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Control of Large Stands of *Phragmites australis* in Great Salt Lake, Utah Wetlands

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CONTROL OF LARGE STANDS OF *PHRAGMITES AUSTRALIS* IN

GREAT SALT LAKE, UTAH WETLANDS

by

Chad R. Cranney

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ABSTRACT

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Phragmites australis (hereafter *Phragmites*) often forms dense monocultures, which displace native plant communities and alter ecosystem functions and services. Managers tasked with controlling this plant need science-backed guidance on how to control *Phragmites* and restore native plant communities. This study took a large-scale approach—to better match the scale of actual restoration efforts—to compare two herbicides (glyphosate vs. imazapyr) and application timings (summer vs. fall). Five treatments were applied to 1.2 ha plots for three consecutive years: 1) summer glyphosate; 2) summer imazapyr; 3) fall glyphosate; 4) fall imazapyr; and 5) untreated control. Dead *Phragmites* following herbicide treatments was mowed in the first two years. Efficacy of treatments and the response of native plant communities were monitored for three years. We report that fall herbicide applications were superior to summer applications. No difference was found between the two herbicides in their ability to reduce *Phragmites* cover. Plant communities switched from emergent to open water communities and were limited by *Phragmites* litter and water depth. Although, some plant communities showed a slow trajectory towards one of the reference sites, cover of important native emergent plants did not increase until year three and remained below 10%. These results suggest that fall is the best time to apply herbicides for effective large-scale control of *Phragmites*. Active restoration (e.g. seeding) may be needed to gain back important native plant communities. Methods to reduce *Phragmites* litter after herbicide applications should be considered.

(99 pages)

PUBLIC ABSTRACT

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Chad R. Cranney

Phragmites (common reed) is a non-native, invasive perennial grass from Eurasia that is taking over wetlands across North America. In Utah, *Phragmites* has expanded to cover tens of thousands of acres in and around the Great Salt Lake (GSL). The GSL and its associated wetlands are recognized regionally and hemispherically as an important bird area (IBA) that provide critical habitat for a wide variety of wetland dependent birds. The invasion and expansion of *Phragmites* has replaced many of the high quality habitats these avian populations rely on. This research aimed to determine the most effective methods to control *Phragmites* and restore native plant species. We took a large-scale approach to evaluate the effectiveness of two herbicides (glyphosate and imazapyr), and application timings (summer and fall), for controlling *Phragmites* to restore native plants and lost bird habitat. After three consecutive years of herbicide application, fall herbicide applications were superior to summer applications and no difference between the types of herbicide used was found. Even with effective control of *Phragmites*, important native plant recovery was slow and limited. In order to gain back the native plants that once dominated before *Phragmites* invaded, re-vegetation efforts such as seeding may be needed.

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CHAPTER 1

INTRODUCTION

Invasive plant species are a global concern and have been identified as major contributors to declining biodiversity (Hobbs & Humphries 1995; Wilcove et al. 1998; Mack et al. 2000). Many invasive plants form dense monocultures that replace structurally and compositionally diverse native plant communities (D'Antonio & Meyerson 2002; Davis et al. 2005). In addition, invasive plant species can have other negative impacts on ecosystems including altered ecological functions and processes such as fire regimes, nutrient cycling, hydrologic cycling, decreased wildlife habitat, and the way people use these ecosystems (Mack & Antonio 1998; Ehrenfeld 2003). The negative impacts associated with invasive plants has led to increased efforts by practitioners to restore these degraded habitats by implementing strategies to eradicate, control, or manage the spread of invasive plants (Hulme 2006; Hobbs & Cramer 2008; Stromberg et al. 2009).

One of the most problematic wetland plant invaders today is *Phragmites australis* (hereafter, *Phragmites*). *Phragmites* is commonly found in alkaline, brackish, and freshwater marshes, along ditches and roadsides (Marks et al. 1994; Kulmatiski et al. 2011). Recent and rapid expansion has been observed in a number of systems including tidal wetlands (Chambers et al. 1999; Bertness et al. 2002), the Great Lakes (Carlson et al. 2009), Great Basin wetlands (Kulmatiski et al. 2011; Kettenring & Mock 2012), and the Gulf Coast (Kettenring et al. 2012a). *Phragmites* often forms dense monotypic stands that reduce light, nutrient, and space availability for desirable plants species, resulting in decreased plant diversity (Marks et al. 1994, Chambers et al. 1999). It is also virtually impenetrable to many wildlife species reducing waterfowl and shorebird use (Benoit & Askins 1999). Other impacts include a decrease in biodiversity of macroinvertebrates (Angradi et al. 2001), a reduction in nursery habitat for fish (Able & Hagan 2000), altered hydrology due to soil accretion (Rooth et al. 2003), altered biogeochemical cycling (Meyerson et al. 1999; Findlay et al. 2003) and direct human impacts such as increased fire hazards, reduced access, and obstructed views (reviewed in Getsinger et al. 2006).

Resource managers across the U.S. are spending considerable amounts of limited resources on strategies to control and restore *Phragmites*-dominated wetlands (Martin & Blossey 2013). Several strategies have been used and studied including: mechanical, hydrologic manipulation, chemical, burning, and biological. However, the success and results of these methods have varied. Some treatments tend to only have temporary success and combination of treatments may be needed for effective control (Marks et al. 1994; Kiviat 2006; Hazelton et al. 2014). The following is a review of different control methods and a number of studies conducted using these methods.

Phragmites Control Methods

Mechanical

Mechanical methods to control *Phragmites* include mowing, cutting with hand tools, disking, and excavating. Often times these approaches are labor intensive as the cut material must be removed from the site to reduce vegetative re-growth (Marks et al. 1994; Kiviat 2006). Studies have reported a decrease in above ground biomass and plant height, but they also report increased stem densities and *Phragmites* dominance after mowing (Weisner & Granéli 1989; Warren et al. 2001; Güsewell 2003; Derr 2008a).

Weisner and Granéli (1989) also reported that cutting in June when rhizome nutrient reserves are at their lowest was the best timing for cutting or mowing. Güsewell (2003) also found that cutting in June and again in September was superior to cutting in September alone. Several studies suggest mowing or cutting over multiple seasons may reduce *Phragmites* dominance (Cross & Flemming 1989; Weisner & Granéli 1989; Marks et al. 1994; Warren et al. 2001; Derr 2008a), but Güsewell (2003) found *Phragmites* still dominated after six years of mowing. Methods of mowing and cutting alone have little impact on the dominance of *Phragmites* but can change stand characteristics such as plant height and above ground biomass. An integrated approach with other control methods may be more useful when trying to control *Phragmites*.

One integrated strategy involves mowing or cutting followed by covering with black plastic. Cutting and covering a *Phragmites* stand in New York resulted in a 90% reduction in *Phragmites* cover. Two years after the treatment *Phragmites* was not present (Marks et al. 1994). Burdick et al. (2010) found that stem density was significantly reduced when covered with plastic compared to plots without plastic (0.1 m^{-2} and 20.7 m^{-2} respectively). Conversely, another study reported no difference in stem density but did find significantly lower rhizome carbohydrate reserves in the plastic covered treatments (Wilcox 2013). Varying results suggest that substrate type and depth of rhizomes may play a large role as plastic treatments may not kill rhizomes deep in the soil (Kiviat 2006; reviewed in Wilcox 2013). See hydrologic manipulations and chemical control sections below for more discussion on combining mowing or cutting with other treatments.

Disking

As with mowing or cutting, disking alone will have little effect on *Phragmites* cover and may in fact stimulate bud production and vegetative growth from cut stems and pieces of rhizome (Marks et al. 1994; Getsinger et al. 2006; Kiviat 2006). Cross and Fleming (1989) suggested that disking in the late summer and early fall would expose and kill rhizomes during the winter freeze. In general, disking is discouraged (reviewed in Marks et al. 1994).

Excavating

Excavation can provide good control as long as all plant material, including rhizomes are removed (Cross & Flemming 1989). Not only is *Phragmites* removed, excavation lowers soil levels and increases water depths above tolerance levels of *Phragmites* (Kiviat 2006). In Connecticut and New Hampshire, excavating below water levels resulted in an increase of native plant communities (reviewed in Hazelton et al. 2014). Excavation can be effective, but is not used often as it is very costly and concerns over mobilization of nutrient and other contaminants have been raised (Kiviat 2006).

Burning

As with mechanical control methods, burning alone is generally not thought of as an effective tool as it has little effect on reducing *Phragmites* dominance (Marks et al. 1994). For example, experimental fires in the Delta Marsh, Manitoba resulted in an increase of *Phragmites* shoot density compared to unburned plots (Thompson & Shay 1985). In the same experiment, nutrient reserves in the rhizomes were at their lowest in June suggesting that multiple years of burning in June may inhibit growth (Thompson &

Shay 1985). Cross and Fleming (1989) also suggested burning in the summer under dry conditions so the fire can burn hot enough to kill the roots and rhizomes. Fire can be used to quickly remove aboveground biomass but unless integrated with other methods such as hydrologic manipulation and herbicides application (Marks et al. 1994), *Phragmites* will continue to persist and in some cases actually increase stand (Van der Toorn & Mook 1982; Thompson & Shay 1985). A discussion on integrating burning with other methods is discussed in the chemical control section below.

Biological

Classic biocontrol methods consist of herbivorous insects found in the invasive plants native range and have detrimental effects on the plants growth and reproduction (Tscharntke 1999; Hazelton et al. 2014). Currently, there are no biocontrols available in North America, but 91% of resource managers report they would release a biocontrol if one were available (Martin & Blossey 2013). Many insects are known to attack *Phragmites* in its native European range, some of which have been inadvertently introduced to the U.S.; however, these insects have had little impact on the spread and growth of *Phragmites* here in the U.S. (Häfliger et al. 2005). Potential biocontrol insects include *Rhizedra lutosa* and *Chilo phragmitella*, which feed on the rhizomes and stem boring moths such as *Archanara* spp. all of which reduce carbohydrate storage (Häfliger et al. 2005). A number of potential biocontrols have been investigated for many years and release of a biocontrol for *Phragmites* could occur in the next couple of years (reviewed in Hazelton et al. 2014).

Grazing

In Europe, grazing has been shown to decrease *Phragmites* density while increasing plant diversity (Vulink et al. 2000; Ausden et al., 2005). In the U.S., few studies have evaluated grazing *Phragmites* (Kiviat 2006). One study evaluated the use of goats in Maryland and reported *Phragmites* density, height, and biomass significantly decreased and in turn species diversity increased (Brundage 2010). Another study in New Jersey evaluated the use of goats for *Phragmites* control and found the goats selected everything but *Phragmites* (Teal & Peterson 2005). In Utah, 49% of managers are using cattle grazing to control *Phragmites* (Kettenring et al. 2012b) and managers report a shift from *Phragmites*-dominated wetlands to *Distichlis spicata* and increased bird use (Hazelton et al. 2014). Little is known about the impacts livestock used for *Phragmites* control are having on the soils, nutrient cycling, and native plant communities (Hazelton et al. 2014).

Hydrological Manipulation

Increasing water depth decreases *Phragmites* ability to photosynthesis and translocate oxygen to the rhizomes (Weisner & Granéli 1989). However, due to the plants large and extensive rhizome system, flooding will have little effect on established stands (Cross & Fleming 1989), but can help control further expansion. For example, water depths of 5 cm and greater prevents seed germination (reviewed in Enlonger 2009). Maintaining water levels >30 cm can prevent stolons from anchoring and establishing (Cross & Fleming 1989). Seedlings and juvenile plants can be killed by raising water levels above the plant when rhizomes are small and nutrient reserves are limited, but older plants with more developed rhizomes can survive anoxic conditions (Armstrong et al. 1999). Mauchamp et al. (2001) reported seedlings that have been growing for 40 days followed by flooding for a month may not kill the plants. In fact, rhizomes have been found to survive anoxic conditions for more than 28 days (Hellings & Gallagher 1992). Marks et al. (1994) suggested flooding over the top of rhizomes at levels >90 cm during the growing season for four months, but also explained this could be detrimental to other desirable plants species. Once established, *Phragmites* can withstand a wide tolerance of flooding depths (0–80 cm), although at depths ≥80 cm stem density is reduced (Coops et al. 1996). Flooding is more detrimental under more reducing substrates (calcareous mud) due to prevailing anaerobic conditions than compared to more oxidized substrates (Marks et al 1994; Weisner & Granéli 2003).

As with cutting or mowing used on its own, it is unlikely that flooding will help restore *Phragmites* dominated wetlands, but integrated with other methods and abiotic factors such as salinity and soil substrates, *Phragmites* dominance may be negatively impacted. For example, Weisner and Granéli (2003) reported that *Phragmites* biomass was significantly decreased when shoots were cut 20 cm below the water surface. However, this was only apparent in low reducing substrates (clay soils) with no decrease in plant biomass in high reducing substrates (sandy soils). Smith (2005) reported 59–99% mortality when *Phragmites* stems were removed below the water surface. Hellings and Gallagher (1992) cut *Phragmites* stems at the base and flooded them with brackish water (10 g/L salinity). After 18 months, they found no living stems above the water surface suggesting control of *Phragmites* could be accomplished by manipulating flooding and salinity levels. In tidal wetlands, removing tidal restrictions, and allowing seawater back into the wetlands has reduced *Phragmites* dominance and increased native vegetation cover (Warren et al. 2001, Chambers et al. 2003). A combination of mowing and flooding may significantly reduce *Phragmites* cover but this method can only be used in areas where water levels can be manipulated, or if managers can be opportunistic when high water levels for an extended period of time are present.

Chemical

Currently, glyphosate and imazapyr are the only herbicides approved by the Environmental Protection Agency (EPA) for use in aquatic systems that have shown any promise for *Phragmites* control (reviewed in Getsinger et al. 2006). According to herbicide labels, glyphosate disrupts the production of enzymes needed for the formation of amino acids which are only found in plants and microorganisms. It is taken into the plant by foliage contact and translocated into the root system. Glyphosate labels also suggest that it does not leave a soil residual and is microbially broken down quickly. Complete breakdown of glyphosate has been reported in <7 days but greenhouse experiments have found it to persist for up to 79 days (reviewed in Hazleton et al. 2014). Imazapyr attacks plant specific amino acid chains in meristematic regions. Imazapyr can be taken in by plants through both foliar exposure and the roots from residual herbicide bound to the soil. In wetter sites imazapyr is broken down by photodegradation in approximately 2 days. In dryer sites and soils where photodegradation does not occur, microbial breakdown is needed and has been reported to range from 1 month to 4 years

(Tu et al. 2001). Both are systematic non-selective herbicides that will have negative impacts to non-target species (Marks et al 1994; Mozdzer et al. 2008).

Traditionally, and as per label recommendations, both herbicides were applied in the late summer or fall after plants have produced inflorescences. During this time *Phragmites* is translocating above ground resources to below ground rhizomes for over-winter storage and the plant carries herbicides to the rhizomes killing both the above and below ground plant material (Marks et al. 1994). Later fall treatments might also limit negative impacts to non-target species since many native wetlands plants have started to senesce by this time (Cross & Flemming 1989; Ailstock et al. 2001). In Maryland, a one-time fall application of glyphosate significantly reduced *Phragmites* cover and resulted in an increase of plant diversity 3-4 years post-treatment (Ailstock et al. 2001). Unfortunately, *Phragmites* was still present 1 year after treatments and continued to increase over the course of the study. By year 5, *Phragmites* cover was substantial and the authors noted that unless follow up treatments are implemented *Phragmites* will return to pre-treatment levels (Ailstock et al. 2001). These findings are consistent with most glyphosate experiments and management efforts that have resulted >80% control after the first year of application but a steady increase of *Phragmites* cover 2-3 years after treatment (Marks et al. 1994; Kiviat 2006).

Studies comparing the two herbicides have found imazapyr to be superior to glyphosate in reducing *Phragmites* cover (Kay 1995; Derr 2008b; Mozdzer et al. 2008; Getsinger et al. 2006). Using glyphosate as a wipe-on application, at a dilution of 25% and 50%, Kay (1995) reported 38% and 33% control, respectively. Using imazapyr at the same rates produced 57% and 75% control, respectively. The wipe-applications were considered sub-optimal, especially when compared to a spray application of 1.25% glyphosate that resulted in 100% control the following year (Kay 1995). Under greenhouse conditions, Derr (2008b) reported glyphosate and imazapyr provided 82% and 93% control, respectively. Getsinger et al. (2006) found that imazapyr was better at controlling *Phragmites* than glyphosate in small treatment sites (40’ x 40’). However, when the treatments were applied to larger sites (several acres), they found no significant difference between the two. Cheshier et al. (2012) also found no difference between the two herbicides with both resulting in >90% control under greenhouse conditions. Although imazapyr has been shown to provide better control compared to glyphosate, slower recovery of native plants in imazapyr treated sites has been reported (Mozdzer et al. 2008).

Contrary to herbicide labels and traditional timing of application (late summer-fall), earlier summer applications of both glyphosate and imazapyr have been shown to be just as or more effective (Derr 2008b; Mozdzer et al. 2008). Comparing June vs. September applications, Mozdzer et al. (2008) found a 20% greater reduction in June glyphosate applications and a 3% greater reduction in June imazapyr applications. Derr (2008b) reported 82% control using glyphosate and 93% control using imazapyr at both June and September applications. Despite these results, little is known how earlier summer applications might be affecting non-target species. Presumably, earlier applications will have greater negative impacts to non-target species since they are actively growing (Mozdzer et al. 2008) compared to fall applications when many of the non-target plants have already started winter dormancy (Marks et al. 1994; Ailstock et al. 2001).

Several experiments have integrated other methods to enhance efficacy of herbicide treatments (Moreira et al. 1999; Ailstock et al. 2001; Getsinger et al. 2006; Carlson et al. 2009; Rapp et al. 2012; Breen et al. 2014). Mowing and burning after herbicide treatments will reduce above ground biomass and allow sunlight to reach the soil surface; therefore promoting native plant germination (Marks et al. 1994; Ailstock et al. 2001). Removing the dead biomass will also aid in monitoring and follow-up treatments the following year (Marks et al. 1994; Ailstock et al. 2001). In a study that compared burned and un-burned herbicide treated *Phragmites*, the burned sites resulted in rapid recolonization of a diverse wetland plant community (Ailstock et al. 2001). The lower number of individual plants and lower diversity in the un-burned sites was attributed to shading affects from the dead, unburned *Phragmites* stems (Ailstock et al. 2001). Another study combined the use of fire and flooding following herbicide treatments, which resulted in 99% control, for up to three years (Getsinger et al. 2006). This treatment also provided the largest increase in non-*Phragmites* cover and even though open water and submergent plants replaced emergent vegetation, (Getsinger et al. 2006). depending on management priorities, these results could be very beneficial. Getsinger et al. (2006) also use a number of other secondary treatments following herbicide applications including; burning, flooding, and mowing, all of which resulted in more *Phragmites* control and increased non-*Phragmites* cover compared to an herbicide application alone.

The use of mechanical methods following herbicide treatments has also shown to provide significant control of *Phragmites*. Moreira et al. (1999) cut and removed *Phragmites* one month prior to fall applications of glyphosate. *Phragmites* was reduced >90% and these results lasted for three years. Rapp et al. (2012) reported that disking and mowing before herbicide applications also provided >90% control for three years. Another study found that more diverse plant communities emerged following herbicide treatments combined with cutting and cutting and removal of dead *Phragmites* stands (Carlson et al. 2009).

Although many methods and combination of methods have been used to control *Phragmites*, the majority of studies report continued maintenance and treatments will be needed to keep *Phragmites* from re-invading (see above) and in most cases complete eradication is unlikely (Turner & Warren 2003). Furthermore, questions still remain about the long-term efficacy of treatments and if actual restoration of native plant communities is occurring (Hazelton et al. 2014). These questions still remain partly because most management efforts and scientific studies lack long-term monitoring, most scientific studies are conducted at very small scales and do not represent the scale of actual on-the-ground restoration efforts, and many studies only report the effects on *Phragmites* (Wagner et al. 2008; Kettenring & Adams 2011; Hazelton et al. 2014).

This study took a large-scale approach to address some of the limitations found in previous research. In particular, we used two different herbicides (glyphosate and imazapyr) and two timings of application (June and September) to evaluate the efficacy of each for reducing *Phragmites* cover. We also evaluated changes in plant communities and native plant recovery to better inform managers about the best strategy to use to decrease *Phragmites* cover and increase native plant cover.

CHAPTER 2

CONTROL OF LARGE STANDS OF *PHRAGMITES AUSTRALIS* IN GREAT SALT LAKE, UTAH WETLANDS

Introduction

Efforts by resource managers and researchers to improve the eradication, control, and spread of invasive plant species have increased substantially over the past three decades (D'Antonio & Meyerson 2002; D'Antonio et al. 2004; Hulme 2006). Management strategies often involve restoring invaded habitats to a more desirable species composition and community structure by reducing the cover of the invasive species and by promoting native plant species establishment (Noss 1990; Hulme 2006; Hobbs & Cramer 2008). Managers rely on scientific research, which should aim to help them prioritize efforts, design strategies and appropriate methods to control, and predict long-term outcomes of invasive plant management (D'Antonio et al. 2004).

Unfortunately, despite the extensive research conducted on invasive plant removal and ecosystem restoration techniques, many restoration efforts by managers have shown highly variable results (Mack et al. 2000; Kettenring & Adams 2011). This discrepancy becomes problematic for managers when trying to choose the most effective management techniques from scientific research for native plant restoration following invasive species control (Mack & D'Antonio 1998; Kettenring & Adams 2011).

Translating results from research to broader scale implementation has often proved challenging because few invasive plant studies have been conducted in an ecological restoration context (Flory 2010). Specifically, many invasive plant control experiments are conducted in pots or mesocosms and those that are conducted in the field often have small plot sizes (often <1 m^2) that do not represent the scale at which actual restoration efforts by resource managers are taking place (Wagner et al. 2008; Flory 2010; Kettenring & Adams 2011). Small-scale experiments may be missing much of the ecological variability and processes involved during large-scale restoration efforts (Petersen et al. 2003; D'Antonio et al. 2004; Erskine Ogden & Rejmanek 2005).

Additionally, the majority of experiments are limited temporally with only a couple years of post-treatment monitoring (Kettenring & Adams 2011; Wagner et al. 2008; Hazelton et al. 2014), and long-term results can differ from short-term initial findings (Blossey 1999), complicating managers' decisions when choosing the most effective strategies for restoration. For example, a study of invasive *Pteridium aquilinum* initially found good control after one application of herbicide, but after five years of monitoring the plant recovered and required additional treatments (Petrov & Marrs 2000). Other long-term studies of *P. aquilinum* revealed that effective control techniques change over time as native plants re-establish from reduced *P. aquilinum* cover (Pakeman et al. 2002; Cox et al. 2007). Therefore, large-scale experiments that incorporate long-term monitoring are needed in order to convey the best techniques for managers.

In addition to spatial and temporal limitations, many invasive plant control experiments only report results of the target invasive species and fail to track the recovery of native plant communities, which is often the ultimate goal of resource managers (Blossey 1999; Kettenring & Adams 2011; Hazelton et al. 2014). Furthermore, certain control techniques that result in good control of the target species may have negative impacts on native plant recovery; especially, when long-term herbicide use is involved (Matarczyk et al. 2002; Wootton et al. 2005; Kettenring & Adams 2011). Evaluations of the impacts to native species are needed to inform managers of best practices to restore the native plants lost by invasive species.

Wetlands are particularly vulnerable to plant invasions in part because they are landscape sinks for plant propagules, sediment, nutrients, and other pollutants coupled with high disturbance rates from floods and dewatering events (Zedler & Kercher 2004). Today, one of the most problematic invasive wetland plant species in North America is *Phragmites australis* (Cav.) Trin. ex Steud. (i.e., common reed; hereafter referred to as *Phragmites*) (Chambers et al. 1999; Kettenring et al. 2012a). *Phragmites* is a perennial clonal grass that is one of the most widely distributed flowering plants in the world (Holm et al. 1977; Rooth & Windham 2000). Although *Phragmites* is indigenous to wetlands throughout North America, an introduced, non-native lineage has rapidly expanded over the last century (Saltonstall 2002). This invasive lineage often forms dense monotypic stands that decrease plant diversity (Chambers et al. 1999; Bertness et al. 2002), reduce habitat quality for wildlife (Benoit & Askins 1999; Able & Hagan 2000; Blossey & McCauley 2000; Fell at al. 2006; Chambers et al. 2012), alter biogeochemical cycling (Meyerson et al. 1999; Findlay et al. 2003), increase fire hazards, and reduce access for recreational opportunities (reviewed in Getsinger et al. 2006).

Phragmites invasions have been very successful because of its rapid growth, flexible reproductive strategies, ability to withstand a broad range of environmental conditions, and its ability to colonize and thrive in disturbed, nutrient rich habitats

(Chambers et al. 1999; Minchinton & Bertness 2003; Mozdzer & Zieman 2010; Kettenring et al. 2015). *Phragmites* can grow to heights exceeding 4 m and densities up to 200 stems/m^2 (Haslam 1972; Marks et al. 1994; Warren et al. 2001). It reproduces both vegetatively, through rhizomes and tillers, and sexually, from seed (Cross & Fleming 1989; Meyerson et al. 2000). Historically, *Phragmites* expansion has been attributed largely to asexual reproduction. However, recent studies have found high genetic diversity within and among *Phragmites* patches, which suggests that sexual reproduction (seed dispersal) contributes to its expansion much more than previously perceived (Belzile et al. 2010; McCormick et al. 2010; Kettenring et al. 2011; Kettenring & Mock 2012; Douhovnikoff & Hazelton 2014).

Controlling the spread of *Phragmites* and restoring native plant-dominated wetlands is a goal of many wetland managers across North America (Marks et al. 1994). Several methods to control *Phragmites* have been used and studied (see Marks et al. 1994; Kiviat 2006; and Hazelton et al. 2014 for complete reviews) yet due to some limitations in previous research questions still remain about the most effective *Phragmites* control strategy. First, *Phragmites* experiments are being conducted at limited temporal and spatial scales and most evaluate treatment effectiveness for only one year (Hazelton et al. 2014) making them unable to track vegetation changes that may take several years to develop (Blossey 1999). Second, the most widely used and researched control method is herbicide application (Kettenring et al. 2012b; Martin & Blossey 2013; Hazelton et al. 2014) but questions remain about the most effective type of herbicide to use, the optimal timing of application, and the impacts to native plant communities.

The active ingredients glyphosate and imazapyr are both non-selective systemic herbicides that have been proven to provide effective control for *Phragmites* (Getsinger et al. 2006; Derr 2008a; Mozdzer et al. 2008). Imazapyr belongs to the herbicide family Imidazolinone and its mode of action impedes the enzyme acetohydroxy acid synthase (AHAS), also known as acetolactate synthase (ALS) (Tu et al. 2001). ALS is a catalyst for the production of the amino acids valine, leucine, and isoleucine, which are required for protein synthesis. Glyphosate belongs to the herbicide family Glycine and its mode of action inhibits the enzyme 5-enolpyruvyl-shikimate-3-phosphate synthase (EPSPS) of the shikimate pathway, which is essential for the production of aromatic amino acids (Tu et al. 2001). Unlike glyphosate, which is the only herbicide known to inhibit EPSPS, many other herbicides exhibit the same mode of action as imazapyr (Duke and Powles 2008).

Plant uptake of glyphosate and imazapyr also differ. While both can be adsorbed by foliar application, imazapyr can also be taken-up by roots (Tu et al. 2001). This potential root uptake can be especially problematic for non-target species as imazapyr can persist in the soil for several months and does not bind strongly to soil particles, thereby leaving it mobile and readily bioavailable in the soil. In contrast, glyphosate strongly binds to soil particles leaving it immobile and unavailable for plant uptake (Tu et al. 2001).

Previous research provides mixed evidence on whether the herbicides imazapyr or glyphosate are more effective at reducing *Phragmites* cover (Kay 1995; Moreira et al. 1999; Ailstock et al. 2001; Getsinger et al. 2006; Derr 2008a; Mozdzer et al. 2008; Cheshier et al. 2012; Lombard et al. 2012), whether summer or fall applications are preferred, and the long-term impacts of herbicide type and timing on native plants (Cross & Flemming 1989; Marks et al. 1994; Derr 2008a; Mozdzer et al. 2008). Finally, the majority of *Phragmites* experiments have been conducted on the Atlantic Coast and Great Lakes region (Kulmatiski et al. 2010) and experiments in other regions—with distinct climate and weather patterns—are needed to develop region-specific control methods such as in the arid Intermountain West, the focal region for the present study.

The broad goal of this study was to address some of the limitations found in previous *Phragmites* control experiments to inform *Phragmites* management decisions. In particular, we took a large-scale approach (several orders of magnitude larger than most experiments) that better represents the scale of actual management efforts, to test the effectiveness of two different herbicides and two timings of application on the reduction of *Phragmites*. We also assessed plant community responses for three years after the initial treatments, compared the response to native reference sites, and looked at factors such as water depth and soil properties to better understand abiotic factors that are likely to influence *Phragmites* control and native plant recovery. Here we address three main questions:

1) How do herbicide treatments affect *Phragmites* cover?

2) Following herbicide applications, are returning plant communities similar in composition to nearby native reference sites?

3) How do water depth and soil properties affect the control of *Phragmites* and the recovery of native plant communities?

Methods

Study sites

Four *Phragmites* control sites, and two reference sites, were selected along the eastern shore of the Great Salt Lake (GSL) and are managed by the Utah Division of Wildlife Resources (UDWR). The four control sites were located at Farmington Bay Waterfowl Management Area (FB1 & FB2), Howard Slough Waterfowl Management Area (HS), and Ogden Bay Waterfowl Management Area (OB) . The two reference sites were located at Farmington Bay (FBref) and Ogden Bay (OBref) Waterfowl Management Areas (Fig. 1). The GSL and its associated wetlands are recognized regionally and hemispherically as an important bird area (IBA) that provide critical habitat for a wide variety of wetland-dependent birds including waterfowl, shorebirds, and waterbirds (Aldrich & Paul 2002). The GSL is also part of the Western Hemisphere Shorebird Reserve Network with 35 million birds visiting the lake each year (Aldrich & Paul 2002). The invasion and expansion of *Phragmites* has replaced many of the high quality habitats these avian populations rely on.

The invasion of *Phragmites* into GSL wetlands is a fairly recent one with the first herbarium record collected in 1993 (Kulmatiski et al. 2011). The cause and exact timing of *Phragmites* establishment around the GSL is not well known. Most land managers suggest *Phragmites* invasions coincided with extensive flooding in 1986 (1284 m above sea level) which left a vast expanse of denuded mudflats (optimal conditions for *Phragmites* establishment) as the salt water receded (Kulmatiski et al. 2011; Randy Berger, UDWR, personal communication). Since that time, *Phragmites* has continued to expand and now encompasses > 9,300 ha. (Long 2014). Native wetland plants commonly replaced by *Phragmites* invasion that are targeted for restoration and management in GSL wetlands are *Bolboschoenus maritimus* (alkali bulrush); *Schoenoplectus acutus* (hardstem bulrush); *Schoenoplectus americanus* (three-square bulrush); *Distichlis spicata* (salt grass); *Salicornia europeae* (pickleweed); and *Typha* spp. (cattails).

Treatments

Herbicide application

At each of the four sites, five treatments were randomly assigned and applied to 1.2 ha plots. The five treatments applied were: (1) summer imazapyr application; (2) summer glyphosate application; (3) fall imazapyr application; (4) fall glyphosate application; and (5) untreated control. The initial treatment was applied in 2012 and follow-up treatments were applied in 2013 and 2014. Summer applications were implemented the last week of June and the first week of July. Fall applications were implemented the last week of August and the first week of September. A Softrak wetland tractor (Loglogic, Mutterton, Cullompton, Devon, EX15 1RW, UK) equipped with a piston-driven sprayer and a boomless nozzle was used to apply herbicides in 2012. In 2013, the same equipment was used except handgun nozzles were used to treat individual plants and patches of *Phragmites* in order to minimize herbicide application to non-target plant species. Glyphosate, under the trade name Aquaneat, and imazapyr, under the trade name Polaris were applied at a rate of 7 L/ ha. Application rates were chosen based on herbicide label recommendations and studies that have shown there is no need to use rates higher than those listed by label instructions (Cheshier 2012). A non-ionic surfactant under the tradename LI-700 was added to help control herbicide drift and with plant absorption. LI-700 was mixed as recommended by the label with 1.89 L/378.54 L of mixed solution. Due to no visibility within *Phragmites* patches during the initial treatment (2012), a Raven Cruizer II (Raven Industries Inc., Sioux Falls, SD) agricultural guidance system was used to guide uniform application of herbicides.

Mowing

All herbicide treated plots were mowed in January 2012 to remove the standing dead biomass. Wetlands were frozen at this time, allowing equipment to access the study sites. Two, ASV, PT-80 tracked skidstears (ASV Inc., Grand Rapids, MN), equipped with front-end hydraulic rotary mowers, were used to mow at three of the sites (FB2, OB, HS). The FB1 site could not be reached with the skidstears due to deeper water and thinner ice, and a Marsh Master (Coast Machinery LLC, Baton Rouge, LA) with a hydraulic rotary motor was used instead. In 2013, the skidstears were used to mow herbicide treated plots at FB2 and HS, while the Marsh Master was used at FB1 and OB.

Data Collection

Plant cover

A systematic sampling design was used to assess vegetation cover in both treatment plots and reference sites. Each plot was divided into thirds with two transects evenly spaced within each plot (Fig. 2). A Softrak and a handheld GPS were used to drive down each transect and measure distance between quadrats. Two 1 m^2 quadrats (1 quadrat perpendicular to each side of the transect) were placed approximately every 9.75

m along each transect, for a total of 20 quadrats per transect and 40 quadrats per treatment plot (Fig. 2). On the ground vegetation sampling consisted of ocular estimations of percent cover of live *Phragmites,* dead *Phragmites*, non-*Phragmites* vegetation, litter, open water, and bare ground in each quadrat, using the following cover classes: 0–1%, 1–5%, 5–25%, 25–75%, and 75–100%. Mid-points of each cover class were used to calculate means. The majority of plants were identified to species level unless identifying features were not yet present, whereas plants were identified to the genus level. Plant identification followed Utah Flora (Welsh et al. 1993) and recent nomenclature followed USDA PLANTS database (http://plants.usda.gov).

To compliment on the ground surveys in such large treatment plots, high resolution 4-band (RGB–red, blue, green, + NIR–near infrared) aerial imagery was used to track changes in *Phragmites* cover over the course of the study. Aerial imagery was contracted from multiple vendors from 2012-2015 (Table 1). Multiple vendors were used due to cost differences and budget constraints. Dates of flights and flight platforms varied across the study. Dates of flights varied due to weather conditions and unforeseen equipment maintenance and downtime. Highest quality imagery was collected during the two most critical years: 2013 after first treatment, and 2015 after final treatment. In 2012, only 3 of the four sites (OB, HS, FB2) were flown due to restrictions on the use of un-maned aerial vehicles (UAVs) in controlled airspace. Only fall data were used for analysis because fall flights represented most of the growing season.

Abiotic factors

Soil samples were collected in June 2012, prior to initial herbicide treatments, using an 8.25 cm diameter core auger. Three samples per transect (every 27 m) were collected for a total of 6 samples per treatment plot. A 30 cm deep sample of mineral soil below the organic layer was collected, placed in plastic bags, and put on ice until they could be transferred to a freezer. At the time of processing, soils were thawed overnight, homogenized, and a sub-sample was sent to the Utah State University Soils Analytical Laboratory (USUAL) for analysis of phosphorous (P; per Olsen & Summers 1982), pH (per Rhoades 1982), and electroconductivity (per Rhoades 1982). Another sub-sample was taken from the remaining soil, dried in an oven overnight, ground by a pestle and mortar, and sent to the Stable Isotope Lab at Utah State University for analysis of total nitrogen (TN). Analysis of TN was determined by continuous-flow direct combustion and mass spectrometry (CF-IRMS).

Water depth was measured at each quadrat, along one side of each transect, for a total of 20 measurements per treatment plot, during the same time as plant cover estimations (Fig. 2).

Data Analysis

Phragmites and non-*Phragmites* percent cover were analyzed separately with a linear mixed model ANOVA with repeated measures using JMP version 12.1.0 (SAS Institute). The statistical model included the fixed effects of treatment (UC, SG, SI, FG, FI), year (2013 summer, 2013 fall, 2014 summer, 2014 fall, 2015 fall), and their interaction. Site (OB, HS, FB1, FB2) and the interaction of site with both treatment and year were random effects in the model. Data within each plot were averaged, and the means were used as data in the analyses to avoid pseudoreplication (Hurlbert 1984). *Phragmites* percent cover values were converted to proportions and a logit transformation was applied to better meet the assumptions of normality and equal variances. Non-*Phragmites* cover values were square-root transformed. Pre-treatment data (2012) are shown in figures but were excluded from analysis because all plots had similar pre-treatment percent cover values and minimal correlation between pre-treatment and post-treatment values. Post-hoc analysis was conducted using contrasts for pertinent comparisons. Analysis of specific plant species, open water, and *Phragmites* litter were unable to meet model assumptions, therefore only descriptive statistics are reported. Means and standard errors presented in figures were calculated directly from the proportion data.

We conducted pilot studies in attempts to automate the identification of *Phragmites* and other plant communities in the aerial imagery. In all years' imagery, ERDAS Imagine 2010 could not effectively differentiate between *Phragmites* and other cover types. The variation in spectral signatures and textures within the *Phragmites* were greater than those between *Phragmites* and other cover types. Other researchers have witnessed this characteristic as well and determined that *Phragmites* is best identified using a combination of multispectral imagery and active remote sensing methods such as LiDAR (Gilmore et al. 2008) or Side Aperture RADAR (Bourgeau-Chavez et al. 2013). In the absence of these additional data sources, we determined that manually digitizing *Phragmites* cover would be the most efficient analysis.

Images were analyzed visually by a single expert observer. A combination of the RGB and NIR bands allowed for differentiation between *Phragmites* and native vegetation. This was then confirmed using texture (patterning, shading, and stature). Subsequent comparison of these methods to known control points within each image confirmed that the method was accurate at determining *Phragmites* near-monocultures as small as 1m^2. Digitized *Phragmites* area within each treatment plot was then used to determine percent cover. A one-way ANOVA model for each year was used to assess the main effects of treatment. Proportion data were logit transformed to better meet the assumptions of normality and equal variances. Post-hoc analysis was conducted using contrasts for pertinent comparisons. Means and standard errors presented in figures were calculated directly from the proportion data.

Plant community data were characterized using non-metric multidimensional scaling (NMDS; Kruskal, 1964) with the vegan package (Oksanen et al. 2013) in R 3.2.4 (R Core Development Team 2016). The *metaMDS* function within vegan was used to standardize the data with a Wisconsin double standardization, and transforms the data using a square-root transformation, calculate a dissimilarity matrix using Bray-Curtis distance, run NMDS multiple times with random starts to avoid local optima, and rotate the axes of the final configuration (Oksanen et al. 2013). Species that occurred in less than 10% of the plots were removed to reduce disproportionate effects of rare species (McCune and Grace 2002).

A second NMDS analysis was used to test whether environmental factors and *Phragmites* litter cover were correlated with plant community composition, using the *envfit* function in the vegan package, with 10,000 permutations. Environmental factors included soil characteristics (TN, P, pH, salinity), and water depth. Soil samples were collected in 2012 and were not collected at reference sites, therefore inferences can only be made to pre-existing soil conditions and plant communities in the treatment plots.

The number of dimensions for each ordination was evaluated by constructing scree plots in order to see the reduction of stress with each additional dimension (McCune and Grace 2002). In both cases, a two-axis solution minimized stress to acceptable levels (< 20; McCune and Grace 2002). To evaluate goodness of fit between sample distance in ordination space and sample distance in the original data, the *stressplot* function in the vegan library was used (McCune and Grace 2002).

Results

Phragmites Cover

A significant treatment × year interaction was found for *Phragmites* percent cover (Table 2a). After the first year of treatments, *Phragmites* cover was decreased from pre-treatment levels of 78–82% to 6% (SG), 5% (SI), 2% (FG), and 1.5% (FI) (Fig. 3a). By the fall of 2015, *Phragmites* cover in the SG and SI treatments significantly increased to 62 and 52%, respectively, and no significant difference was found between the SG and SI treatments compared to the UC treatment. In contrast, *Phragmites* cover in the FG and FI treatments decreased to 16 and 12%, respectively, and was significantly different from the UC treatment in 2015. The main effect of season of application (summer vs. fall) was significant, with fall herbicide applications resulting in significantly lower *Phragmites* cover. The main effect of herbicide and the interaction between season of herbicide application and type of herbicide used (season × herbicide) were not significant.

Phragmites percent cover estimated from the aerial photos resulted in similar patterns to on-the-ground estimate (Fig. 3b). The main effect of treatment resulted in significant differences in *Phragmites* cover in 2013, 2014, and 2015 (Table 2c). After the first year of treatments, *Phragmites* covers were greatly reduced from pre-treatment levels of 90% or greater to 28% (SG), 27% (SI), 19% (FG), and 5% (FI). By 2015, *Phragmites* cover in all herbicide treatments was significantly lower than the UC treatment. Fall treatments resulted in significantly lower *Phragmites* cover compared to summer treatments. Digitized maps of each treatment and year are provided in the appendix.

Non-Phragmites Cover

Non-*Phragmites* percent cover increased significantly across all herbicide treatments over the course of the study with significant effects of treatment and year (Table 2; Fig. 4). In 2012, average non-*Phragmites* cover for all treatments was less than 2%. By 2015, non-*Phragmites* cover increased to 17% (SG), 50% (SI), 48% (FG), and 66% (FI). There was no significant difference between the type of herbicide used, season of application, or the interaction between season of application and herbicide. In 2015, all herbicide treatments resulted in significantly higher estimates of non-*Phragmites* cover compared to the UC treatment. A portion of the increase in non-*Phragmites* cover within the herbicide treatments was attributed to increases in *Lemna* spp. cover (Fig. 5). By the fall of 2015, *Lemna* spp. cover accounted for 10% (SG), 27% (SI), 28% (FG), and 36%

(FI) of the increase in non-*Phragmites* cover. Not including *Lemna* spp., increases in non-*Phragmites* cover was much less, with only 7% (SG), 23% (SI), 20% (FG), and 30% (FI) (Fig. 6). Non-*Phragmites* cover (minus *Lemna* spp.) was slower to recover, with minimal increase until the fall of 2014. Bulrush species (*Bolboschoenus maritimus*, *Schoenoplectus acutus*, and *Schoenoplectus americanus*) showed no increase in the SG and SI treatments. These species were slow to recover and remained below 5 and 10% in the FG and FI treatments, respectively (Fig. 7). *Typha* spp. was also slow to recover with little to no increase until the fall 2014 and 2015; although, estimates remained below 15% for all years (Fig. 8).

In 2013, after the first year of treatments, *Phragmites* litter cover remained at, or near, pre-treatment levels. Litter cover decreased monotonically in subsequent years, but did not decrease to below 20% until the fall of 2014 (Fig. 9).

Open water cover increased to 20% in the SG treatment and 25% in the SI treatment in the summer of 2014, but then decreased to 6 and 3%, respectively, by the fall of 2015 (Fig. 10). Open water cover in the FG and FI treatments remained at, or below 10% until the summer of 2014 where it increased above 20% and remained above 20% in the fall of 2015.

Plant Communities

NMDS analysis comparing plant communities within treatment plots and plant communities within reference plots reached a stable solution after 30 iterations (stress = 14.68). The two-axis solution produced a non-metric fit of $R^2 = 0.98$ and a linear fit of 0.92. NMDS 1 axis scores were higher for plots with less *Phragmites* and *Phragmites* litter. NMDS 2 scores were higher for plots associated with plant species that are found in shallower water depths and less frequently flooded conditions.

Pre-treatment (2012) plant communities were similar in composition with all treatment plots clustered around *Phragmites* (Fig. 11). After the first year of treatments (2013), plant communities in the herbicide plots changed from pre-treatment levels and resulted in a higher abundance of litter with very little other vegetation observed. In the fall of 2013, some of the herbicide treatment plots were still mostly clustered around litter. A few plots were starting to show a shift to open water and plant communities consisting of higher abundances of *Lemna* spp. and *Typha* spp. In the summer of 2014, herbicide plots resulted in less abundance of litter and were mainly clustered around open water, algae, with some *Typha* spp. and *Lemna* spp. A couple plots resulted in a higher abundance of *Ranunculus* spp. and *Rumex* spp. Reference sites were associated with higher abundances of *B. maritimus* (OBref) and *S. americanus* (FBref). A few of the herbicide plots were starting to show a trajectory towards similar plant composition as the FBref plot. In the summer of 2014, no treatment plots showed a trajectory toward the OBref plot. Similar results were found in the fall of 2014 with even more of the herbicide plots showing a trajectory towards the FBref plot, but still consisted of higher abundances of *Typha* spp. and *Lemna* spp. By the fall of 2015, plant composition in the herbicide plots still showed little resemblance to the OBref plot with some plots still showing a slow trajectory towards the FBref plots. Most of the summer treatments showed a trajectory back towards the UC treatments and pre-treatment plant composition with higher abundances of *Phragmites*.

NMDS analysis of plant composition within treatment plots and how environmental factors are influencing composition reached a stable solution after 43 iterations (stress = 16.13). The two-axis solution produced a non-metric fit of $R^2 = 0.97$ and a linear fit of $R^2 = 0.88$. NMDS 1 axis scores were higher for plots with less *Phragmites*. NMDS 2 scores were higher for plots associated with plant species that are found in deeper water. Factors that were significantly correlated with plant community composition included salinity, pH, litter, and water depth (Fig. 12). Measured values for soil chemistry factors and water depth are presented in table three. The length of vectors showed that litter ($r^2 = 0.47$) and water depth ($r^2 = 0.32$) explained most of the variation in plant community composition. Litter and water depth also showed divergent vectors suggesting a negative correlation, whereas the amount of litter decreases as water depth increases. Most plant species, with the exception of *Phragmites*, were associated with less litter. *Lemna* spp. and *S. americanus* were associated with treatment plots in deeper water. Treatment plots with less litter and shallow water depths were associated with *Hordeum jubatum*, *Polypogon monspeliensis*, and *Rumex* spp. Treatment plots with intermediate water depth and less litter were associated with *B. maritimus*, *Typha* spp., algae, and open water.

Discussion

In this study, we found that large-scale control of *Phragmites* was much more effective with the fall herbicide treatments compared to summer treatments. In addition, we found no difference between the two herbicides when they were compared for each application timing; fall treatments were equally successful and summer treatments were equally unsuccessful. However, native plant recovery (including important emergent plants found in this region) was slow, and limited, even when *Phragmites* was effectively controlled. Plant community variation in composition was mainly driven by water depth and the amount of *Phragmites* litter covering the area. Plant communities within each *Phragmites* treatment plot differed greatly from reference sites, although some plots showed a slow trajectory towards one of the reference plant communities. These results suggest that applying glyphosate or imazapyr in the fall can greatly reduce *Phragmites* cover after three years of consecutive treatments, but recovery of native plants is minimal and may need to be addressed through additional restoration actions.

Past *Phragmites* control studies focusing on herbicide treatments have left managers with unanswered questions and conflicting information about the best time of year to apply herbicides for maximum efficacy. Contrary to recent studies by Mozdzer et al. (2008) and Derr (2008b) that found summer treatments were just as, or more effective at controlling *Phragmites* than fall applications, our study indicates that fall applications provide significantly greater longer-term control of *Phragmites* than summer applications. One reason for this discrepancy could be different duration of monitoring in the present study versus previous work, and the long-term effects of herbicides. Both of the previous studies monitored *Phragmites* control for only one year after treatments, whereas our study monitored for three years after the initial treatment and one year after the final follow-up treatment. Our findings suggest that *Phragmites* sprayed in the summer may temporarily reduce aboveground growth for a couple years, but summer herbicide applications may not be killing rhizomes. Our results corroborate earlier studies and herbicide label recommendations that suggest good to excellent control of *Phragmites* is achieved by applying herbicides late in the summer or fall (Ailstock et al. 2001; Carlson et al. 2009; Lombard et al. 2012). During this time of the growing season, *Phragmites* is translocating nutrients to belowground parts, therefore simultaneously translocating herbicides and killing the rhizomes (Cross & Flemming 1989; Marks et al. 1994). In addition, fall herbicide applications can be applied after many native plants have initiated dormancy, thereby decreasing deleterious effects to non-target plants (Marks et al. 1994; Ailstock et al. 2001). Our results are confirmed with a seven-year study that treated *Phragmites* with glyphosate and found higher success once herbicide applications switched from summer to fall (Lombard et al. 2012).

Past *Phragmites* control studies have also resulted in conflicting advice about the most effective herbicide to use. Here we found no significant difference between glyphosate and imazapyr in their ability to control *Phragmites*. Conversely, some studies have found imazapyr to be superior to glyphosate (Kay 1995; Getsinger et al. 2006; Derr 2008b; Mozdzer et al. 2008). However, Kay (1995) reported the difference in effectiveness only lasted for one year and Getsinger et al. (2006) reported that when applied to large patches (several acres) the difference was negligible. In the current study, imazapyr did provide slightly lower *Phragmites* cover estimates each year in the fall treatments; yet, this difference was minimal and does not support that imazapyr should be used over glyphosate. Furthermore, the use of imazapyr has been shown to negatively impact the recovery of non-target species when compared to glyphosate, which may be detrimental to restoration efforts and should be considered when deciding on the type of herbicide to use (Mozdzer et al. 2008).

Comparisons between treatment plots and carefully selected reference plots establish standards that can be used to make strong inferences as to whether treatments are successful (reviewed in Neckles et al. 2002). Until now, only one *Phragmites* control study has made these comparisons (see Moore et al. 2012), even though the goal of many invasive plant restoration programs is to decrease invasive plant cover and increase desirable native plant communities (Noss 1990; D'Antonio & Meyerson 2002; Kettenring & Adams 2011). Unfortunately, restoration of invaded systems often results in either short-term control of the target invasive species, establishment of other invasive plants, and/or limited establishment of desirable native plant species (D'Antonio & Meyerson 2002; Kettenring & Adams 2011).

In our study, we found that plant communities in the majority of treatment plots, showed little resemblance to the two reference plots, with only a few plots showing a slow trajectory towards the FBref plot. *Bolboschoenus maritimus* accounted for 25% of the vegetation cover in the OBref plot, while *Schoenoplectus americanus* accounted for 40% of the vegetation cover in the FBref plot. We found very limited recovery of these species in our study. In fact, cover estimates of important native emergent species found in this region—*B. maritimus*, *S. acutus*, and *S. americanus*—were <10% combined in most treatment plots.

Even though we saw low recovery of important native emergent vegetation, we did see a shift from emergent plant communities to open water and open water plant communities. Two factors that strongly influenced the returning plant communities was *Phragmites* litter and water depth. Deeper water resulted in less litter and plant communities with higher abundances of *Lemna* spp. and included some *S. americanus*. Plant communities at intermediate water depths resulted in the return of some *Typha* spp., algae, and *B. maritimus*. Shallower water depths resulted in the return of *Rumex* spp. and two grasses (*Hordeum jubatum*, *Polypogon monspeliensis*).

Both *Lemna* spp. and open water increased greater than 20% in the fall herbicide treatments by 2015. Getsinger et al. (2006) found a similar shift in emergent vegetation to open water plant communities when *Phragmites* herbicide treatments were followed by burning and subsequent flooding. Depending on management goals and objectives, these changes could be desirable, especially in areas where waterfowl management is a priority, as is the case in this study (Randy Berger, UDWR, personal communication). Nonetheless, it is unknown if these less vegetated open water areas will be more susceptible to *Phragmites* re-invasion compared to dense native emergent plant communities. Results from an outdoor mesocosm experiment suggest that flooding does limit *Phragmites* invasion from seed, but also lowers biotic resistance (plant competition) of the native resident community due to fewer plant species having the ability to withstand anaerobic conditions (Byun et al. 2015). Intact and diverse native plant communities might be able to resist, or at least slow down, the re-invasion of *Phragmites* by limiting the availability of space and resources (Kennedy et al. 2002; Kettenring & Adams 2011; Byun et al. 2013). Field experiments have demonstrated that native plants are capable of competing with *Phragmites* seedlings (Minchinton 2002; Minchinton &

Bertness 2003; Kettenring et al. 2015) and can reduce the amount of *Phragmites* emerging from rhizomes (Konisky & Burdick 2004; Peter & Burdick 2010). In fact, when native species were planted with *Phragmites* rhizomes, increased species richness significantly reduced *Phragmites* density, biomass, and survival (Peter & Burdick 2010). When few native plant communities are returning following *Phragmites* control, as was the case here, active revegetation may be needed in order to re-establish desirable native species and reduce the possibility of re-invasion.

Passive restoration (i.e. without actively re-vegetating) relies heavily on intact and diverse seedbanks along with the ability of adjacent desirable vegetation to supply plant propagules to the restored site. Studies have shown that diverse native seedbanks can be found under *Phragmites* monocultures (Ailstock et al. 2001; Minchinton et al. 2006; Hallinger & Shisler 2009; Baldwin et al. 2010); however, recruitment of non-*Phragmites* cover following *Phragmites* removal depends on the type of method used. For instance, more diverse and rapid re-colonization of non-*Phragmites* cover was found when the dead biomass was removed either by fire (Ailstock et al. 2001) or by cutting and raking (Carlson et al. 2009). One factor that likely played a major role in the limited re-establishment of native emergent plants in the present study was the amount of litter left behind following the mowing treatments. This litter layer most likely prevented sunlight from reaching the soil surface, therefore prohibiting the re-establishment of many plants. Our results indicate that the amount of litter was a significant factor contributing to the composition of plant communities following herbicide application. Higher amounts of litter cover led to a reduced number of emerging species while areas with less litter had a greater number of different species established. However, total cover of these species was low. We also found that non-*Phragmites* cover, excluding *Lemna* spp., did not increase until the litter layer substantially decreased. Managers implementing *Phragmites* control programs need to consider methods that can reduce or facilitate quicker decomposition of *Phragmites* litter, especially in areas where prescribed fires are limited (e.g., due to air quality concerns or proximity to housing developments).

In addition, the spatial scale of *Phragmites* infestation and the scale of control efforts in our study most likely also played a role in the limited establishment of non-*Phragmites* cover following treatments. Our treatment sites consisted of large, monotypic stands of *Phragmites* that had persisted for several years. The longer *Phragmites* has been present the longer *Phragmites* has contributed to the seed bank (D'Antonio & Meyerson 2002). Also, large-scale *Phragmites* invasions can mean that remnant native plant communities are sufficiently far away such that they are unable to supply new propagules needed for establishment. For example, Erskine Ogden and Rejmanek (2005) used a small-scale pilot study to choose the best treatment for recovery of native plants in a *Foeniculum vulgare* (fennel)-dominated system and then applied that treatment at the landscape scale. The pilot study resulted in a significant increase in native plant richness, but when applied to a landscape scale, an introduced grass dominated. They attributed this discrepancy to the fact that the small-scale pilot study included more diverse plant communities nearby whereas in the large-scale invasion, propagule sources were too far away to drive native plant colonization.

As with any restoration activity, longer-term monitoring of *Phragmites* control methods is essential to ensure goals and objectives are being met and that adaptive management can occur if they are not; yet many studies and management practices fail to monitor invasive plant control activities for more than two years (Kettenring & Adams 2011; Hazelton et al. 2014). Failure to monitor for several years could lead to misplaced assumptions and inferences that may not reflect plant community or ecosystem change over the longer-term (Blossey 1999). This finding was especially true in our study, where if monitoring only lasted for a couple years, results would have implied that summer and fall treatments were equally effective at reducing *Phragmites* cover. Limited monitoring would have also left us with a bleak picture of native plant recovery. In the final year of monitoring, we saw a slight increase in important emergent vegetation and in some instances plant communities were starting to show a trajectory toward native reference plots. Whether this trajectory continues, creates a new novel plant community, or reverts back to a *Phragmites*-dominated community will only be answered by continued monitoring. Finally, the fact that we and others (Ailstock et al. 2001; Warren et al. 2002; Getsinger et al. 2006; Carlson et al. 2009; Lombard et al. 2012) found that consecutive years of herbicide treatments are necessary to effectively reduce *Phragmites* cover, and that *Phragmites* persists even after such treatments, further elucidates the need for longer-term monitoring and possibly, continual control.

Management Implications and Conclusions

The results of our study provide a number of science-based recommendations for large-scale management of *Phragmites* and efforts to restore native plant communities in

Phragmites-dominated wetlands. First, managers should apply herbicides in the fall for the most effective longer-term control. Glyphosate is the best candidate herbicide as it is less expensive, just as effective, and may have less of a negative impact on native plant recruitment compared to imazapyr. However, the continued use of herbicides with the same mode of action can result in herbicide resistant plants (Tu et al. 2001). We found no significant difference between glyphosate and imazapyr in their ability to reduce *Phragmites* cover; therefore, switching between the two herbicides, or mixing the two together—due to their different modes of action—may help reduce the risk of herbicide resistance (Green 2007). Second, native emergent plant communities are slow to return and may be limited by seed bank diversity, propagule dispersal from remnant propagule source communities, and *Phragmites* litter. Due to these limitations, large-scale control efforts may need to be supplemented with active revegetation in order to facilitate native plant growth and decrease the opportunity for *Phragmites* or other invasive plant re-invasion. Third, the standing dead material and litter following herbicide treatments is an issue that needs to be addressed through additional management actions. Due to increased restrictions on the use of prescribed burns in GSL wetlands, we used rotary mowers that unfortunately did not mulch the dead *Phragmites* as much as we hoped. An alternative approach could be the use of a flail mower which is much more effective at cutting *Phragmites* into very small pieces (Chad Cranney, personal observation) and might aid in quicker decomposition of the litter left behind. Another option, that is very costly and logistically complicated, is the complete removal of the dead *Phragmites* biomass. This option would also provide better conditions that are conducive to active revegetation.

However, caution should be exercised as exposed bare ground provides prime conditions for *Phragmites* seed germination. Fourth, sites with deeper water have reduced *Phragmites* litter but nonetheless have transitioned from emergent to open water communities. Depending on the specific management goals, open water habitat may be a desirable result, but it is not known how long these conditions will last. Lastly, longer-term or continual monitoring is essential, especially when dealing with a plant that requires at least three consecutive years of herbicide applications and will most likely never be eradicated from large infestations.

CHAPTER 3

SUMMARY AND CONCLUSIONS

The negative impacts associated with invasive plant species, such as declining biodiversity, altered fire regimes, changes in nutrient and hydrological cycling, decreased wildlife habitat, and changes in the way people use these ecosystems, has led to increased efforts by resource managers to restore these degraded habitats (Hobbs & Humphries 1995; Wilcove et al. 1998; Mack et al. 2000; Ehrenfeld 2003). Invasive plant species control programs often aim to restore invaded habitats to a more desirable species composition and community structure by promoting native species establishment. Unfortunately, implementation of control techniques and restoration efforts by resource managers has shown highly variable results despite extensive scientific research and experiments regarding invasive plant species (Mack et al. 2000; Kettenring & Adams 2011). This discrepancy may be due to some of the limitations found in previous invasive plant control studies.

The goal of our study was to address some of the limitations found in previous *Phragmites* control experiments in order to better inform management decisions concerning effective methods and strategies to reduce *Phragmites* cover and increase native plant cover. The limitations in past research includes: 1) limited temporal and spatial scales that do not represent the scale at which actual management efforts are implemented; 2) studies that focus on efficacy of treatments in terms of invasive plant mortality, rather than changes in native plant recovery; and 3) short-term monitoring (often <2 years) that misses successional changes in plant communities that may take several years (Blossey 1999; Wagner et al. 2008; Stromberg et al. 2009; Kettenring & Adams 2011; Hazelton et al. 2014). These limitations make translating results from research to broader scale implementation difficult and managers are in need of applicable science-based information that will improve success of *Phragmites* control efforts (Kettenring & Adams 2011).

Here we took a large-scale approach that better represents the scale of actual *Phragmites* management efforts, to evaluate the effectiveness of glyphosate and imazapyr herbicide applications. We specifically tested which herbicide was most effective at reducing *Phragmites* cover, what timing of application (summer vs. fall) was more effective, how these treatments affected native plant communities, and how soil chemistry properties and water depth affected returning plant community composition.

We found the most effective time to apply herbicides for large-scale control of *Phragmites* was in the fall. After three consecutive years of herbicide applications, fall treatments resulted in significantly lower percent cover of *Phragmites* (<20%) compared to summer treatments (>50%) and the UC treatment (>50%). However, we did not find any difference when we compared the use of glyphosate and imazapyr and their effectiveness at reducing *Phragmites* cover. The use of imazapyr in other studies has shown to delay the recovery of native plants (Mozdzer et al. 2008). Additionally, others have suggested that earlier summer applications might be more detrimental to native plants compared to fall applications when many of the native plants have entered dormancy (Marks et al. 1994; Ailstock et al. 2001). We had hoped to test how these two different herbicides and their timing of application affected native plant recovery, but unfortunately, with such minimal recovery across all treatments we were unable to do so. Future research is needed in order to test whether or not imazapyr and summer applications of either herbicide are negatively affecting the recovery of native plants.

The re-establishment of native plant communities is a high priority for many invasive plant control programs, but unfortunately, reviews of invasive plant control studies have found that in many cases native plants are not returning (Kettenring & Adams 2011; Hazelton et al. 2014). Our study also found very low cover estimates of native plant cover, especially for important emergent plants in this region. However, we did see an increase in open water plant communities and in open water areas. We also started to see some plant communities within the treatments becoming more similar to one of the native reference plots. How long open water communities will persist and whether or not some of the treatments will continue a trajectory towards the reference site are undetermined, and will only be answered with continued monitoring.

Two factors that most likely limited the return of native plants in our study was light availability due to the large litter layer and the fact that no native plant communities were nearby. Although, we did not test the seedbank in this study, other studies have found that diverse seedbanks do reside under *Phragmites* monocultures but the right conditions are needed in order for them to propagate. In a complimentary study to ours, diversity of native plant recovery following *Phragmites* control methods was directly related to diversity of the seedbank (Rohal et al. unpublished data). Seedbanks themselves are limited by the amount of time *Phragmites* has persisted and how long native plants have been displaced from a particular site. The current condition of seedbanks should be assessed before passive or active re-vegetation methods are implemented.

With this study, we provide quantitative evidence in regards to the most effective strategies for large-scale control of *Phragmites* and the re-establishment of native plant communities. Resource managers will be able to use these results to choose the most appropriate timing of application and type of herbicide to use. Managers can also use our results in order to take steps concerning the limited native plant recovery and to try to manipulate factors that contribute to plant community composition such as water depth and *Phragmites* litter. Furthermore, our results can inform future research to investigate strategies to minimize *Phragmites* litter after spraying, and re-vegetation strategies that are applicable and logistically feasible at such large scales.

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Table 1. Vendors, platforms, dates, types of imagery, and resolutions of aerial photographs used to estimate *Phragmites* cover.

Source	**2012**	**2013**	**2014**	**2015**
Vendor	AggieAir Flying Circus	Aerographics	Utah State University Remote Sensing Lab	Aerographics
Platform	UAV	Fixed wing	Fixed wing	Fixed wing
Summer date	27 June 2012	8 July 2013	1 July 2014	31 August 2015
Fall date	No flight	5 October 2013	3 October 2014	No flight
Imagery type	RGB + NIR	4-band	4-band	4-band
Pixel resolution	RGB = 7cm NIR = 6 cm	5 cm	6 cm	5 cm

Table 2. ANOVA results for *Phragmites* (both ground and aerial estimates) and non-*Phragmites* percent cover, assessing treatment (SG = summer glyphosate, SI = summer imazapyr, FG = fall glyphosate, FI = fall glyphosate) and year (2013 summer, 2013 fall, 2014 summer, 2014 fall, 2015 fall) effects. The 2015 fall data were used for treatment contrasts. Values in bold are significant at α = 0.05.

Effect	***df***	***F*-value**	***p*-value**
(a) *Phragmites* percent cover			
Treatment	4,12	12.22	<**0.001**
Year	4,12	11.28	<**0.001**
Year × Treatment	16,48	3.15	**0.001**
Contrasts			
SG vs. UC	1,32.72	0.07	0.790
SI vs. UC	1,32.72	0.52	0.478
FG vs. UC	1,32.72	8.12	**0.008**
FI vs. UC	1,32.72	11.34	**0.002**
Glyphosate vs. Imazapyr	1,32.72	0.47	0.498
Season vs. Herbicide	1,32.72	0.002	0.962
Fall vs Summer	1,32.72	13.68	<**0.001**
(b) Aerial *Phragmites* percent cover			
2012 Treatment	4,8	1.32	0.340
2013 Treatment	4,12	35.14	**<0.001**
2014 Treatment	4,12	13.84	**<0.001**
2015 Treatment	4,12	18.74	**<0.001**
Contrasts			
SG vs. UC	1,12	4.99	**0.045**
SI vs. UC	1,12	12.30	**0.004**
FG vs. UC	1,12	41.35	**<0.001**
FI vs. UC	1,12	58.34	**<0.001**
Fall vs Summer	1,12	34.65	**<0.001**
(c) Non-*Phragmites* percent cover			
Treatment	4,12	6.85	**0.004**
Year	4,12	4.60	**0.018**
Year × Treatment	16,48	1.59	0.108
Contrasts			
SG vs. UC	1,12	10.49	**0.007**
SI vs. UC	1,12	16.75	**0.002**
FG vs. UC	1,12	13.04	**0.004**

Table 2. (cont.)

FI vs. UC	1,12	22.88	**<0.001**
Glyphosate vs. Imazapyr	1,12	2.05	0.177
Season vs. Herbicide	1, 26.97	0.16	0.694
Fall vs. Summer	1,12	0.56	0.468

Table 3. Summary of abiotic factors ($\overline{Y}$ ± SE) at each study site. Soil chemistry factors were measured in 2012. Mean water depth was calculated from all sampling periods. Electroconductivity (dS/m) was converted to ppt (parts per thousand) using (EC × 640) / 1000. TN = total nitrogen, P = phosphorus.

Site	TN (µg)	P (mg/kg)	pH	Salinity (ppt)	Water Depth (cm)
OB	66.37 ± 6.98	37.32 ± 3.50	7.81 ± 0.05	11.65 ± 1.53	16.55 ± 1.12
HS	52.43 ± 9.30	22.96 ± 0.95	7.89 ± 0.02	19.16 ± 1.31	10.86 ± 0.51
FB2	57.41 ± 5.04	39.04 ± 3.56	7.89 ± 0.02	17.05 ± 2.26	7.09 ± 1.30
FB1	87.16 ± 6.98	44.35 ± 1.37	7.88 ± 0.01	13.97 ± 0.78	24.48 ± 0.68
OBref	–	–	–	–	2.33 ± 2.33
FBref	–	–	–	–	26.63 ± 2.66

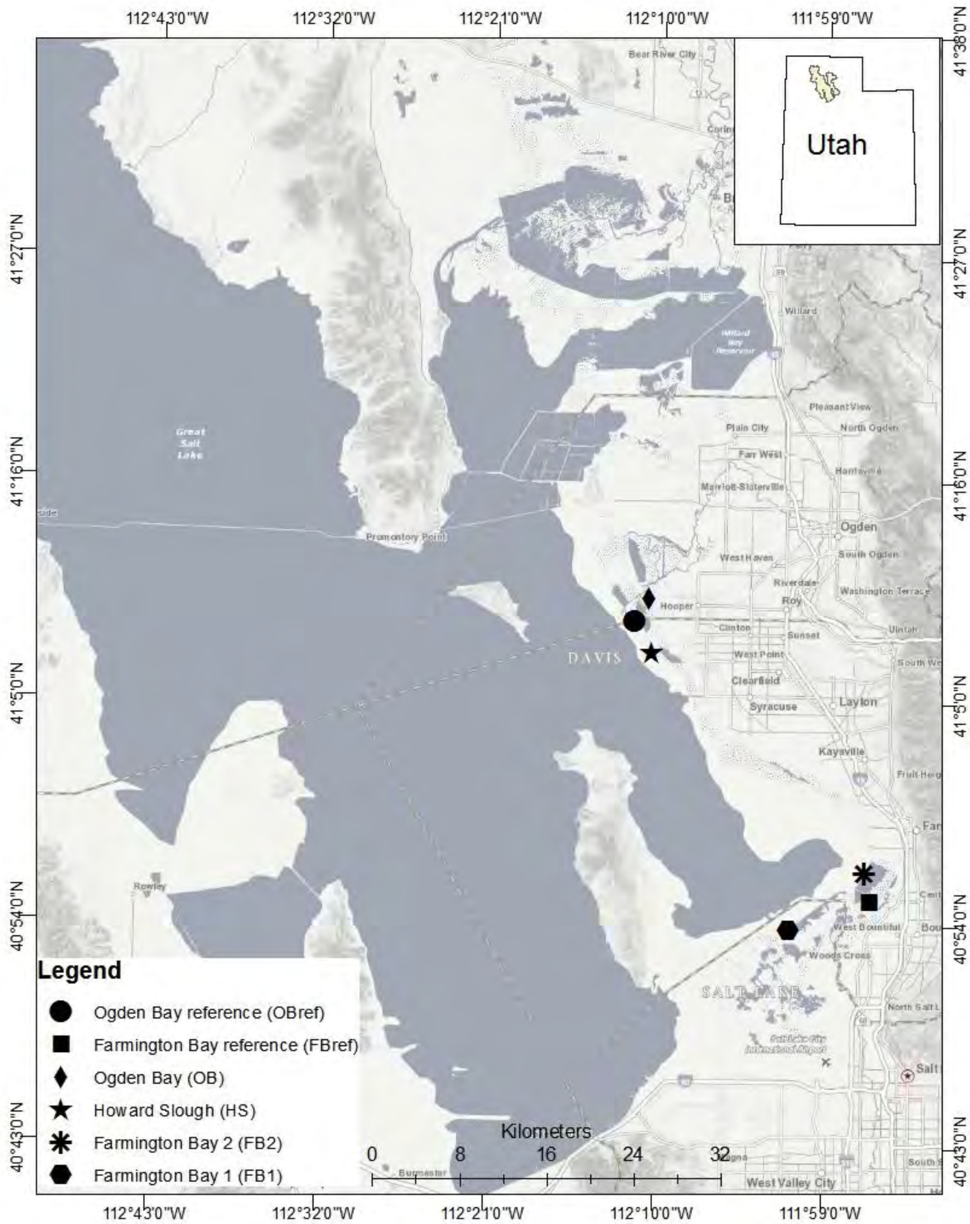


Figure 1. Geographic locations of *Phragmites* control sites (OB, HS, FB1, and FB2) and reference sites (OBref and FBref) along the eastern shore of the Great Salt Lake, Utah.

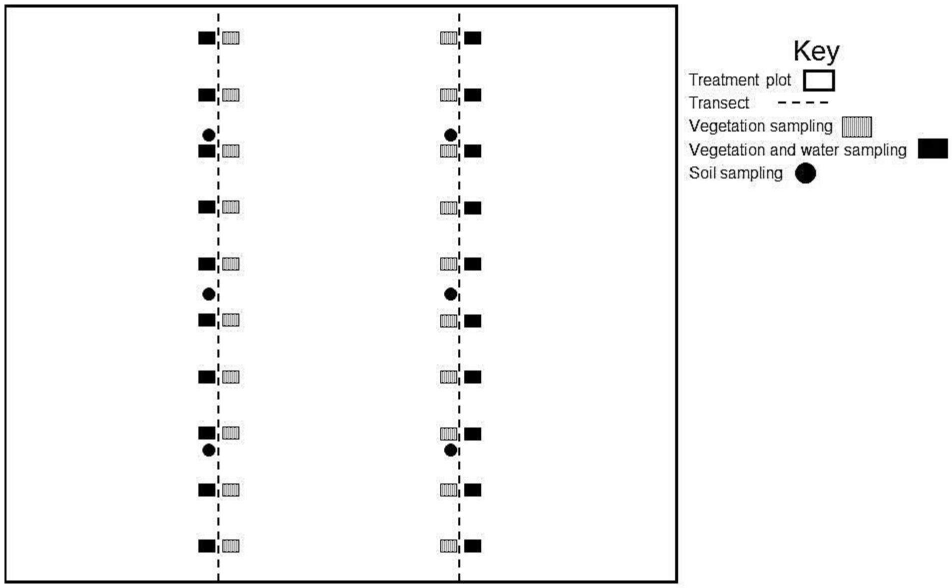
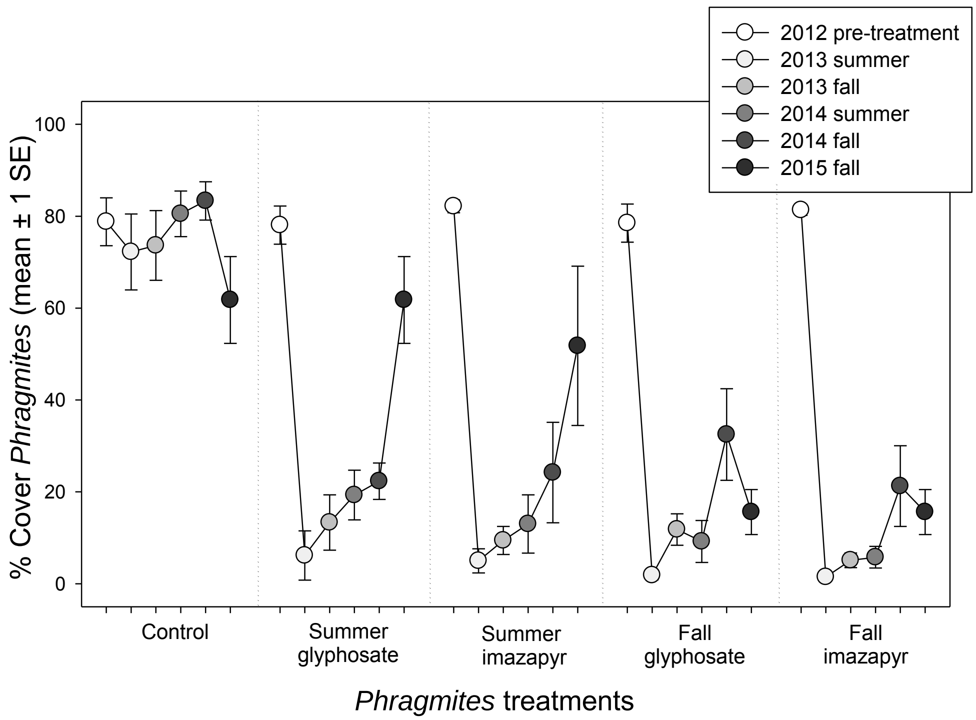


Figure 2. Schematic of one treatment plot illustrating the vegetation, soil, and water depth sampling locations. Each site had five treatment plots (one plot per *Phragmites* treatment). Reference site data collection followed the same sampling scheme except no soil samples were collected.

a



b

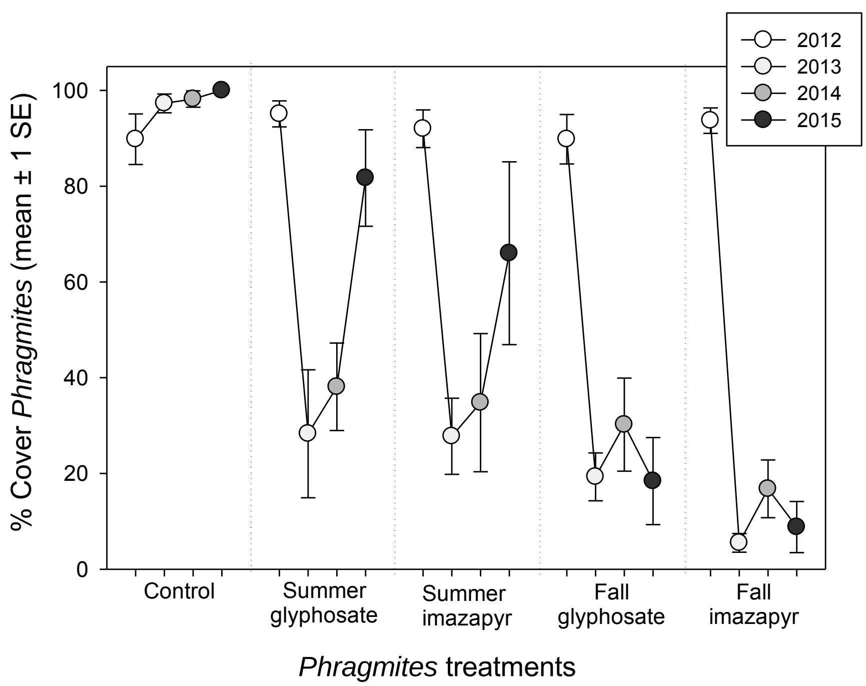


Figure 3. a) *Phragmites* cover by *Phragmites* treatment estimated from on-the-ground sampling over the course of the study. b) *Phragmites* cover by *Phragmites* treatment from aerial photos over the course of the study.

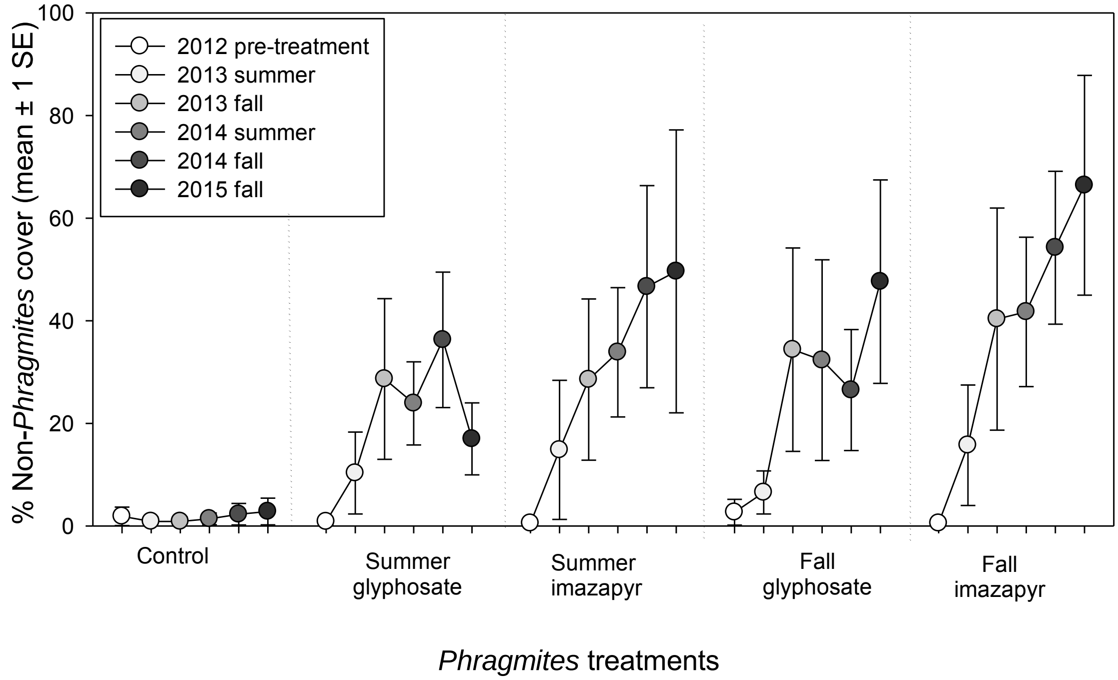


Figure 4. Non-*Phragmites* cover by *Phragmites* treatment over the course of the study. Cover estimates include all vegetation except *Phragmites*.

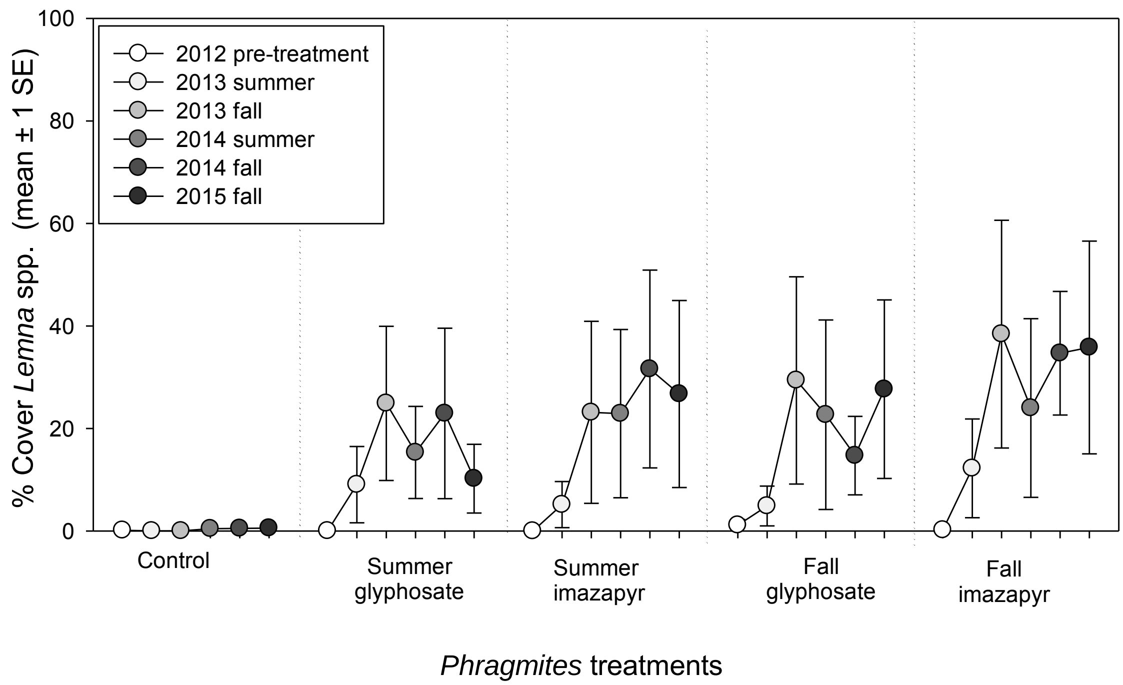


Figure 5. *Lemna* spp. cover by *Phragmites* control treatment over the course of the study.

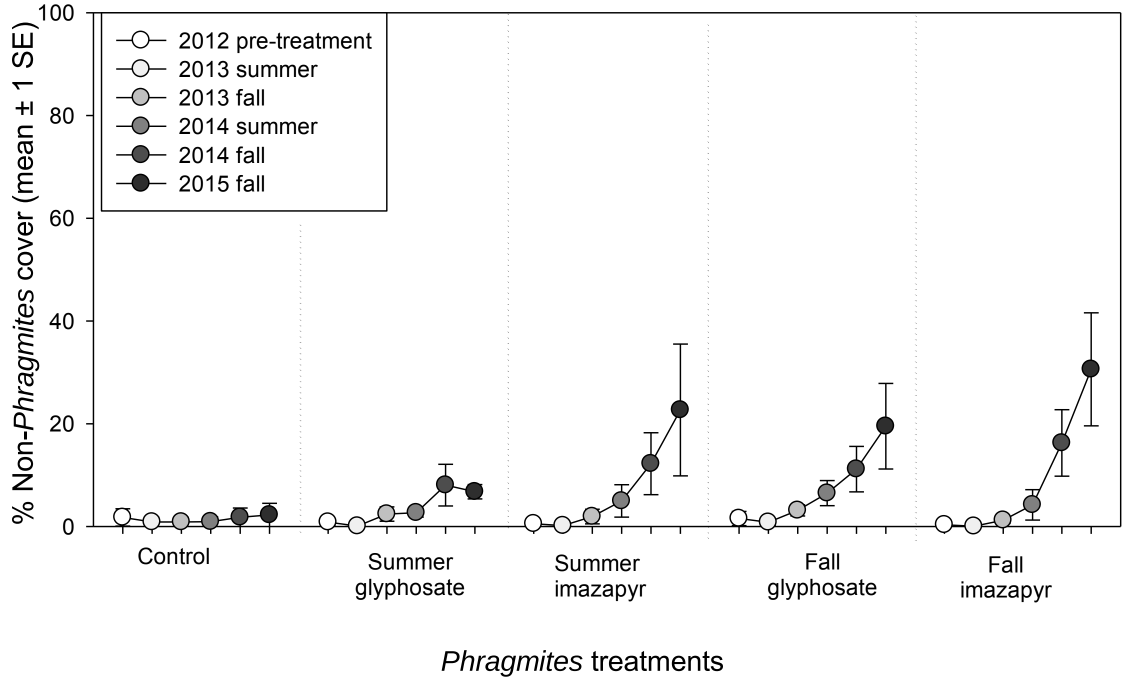


Figure 6. Non-*Phragmites* cover, excluding *Lemna* spp. cover, by *Phragmites* treatment over the course of the study.

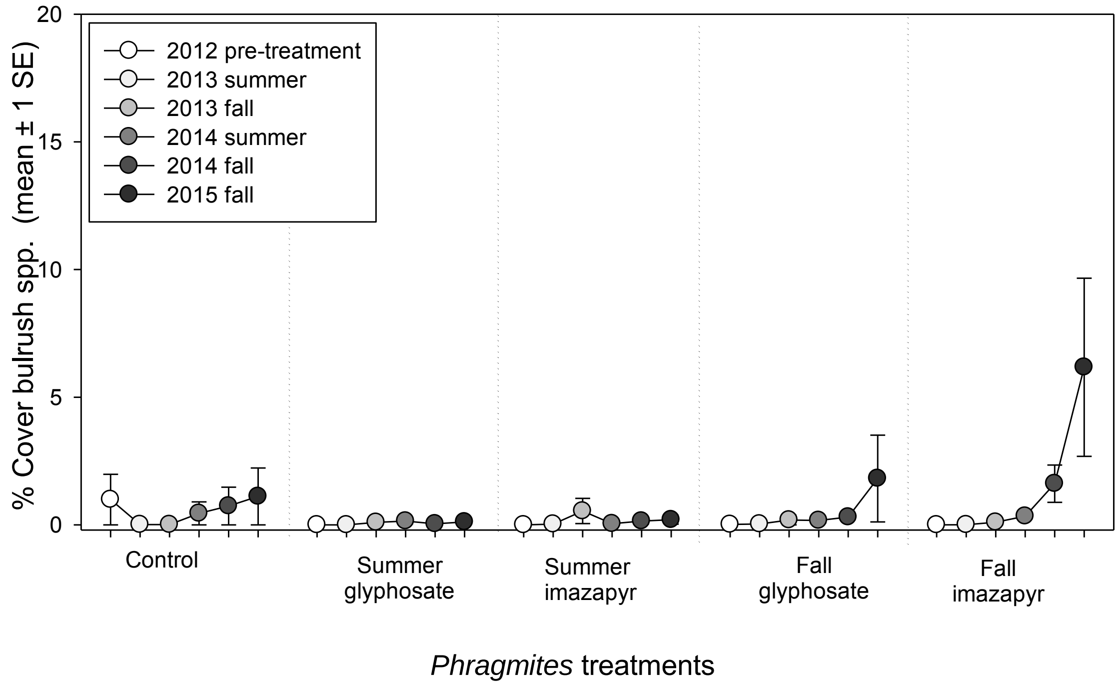


Figure 7. Percent cover of three bulrush species (*Bolboschoenus maritimus*, *Schoenoplectus acutus*, and *Schoenoplectus americanus*) cover by *Phragmites* treatment over the course of the study.

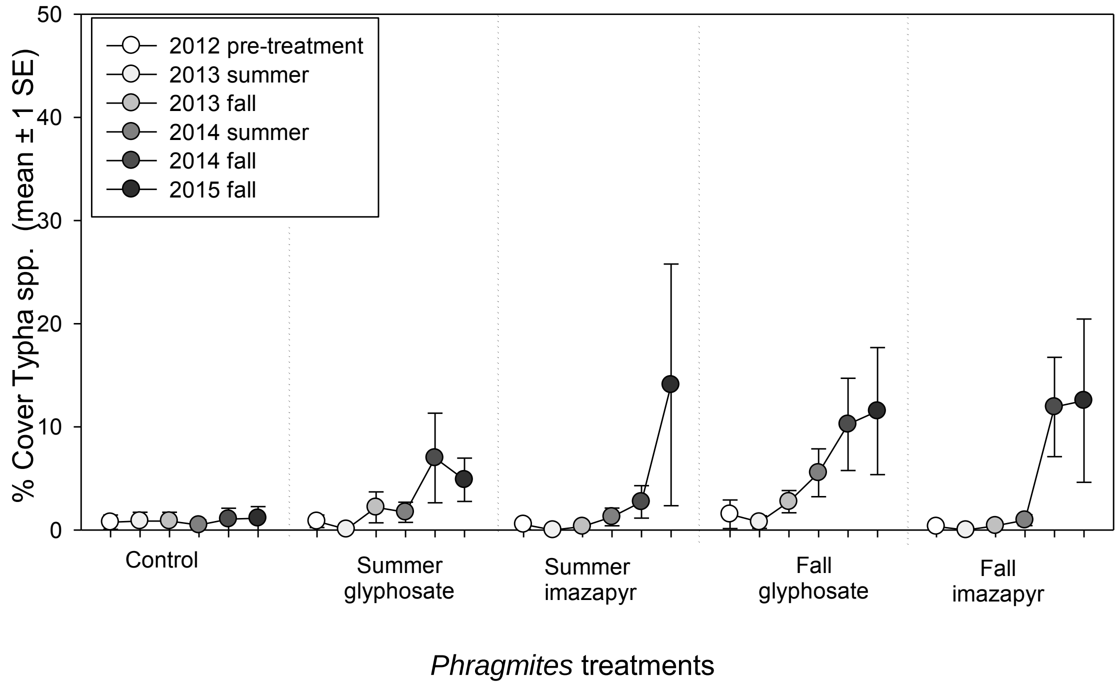


Figure 8. *Typha* spp. cover by *Phragmites* treatment over the course of the study.

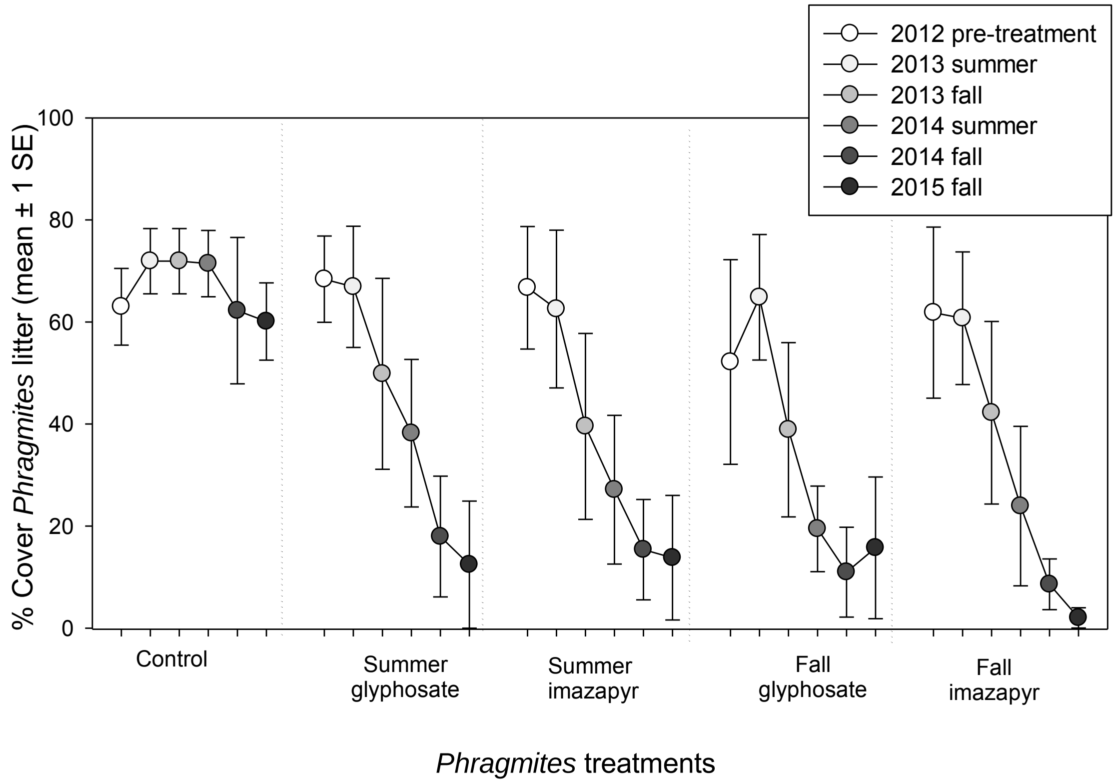


Figure 9. *Phragmites* litter cover by *Phragmites* treatment over the course of the study.

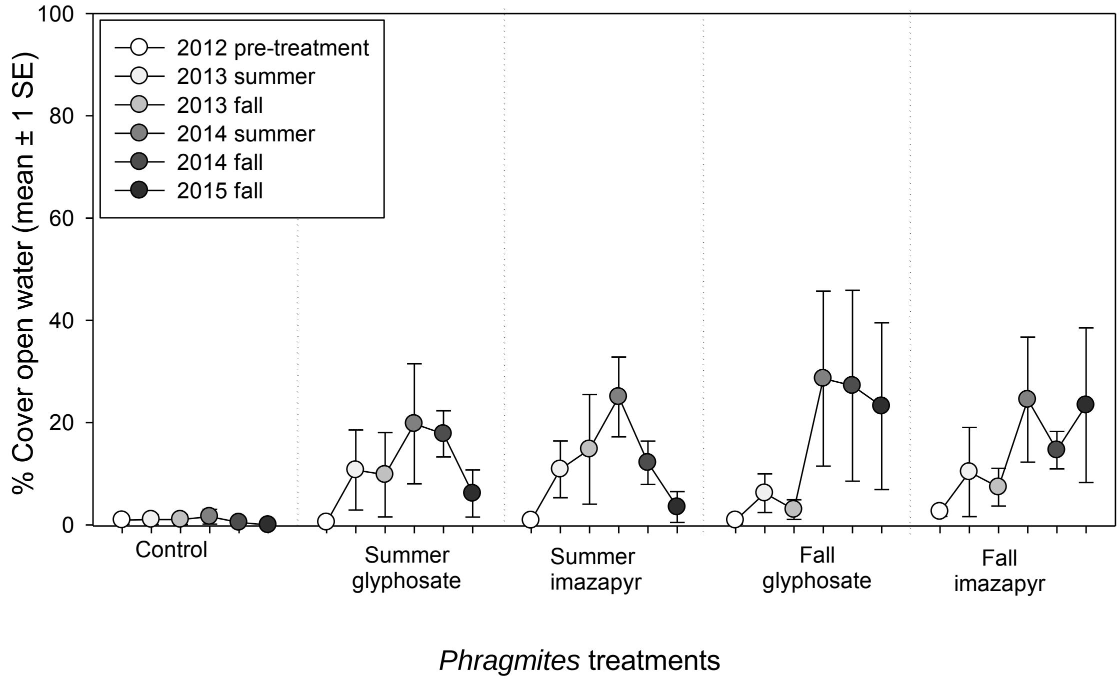


Figure 10. Open water cover by *Phragmites* treatment over the course of the study.

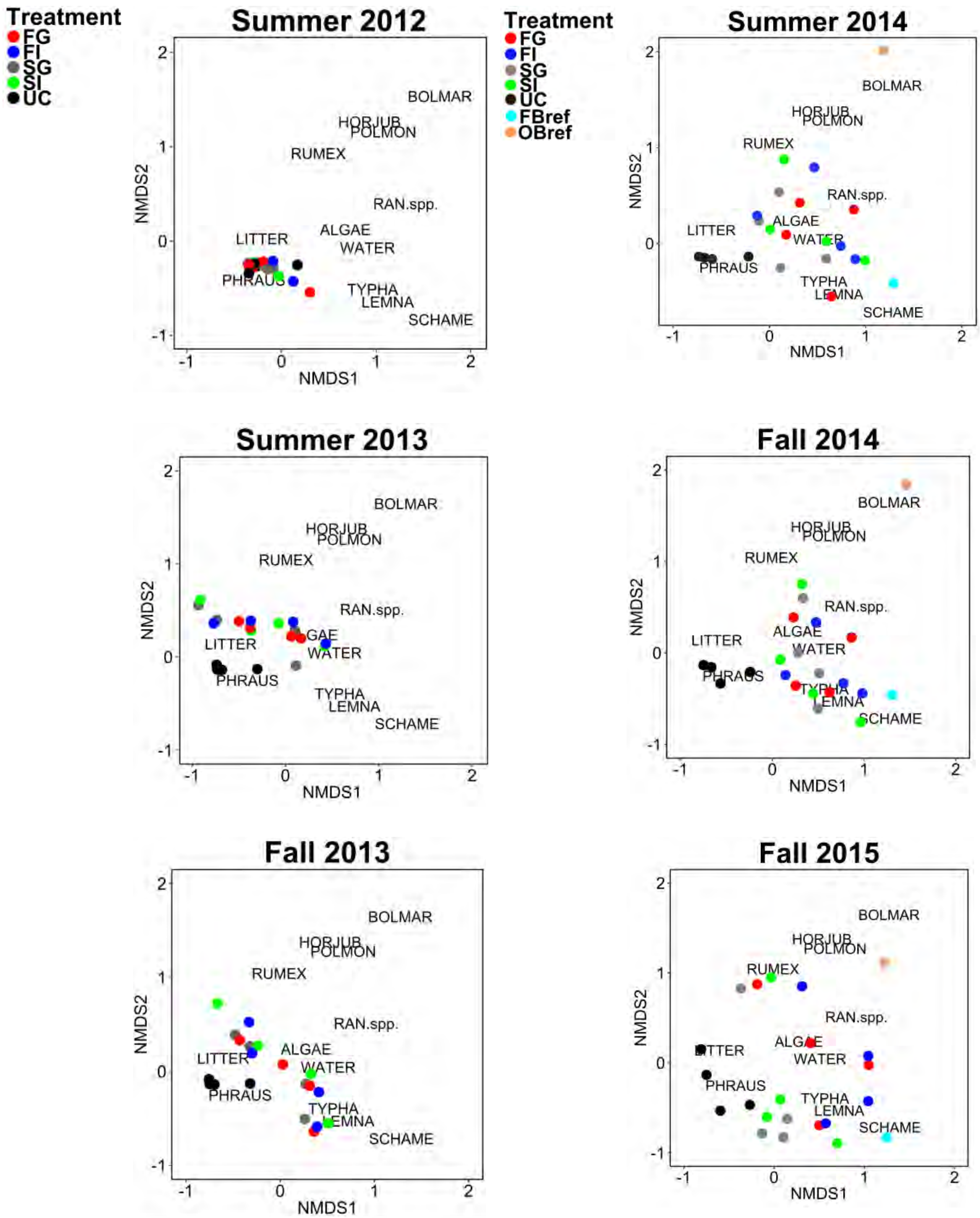


Figure 11. NMDS ordination showing plant community composition for plots by treatment. A single NMDS was run and separated by year (axis=2, stress=14.69). Plant communities were compared to reference plots starting in the summer of 2014.

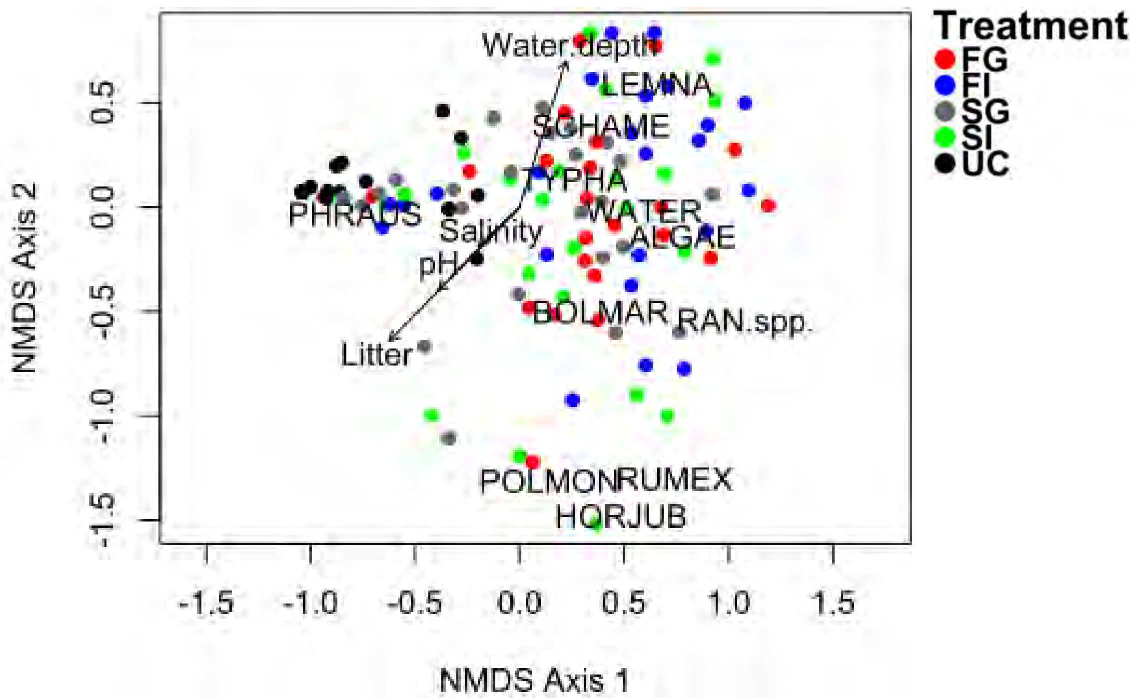


Figure 12. NMDS ordination of plant community composition (excluding litter) by treatment (axis = 2, stress = 16.13). Overlaid vectors were significantly correlated with plant composition (salinity: $p = 0.069$, water depth: $p = \leq 0.001$, pH: $p = \leq 0.001$, litter: $p = \leq 0.001$).

APPENDIX

Digitized Maps of *Phragmites australis*

Cover from Aerial Imagery

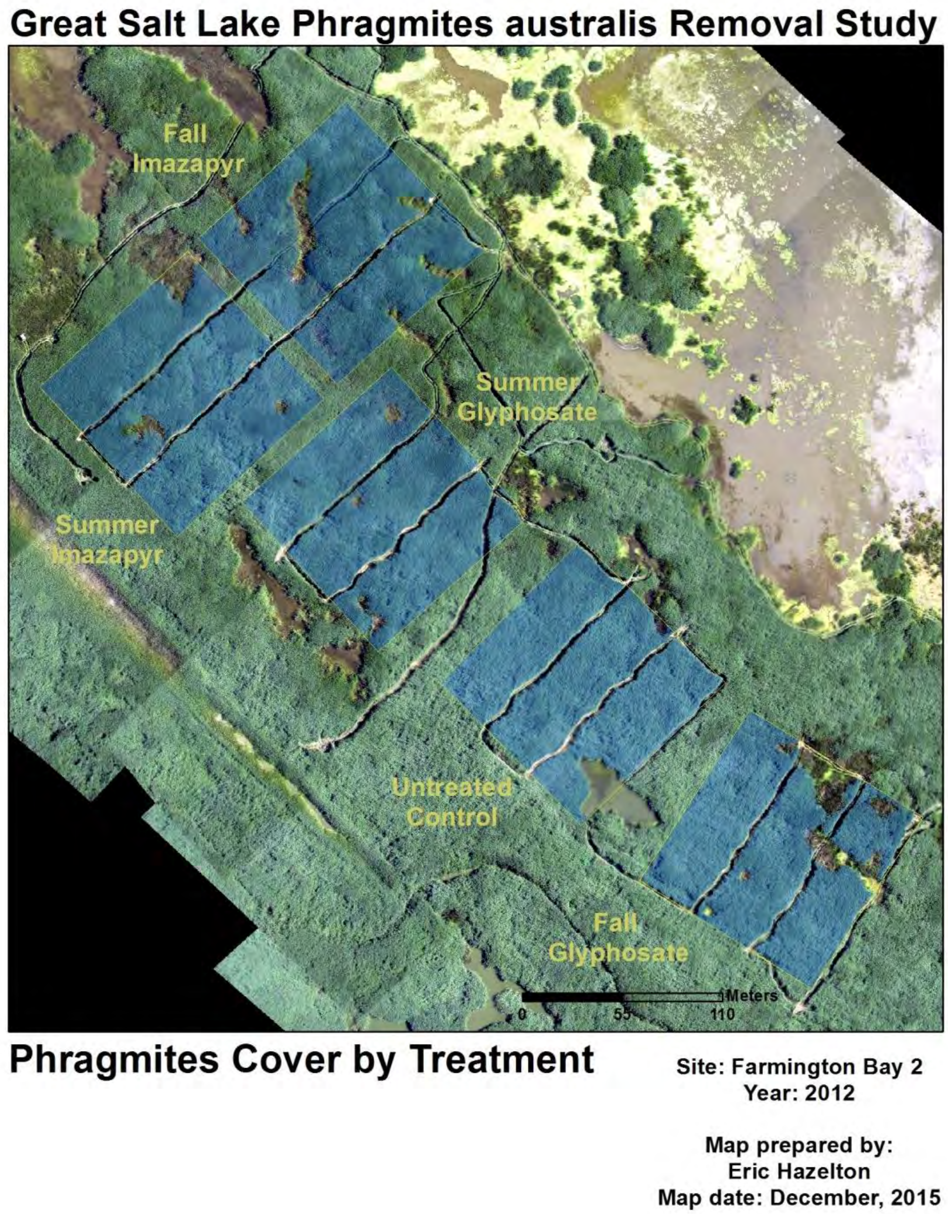


Figure 13. Digitized map of *Phragmites* cover (in blue) at Farmington Bay 2, 2012.

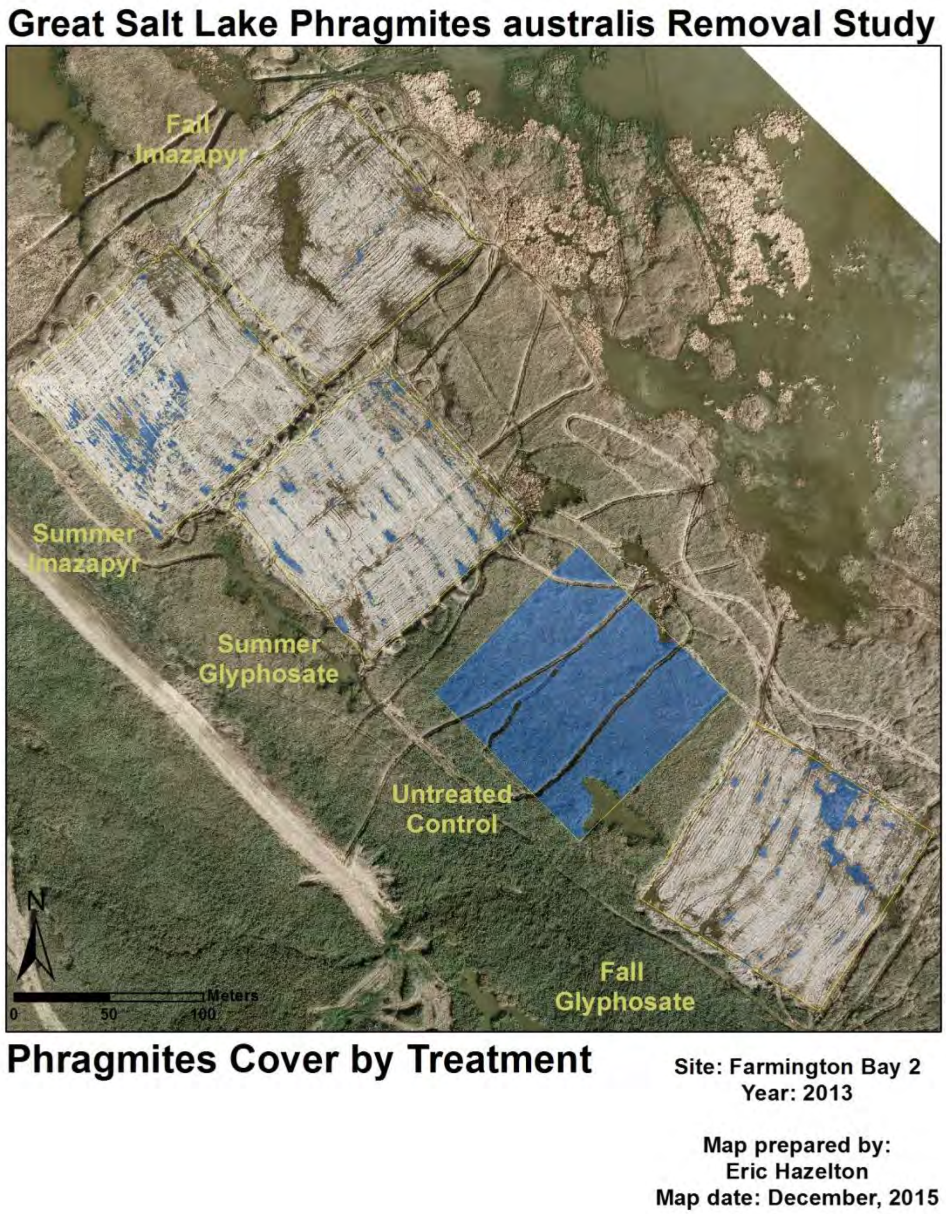


Figure 14. Digitized map of *Phragmites* cover (in blue) at Farmington Bay 2, 2013.

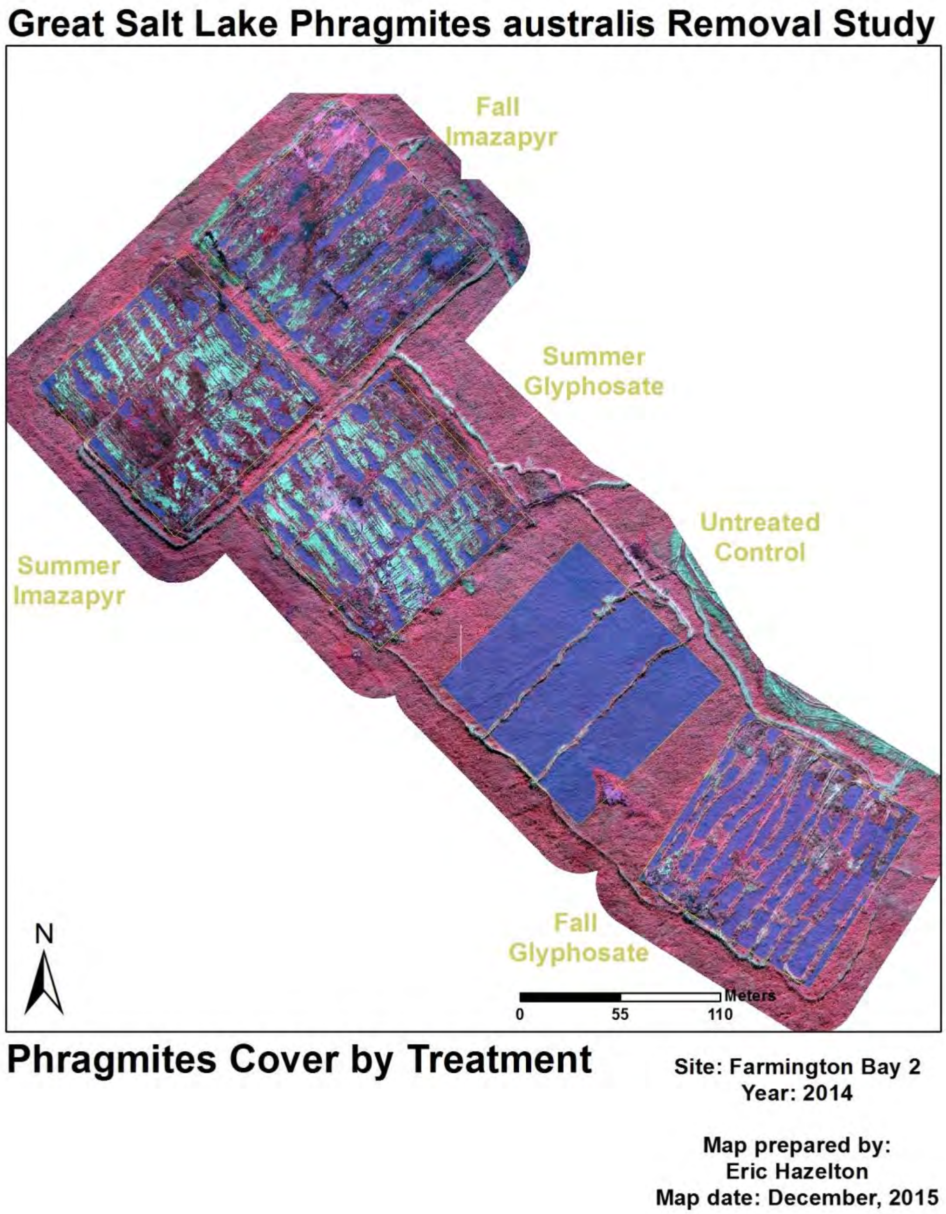


Figure 15. Digitized map of *Phragmites* cover (in blue) at Farmington Bay 2, 2014.

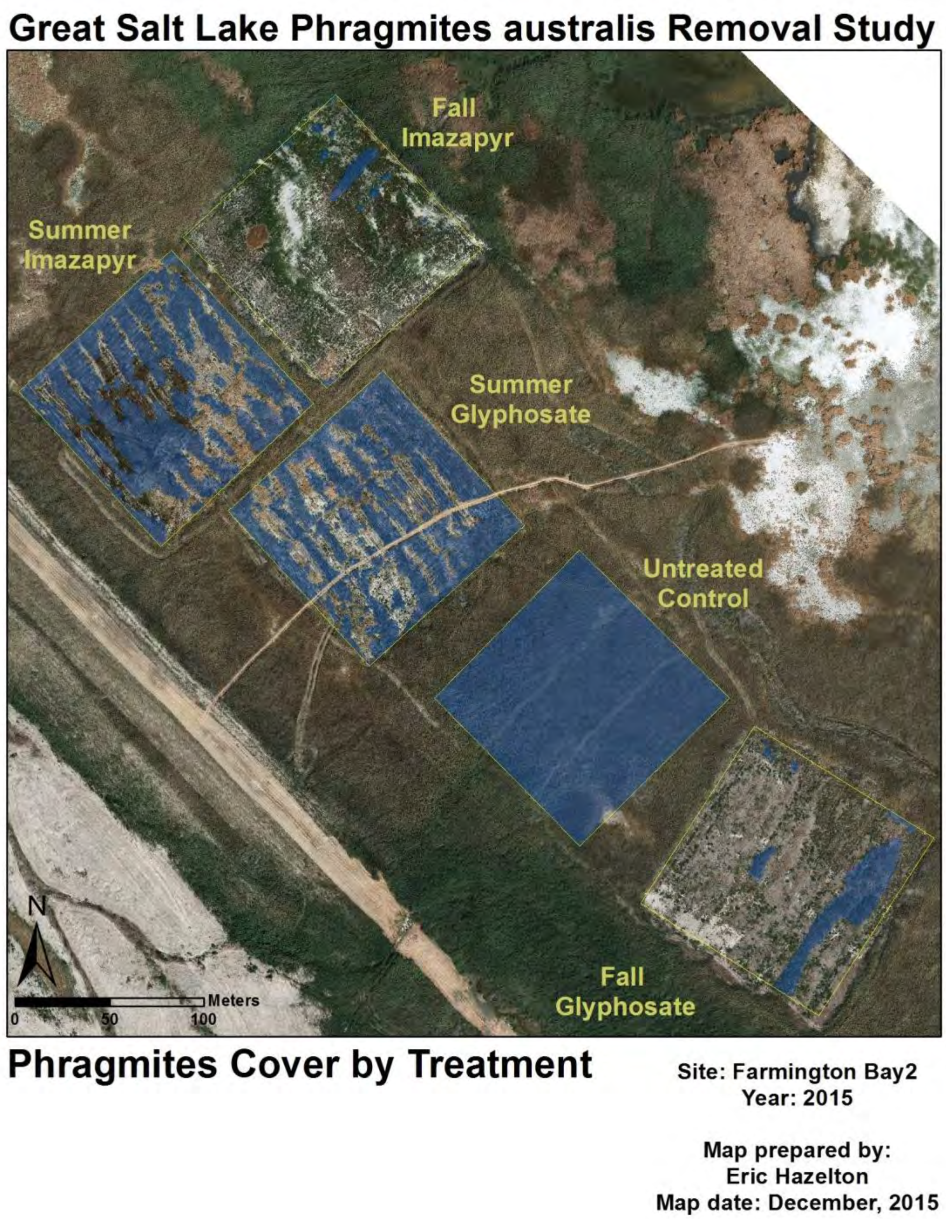


Figure 16. Digitized map of *Phragmites* cover (in blue) at Farmington Bay 2, 2015.

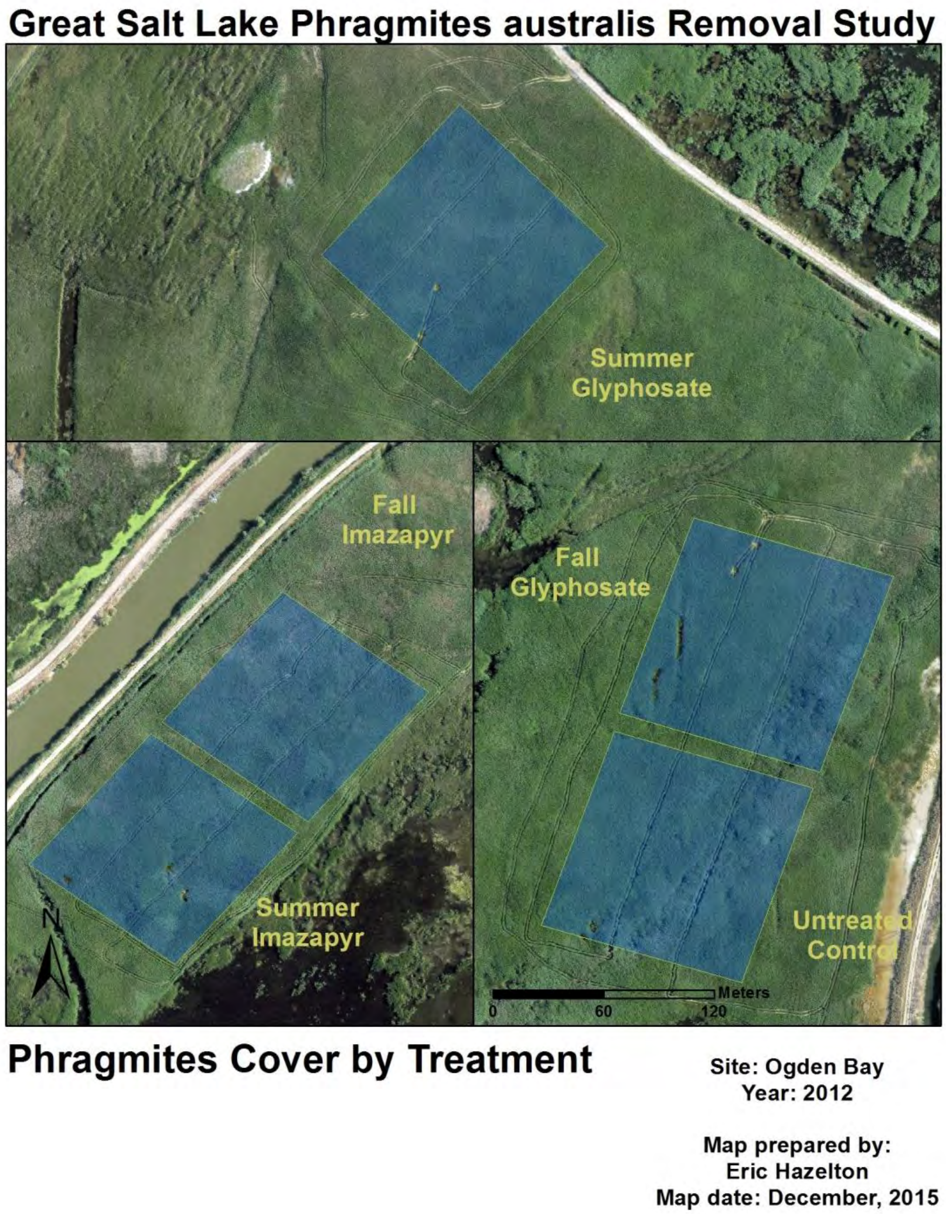


Figure 17. Digitized map of *Phragmites* cover (in blue) at Ogden Bay, 2012.

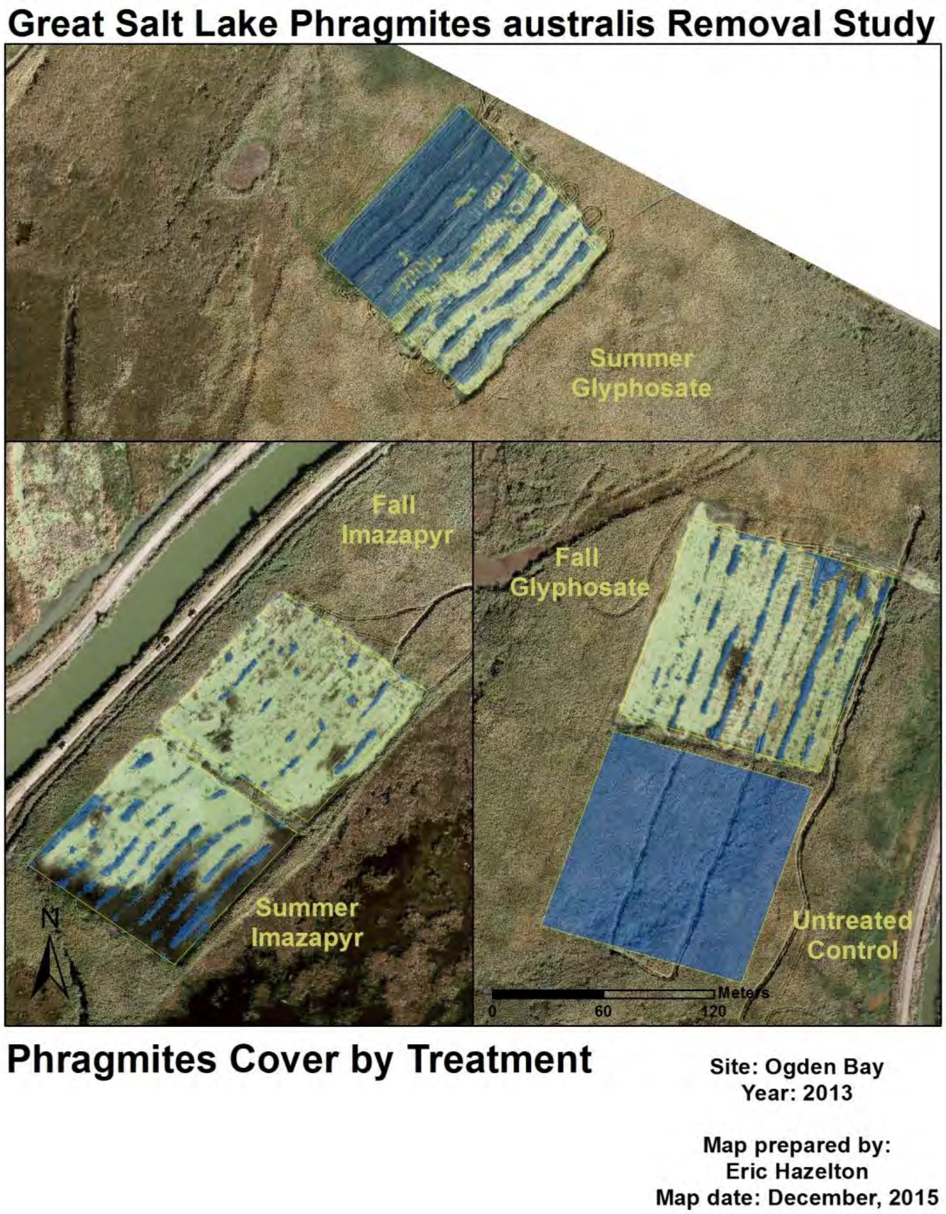


Figure 18. Digitized map of *Phragmites* cover (in blue) at Ogden Bay, 2013.

Figure 19. Digitized map of *Phragmites* cover (in blue) at Ogden Bay, 2014.

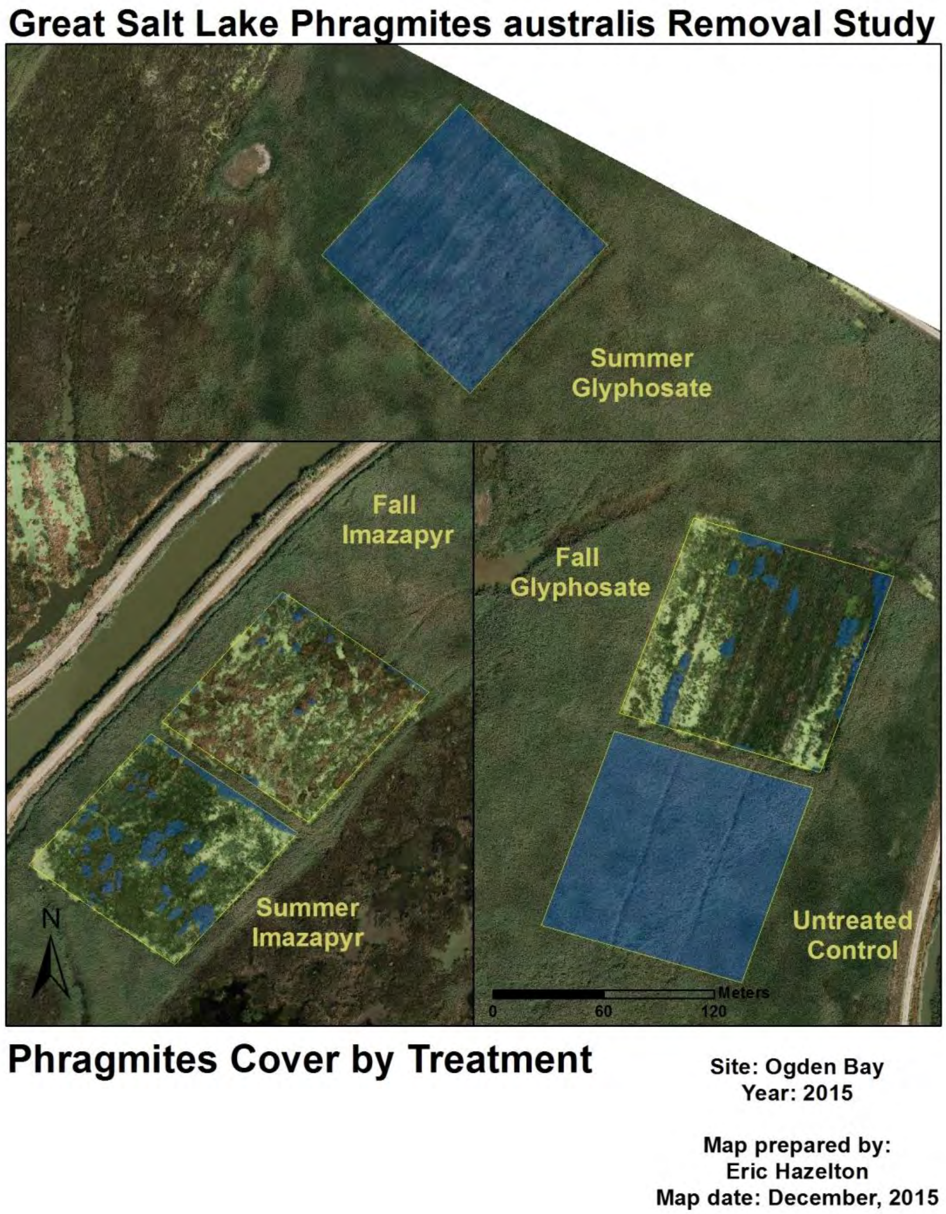


Figure 20. Digitized map of *Phragmites* cover (in blue) at Ogden Bay, 2015.

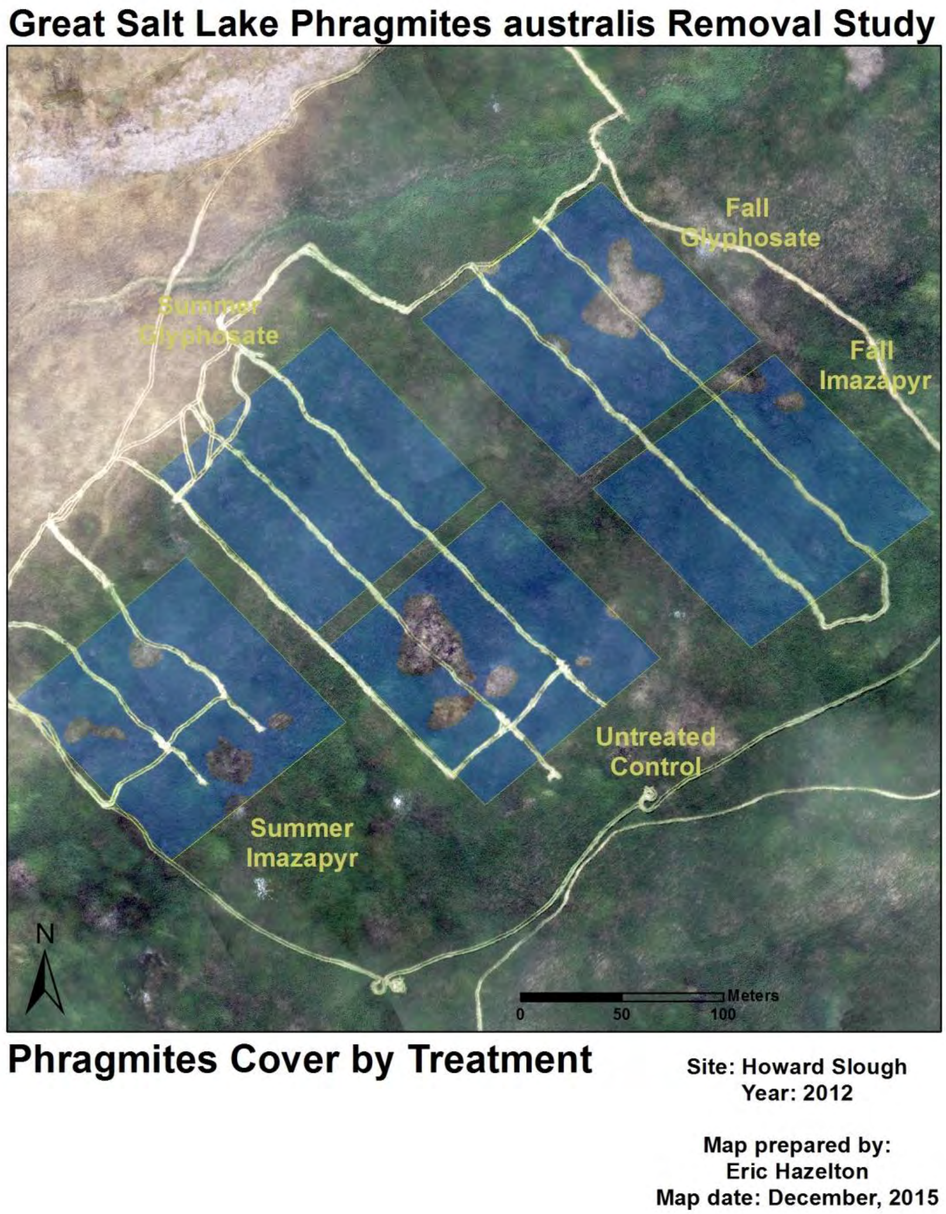


Figure 21. Digitized map of *Phragmites* cover (in blue) at Howard Slough, 2012.

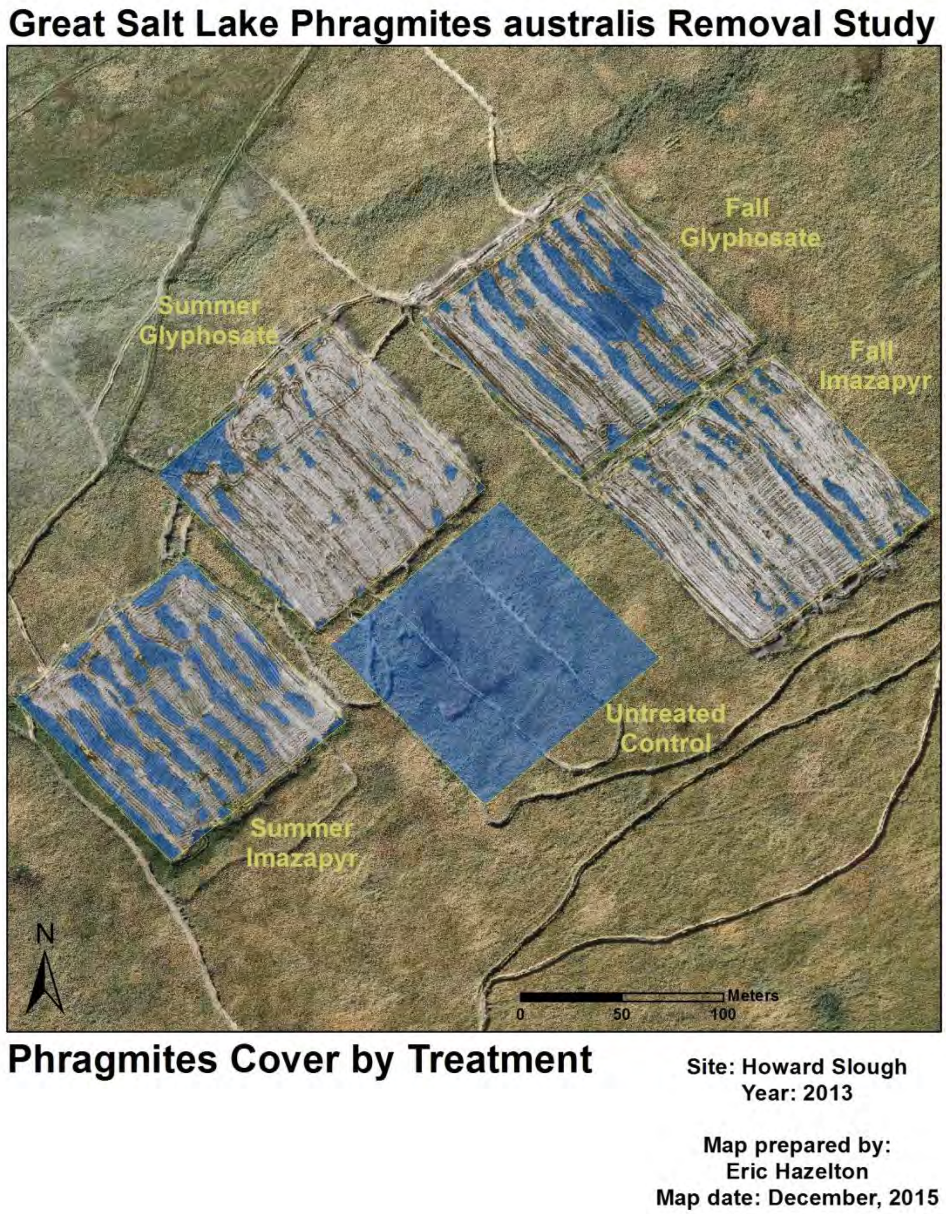


Figure 22. Digitized map of *Phragmites* cover (in blue) at Howard Slough, 2013.

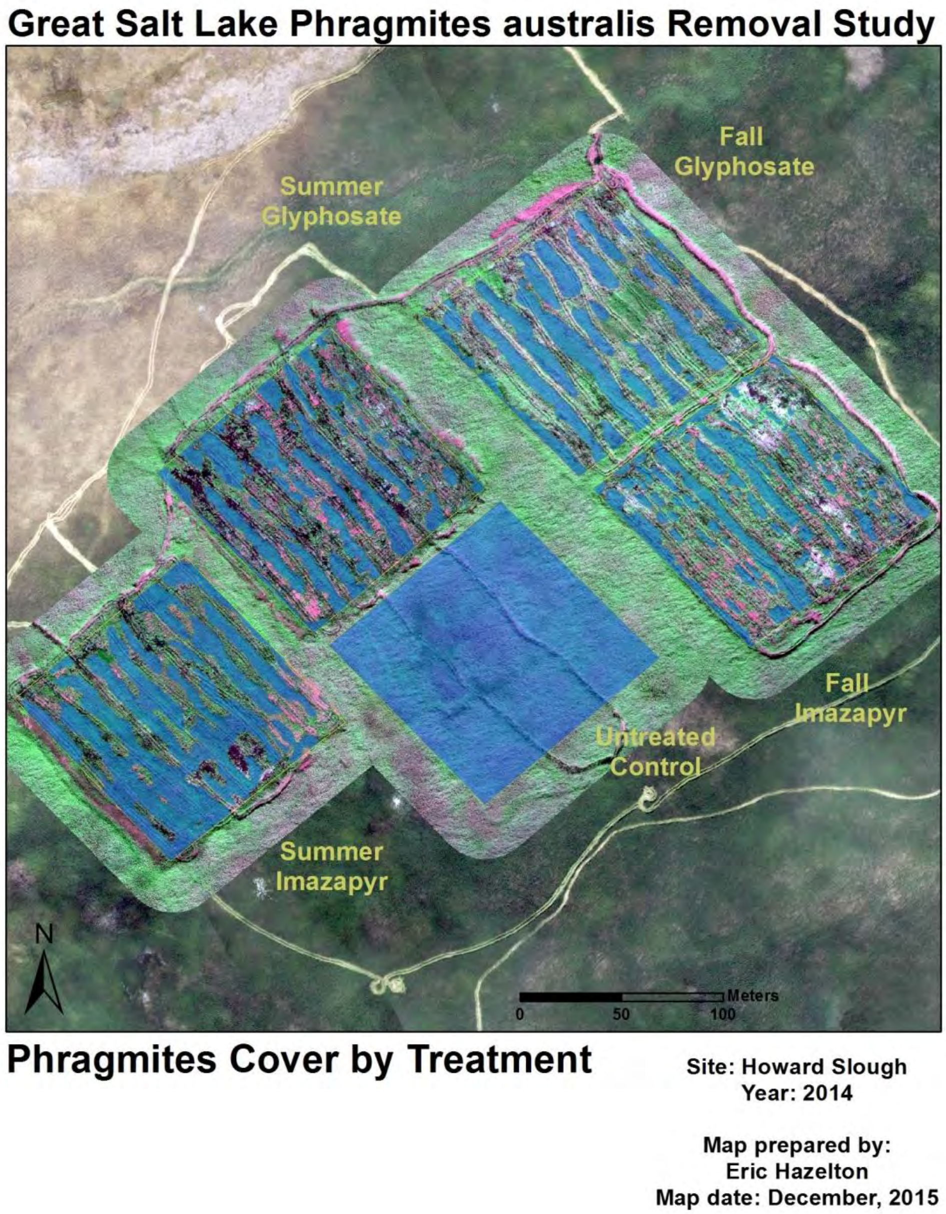


Figure 23. Digitized map of *Phragmites* cover (in blue) at Howard Slough, 2014.

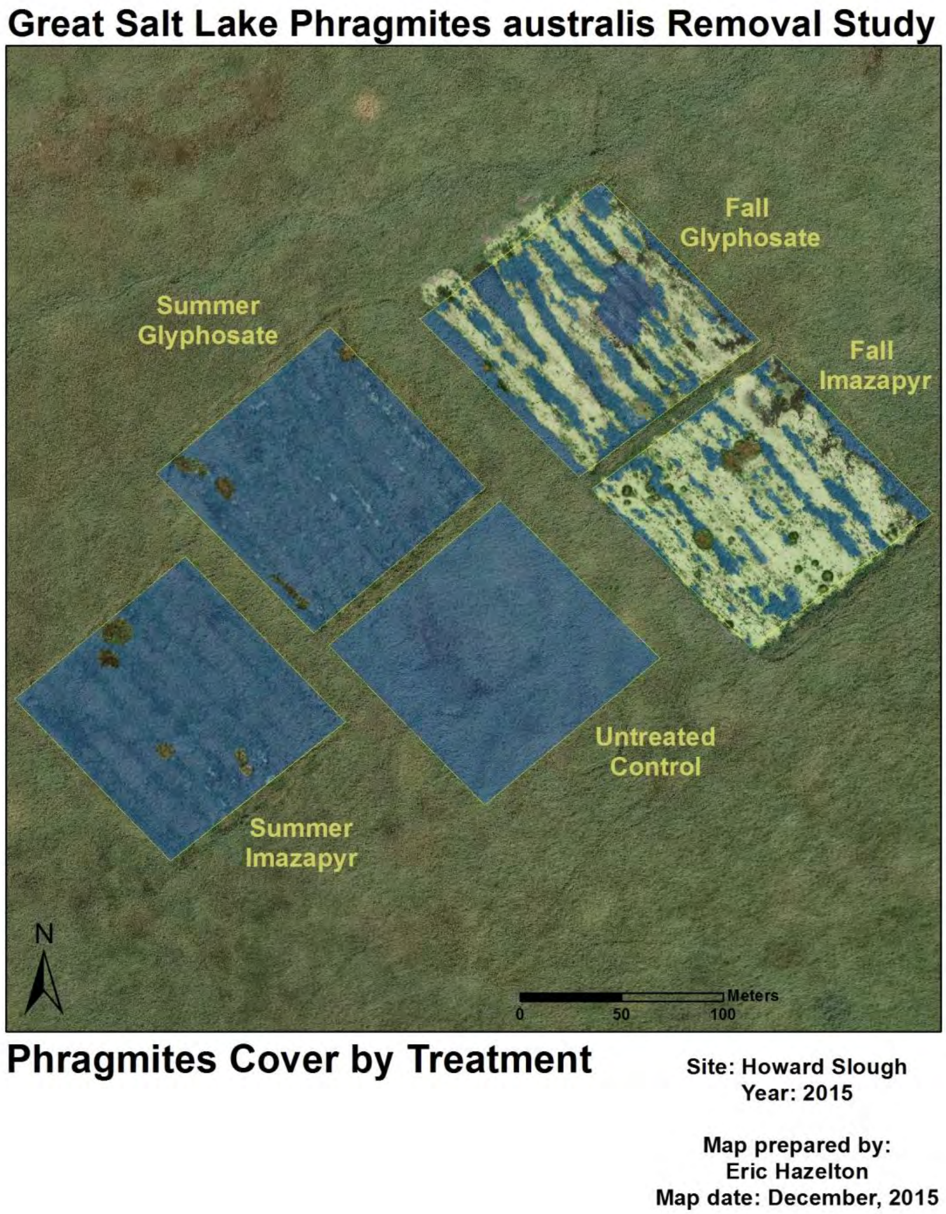


Figure 24. Digitized map of *Phragmites* cover (in blue) at Howard Slough, 2015.

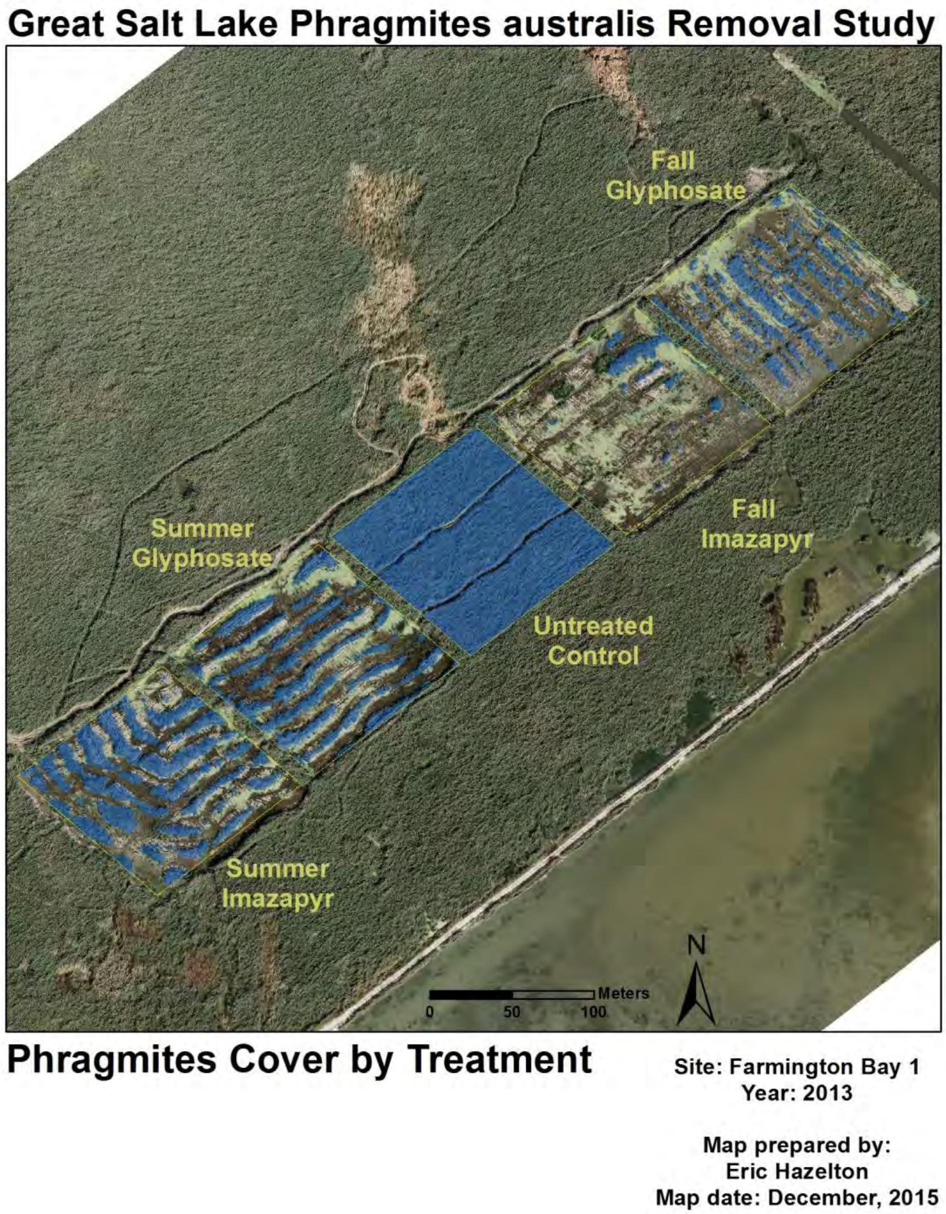


Figure 25. Digitized map of *Phragmites* cover (in blue) at Farmington Bay 1, 2013.

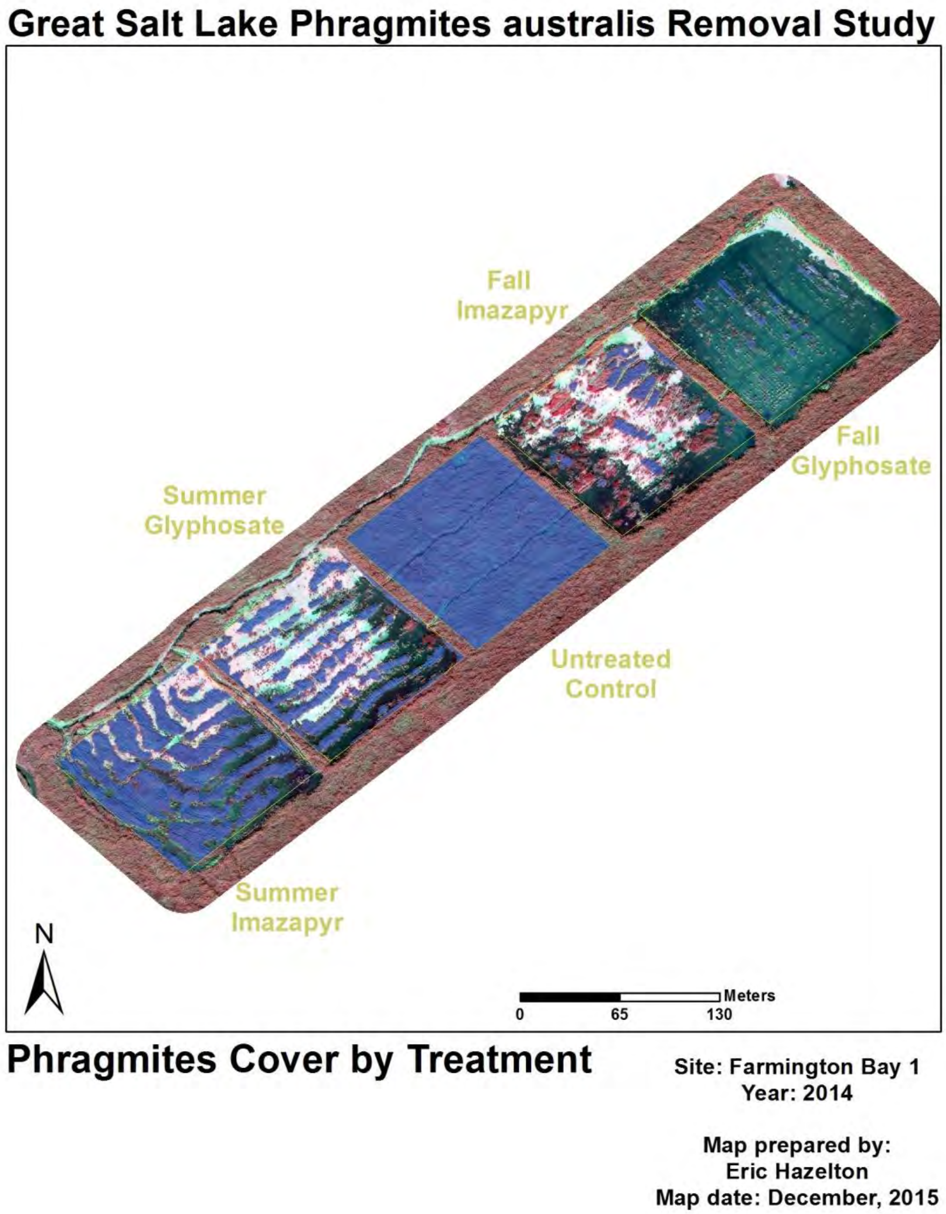


Figure 26. Digitized map of *Phragmites* cover (in blue) at Farmington Bay 1, 2014.

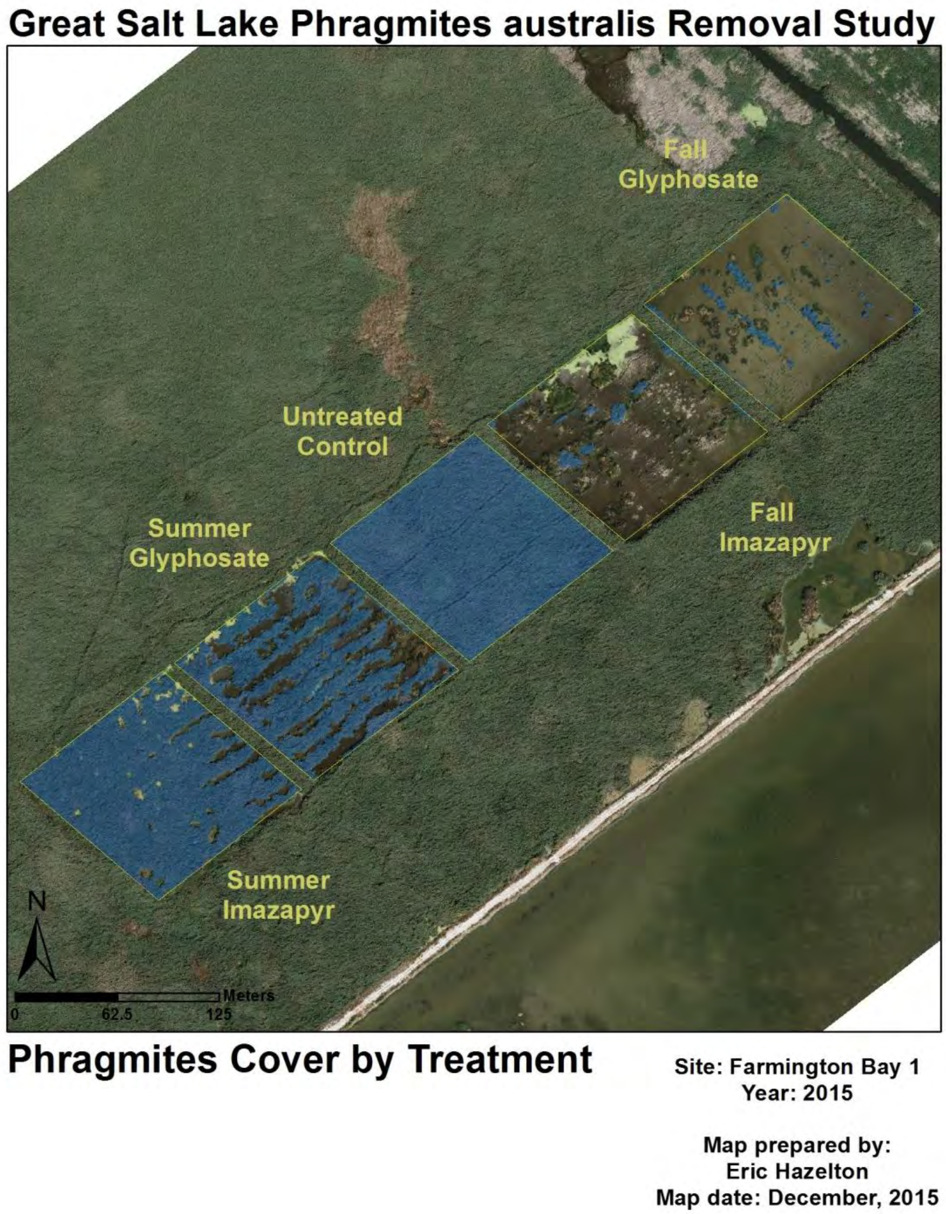


Figure 27. Digitized map of *Phragmites* cover (in blue) at Farmington Bay 1, 2015.