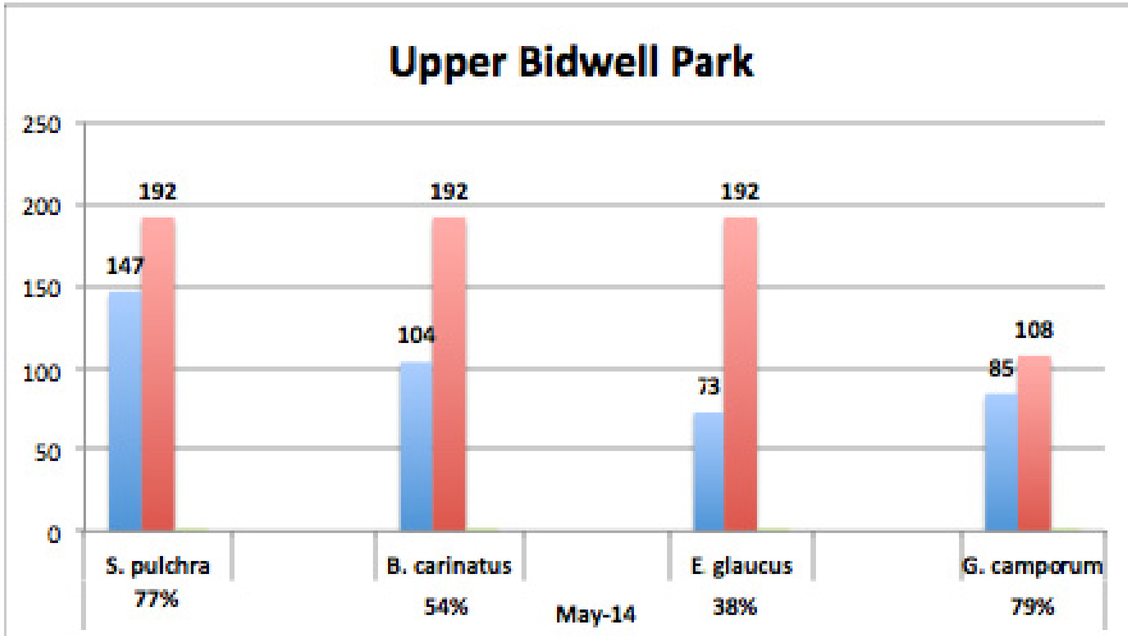

Figure 6. Percent survival of each of the out-planted perennial plugs for the BIDWELL site in May of 2014. The red column represents the total planted and the blue shows the total survived.

Statistical Analysis of Perennial Grass Plug
Survival for May 2014

BCCER

Elymus glaucus was found to have the highest survival out of all three perennial grass treatments in May of 2014 followed by *B. carinatus* and *S. pulchra* (Figure 9). The ANOVA showed that the three perennial native grass plug treatments survival totals were found to be significantly different ($F = 4.08$, $P = .026$). The Tukey pairwise comparisons ($\alpha = 0.05$) found the mean survival totals varied with *E. glaucus* being significantly different from *S. pulchra,* while *B. carinatus* was not found to be significantly different from either of the other perennial grass plug treatments (Table 6).

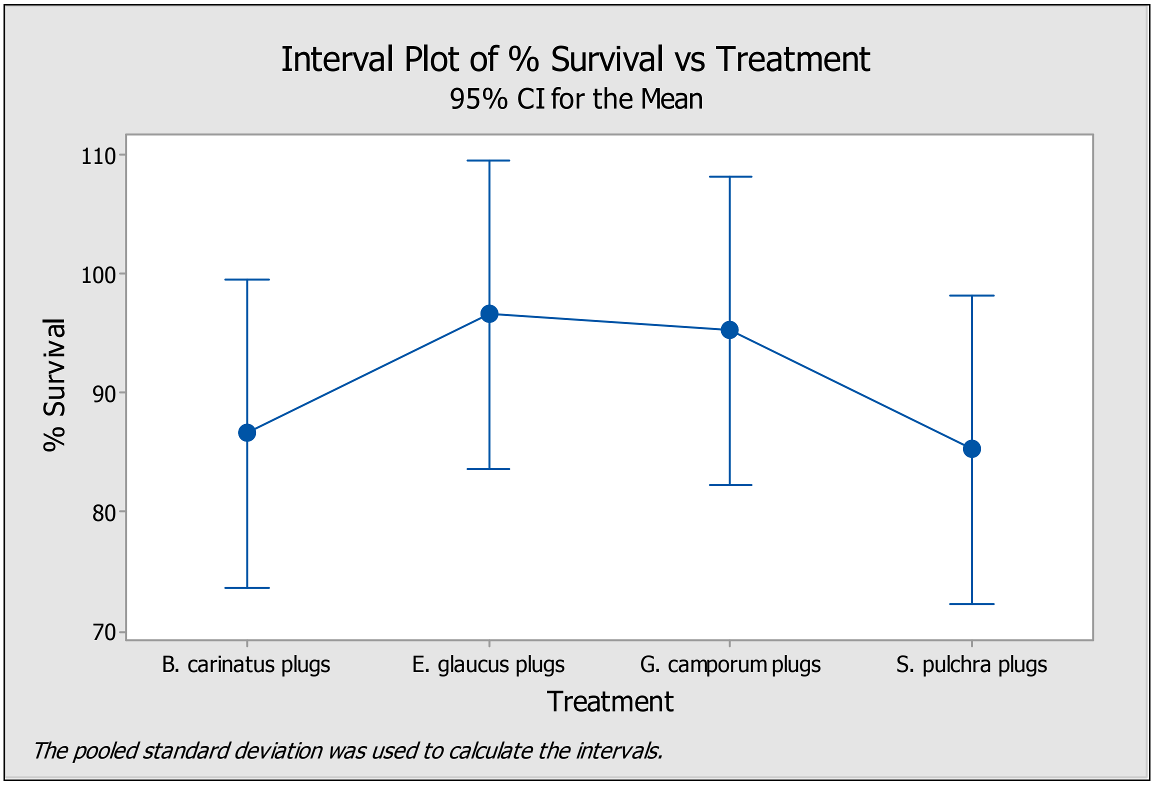

Figure 7. Variation in the mean percent survival for all out-planted perennial plug treatments for BCCER in May of 2014.

Table 4. Tukey pairwise comparisons (α = 0.05) between the BCCER four out-planted plugs mean percent survival with standard deviation in May 2014. Treatments that share the same letter are not significantly different.

	BCCER	
Treatment	*Mean Survival Totals*	*Groupings*
E. glaucus	96.67 (±1.528)	A
G. camporum	95.33 (±5.69)	A
B. carinatus	86.67 (±14.74)	A
S. pulchra	85.33 (±11.24)	A

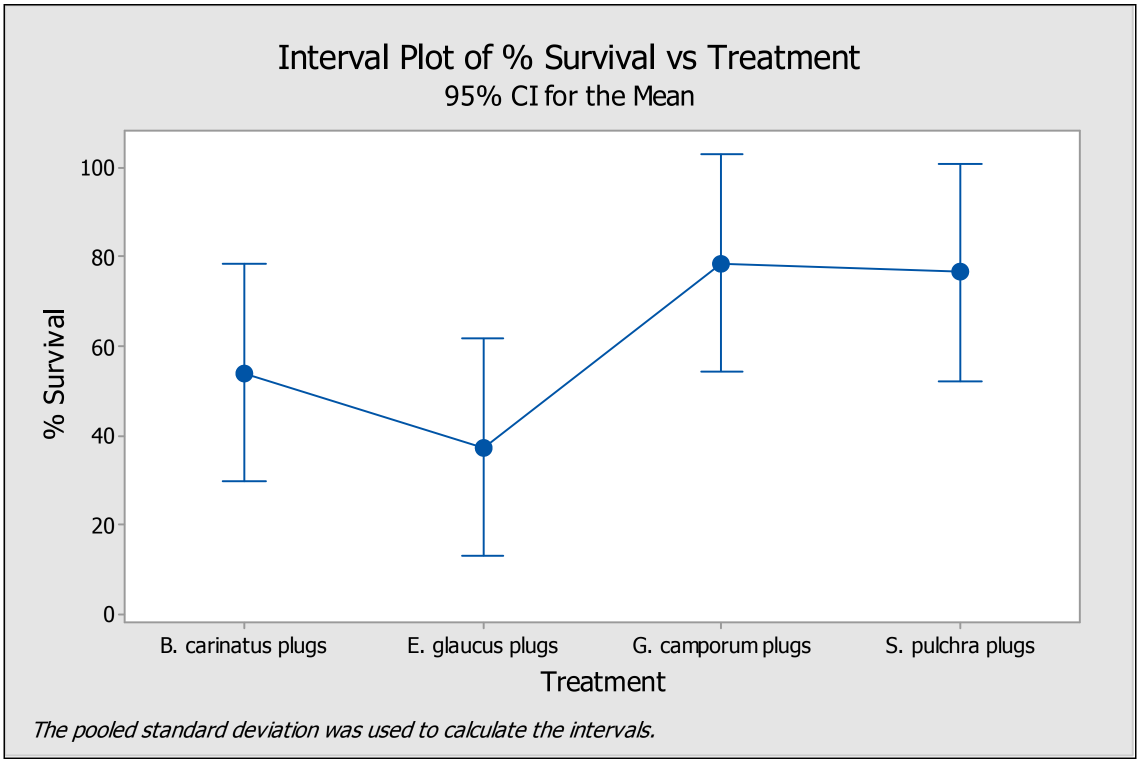

Figure 8. Variation in the mean percent survival for all out-planted perennial plug treatments for BIDWELL in May of 2014.

Table 5. Tukey pairwise comparisons (α = 0.05) between the BIDWELL four out-planted plugs mean percent survival with standard deviation in May 2014. Treatments that share the same letter are not significantly different.

	BIDWELL	
Treatment	*Mean Survival Totals*	*Groupings*
G. camporum	78.7 (±19.8)	A
S. pulchra	76.7 (±18.0)	A
B. carinatus	54.0 (±17.3)	A
E. glaucus	37.3 (±18.0)	A

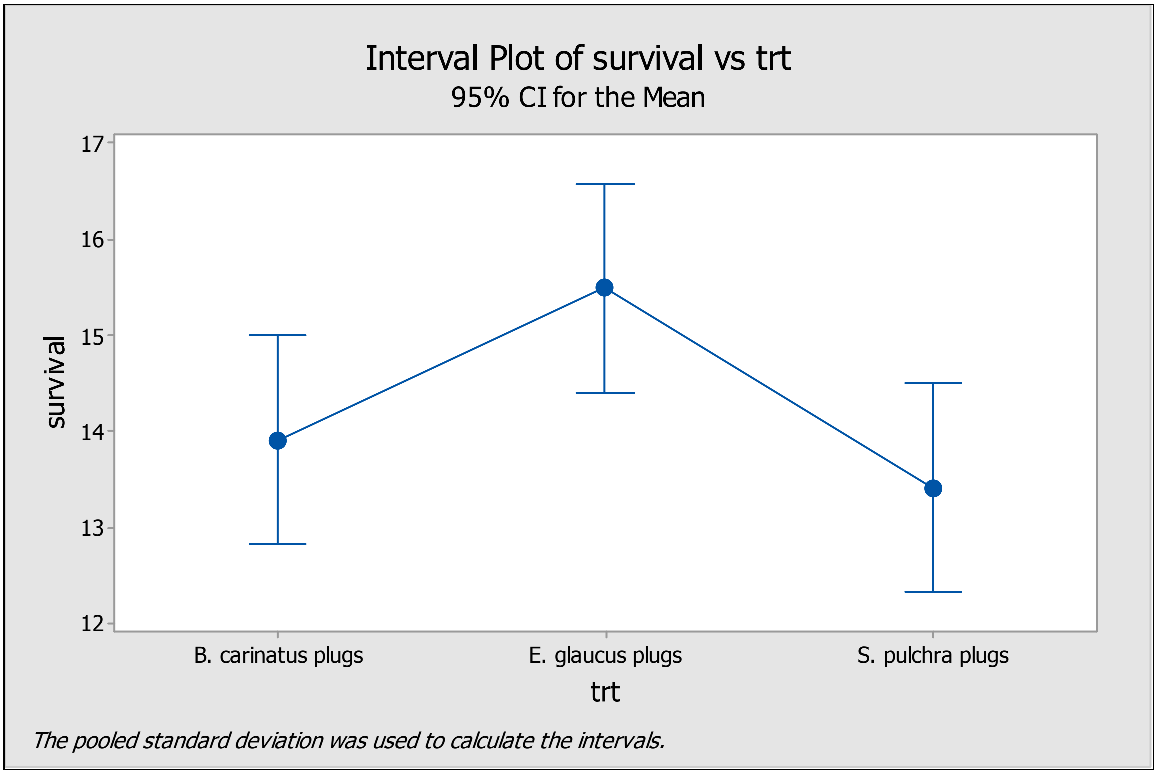

Figure 9. Mean survival total values between perennial plug treatments (*S. pulchra*, *B. carinatus* and *E. glaucus*) for BCCER.

Table 6. Tukey pairwise comparisons between the mean survival totals with standard deviation for perennial grass plug treatments (*S. pulchra*, *B. carinatus* and *E. glaucus*) in BCCER out of a total of 16 planted plugs per 1m^2 plot. Treatments that share a common letter are not significantly different.

	BCCER	
Treatment	*Mean Survival Totals*	*Groupings*
E. glaucus	15.500 (±0.674)	A
B. carinatus	13.917 (±2.539)	A B
S. pulchra	13.417 (±1.881)	B

<u>BIDWELL</u>

Stipa pulchra had the highest survival out of all three perennial grass treatments in May of 2014 followed by *B. carinatus* and *E. glaucus* (Figure 10). The ANOVA showed that the three perennial native grass plug treatment survival totals were significantly different ($F = 8.65$, $P = 0.001$). The Tukey pairwise comparisons ($\alpha = 0.05$) found the mean survival total varied with *E. glaucus* being significantly different from *S. pulchra*, however, *B. carinatus* was not found to be significantly different than either of them (Table 7).

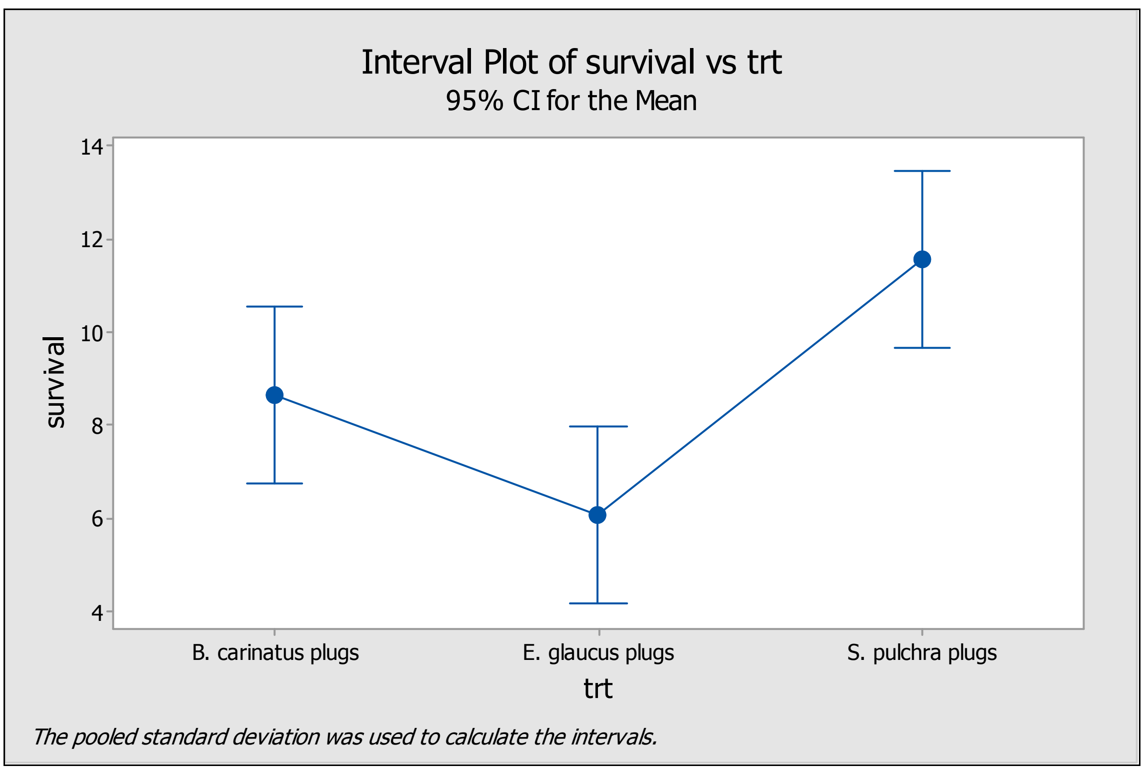

Figure 10. Mean survival total values between perennial plug treatments (*S. pulchra*, *B. carinatus* and *E. glaucus*) for BIDWELL.

Table 7. Tukey pairwise comparisons (α = 0.05) between the mean survival totals with standard deviation for perennial grass plug treatments (*S. pulchra*, *B. carinatus* and *E. glaucus*) in BIDWELL out of a total of 16 planted plugs per $1m^2$ plot. Treatments that share a common letter are not significantly different.

BIDWELL		
Treatment	*Mean Survival Totals*	*Groupings*
S. pulchra	11.583 (±3.147)	A
B. carinatus	8.667 (±2.871)	A B
E. glaucus	6.08 (±3.65)	B

Percent Cover Values from
March 2013-May 2014

BCCER

Treatments. *B. carinatus* and *E. glaucus* account for the highest treatment cover values from the start of monitoring in March through May of 2013, with *S. pulchra* and *G. camporum* following closely behind. During this period of initial growth the direct seeded treatments (mixed grasses & *M. elegans*) cover values remain competitive to the out-planted plug treatments (Figure 11; Appendix B). Through the summer months while *B. carinatus* and *E. glaucus* continue to account for the highest treatment cover values, they along with *S. pulchra* steadily decline until eventually reaching dormancy in September. *Grindelia camporum* cover values however, continue to increase as well through the summer months. The direct seeded treatments eventually become surpassed by all of the plugs treatments through the summer; with the exception of *S. pulchra* with the lowest percent cover values. By the following September *G. camporum* and *M. elegans* are the treatments with remaining cover vales (Figure 11; Appendix B).

With the onset of fall in October *S. pulchra* shows the most dramatic increase in growth from complete dormancy, along with the renewed growth from all of the other

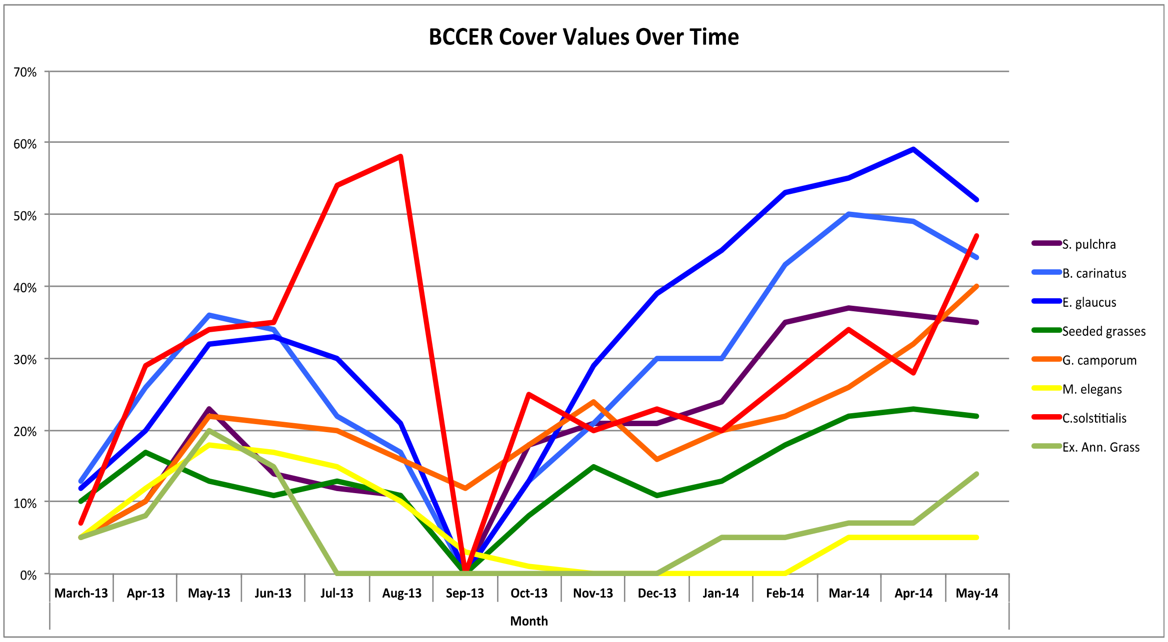

Figure 11. Cover values for native treatments along with the cover of *C. solstitialis* and non-native annual grasses found in control plots over the course of 15 months of monitoring at the BCCER site.

grass treatments. Each of the grass treatments cover values continue to increase steadily through the wet season (November-February) again lead by *B. carinatus* and *E. glaucus,* but in this case *E. glaucus* accounts for the highest values throughout. Vegetative growth continues to increase through spring of 2014 with the grass plugs accounting for the highest cover values recorded throughout the experiment. *Grindelia camporum* follows after not exhibiting similar growth to the grass treatments during the winter months. The seeded treatments show mixed results with the grass mix steadily increasing from October to May of 2014. *Madia elegans* as the only annual treatment accounts for little to no cover through the winter as expected from November to February, but then is the only treatment to show decreased cover values from the previous year (Figure 11; Appendix B).

Centaurea solstitialis. Following the burn in October of 2012 *C. solstitialis* cover values begin to reestablish by March of 2013. *Centaurea solstitialis* cover values within the establishing treatment plots generally remain slightly lower to the control plots during the initial growth cycle following the burn. An exception occurs in the seeded treatments of mixed grasses and *M. elegans* which were raked in, hampering the development of *C. solstitialis* seedlings in the process, and thus exhibit a more dramatic reduction of *C. solstitialis* cover values throughout the initial growth period in the Spring of 2013. *Centaurea solstitialis* cover values then continue to increase through its most dramatic growth period in the summer months (Figure 11; Appendix B).

The next generation of *C. solstitialis* seedlings quickly established with the fall rains in October 2013, and remain in the mid to low vegetative cover in control plots through April of 2014. October also had the greatest variation between *C. solstitialis* and treatments showing a minimum of a 10% decrease (Figure 11; Appendix B). *Centaurea solstitialis* cover values then remain fairly constant between treatments and controls from November through April of 2014. The first major shift in *C. solstitialis* growth occurs from April to May in the control plots with an increase of nearly 20% (Figure 11; Appendix B). The seeded treatments initially show slightly lower *C. solstitialis* values than plug treatments during January and February, but then are surpassed by the plug treatments in the spring from March through May of 2014. By May of 2014 the plug treatments show the greatest reduction in *C. solstitialis* cover from in the controls with a reduction of nearly 20-30%, while the seeded treatments showed a reduction of 10-15% (Figure 12). *Elymus glaucus* was found to be the most effective of all treatments reducing

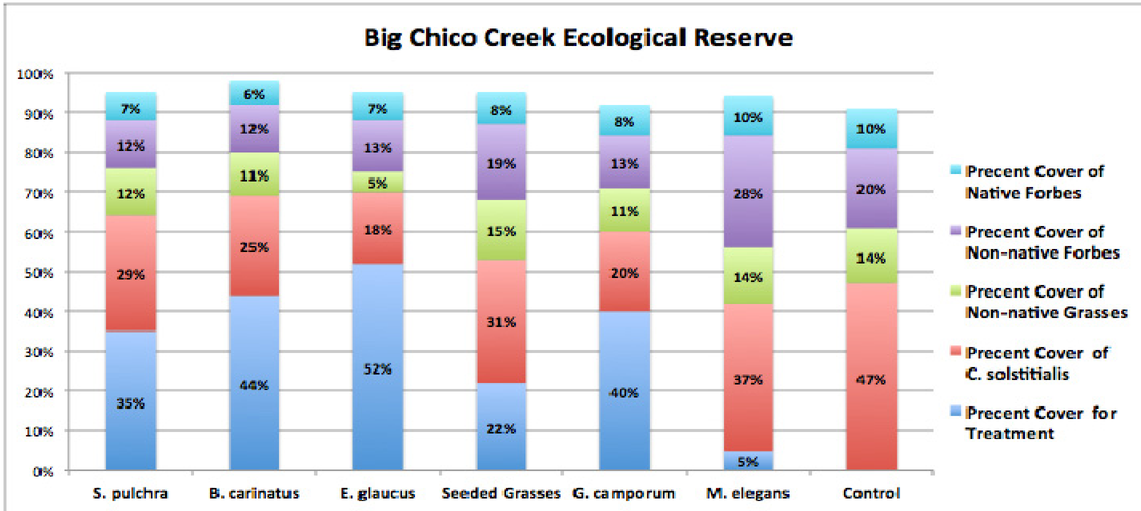

Figure 12. Shows the percent cover value distribution among all five classes for each of the seven treatments during the final month of monitoring in May of 2014. With the out-planted perennial plug treatments showing nearly a 20% reduction in *C. solstitialis* compared to the control plots.

C. solstitialis compared to its cover in the control plots by 17% in March, by 15% in April, and by 19% in May (Figure 11; Appendix B).

Non-native Annual Grasses. Non-native annual grass cover values were found to be low upon monitoring in March of 2013 following the control burn in October of 2012. Although they were initially found to be higher in some of the treatment plots than the controls. This can be seen in March and April of 2013. The transition to favor the treatments eventually occurs in May and again in June before the annual grasses complete their life cycle and senesce in July of 2013. The next generation of non-native annual grass seedlings began to emerge in January of 2014 with cover values remaining low through April of 2014 and by the final month of monitoring in May 2014 they show a slight increase (Figure 11). In each case, there is little variation between the treatment and control plots (Appendix B).

Non-native Forbs. Non-native forb cover values were found to be low, remaining below 10% upon the start of monitoring in March and April of 2013, until they show a slight increase to between in May and June of 2013. Most of the non-native forbs have senesced and set seed by July, and account for little to no cover values through December of 2013 when they return. Cover values remain low in January, and show a slight increase in February, before some of the treatments account for as much as 20% in March, April, and May of 2014. In each case during the final three months of monitoring, the *M. elegans* plots with the least amount of vegetative cover of all treatments contained the highest non-native forb cover values second to only the control plots, followed by the seeded grasses. The perennial grass plugs collectively were found to have the lowest amount of non-native forb cover through the final months of monitoring (Appendix B).

Native Forbs. Native forb values were found to make up a small amount of vegetative cover within the treatment plots beginning in March of 2013. Native forb cover values increases in both the treatment and control plots during their peak month of growth in April and begin to decline from May to July before senescing in August a month later than most of the non-native forbs. The native forbs also were found to begin germination a month after the non-native forbs in January of 2014. Native forbs then maintained low cover values in all treatments during the winter months of January and February and increase with the onset of spring in before slightly decreasing again in May of 2014. In each case little to no variation in native forb cover values were found between the treatments or control plots (Appendix B).

BIDWELL

Treatments. Following the initial planting in January of 2013 the direct seed treatments of both mixed grasses and *M. elegans* show the highest cover values in both March and April of 2013. The out-planted plug treatments remain fairly low initially in March and April before accounting for move vegetative cover than the seed treatments during the summer months of May through August of 2013. *Bromus carinatus* accounts for the highest cover values among all of the plug treatments from March through June, before it is surpassed by *E. glaucus* with slightly higher values in July and August (Figure 13; Appendix B).

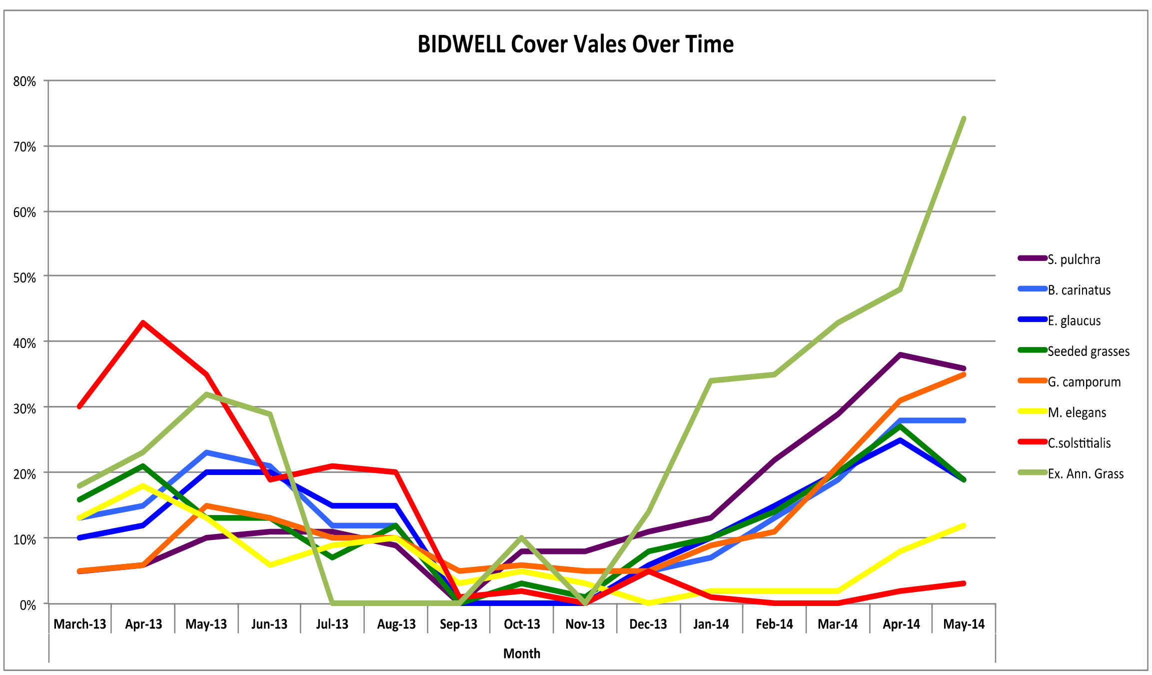

Figure 13. Cover values for native treatments along with the cover of *C. solstitialis* and non-native annual grasses found in control plots over the course of 15 months of monitoring at the BIDWELL site.

In September of 2013 once all of the perennial grasses have gone dormant, *G. camporum* and *M. elegans* account for the only vegetative cover values. Although its cover values are low *S. pulchra* along with its seedlings in the direct seeded grass mix treatment cover values begin to return in October and November, while *E. glaucus* and *B. carinatus* remained completely dormant accounting for 0% vegetative cover through December of 2013 (Figure 13; Appendix B). In January of 2014, all of the treatment cover values return along with *B. carinatus* and *E. glaucus*. *Stipa pulchra* cover values continue to remain the highest of all treatments throughout the completion of the experiment in May of 2014 (Figure 14; Appendix B). The seeded grass treatments cover

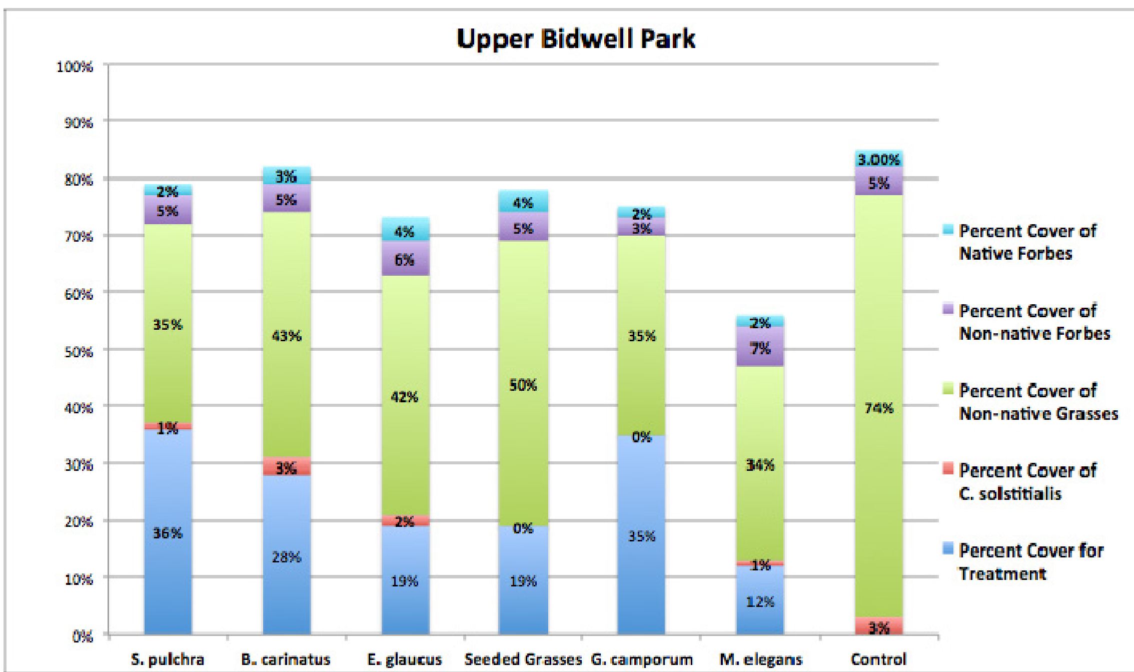

Figure 14. Shows the percent cover value distribution among all five classes for each of the seven treatments during the final month of monitoring in May of 2014. With the out-planted perennial plug treatments of *S. pulchra* and *G. camporum* accounting for the highest cover values. While *C. solstitialis* cover has been was dramatically reduced through the use of herbicides annual grasses cover greatly increased with 25% less in each of the treatments compared to the controls.

values remain fairly consistent gradually increasing from February through April. *Madia elegans* cover values also increase from January through May of 2014. *Grindelia camporum* separates from the other treatments in May as well and finished with second highest amount of vegetative cover just below *S. pulchra* (Figure 14; Appendix B).

Centaurea solstitialis. Following the burn in October of 2012, *C. solstitialis* cover values quickly reestablish by March of 2013. *Centaurea solstitialis* cover values were found to be lower within the out-planted treatment plots, and much lower in the direct seed treatment plots (Appendix B). *Centaurea solstitialis* cover values continue increase in April in the control plots, with lower cover values in both the plug and seed treatments, before being treated with herbicide in May and again in June of 2013. Cover values subsequently begin to decline from May through August in the controls as well as the treatment plots before nearly reaching total dormancy in September through December of 2013 (Figure 13; Appendix B). In January of 2014, cover values of *C. solstitialis* return for all treatments before diminishing again from February through May with slightly lower cover values in the treatment plots compared to the controls. During the final month of monitoring in May of 2014, *C. solstitialis* accounted for slightly more cover in the control plots than most of the treatment plots (Figure 14).

Non-native Annual Grasses. Non-native annual grasses accounted for nearly 20% cover values within the control plots in BIDWELL during March of 2013, following the control burn in October of 2012. Non-native annual grass cover values were found to be lower within the out-planted plug treatment plots than the control and more so within the direct seeded plots from March through and June. The annual grasses finish setting seed by July of 2013 and senesce accounting for no vegetative cover (Appendix B).

The next generation of non-native annual grass seedlings began to emerge with the first rains in October, then die back in November, and resurface again in December of 2013. Initially in December, the seeded treatments with again showed slightly lower non-native annual grass cover values than the plug treatments. This trend between the direct seeded and out-planted plug treatments changes in 2014 to favor the plug treatments when non-native annual grass cover values continue to increase each month from January through May. Although, both treatments maintain lower amounts of non-native annual grass cover throughout the final stretch from January to May of 2014. The final month of monitoring in May of 2014 showed the greatest disparity between the treatment and control plots (Figure 14). Non-native annual grass cover values were the highest of all treatments in the seeded native grass mix where they accounted for as much as 50% cover. The highest non-native grass cover among plug treatments occurred within the *B. carinatus* treatment, while the lowest values among the plug treatments were found within *S. pulchra* and *G. camporum*, which had a 39% reduction in non-native annual grass cover from control plots upon the completion of the experiment (Figure 14).

Non-native Forbs. The non-native forbs maintained lower cover values in the seeded treatments than the out-planted plug treatments during the initial months of monitoring from March through June, before senescing in July of 2013. Non-native forb cover along with the annual grasses begin to return in October, fall dormant in November, and reoccur in December of 2013. Values remain below 15% through the duration with the month of April 2014 having the highest cover values. The seeded grass treatments maintain the lowest non-native forb cover values during this period from December through May 2014 (Appendix B).

Native Forbs. Native forbs account for the lowest percent cover values of all five-cover classes in BIDWELL. Cover values remained below 10% from March through June of 2013 with April having the highest amount of native forb cover. Many of the native forbs have senesce by July accounting for little to no vegetative cover. Along with the non-native forbs and annual grasses, cover values return briefly in October, and then drop out in November. Native forbs cover then returns from December through January of 2014. Cover values show a slight increase from February through May of 2014, but remain quite low between 0-5% with little to no variation between the treatment and control plots (Appendix B).

Statistical Analysis of Treatment Mean Cover Values for May 2014

BCCER

The one-way ANOVA showed the treatments mean cover values to be statistically significantly ($F = 44.75$, $P = <0.0001$) for the final month of monitoring in May 2014. The Tukey pairwise comparisons ($\alpha = 0.05$) found the mean cover values to vary between treatments (Table 8). The out-planted plug treatments accounted for higher mean cover values than the direct seeded treatments (Figure 15). The *E. glaucus* plug treatment ended with the highest mean cover value of 52% of the $1m^2$ plots. *Bromus carinatus* and *G. camporum* treatments followed with 43% and 40% and were statistically similar to each other as well as *E. glaucus* and the *S. pulchra* treatments. *Elymus glaucus* (52%) however was found to be significantly different than *S. pulchra* (33%). The seeded grass treatments were found to be statistically similar to the *S. pulchra* treatments but were significantly different than the *M. elegans* treatment, and were found

Table 8. Mean cover values for 5/14 from the Tukey pairwise comparisons for all treatments for BCCER. Treatments with the same letter were not found to be significantly different from each other.

May 2014	*BCCER*	
Treatment	*Mean CV*	*Groupings*
E. glaucus	52.08 (±7.53)	A
B. carinatus	43.75 (±12.99)	A B
G. camporum	40.42 (±16.58)	A B
S. pulchra	33.33 (±7.78)	B C
Seeded grass	21.67 (±13.37)	C
M. elegans	5.00 (±0.00)	D

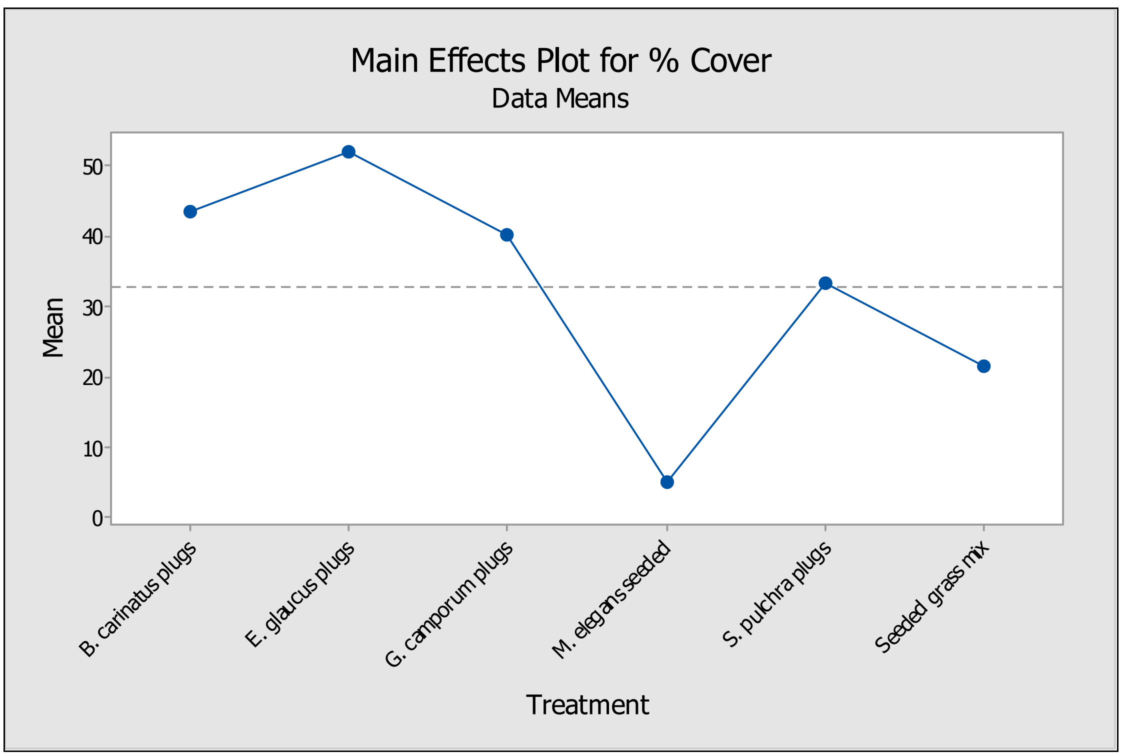

Figure 15. Mean cover values for all treatments upon the completion of the experiment in May of 2014 for BCCER.

to be significantly lower than all other treatments with a mean cover value of 5% (Table 8).

BIDWELL

The one-way ANOVA showed that the mean cover values varied significantly between treatments BIDWELL (F = 11.06, P = <0.0001) during the final month of monitoring in May 2014. The Tukey pairwise comparisons (α = 0.05) found the mean cover values to vary between treatments (Table 9). The out planted plug treatments accounted for higher mean cover values than the direct seeded treatments (Figure 16). *Stipa pulchra* accounted for the highest mean cover value occupying 37% of the $1m^2$ plots. *Grindelia camporum* and *B. carinatus* followed with lower mean cover values of 35% and 28% and were statistically similar to the *S. pulchra* and *E. glaucus* treatments. *Stipa pulchra* and *E. glaucus* were found to be significantly different from each other with *E. glaucus* in this case showing the lowest mean cover values of the out planted plug treatments. The seeded grass treatment was found to be significantly lower than both the *S. pulchra and G. camporum* treatments, but was not significantly different than any of the other treatments. The *M. elegans* treatment was not significantly different from the seeded grass treatment or the *E. glaucus* plug treatment, but was significantly lower than the rest (Table 9).

Statistical Analysis of *C. solstitialis* Mean Cover
Values for May 2014

BCCER

The one-way ANOVA confirmed that treatments did have an effect on the mean cover values of *C. solstitialis* (F = 6.21, P = 0.000). Treatments were shown to reduce the mean cover values of *C. solstitialis* by at least 10% and some by over half when compared to the control (Figure 17). Tukey pairwise comparisons (α = 0.05)

Table 9. Mean cover values for 5/14 from the Tukey pairwise comparisons for all treatments for BIDWELL. Treatments with the same letter were not found to be significantly different from each other.

May 2014	*BIDWELL*	
Treatment	*Mean CV*	*Groupings*
S. pulchra	36.25 (±8.29)	A
G. camporum	35.42 (±15.29)	A B
B. carinatus	28.33 (±17.36)	A B C
E. glaucus	20.00 (±12.25)	B C D
Seeded grass	18.75 (±11.10)	C D
M. elegans	11.67 (±15.72)	D E

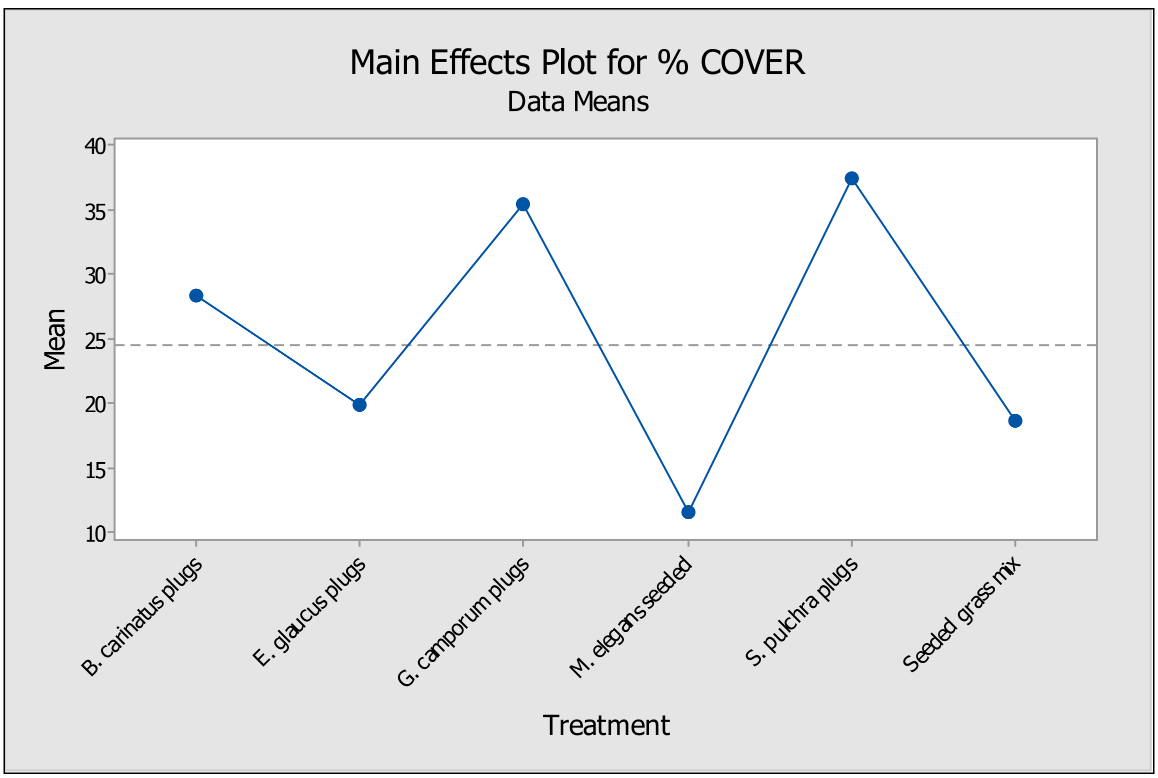

Figure 16. Mean cover values for all treatments upon the completion of the experiment in May of 2014 for BIDWELL.

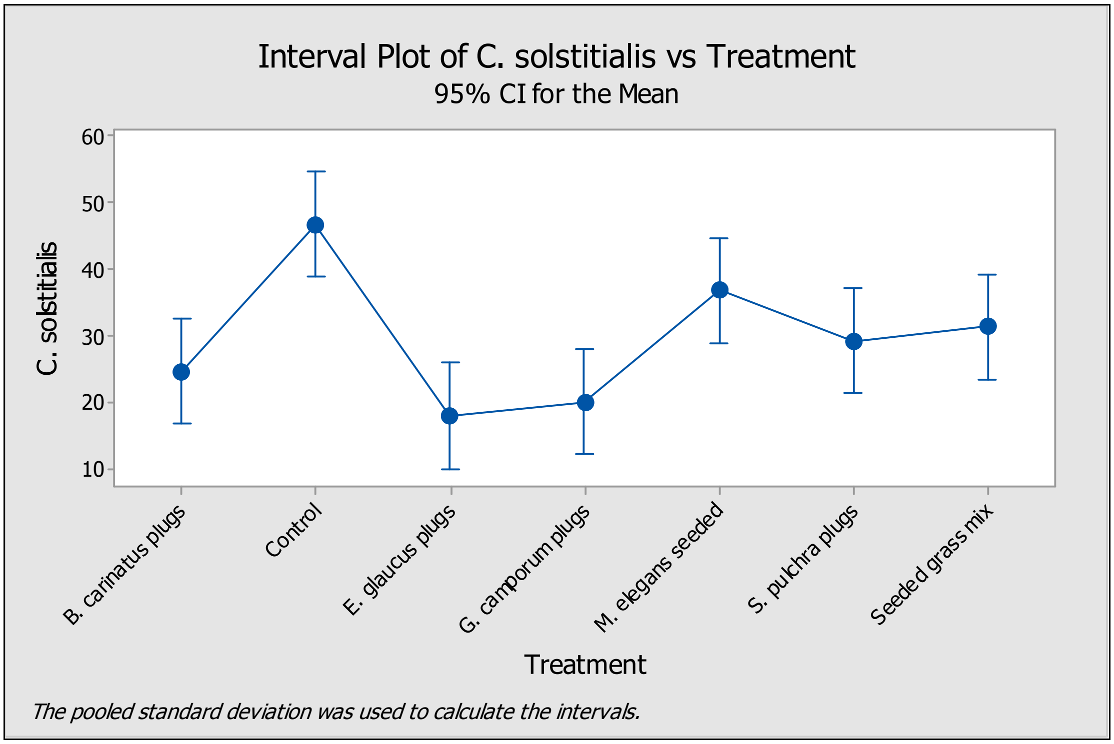

Figure 17. Show the variations between given treatments and the mean vegetative cover values for *C. solstitialis* during April of 2014 in BCCER.

showed significant differences were found between each of the out-planted plug treatments and the control while the seeded treatments were found to be statistically similar to the control (Table 10).

Table 10. Mean vegetative cover values with treatments with the lower *C. solstitialis* vegetative cover values on top and higher amounts below during May of 2014 in BCCER.

April 2014	*BCCER*			
Treatment	*C. solstitialis*	*Groupings*		
E. glaucus	17.92 (±8.91)			C
G. camporum	20.00 (±10.44)		B	C
B. carinatus	24.58 (±15.44)		B	C
S. pulchra	29.17 (±11.65)		B	C
Seeded grass	31.25 (±13.16)	A	B	C
M. elegans	36.67 (±18.38)	A	B	
Control	46.67 (±16.56)	A		

BIDWELL

The one-way ANOVA showed the treatments to have a significant effect on the mean cover values of *C. solstitialis* ($F = 2.32$, $P = 0.041$). Tukey pairwise comparisons ($\alpha = 0.05$) however, showed no significant variation between treatments and *C. solstitialis* (Table 11). Here the mean cover values and scale are greatly reduced for *C. solstitialis* to below 4% from herbicide applications in May and again in June of 2013 to the BIDWELL site (Figure 18).

Table 11. Mean vegetative cover values with treatments with the lower *C. solstitialis* vegetative cover values on top and higher amounts below during May of 2014 in BIDWELL.

April 2014	*BIDWELL*	
Treatment	*C. solstitialis*	*Groupings*
G. camporum	0.00 (±0.00)	A
Seeded grass	0.00 (±0.00)	A
S. pulchra	0.833 (±1.946)	A
M. elegans	1.250 (±2.261)	A
E. glaucus	2.083 (±2.575)	A
B. carinatus	2.5 (±3.371)	A
Control	2.5 (±3.99)	A

Statistical Analysis of Non-Native Annual Grass Mean
Cover Values for April 2014

BCCER

The one-way ANOVA and Tukey pairwise comparisons ($\alpha = 0.05$) showed that the treatments did not have an effect on non-native annual grasses mean cover values ($F = 0.93$, $P = 0.481$) (Table 12). The annual grasses cover values remained fairly low with under 10% in April of 2014. The mean cover values were found to remain fairly low ranging between 6-10% in treatments (Figure 19).

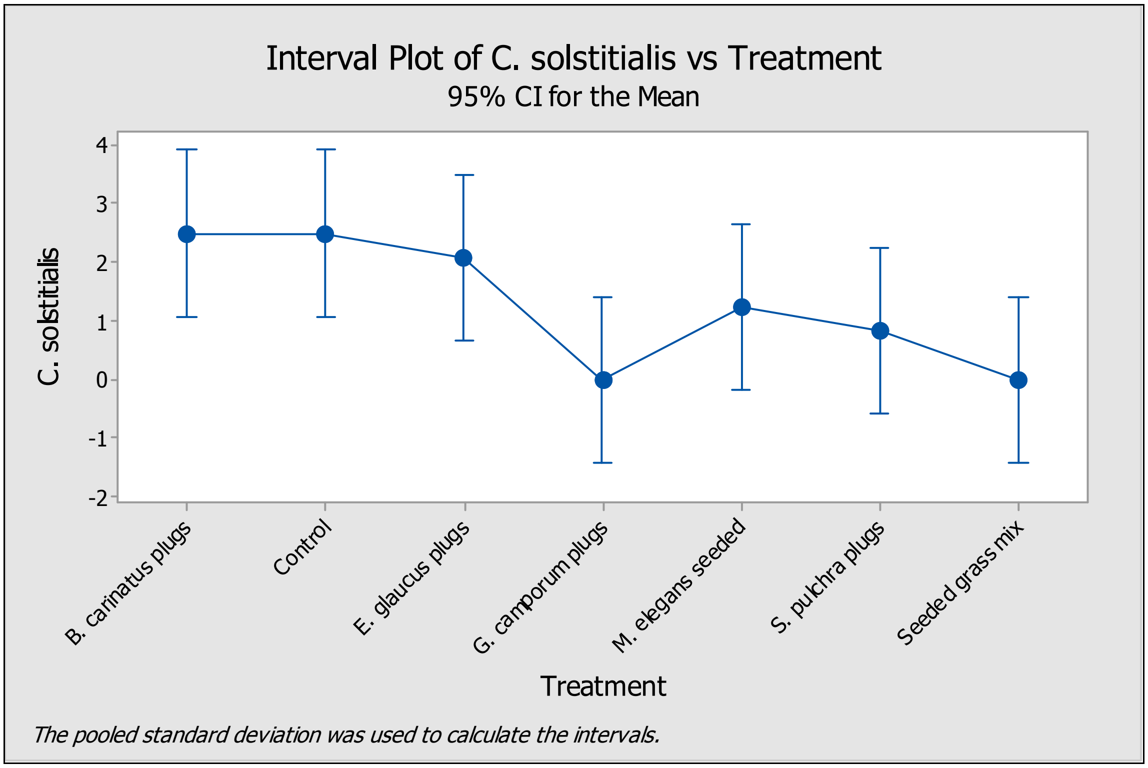

Figure 18. Show the variations between given treatments and the mean vegetative cover values for *C. solstitialis* during April of 2014 in BIDWELL.

Table 12. Non-native annual grasses mean vegetative cover values with treatments with the lower amounts on top and higher amounts below during April of 2014 for BCCER.

April 2014	*BCCER*	
Treatment	*Annual Grass CV*	*Groupings*
E. glaucus	6.250 (±2.261)	A
M. elegans	6.667 (±3.257)	A
Control	7.083 (±3.343)	A
G. camporum	7.500 (±2.611)	A
S. pulchra	7.500 (±2.611)	A
Seeded grass	7.500 (±2.611)	A
B. carinatus	8.750 (±3.108)	A

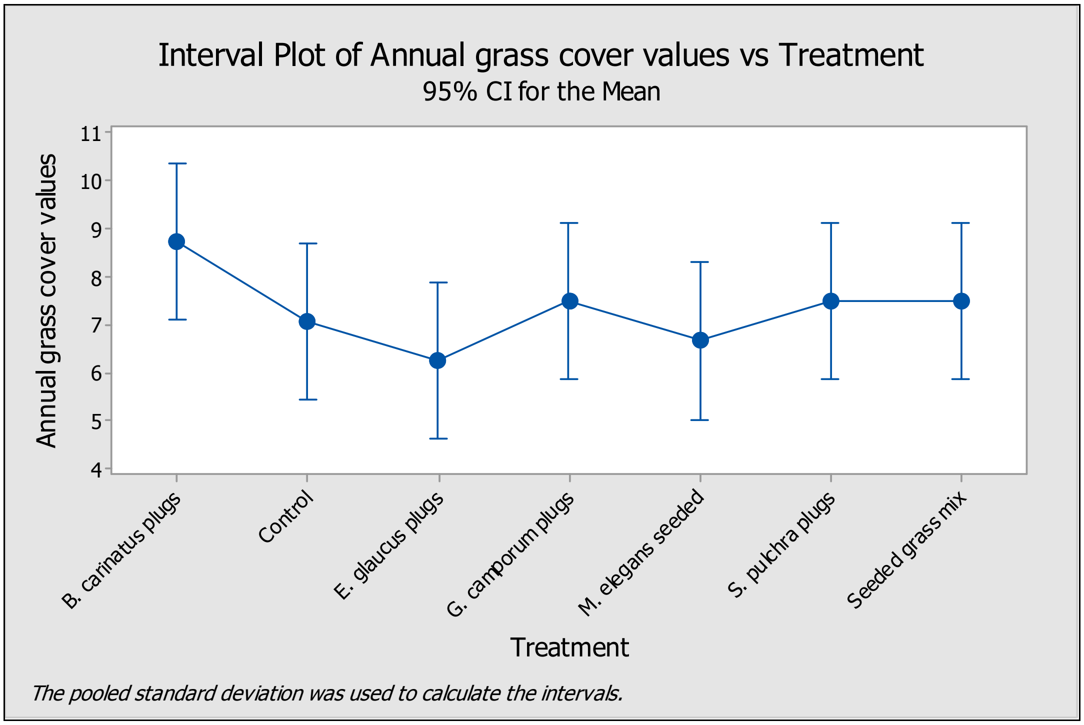

Figure 19. Variations between given treatments and the mean vegetative cover values for non-native annual grasses during April of 2014 in BCCER.

BIDWELL

The one-way ANOVA showed that the treatments had a significant effect on non-native annual grasses mean cover values ($F = 4.19$, $P = 0.001$). Non-native annual grasses mean cover values ranged between 20-40% in April of 2014 (Figure 20). The Tukey pairwise comparisons ($\alpha = 0.05$) also showed significant differences between the seeded treatments and non-native grass cover. *Madia elegans* plots maintained the lowest mean cover values (20.83%), while the seeded grass treatment had the highest mean cover value (38.33%). All of the treatments were found to have a lower amounts of non-native annual grass than the control having nearly 48% cover but only the *M. elegans* treatment was found to be significantly different in the Tukey pairwise comparison (Table 13).

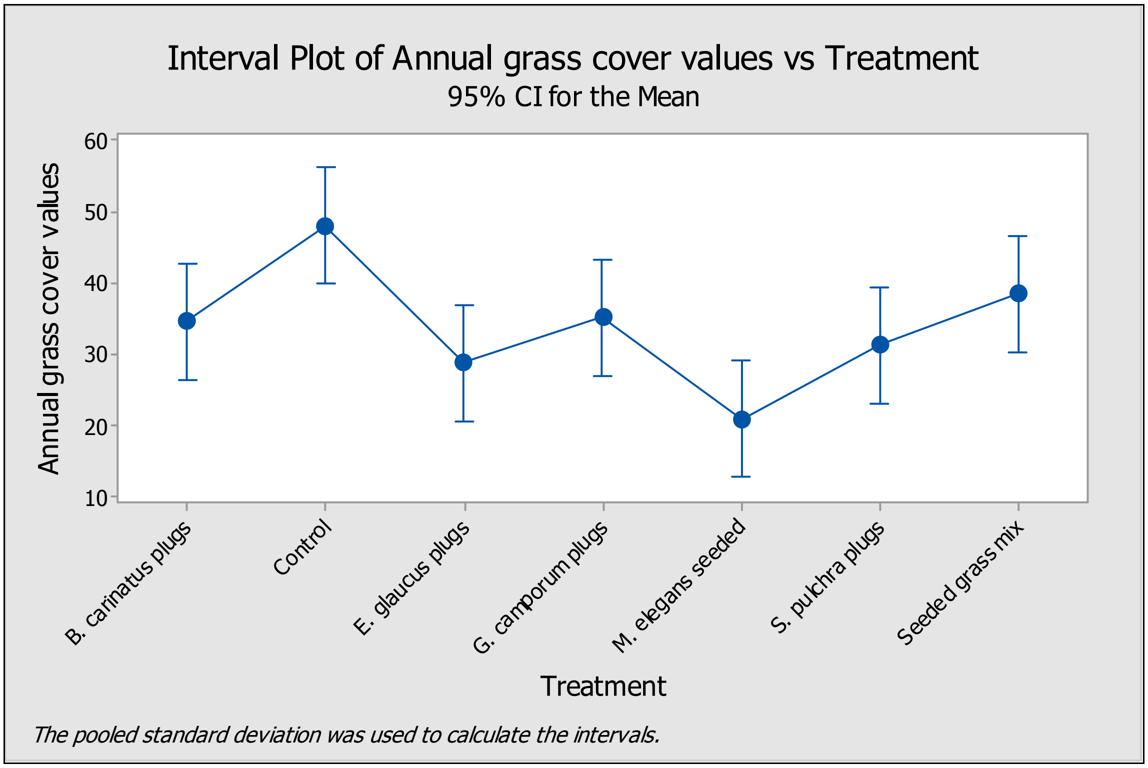

Figure 20. Variations between given treatments and the mean vegetative cover values for non-native annual grasses during April of 2014 in BIDWELL.

Table 13. Non-native annual grasses means vegetative cover values with treatments with the lower amounts on top and higher amounts below during April of 2014 for BIDWELL.

April 2014	*BIDWELL*		
Treatment	*Annual Grass CV*	*Groupings*	
M. elegans	20.83 (±9.00)	A	
E. glaucus	28.75 (±13.16)	A	B
S. pulchra	31.25 (±16.69)	A	B
B. carinatus	34.58 (±15.73)	A	B
G. camporum	35.00 (±12.79)	A	B
Seeded grass	38.33 (±12.49)		B
Control	47.92 (±18.52)		B

Statistical Analysis of Non-native Forb Mean Cover Values for April 2014

BCCER

The one-way ANOVA revealed that all of the treatments as well as the control did not have a significant effect on the cover value of non-native forbs ($F = 1.30$, $P = 0.265$). The Tukey pairwise comparisons ($\alpha = 0.05$) also found no significant correlation between all of the treatments and the mean cover values of non-native forbs (Table 14). However, the seeded and control plots showed higher mean cover values for non-native forbs than all of the plot treatments, but again the difference was not found to be significant (Figure 21).

Table 14. Non-native forb mean vegetative cover values during April of 2014 in BCCER

April 2014	*BCCER*	
Treatment	*Non-Native Forb CV*	*Groupings*
E. glaucus	11.67 (±3.89)	A
S. pulchra	13.75 (±4.33)	A
B. carinatus	13.75 (±4.33)	A
G. camporum	15.42 (±4.50)	A
Seeded grass	17.08 (±12.33)	A
Control	18.75 (±10.25)	A
M. elegans	20.00 (±15.67)	A

BIDWELL

The one-way ANOVA revealed that all of the treatments as well as the control did not have a significant effect on the cover value of non-native forbs during April of 2014 ($F = 2.08$, $P = 0.065$). The Tukey pairwise comparisons ($\alpha = 0.05$) found no significant correlation between all of the treatments and the mean cover values of non-native forbs (Table 15). The seeded *M. elegans* and control treatments showed higher

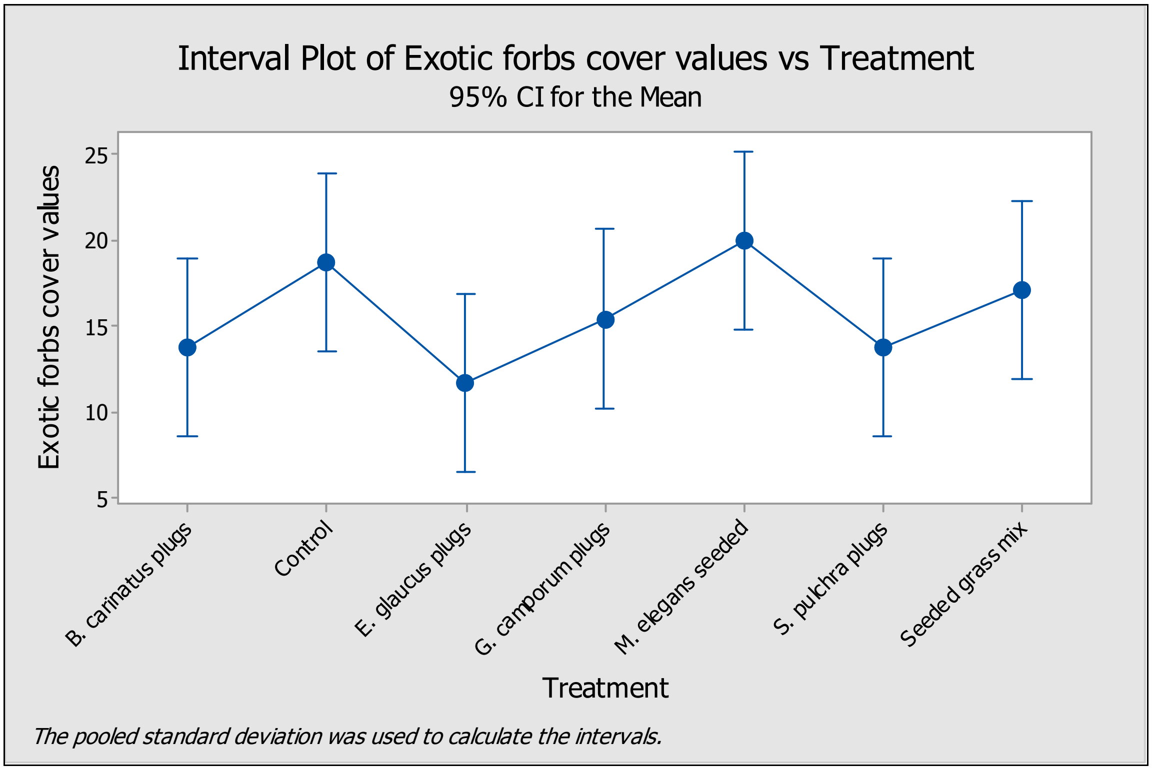

Figure 21. Variations between given treatments and the mean vegetative cover values for non-native forbs during April of 2014 in BCCER.

mean cover values for non-native forbs than all of the other treatments with the exception of *E. glaucus* which had the highest overall non-native forb mean cover value, but again the difference was not significant (Figure 22).

Table 15. Non-native forb mean vegetative cover values during April of 2014 in BIDWELL.

April 2014	*BIDWELL*	
Treatment	*Non Native Forb CV*	*Groupings*
Seeded grass	7.917 (±5.97)	A
B. carinatus	7.92 (±3.96)	A
G. camporum	8.75 (±4.33)	A
S. pulchra	9.17 (±5.97)	A
M. elegans	11.25 (±4.33)	A
Control	12.08 (±7.82)	A
E. glaucus	13.75 (±6.78)	A

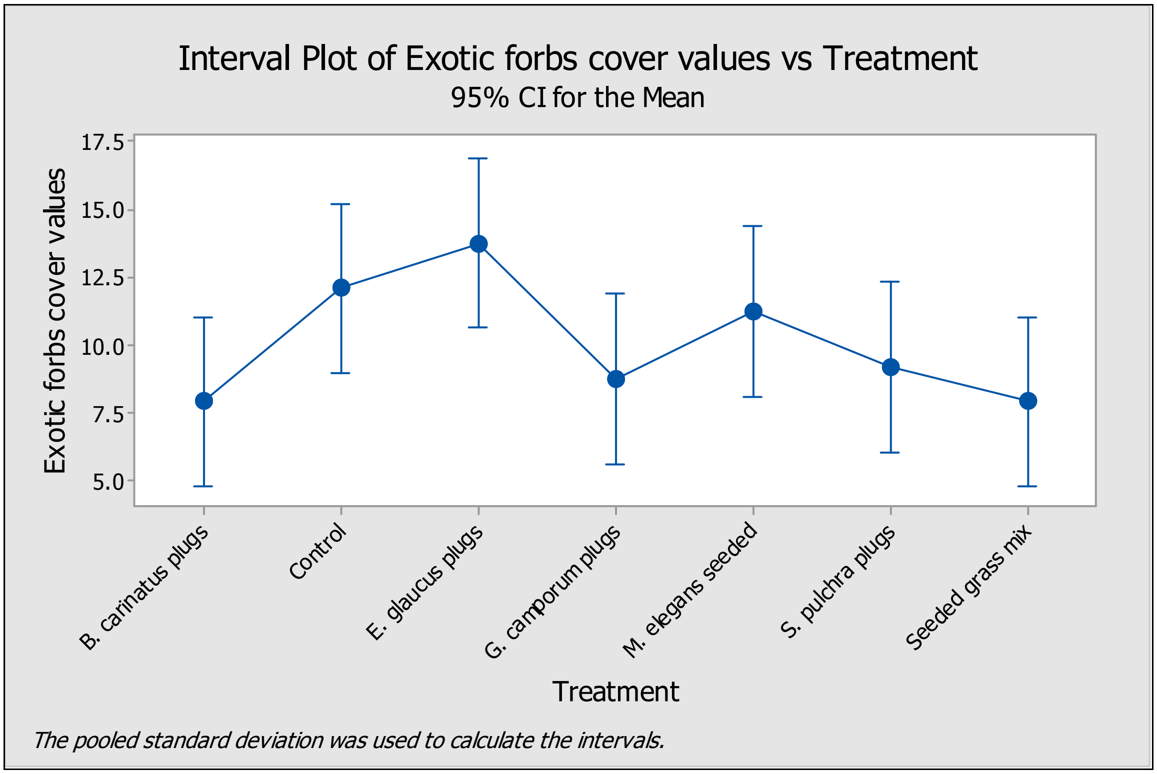

Figure 22. Variations between given treatments and the mean vegetative cover values for non-native forbs during April of 2014 in BIDWELL.

Native Forb Mean Cover Values for
April 2014

BCCER

The one-way ANOVA revealed that the treatments were found to have a significant effect on the vegetative cover of native forbs with a (F = 5.14, P = <0.0001). The Tukey pairwise comparisons (α = 0.05) found the mean cover values to vary between treatments. Native forb mean cover values were found to be higher in the seeded treatments and control than all of the out-planted plug treatments (Figure 23). The *M. elegans* treatment was found to be significantly higher than each of the perennial grass plug treatments (Table 16).

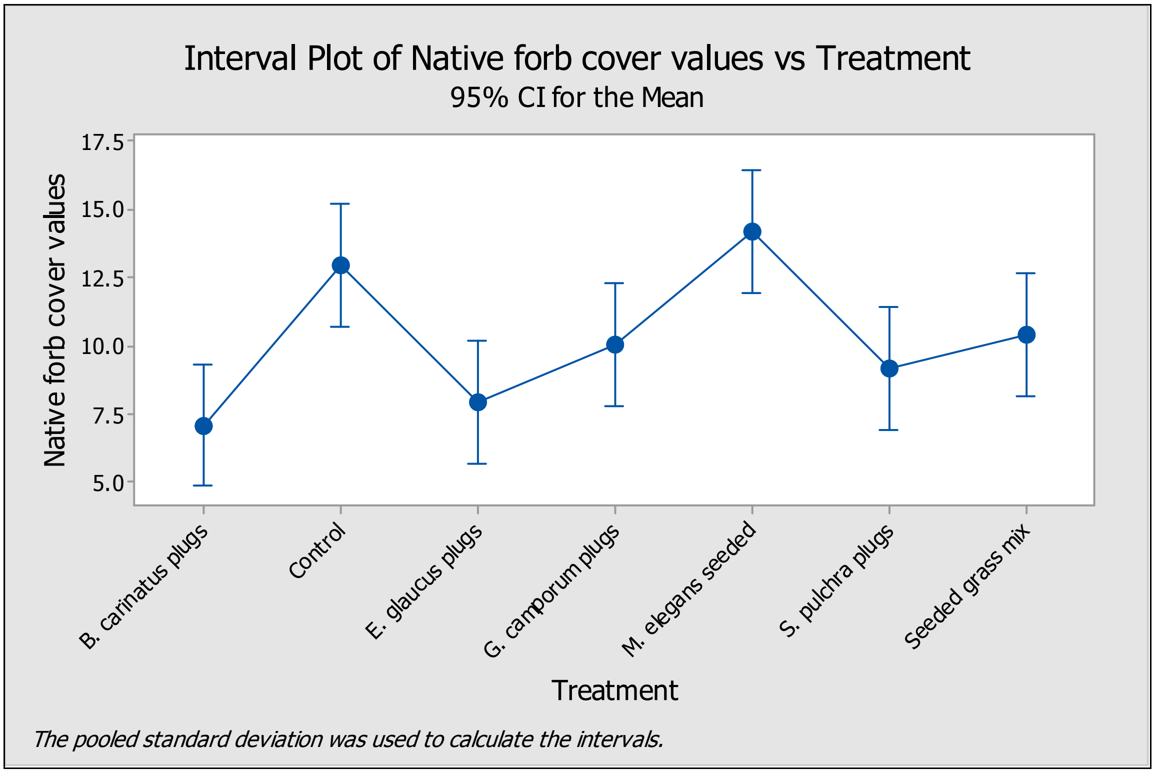

Figure 23. Variations between the treatments and native **forb** mean vegetative cover values during April of 2014 in BCCER

Table 16. Native forb mean vegetative cover values among the different treatments listed from those with the most to least during April of 2014 for BCCER.

April 2014	*BCCER*	
Treatment	*Nat. Forb CV*	*Groupings*
M. elegans	14.17 (±7.78)	A
Control	12.92 (±4.50)	A B
Seeded grass	10.42 (±4.50)	A B C
G. camporum	10.00 (±3.015)	A B C
S. pulchra	9.167 (±2.887)	B C
E. glaucus	7.917 (±3.343)	C
B. carinatus	7.083 (±2.575)	C

BIDWELL

The one-way ANOVA revealed the treatments were not found to have a significant effect on the vegetative cover of native forbs with a ($F = 0.71$, $P = 0.640$). The Tukey pairwise comparisons ($\alpha = 0.05$) showed no significant difference between any of the treatments or control (Table 17). The mean native forb cover values were found to occupy as little as 1-5% vegetative cover in all of the treatments in April 2014 (Figure 24).

Table 17. Native forb mean vegetative cover values among the different treatments listed from those with the most to least during April of 2014 for BIDWELL.

April 2014	*BIDWELL*	
Treatment	*Nat. Forb CV*	*Groupings*
B. carinatus	5.00 (±6.40)	A
E. glaucus	4.58 (±6.20)	A
Seeded grass	3.75 (±6.44)	A
Control	3.33 (±3.89)	A
M. elegans	2.92 (±5.42)	A
G. camporum	2.50 (±3.99)	A
S. pulchra	1.250 (±3.108)	A

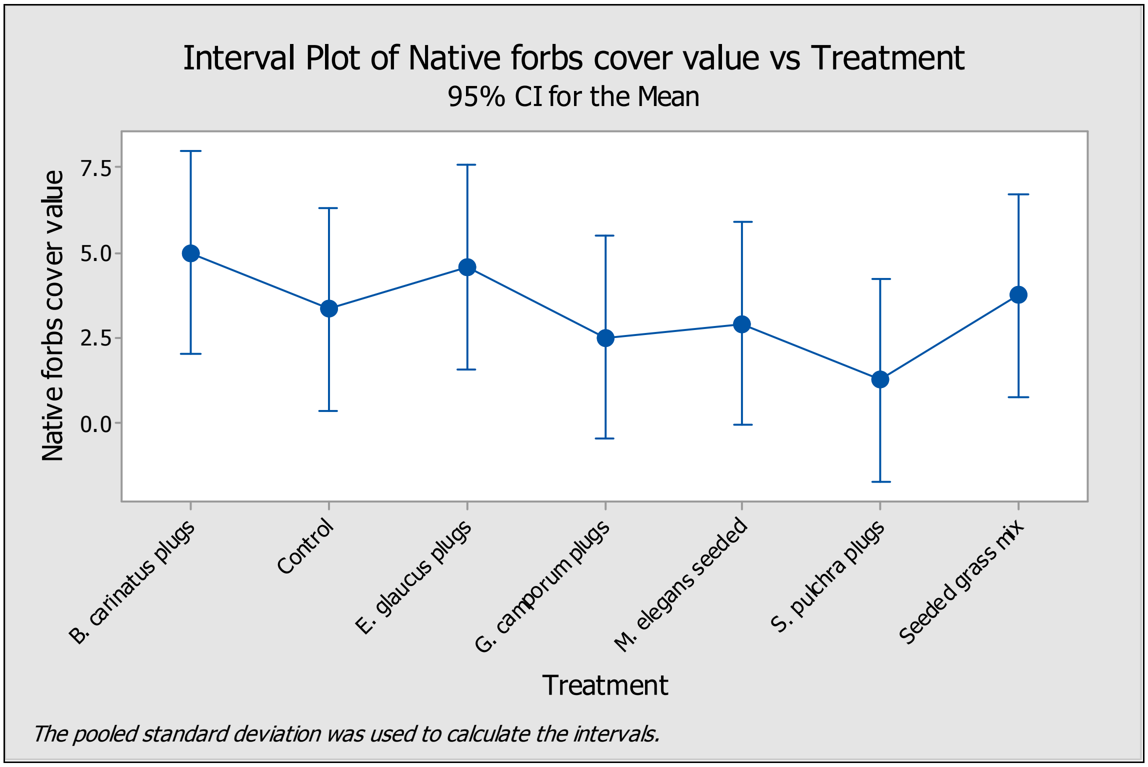

Figure 24. Variations between the treatments and native forb mean vegetative cover values during April of 2014 in BIDWELL.

CHAPTER V

DISCUSSION

The field of restoration ecology is rapidly growing with its continued need for application to degraded ecosystems. Ecological restoration in conjunction with ecological theory and scientific study present an opportunity to increase the understanding of the mechanisms behind the successes and failures of the restoration work. Studies focused on the use of restoration techniques that are designed to monitor and assess the management strategies directly contribute toward the advancement of the field. This process will add to the body of knowledge surrounding the ecological principals of restoration and provide necessary insights to restore the health and function of ecosystems. This information can be readily applied for the stewardship of California grasslands that have been altered through land use and the introduction of non-native species. While methods to reduce invasive species have been found, their long-term success has been limited. The revegetation of native species following the reduction of invasive species may help to shift the competitive advantage and provide long-term stability. Here, I focused on assessing the effectiveness of revegetation using native grassland taxa and monitored their interactions with non-native species following a prescribed burn.

Out-planted species are often most vulnerable during the early stages of development (Corbin et al. 2007b). Greenhouse propagation along with nutrients in potting soils and regular watering conditions optimize growth, but can lead to the development of quick growing softer tissues that make them more susceptible to herbivory in the field. The trend of increased herbivory tends to perpetuate if out-planted species are given supplemental irrigation. Often restoration occurs in areas where irrigation is not easily accessible or within the budget and scope of a project. In this scenario the timing of planting is paramount to its success. Planting efforts that are carried out after the first few rains in the fall allow the out-planted natives time to allocate energy towards root development over the course of the winter (Jackson & Roy 1986; Holmes & Rice 1996) and prepare them for the summer dry season are often the most effective. Once out-planted native species have endured their first summer and had a year to establish, survival rates have been shown to increase (Corbin et al. 2007b).

The perennial out-planted plugs were found to be quite effective with high survival percentages 17 months following their initial planting in January of 2013. The BCCER locations survival ranged from (85-97%), with a total of 620 out of the original 684 (91%) alive in May of 2014, while the BIDWELL site ranged from (38-79%) and finished with 409 out of the 684 (60%) surviving (Figure 4 and 6). These rates are remarkable considering they were not given irrigation during a record year of drought in 2013. Survival numbers varied throughout the 15 months of data collection depending on the season and climate during recording which points to the importance of taking into account the timing and duration of monitoring when evaluating data to determine the results of restoration efforts (Figures 3 and 5; Appendix A).

The invaded grassland site at BCCER responded quite well to revegetation efforts following the prescribed burn. Plants established quickly in the conditions of the site in loose loam soils with slightly more precipitation and cooler temperatures upstream entering the foothills. Each of the out-planted plug treatments finished with greater than 85% survival in May of 2014. While all of the out-planted plugs performed quite well *E. glaucus* (97%) and *G. camporum* (95%) ended with the highest survival values (Figure 4). *Elymus glaucus* (52%) and *B. carinatus* (44%) exhibited quick growth that enabled them to occupy the highest cover of the perennial grasses within the plots (Figure 11). *Elymus glaucus* generally performs well in grassland restoration projects; however, I was surprised to see the success of *B. carinatus* which is less commonly used. It is worth noting that *B. carinatus* was one of the only perennial grasses which dispersed seeds within the scope of the study with the next generation of seedlings occurring within plots during the second year of monitoring.

The invaded grassland site at BIDWELL downstream closer to the valley floor had harder clay soils with slightly less precipitation and warmer temperatures, which along with the accidental herbicide treatments contributed to mixed results. *Stipa pulchra* (77%) and *G. camporum* (79%) ended with the highest survival rates while *B. carinatus* and *E. glaucus* values fell (Figure 6). *Stipa pulchra* (36%) and *G. camporum* (35%) also accounted for the most vegetative cover of all of the treatments (Figure 14). In addition, *S. pulchra* was also the first of the perennial native grasses to return from dormancy in October, while *B. carinatus* and *E. glaucus* seemed to require more saturation in order to send up their first shoots, which did not emerge until December (Figures 5 and 13; Appendices A and B).

Unfortunately, no true comparisons can be made between sites due to the accidental herbicide application to the BIDWELL location in May and June of 2014, but it does present another factor to evaluate. Although the direct treatment of *C. solstitialis* dramatically reduced competition in the BIDWELL site, it also increased the amount of light exposer and heat into the plots, which along with the effect of the herbicide itself may have hampered treatments development. The morphological traits of the perennial grass species led to differences in growth habits that may have also caused a variation. While comparisons have been made between the seasonal growth rates and water use of non-native annual and native perennial grasses (Holmes & Rice 1996; Dyer & Rice 1999), less is known about variations between native perennials grasses themselves. Although the native perennials used in this study are all ‘cool season’ C3 grasses, the dense thin leaves of ‘needle grass’ may have enabled them to maintain more water than the thicker leaves of *B. carinatus* and *E. glaucus* with presumably higher rates of transpiration (Parkurst & Loucks 1972; Smith & Geller 1980). This could account for the fact that *S. pulchra* had higher survival and was able to re-establish from dormancy several months before the other perennial grass at the BIDWELL site.

Another interesting comparison is between the out-planted grass plugs and the direct seeded mixed grass treatments. In May of 2014 the cover values in both sites favored the out-planted plugs, although more so in the BCCER site [plugs (35-52%), mixed seed (22%)], than BIDWELL [plugs (19-36%), mixed seed (19%)]. Again just as with the out-planted plugs, *S. pulchra* seedlings outperformed the other two perennials grasses which accounted for more cover within the mixed seeded grass plots in the BIDWELL location. Although the results favor the out-planted plugs in each case, the role of direct seeding cannot be under estimated in grassland restoration. Direct seeding techniques allow for broader applications to open areas with the use of seeding and tilling equipment. However, it may not be the best choice for locations with varying terrain and access points. Grassland restoration using direct seeding can also require more time; the seedlings of *S. pulchra* have been shown to take as long as seven years to establish (Stromberg 2007a). Seeded grasses have also benefit from continued weeding in the early stages in order to get the best results (Bugg et al. 1997), which was not within the scope of my project. Once seeded perennial native grasses have had time to establish they become less affected by non-native annual grasses after two to four years of growth (Corbin & D'Antonio 2004). So more time may be required to assess the success of direct seeding techniques over the long-term.

Just as the timing of out-planting is paramount to its success, the timing of the sowing of seeds may provide an opportunity to benefit from the variation in life stages between native and invasive species (Stromberg et al. 2007a). Land use history and the duration of the invasive species presence will affect the amount of their seed-bank and initially, prescribed burns may actually be to their benefit. The release of nutrients along with decreased competition following the burn can lead to a resurgence of invasive seedlings and potentially increase their populations. This period of invasive reestablishment may however present a window of time for eradication, preventing seed formation and greatly diminishing their seed-bank. In fact, the seed-bank of non-native annual grasses are short lived (Rice 1989), and properly timed tilling has been used as an effective means to reduce their seed-bank (Stromberg et al. 2002; Stromberg et al. 2007a). *Centaurea solstitialis* seedlings emerged hardily following the fall burns in 2012 (Appendix B). In December of 2012 two months after the prescribed burn the sowing of the direct seed treatments coincided with the first re-sprouts of annual grasses and *C. solstitialis*. This had the additional benefit of wiping out invasive seedlings while raking in the native seeds and is seen in the initial reduction of non-native grasses and *C. solstitialis* cover following the direct seed treatments in spring of 2013 (Appendix B).

Consecutive treatments are imperative to successfully reduce invasive seed banks in grasslands (DiTomaso et al. 1999; Kyser & DiTomaso 2002). Following the prescribed burn in October of 2012, the use of herbicide in the summer of 2013 proved to be quite effective in reducing the percentage of *C. solstitialis* in BIDWELL (Figure 13 and Appendix B). Integrated approaches using herbicide following prescribed fire have been found to be effective in other field studies (DiTomaso et al. 2003; Torrence 2003; DiTomaso et al. 2007a). Herbicide treatments following fire over a six year period were successful in the suppression of non-native annual grasses and *C. solstitialis*, however areas which were not seeded or planted with another competitor following treatments reverted to annual grasses and eventually *C. solstitialis* (Enloe et al. 2005). A three-year consecutive burn study for *C. solstitialis* resulted in a 99% decrease, but without continued treatments it reestablished over the course of four years to preexisting levels. (DiTomaso et al. 1999; Kyser & DiTomaso 2002).

The herbicide application to the BIDWELL site, as with other ‘disturbances’ provided an opportunity for early successional colonizers to move in; in this case the non-native annual grasses. In many cases this is perceived as supporting the lesser of two evils, while in fact one facilitates the spread of the other by setting the stage for further *C. solstitialis* invasion. In May 2013, before the herbicide application in BIDWELL, non-native annual grasses accounted for 32% cover in control plots and by May 2014 the amount had risen to 74% (Figure 13; Appendix B). In May 2014, just a year following the herbicide treatments *C. solstitialis* had reestablished to 3% cover in the BIDWELL control plots. While this may not seem like a lot, with one plant capable of producing 75,000 seeds (DiTomaso & Healy 2007), without continued control it would not take long for their reestablishment. This cycle of invasion has contributed to the repeated use of herbicides in California's grasslands for *C. solstitialis* control.

The primary goal of my study was to test whether the competitive effect of native grassland species on invasive species could aid in their control. The use of herbicide on *Centaurea solstitialis* at the BIDWELL site unfortunately disrupted that process, but the effect of the native treatments on non-native annual grasses can still be evaluated. In May of 2014, the treatment plots finished with 34-50% annual grass cover, while the controls amounted to 74%, an average reduction of 33% (Figure 14). *The Madia elegans* treatment was found to have significantly less non-native annual grasses when compared to the controls in April of 2014 and had 40% less during May (Appendix B). In the BCCER location where the effect of the treatments on *C. solstitialis* could be assessed, all of the out-planted plug treatments were found to be significantly different than the control plots, with an average reduction of *C. solstitialis* cover of 24% (Figure 12). The *Elymus glaucus* treatment was found to be the most effective of all the out-planted plugs, reducing the cover of *C. solstitialis* by as much as 28% (Figure 12). These data support that not only can native grassland species survive through the process of restoration, but they can also hamper the growth and development of both non-native annual grasses as well as *C. solstitialis*.

While perennial grasses are effective competitors in invaded grasslands there have been fewer studies involving the use of native forbs (Cook 1965; Carlsen et al. 2000; Gillespie & Allen 2004). A diverse spectrum of native herbaceous species could allow for greater resource capture throughout the duration of the year (Sheley et al. 1999; Dukes 2001; DiTomaso et al. 2007). *Croton setigerus* (dove weed) a native annual was found to use more water than *C. solstitialis* during the late summer months (Gerlach 2004) and *Hemizonia congesta* (hayfield tarweed) was found to reduce the growth of *C. solstitialis* (Dukes 2001). The term 'guardian species' has been applied to native species with traits to similar invasive species and may impede the establishment. I used members of the Asteraceae found within the watershed of Big Chico Creek and have broad ranges in California grasslands, the perennial *Grindelia camporum* and the annual *Madia elegans*. Although accounting for a minimal amount of cover, they both were the only two treatments to account for any cover values in both sites during the driest time of year in September when all of the grasses were dormant. Unfortunately, broadleaf herbicide also affects native species (DiTomaso et al. 2007) as was the case in the BIDWELL site. *Madia elegans* after having the highest cover values of 18% before the spray in April of 2013 fell to 8% the following year in April of 2014 (Figure 13; Appendix B). The leaves of *G. camporum* appeared to be damaged from the spray as well with cover values remaining low through the summer (5-10%) and fall (5-11%); however, it dramatically increased the following spring with values between 21-35% and finished with the highest survival percentage of all of the out-planted plugs with 79% (Appendix B; Figure 6). *Grindelia camporum* performed very well in the BCCER location as well, with the highest survival percentage of all out-planted plugs with 97% and a final cover value of 40% and (Figure 4). In addition, *G. camporum* was second only to *E. glaucus* in the reduction of *C. solstitialis* cover by 27% in May of 2014 (Figure 4). The use of *G. camporum* in restoration has been supported for weed suppression in large-scale restoration sites (T. Griggs, Cal IPC conference CSU Chico 2014). Perhaps the most consequential factor may be the fact that in both cases the flowering period overlapped with that of *C. solstitialis* during the summer months. This not only presents additional resources for pollinators, but may also deplete phosphorus and other nutrients used for flower production.

Lastly, and perhaps most importantly, is the topic of the long-term maintenance and management of grasslands where restoration efforts have been conducted. As mentioned timing is quintessential for success of management efforts and consecutive treatments are crucial in order to minimize the spread of invasive species in grasslands. While it is important to control invasives before attempting revegetation, without the use of restoration, invasives will eventually return. Control methods of invasive species should coincide with the seed collection and propagation of native species to eventually fill their void. It may take several years for the invasive seed banks to subside, but eventually invasive re-sprouts become more manageable with the use of targeted hand weeding, mowing or spot herbicide treatments before they can flower or set seed. The implementation of large-scale native out-plantings and direct seeding will then help to control their growth and reestablishment.

It is the nature of grasslands to be continually managed as they were for thousands of years before European colonization and the role of fire may be the key component. Repeated burns have been carried out to treated sites in the BCCER location with great success with out-planted grasses returning with more vigor. The use of fire may actually increase native forbs (DiTomaso et al. 1999) and aid in the health of native bunch grasses in the process (Menke 1992; Dyer & Rice 1997; Seabloom et al. 2005;

Hankins 2013). *Stipa pulchra* was found to produce larger seeds with increased germination rates in burned sites than unburned (Dyer 1993) and its seedlings were found to grow more densely within burned plots (Dyer 1996). *Stipa pulchra* was found to exhibit greater growth in the second year following fire (Langstroth 1991, Klinger & Messer 2001); however, each situation is site specific with varied results (Reiner 2007) and not all grasslands can be managed with the use of fire. The use of fall burns, when properly timed can reduce emerging non-native annual grasses and *C. solstitialis* seedlings before native grasses or forbs have germinated. One must become closely acquainted with the habits and cycles of both the invasive and native species within a particular grassland before management efforts are carried out to ensure the success of restoration practices.

CHAPTER VI

CONCLUSIONS

I was fortunate to be able to carry out my research in the unique environment of the watershed of Big Chico Creek. While my background in restoration has been mainly in the coast range I thought this would present an opportunity to get to know an ecosystem I was less familiar with. Unlike many of the areas I have done my field work the riparian forests, chaparral, foothill forest, and emergent pine communities fortunately appear be largely intact within both the BIDWELL and BCCER sites. It is the grassland understory that have been most dramatically altered and in need of attention.

Although, much native diversity remains and many native species can still be found within these invaded fields. During my short time in the grasslands I encountered numerous native species for example geophytes such as *Allium* spp., *Brodiaea* spp., *Calochortus* spp., *Chlorogalum pomeridianum*, *Dichelostemma* spp., *Tritelia* spp, *Lomatium* spp., *Sanicula* spp. and *Toxicoscordion* spp., as well as many native annual forbs such as *Eschscholzia* spp., *Lupinus* spp., *Clarkia* spp., *Claytonia* spp., *Antirrhinum cornutum*, *Calandrinia ciliata, Calycadenia* spp., *Castilleja* spp., *Clarkia* spp., *Croton setigerus*, *Deinandra spp.*, *Gilia spp.*, *Githopsis specularioides*, *Grindelia camporum*, *Hemizonia* spp., *Lasthenia* spp., *Layia* spp., *Lithophragma* spp., *Madia* spp., *Nemophila* spp., *Odontostomum hartwegii*, *Plagiobothrys* spp., *Trichostema lanceolatum* and *Viola* spp. Before the management of the invasive species is carried out, native plant species must be thoroughly inventoried. Those populations, which are threatened, rare, isolated, or dwindling, should be mapped and prioritized for specialized preservation efforts, including careful seed collection and propagation. If the ultimate goal of natural resource management is to preserve and protect biodiversity, these resources should be properly cared for, especially if they can be degraded or lost all together through invasive control efforts. As may be the case in the BIDWELL location where control efforts have been ongoing and have included the use of herbicide and goats where far fewer native forbs occur (Appendix B).

Once areas with sensitive species have been identified and clear management objectives including the monitoring of those efforts have been implemented the focus can shift towards addressing the big picture. The grasslands of the Big Chico Creek encompass such a large-scale that their management is quite complex and would require a multifaceted tactic. Since an area of this magnitude an entire watershed approach would be overwhelming I recommend breaking the area down by dividing it into separate parcels using the ephemeral/year round drainages as the boarders for each. The perimeters of each parcel should then be mapped, labeled, and assigned a site specific management plan. The next step would be to map significant native perennial grass resources and invasive threats using polygons within each of the sites. The remaining native grasses include *B. carinatus*, *Bromus laevipes*, *Elymus elymoides*, *E. glaucus*, *Elymus triticoides*, *Melica* spp., *Mulenbergia rigens*, and *S. pulchra,* while the top invasive threats currently are *C. solstitialis*, *T. caput-medusae*, and potentially *A. triuncialis*. Once the biggest invasive threats and native resources have been identified, the protocol for each of the sites can be designed.

Management should begin with the seed collection of native grasses and forbs found with in each of the sites, followed by invasive control efforts. If no native grasses remain, seed collection from the most abundant sources within the watershed should be used. Allocating control efforts in sites that will ultimately become re-seeded and invaded from above wastes both energy and resources. Invasive plants tend to begin along roadways and are then transported through trails so that is a good place to reduce their encroachment. For larger areas prescribed burns are the best tool available, followed by properly timed mowing, and the use of herbicides as a last resort.

It is important that the management of each of the sites only be carried out once the resources for consecutive invasive treatments along with monitoring and maintenance has been secured in order to see them through. This includes funding for the personnel to conduct control methods, as well as for the seed collection and propagation of native species. The mapping, seed collection, and plant propagation can often be carried out with the help of interns, students, or volunteers. Outreach through local groups and schools along with the use of AmeriCorps volunteers can be a great way to get involvement from the community. While the control of invasive species can be more rigorous involving the use of herbicides, mowing equipment, and prescribed burns and may be better accomplished through seasonal staff, conservation corps crews, or prison inmate programs. Through this strategic approach and a lot of hard work, I believe the grasslands of Big Chico Creek offer an incredible opportunity for habitat restoration and natural resources enhancement in the north state.

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APPENDIX A

Appendix A tracks each of the out planted plug treatments percent survival average over 15 months of monitoring from March 2013- May 2014. Values are shown for both sites the Big Chico Creek Ecological Reserve (BCCER) and Upper Bidwell Park (BIDWELL). Note that the accidental herbicide application to the BIDWELL site occurred in May and again in June of 2013. Many of the perennial grass plugs fell dormant during the driest time of the year in September and returned later in the rainy season.

*These "survival" values were then shown with the letter "D" to represent dormancy.

2013

March 2013	**Site:**	**BCCER**		**BIDWELL**	
Treatment	# Planted	# Alive	Survival %	# Alive	Survival %
S. pulchra	192	189	98%	184	96%
B. carinatus	192	191	99%	190	99%
E. glaucus	192	192	100%	191	99%
G. camporum	108	107	99%	107	99%

April 2013	**Site:**	**BCCER**		**BIDWELL**	
Treatment	# Planted	# Alive	Survival %	# Alive	Survival %
S. pulchra	192	187	97%	179	93%
B. carinatus	192	191	99%	182	95%
E. glaucus	192	192	100%	189	98%
G. camporum	108	107	99%	106	98%

May 2013	**Site:**	**BCCER**		**BIDWELL-Sprayed**	
Treatment	# Planted	# Alive	Survival %	# Alive	Survival %
S. pulchra	192	187	97%	174	91%
B. carinatus	192	189	98%	178	93%
E. glaucus	192	192	100%	188	98%
G. camporum	108	106	98%	106	98%

June 2013	**Site:**	**BCCER**		**BIDWELL-Sprayed**	
Treatment	# Planted	# Alive	Survival %	# Alive	Survival %
S. pulchra	192	189	94%	169	88%
B. carinatus	192	191	98%	170	89%
E. glaucus	192	192	99%	180	94%
G. camporum	108	107	98%	105	97%

July 2013	**Site:**	**BCCER**		**BIDWELL**	
Treatment	# Planted	# Alive	Survival %	# Alive	Survival %
S. pulchra	192	168	88%	135	70%
B. carinatus	192	186	97%	145	76%
E. glaucus	192	189	98%	163	85%
G. camporum	108	105	97%	90	83%

August 2013	**Site:**	**BCCER**		**BIDWELL**	
Treatment	# Planted	# Alive	Survival %	# Alive	Survival %
S. pulchra	192	148	77%	92	48%
B. carinatus	192	155	80%	110	57%
E. glaucus	192	178	92%	137	71%
G. camporum	108	105	97%	72	67%

September 2013	**Site:**	**BCCER**		**BIDWELL**	
Treatment	# Planted	# Alive	Survival %	# Alive	Survival %
S. pulchra	192	*D	0%	1	0%
B. carinatus	192	*D	0%	*D	0%
E. glaucus	192	*D	0%	*D	0%
G. camporum	108	92	85%	42	39%

October 2013	**Site:**	**BCCER**		**BIDWELL**	
Treatment	# Planted	# Alive	Survival %	# Alive	Survival %
S. pulchra	192	146	76%	88	46%
B. carinatus	192	122	64%	*D	0%
E. glaucus	192	107	56%	*D	0%
G. camporum	108	102	94%	39	36%

November 2013	**Site:**	**BCCER**		**BIDWELL**	
Treatment	# Planted	# Alive	Survival %	# Alive	Survival %
S. pulchra	192	165	86%	73	38%
B. carinatus	192	155	81%	*D	0%
E. glaucus	192	182	95%	*D	0%
G. camporum	108	105	97%	36	33%

December 2013	**Site:**	**BCCER**		**BIDWELL**	
Treatment	# Planted	# Alive	Survival %	# Alive	Survival %
S. pulchra	192	182	95%	133	69%
B. carinatus	192	173	90%	44	23%
E. glaucus	192	190	99%	72	38%
G. camporum	108	106	98%	79	73%

2014

January 2014	**Site:**	**BCCER**		**BIDWELL**	
Treatment	# Planted	# Alive	Survival %	# Alive	Survival %
S. pulchra	192	180	94%	143	74%
B. carinatus	192	175	91%	90	47%
E. glaucus	192	190	99%	99	52%
G. camporum	108	106	98%	82	76%

February 2014	**Site:**	**BCCER**		**BIDWELL**	
Treatment	# Planted	# Alive	Survival %	# Alive	Survival %
S. pulchra	192	180	94%	145	76%
B. carinatus	192	175	91%	110	57%
E. glaucus	192	190	99%	101	53%
G. camporum	108	106	98%	86	80%

March 2014	**Site:**	**BCCER**		**BIDWELL**	
Treatment	# Planted	# Alive	Survival %	# Alive	Survival %
S. pulchra	192	178	93%	145	76%
B. carinatus	192	174	91%	112	58%
E. glaucus	192	188	98%	101	53%
G. camporum	108	106	98%	85	79%

April 2014	**Site:**	**BCCER**		**BIDWELL**	
Treatment	# Planted	# Alive	Survival %	# Alive	Survival %
S. pulchra	192	176	92%	141	73%
B. carinatus	192	172	90%	115	60%
E. glaucus	192	188	98%	90	47%
G. camporum	108	104	96%	85	79%

May 2014	**Site:**	**BCCER**		**BIDWELL**	
Treatment	# Planted	# Alive	Survival %	# Alive	Survival %
S. pulchra	192	164	85%	147	77%
B. carinatus	192	167	87%	104	54%
E. glaucus	192	186	97%	73	38%
G. camporum	108	103	95%	85	79%

APPENDIX B

Appendix B shows the combined and averaged vegetative percent cover values for each of the seven treatments along with the five cover classes in each of the treatment plots. Values were recorded over the course of fifteen months from March of 2013 through May 2014.

2013

March 2013	**Treatment**		***C. solstitialis***		**Ann. Grass**		**Non-Nat. Frb**		**Nat. Forb**	
Site:	BCC	BID	BCC	BID	BCC	BID	BCC	BID	BCC	BID
S. pulchra	5%	5%	16%	16%	13%	13%	8%	8%	2%	2%
B. carinatus	13%	13%	13%	13%	13%	13%	9%	9%	5%	5%
E. glaucus	12%	10%	3%	18%	6%	10%	4%	9%	9%	3%
Seeded grass	10%	16%	3%	7%	3%	5%	3%	5%	7%	3%
G. camporum	5%	5%	7%	23%	4%	18%	3%	10%	9%	3%
M. elegans	5%	13%	3%	5%	3%	3%	3%	5%	9%	4%
Control	X	X	7%	30%	5%	18%	3%	11%	13%	3%

April 2013	**Treatment**		***C. solstitialis***		**Ann. Grass**		**Non-Nat. Frb**		**Nat. Forb**	
Site:	BCC	BID	BCC	BID	BCC	BID	BCC	BID	BCC	BID
S. pulchra	10%	6%	23%	32%	10%	18%	9%	18%	14%	5%
B. carinatus	26%	15%	19%	24%	9%	18%	8%	16%	9%	7%
E. glaucus	20%	12%	16%	29%	8%	13%	7%	15%	10%	8%
Seeded grass	17%	21%	9%	17%	5%	11%	4%	13%	9%	5%
G. camporum	10%	6%	25%	35%	8%	19%	8%	19%	12%	5%
M. elegans	12%	18%	8%	14%	5%	6%	4%	12%	12%	5%
Control	X	X	29%	43%	8%	23%	8%	18%	15%	4%

May 2013	**Treatment**		***C. solstitialis***		**Ann. Grass**		**Non-Nat. Frb**		**Nat. Forb**	
Site:	BCC	BID	BCC	BID	BCC	BID	BCC	BID	BCC	BID
S. pulchra	23%	10%	25%	18%	18%	28%	9%	22%	9%	4%
B. carinatus	36%	23%	27%	20%	15%	31%	9%	17%	8%	4%
E. glaucus	32%	20%	25%	23%	15%	22%	7%	18%	8%	4%
Seeded grass	13%	13%	13%	13%	14%	26%	7%	13%	9%	4%
G. camporum	22%	15%	32%	22%	17%	28%	10%	24%	8%	4%
M. elegans	18%	13%	13%	13%	11%	17%	8%	16%	13%	5%
Control	X	X	34%	35%	20%	32%	11%	20%	10%	3%

2013 (continued)

June 2013	Treatment		*C. solstitialis*		Ann. Grass		Non-Nat. Frb		Nat. Forb	
Site:	BCC	BID	BCC	BID	BCC	BID	BCC	BID	BCC	BID
S. pulchra	14%	11%	33%	13%	12%	25%	14%	12%	8%	1%
B. carinatus	34%	21%	25%	15%	12%	22%	10%	13%	8%	1%
E. glaucus	33%	20%	25%	15%	12%	18%	9%	11%	9%	2%
Seeded grass	11%	13%	18%	7%	12%	15%	7%	9%	9%	2%
G. camporum	21%	13%	32%	13%	11%	26%	12%	15%	9%	2%
M. elegans	17%	6%	19%	10%	9%	10%	8%	10%	10%	3%
Control	X	X	35%	19%	15%	29%	15%	17%	11%	1%

July 2013	Treatment		*C. solstitialis*		Ann. Grass		Non-Nat. Frb		Nat. Forb	
Site:	BCC	BID	BCC	BID	BCC	BID	BCC	BID	BCC	BID
S. pulchra	12%	11%	51%	13%	0%	0%	4%	0%	3%	0%
B. carinatus	22%	12%	44%	11%	0%	0%	0%	0%	2%	0%
E. glaucus	30%	15%	43%	15%	0%	0%	4%	0%	3%	0%
Seeded grass	13%	7%	33%	7%	0%	0%	0%	0%	8%	0%
G. camporum	20%	10%	48%	10%	0%	0%	0%	0%	4%	0%
M. elegans	15%	9%	26%	7%	0%	0%	5%	0%	10%	0%
Control	X	X	54%	21%	0%	0%	0%	0%	8%	0%

August 2013	Treatment		*C. solstitialis*		Ann. Grass		Non-Nat. Frb		Nat. Forb	
Site:	BCC	BID	BCC	BID	BCC	BID	BCC	BID	BCC	BID
S. pulchra	11%	9%	54%	10%	0%	0%	0%	0%	0%	0%
B. carinatus	17%	12%	40%	10%	0%	0%	0%	0%	0%	0%
E. glaucus	21%	15%	43%	10%	0%	0%	0%	0%	0%	0%
Seeded grass	11%	12%	32%	5%	0%	0%	0%	0%	0%	0%
G. camporum	16%	10%	53%	9%	0%	0%	0%	0%	0%	0%
M. elegans	10%	10%	27%	4%	0%	0%	0%	0%	0%	0%
Control	X	X	58%	20%	0%	0%	0%	0%	0%	0%

September 2013	Treatment		*C. solstitialis*		Ann. Grass		Non-Nat. Frb		Nat. Forb	
Site:	BCC	BID	BCC	BID	BCC	BID	BCC	BID	BCC	BID
S. pulchra	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
B. carinatus	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
E. glaucus	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
Seeded grass	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
G. camporum	12%	5%	0%	0%	0%	0%	0%	0%	0%	0%
M. elegans	3%	3%	0%	0%	0%	0%	0%	0%	0%	0%
Control	X	X	0%	1%	0%	0%	0%	0%	0%	0%

October 2013	Treatment		*C. solstitialis*		Ann. Grass		Non-Nat. Frb		Nat. Forb	
Site:	BCC	BID	BCC	BID	BCC	BID	BCC	BID	BCC	BID
S. pulchra	18%	8%	11%	0%	0%	3%	0%	2%	0%	1%
B. carinatus	13%	0%	12%	0%	0%	7%	0%	2%	0%	1%
E. glaucus	13%	0%	12%	0%	0%	1%	0%	2%	0%	0%
Seeded grass	8%	3%	15%	1%	0%	6%	0%	3%	0%	1%
G. camporum	18%	6%	12%	1%	0%	9%	0%	2%	0%	0%
M. elegans	1%	5%	14%	1%	0%	2%	0%	2%	0%	0%
Control	X	X	25%	2%	0%	10%	0%	2%	0%	0%

2013 (continued)

November 2013	Treatment		*C. solstitialis*		Ann. Grass		Non-Nat. Frb		Nat. Forb	
Site:	BCC	BID	BCC	BID	BCC	BID	BCC	BID	BCC	BID
S. pulchra	21%	8%	6%	0%	0%	0%	0%	0%	0%	0%
B. carinatus	21%	0%	7%	0%	0%	0%	1%	0%	0%	0%
E. glaucus	29%	0%	7%	0%	0%	0%	0%	0%	0%	0%
Seeded grass	15%	1%	8%	0%	0%	0%	0%	0%	0%	0%
G. camporum	24%	5%	6%	0%	0%	0%	0%	0%	0%	0%
M. elegans	0%	3%	15%	0%	0%	0%	0%	0%	0%	0%
Control	X	X	20%	0%	0%	0%	0%	0%	0%	0%

December 2013	Treatment		*C. solstitialis*		Ann. Grass		Non-Nat. Frb		Nat. Forb	
Site:	BCC	BID	BCC	BID	BCC	BID	BCC	BID	BCC	BID
S. pulchra	21%	11%	20%	5%	0%	12%	5%	5%	0%	0%
B. carinatus	30%	5%	18%	5%	0%	11%	6%	6%	0%	1%
E. glaucus	39%	6%	17%	5%	0%	8%	6%	7%	0%	0%
Seeded grass	11%	8%	17%	5%	0%	9%	7%	5%	0%	1%
G. camporum	16%	5%	19%	5%	0%	11%	5%	6%	0%	0%
M. elegans	0%	0%	16%	5%	0%	8%	8%	5%	0%	0%
Control	X	X	23%	5%	0%	14%	7%	6%	0%	1%

January 2014	Treatment		*C. solstitialis*		Ann. Grass		Non-Nat. Frb		Nat. Forb	
Site:	BCC	BID	BCC	BID	BCC	BID	BCC	BID	BCC	BID
S. pulchra	24%	13%	20%	2%	6%	23%	9%	5%	5%	0%
B. carinatus	30%	7%	17%	1%	5%	25%	8%	7%	5%	1%
E. glaucus	45%	10%	16%	0%	4%	17%	8%	7%	5%	1%
Seeded grass	13%	10%	13%	0%	5%	18%	7%	4%	5%	1%
G. camporum	20%	9%	15%	0%	5%	26%	8%	6%	5%	1%
M. elegans	0%	2%	14%	3%	5%	7%	8%	4%	5%	0%
Control	X	X	20%	1%	5%	34%	9%	7%	5%	1%

2014

February 2014	Treatment		*C. solstitialis*		Ann. Grass		Non-Nat. Frb		Nat. Forb	
Site:	BCC	BID	BCC	BID	BCC	BID	BCC	BID	BCC	BID
S. pulchra	35%	22%	21%	0%	5%	18%	11%	4%	5%	0%
B. carinatus	43%	13%	20%	0%	6%	20%	10%	4%	5%	1%
E. glaucus	53%	15%	14%	0%	5%	13%	9%	5%	5%	3%
Seeded grass	18%	14%	11%	0%	5%	20%	12%	4%	5%	0%
G. camporum	22%	11%	15%	0%	6%	23%	12%	5%	5%	1%
M. elegans	0%	2%	12%	0%	5%	8%	14%	4%	5%	2%
Control	X	X	27%	0%	5%	35%	18%	5%	6%	1%

March 2014	Treatment		*C. solstitialis*		Ann. Grass		Non-Nat. Frb		Nat. Forb	
Site:	BCC	BID	BCC	BID	BCC	BID	BCC	BID	BCC	BID
S. pulchra	37%	29%	24%	0%	7%	39%	13%	8%	12%	2%
B. carinatus	50%	19%	18%	0%	7%	35%	12%	8%	8%	5%
E. glaucus	55%	20%	17%	0%	6%	31%	12%	10%	9%	3%
Seeded grass	22%	20%	22%	0%	6%	34%	16%	7%	9%	3%
G. camporum	26%	21%	20%	0%	6%	42%	16%	9%	9%	1%
M. elegans	5%	2%	19%	0%	7%	21%	20%	8%	10%	3%
Control	X	X	34%	0%	7%	43%	19%	10%	11%	3%

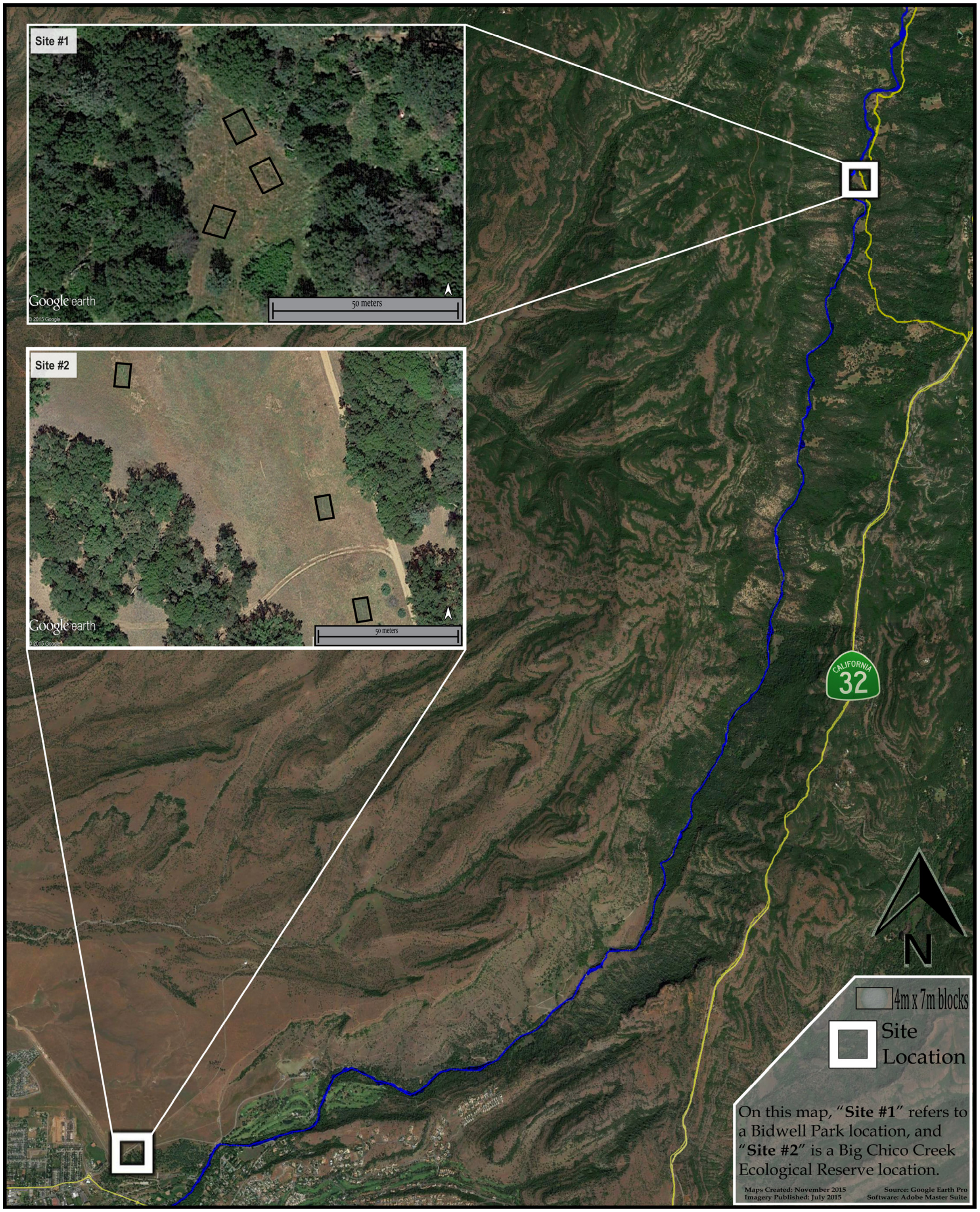
2014 (continued)

April 2014	**Treatment**		***C. solstitialis***		**Ann. Grass**		**Non-Nat. Frb**		**Nat. Forb**	
Site:	BCC	BID	BCC	BID	BCC	BID	BCC	BID	BCC	BID
S. pulchra	36%	38%	23%	1%	8%	31%	14%	9%	9%	1%
B. carinatus	49%	28%	18%	1%	9%	35%	14%	8%	7%	5%
E. glaucus	59%	25%	15%	1%	6%	29%	12%	14%	9%	5%
Seeded grass	23%	27%	20%	0%	8%	38%	17%	8%	10%	4%
G. camporum	32%	31%	18%	1%	8%	35%	15%	9%	10%	3%
M. elegans	5%	8%	22%	1%	7%	21%	20%	11%	14%	3%
Control	X	X	28%	2%	7%	48%	19%	12%	13%	3%

May 2014	**Treatment**		***C. solstitialis***		**Ann. Grass**		**Non-Nat. Frb**		**Nat. Forb**	
Site:	BCC	BID	BCC	BID	BCC	BID	BCC	BID	BCC	BID
S. pulchra	35%	36%	29%	1%	12%	35%	12%	5%	7%	2%
B. carinatus	44%	28%	25%	3%	11%	43%	12%	5%	6%	3%
E. glaucus	52%	19%	18%	2%	8%	42%	13%	6%	7%	4%
Seeded grass	22%	19%	31%	0%	15%	50%	19%	5%	8%	4%
G. camporum	40%	35%	20%	0%	11%	35%	13%	3%	8%	2%
M. elegans	5%	12%	37%	1%	14%	34%	28%	7%	10%	2%
Control	X	X	47%	3%	14%	74%	20%	5%	10%	3%

APPENDIX C

Location of both research sites along the watershed of Big Chico Creek



APPENDIX D

Appendix D shows the points latitude and longitude points of all four corners for each of the three blocks in both sites. These GPS points can be used to locate them for future research.

	BCCER		**BIDWELL**
	Block 1		**Block 1**
SW-	N- 39.86176°	SW-	N- 39.76364°
	W- 121.70818°		W- 121.80040°
SE-	N- 39.86174°	SE-	N- 39.76395°
	W- 121.70813°		W- 121.80038°
NE-	N- 39.86183°	NE-	N- 39.96372°
	W- 121.70812°		W- 121.80042°
NW	N-39.86183°	NW-	N- 39.76371°
	W-121.70818°		W- 121.80046°

	Block 2		**Block 2**
SW-	N- 39.86210°	SW-	N- 39.76375°
	W- 121.70826°		W- 121.80029°
SE-	N- 39.86211°	SE-	N- 39.76377°
	W- 121.70826°		W- 121.80023°
NE-	N- 39.86215°	NE-	N- 39.76381°
	W- 121.70830°		W- 121.80028°
NW-	N- 39.86214°	NW-	N- 39.76381°
	W- 121.70836°		W- 121.80032°

	Block 3		**Block 3**
SW-	N- 39.86249°	SW-	N- 39.76385°
	W- 121.70930°		W- 121.80039°
SE	N- 39.86249°	SE-	N- 39.76388°
	W- 121.70924°		W- 121.80035°
NE	N- 39.86256°	NE-	N- 39.76395°
	W- 121. 70923°		W- 121.80038°
NW	N- 39.86258°	NW-	N- 39.76393°
	W- 121.70929°		W- 121.80042°