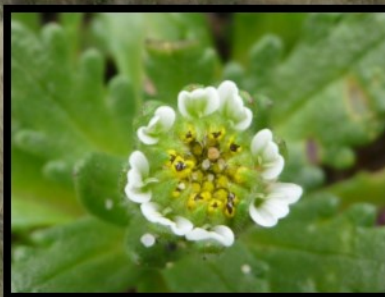




Coastal Dune Restoration Vegetation Monitoring Report



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SUMMARY

Native dune habitat in Point Reyes National Seashore (Seashore) provides critical habitat for four federally listed and several additional rare and unique species of plants and animals and represents some of the largest expanses of rare native dune plant communities remaining in California. However, by the late 1990s, more than 60% (1,400 acres) of the park's roughly 2,200 acres of coastal dune, bluff, and scrub habitat has been invaded by European beachgrass (*Ammophila arenaria*) and iceplant (*Carpobrotus* spp.), which establish vast monocultures that crowd out native plant species, as well as threaten listed animal species that also depend on dune habitat such as Western snowy plover (*Charadrius alexandrinus nivosus*) and Myrtle's silverspot butterfly (*Speyeria zerene myrtleae*). These impacts have spurred the Seashore to embark on an ambitious coastal dune restoration program since 2001. Three of the projects recently conducted include Abbotts Lagoon Coastal Dune Restoration Project (initiated in 2011), B Ranch North Restoration Project (initiated in 2013), and AT&T Coastal Dune Restoration Project (initiated in 2015). These projects relied on several different methods to remove the primary invasives, European beachgrass and iceplant, including manual removal, mechanical removal, and herbicide treatment. Since 2001, the Seashore has removed approximately 269 acres of non-native invasive plants from approximately 524 acres of coastal dunes.

Vegetation monitoring conducted pre- and post- restoration of these projects has discerned some commonalities among these different types of dune restoration approaches. All of the methods – manual, mechanical, and herbicide – result in a decrease in primary invader species, whether it be European beachgrass or iceplant. There are some exceptions to this. One of the earliest restoration projects near the mouth of Abbotts Lagoon attempted to eradicate European beachgrass through manual removal, but the extremely deep rhizome system (3- 12 feet deep in foredunes) and the need for frequent re-treatment rendered this approach ineffective at the Seashore, although it has been very successful in Humboldt (Pickart and Sawyer 1998). Subsequent manual removal projects have focused more on areas where invasion intensity is low (<30% cover). The other two restoration approaches – mechanical removal and herbicide treatment -- resulted in almost immediate reductions in primary invasives cover, with regrowth or new establishment less than 3- 5% even by just the second year. However, they created startlingly different ecological environments post-restoration. Herbicide treatment converted dense monocultural stands of live beachgrass into dense stands of dead biomass that take more than 3 to 5 years to even begin to decompose, with decomposition generating large amounts of thatch and detritus that eliminated most bareground or open space. In contrast, mechanical removal instantaneously converted dense stands of beachgrass into vast expanses of unvegetated sand that, under conditions of drought, ended up being largely nothing but bareground or open space for 5 years or more.

Both of these disturbed environments proved vulnerable to invasion by a whole new suite of invasives, so-called "secondary invaders." Secondary invaders are classified as species that are not the primary removal target, but that take advantage of decreases in cover of primary target species and increases in available space to gain a foothold and potentially oust native species (Pearson et al. 2016). In certain instances, secondary invaders can pose more of a risk than even

the primary invaders to restoration projects focusing on invasives removal (Pearson et al. 2016). At the Seashore, the suite of secondary invaders appears to vary depending on restoration approach and/or location within the coastal dune ecosystem. Mechanical removal areas (foredune and backdune) and herbicide-treated foredunes support a mix of European searocket (*Cakile maritima*), along with New Zealand spinach (*Tetragonia tetragonoides*), European black nightshade (*Solanum nigrum*), and iceplant (*Carpobrotus* spp.). At Abbotts, European searocket was a relatively infrequent occurrence prior to and immediately after mechanical removal in 2011, but by summer 2016, European searocket numbers had exploded, with moderate to dense cover of this species in all of the mechanically restored areas, as well as in adjacent native dunes affected by inland movement of destabilized sands following restoration. In certain areas, searocket literally blanketed colonies of the federally endangered Tidestrom's lupine (*Lupinus tidestromii*) in mechanically restored areas. A different suite of secondary species have impacted herbicide-treated backdune areas, including riggut brome (*Bromus diandrus*), fescue brome (*Festuca bromoides*), common groundsel (*Senecio sylvestris*), and sow thistle (*Sonchus* spp.). These species have also encroached on dune restoration areas other than Abbotts, including at B Ranch North and AT&T, but the Seashore has been proactively trying to counter these threats through both manual removal and herbicide treatment.

Ultimately, what will define these dune restoration projects as being successful is not whether the park can eliminate both primary and now secondary invaders, but whether it can reestablish native dune ecosystem conditions, processes, and functions that support special status plants and wildlife. Initially, the answer to this question appear to be mixed. Tidestrom's lupine and western snowy plover moved in on the Abbotts Lagoon project area even before excavators and bulldozers had been removed. However, in terms of promoting a native dune vegetation community, the early monitoring results painted a pretty dismal picture. Both mechanical removal and herbicide treatment caused cover and richness of native dune plant species to plummet dramatically. In the case of mechanical removal, the re-setting of the successional timeline was almost too extreme, with soil conditions and dune processes either too primordial or too severe to support establishment of plants other than pioneer or nursery species such as Tidestrom's lupine, while in the case of herbicide treatment, restoration does not appear to have been extreme enough to provoke a successional reversion to native dune – except where sand overwash occurs. Re-establishment of native vegetation was also confounded by longer-term climatic conditions, with drought conditions persisting between 2012 and fall 2016, when many of these projects were initiated.

Post-treatment corrective measures explored to date do not appear to be the “magic bullet” for reversing these issues. Even if lack of native species propagules appeared to be a problem -- which, in this case, it does not -- active revegetation of at least mechanical removal areas relies on protection of plantings or seeding from high winds and appropriate nutrient/soil biota conditions for success. In terms of accelerating decomposition of dead beachgrass stands in herbicide-treatment areas, mowing and scraping did increase beachgrass decomposition rates and bareground cover, but did not increase native species cover and, in the case of dozing, actually decreased native species richness (Ender and Parsons 2017). Measures such as

mowing, scraping, or prescribed burning may introduce an element of disturbance that could actually disrupt, rather than accelerate, conversion to more native habitats.

Based on recent vegetation results, there are signs, however, that native vegetation establishment is beginning to occur in the older mechanical removal and herbicide treatment projects dating back to between 2011 and 2014. Cover of native species is increasing, and so is total -- and in some instances, native -- species richness. Recruitment may have been aided by changes in climatic patterns, with the previous drought ending in 2016-2017, which had roughly twice the amount of normal rainfall. Whether these restored areas are on the right path towards evolving into native Dune Mat or Dune Scrub or reversing some of the premature successional aging of remnant native dune habitat remains in question. Multivariate analyses indicate that species composition is not comparable yet with reference systems, whether Dune Mat or Dune Scrub. But results point to at least some progress in terms of achieving vegetation-related restoration objectives should restoration efforts not be derailed by re-invasion of non-native species. Vigilant re-treatment of primary and secondary invaders will be key, ultimately, to success of these projects, and as Pickart and others have pointed out, re-treatment may not be a short-term obligation, but a long-term commitment, as new invaders brought by water, wind, or human will continue to try and establish in this naturally disturbance-prone environment.

Even if invasives are successfully removed and excluded, this may not be enough to fully restore dune ecosystems due to “legacy” effects of these invasives on both abiotic and biotic conditions of soils, dune structure, and other biological mutualisms of dune ecosystems still not fully understood (Suding et al. 2004, Suding and Hobbs 2009, Yelenik and Levine 2010, Yelenik and D’Antonio 2013). A number of restoration projects have shown that removal of areas long invaded by iceplant does not necessarily result in immediate establishment by natives, which researchers have surmised is due to any number of factors, including allelopathic effects of iceplant litter; changes in soil pH, organic matter, nutrient levels, and water-holding capacity; and adverse changes in soil biota (Andreu et al. 2010, Novoa et al. 2012, Novoa et al. 2013, Winsemius 2013). In many cases, these “restored” areas end up being colonized primarily by weedy, opportunistic non-native species (Andreu et al. 2010, Hamingson and Voeller 2011, Novoa et al. 2013). Legacy effects of European beachgrass are less well-understood, but it definitely can change foredune structure (Pickart and Wiedemann 1997) and slow, if not possibly preclude, efforts to re-establish native vegetation due to its slow decomposition-related dynamics. We do not have enough information yet to predict how persistent these decomposition-legacy effects will be, but bush lupine (*Lupinus arboreus*), which is believed to be non-native in Marin County, has well-documented, long-term legacy effects on soils from decomposition of nitrogen-enriched litter that promotes colonization of weedy annuals such as we have observed in herbicide-treated backdune areas (Zink et al. 1995, Maron and Connors 1996). Restoration project managers may need to find effective ways to tackle these legacy effects if restoration efforts are ultimately to be successful.

Despite these habitat-creation challenges, it’s difficult to characterize these restoration efforts as unsuccessful given the overwhelmingly positive response from threatened and endangered plant species. The sheer numbers of Tidestrom’s lupine plants within the mechanically restored area at Abbots is in and of itself a tremendous success story, giving recovery efforts of this

endangered species a tremendous boost. While herbicide treatment areas offer less opportunity for plant establishment due to slow decomposition of beachgrass, movement of sand into both foredunes and backdunes has greatly increased the number of Tidestrom's lupine, beach layia (*Layia carnosa*; FE), and curlyleaf monardella (*Monardella sinuata* ssp. *nigrescens*; CNPS List 1B.1) within treated areas, with beach layia numbers climbing dramatically in recent years. Along with this has also come increased use of the Abbotts Lagoon restored dunes by Western snowy plovers and possibly Myrtle's silverspot butterfly, although plover use has dropped off in recent years due to the prevalence of ravens within the restored area. As with re-establishment of native vegetation community, simply removing the non-native invasive species that encroached on nesting areas and provided cover for predators may not be enough to create fully successful nesting and rearing habitat for plovers. Additional adaptive restoration measures may be needed, such as nest enclosures (already ongoing) and predator control.

In addition to providing new habitat for establishment of Tidestrom's lupine and other rare plant species, restoration at Abbotts also benefitted the existing Tidestrom's lupine population in the adjacent native Dune Mat habitat by decreasing predation of seed by native deer mice. Predation rates dropped from between 44.8% and 93.9% prior to restoration to between 3.7% and 9.4% immediately after restoration (Pardini and Knight 2013; Pardini et al. 2018). The same benefits, however, have not been realized from herbicide treatment of European beachgrass stands at B Ranch North that encircle a native Dune Mat habitat area that supports a critically imperiled Tidestrom's lupine population. Prior to restoration, predation rates averaged 70%; since restoration, predation rates have dropped minimally, averaging 58%, although they did decrease in 2018 to 33.2% (WU, unpub. data). Differences in benefits to existing lupine populations may come down to the fact that mechanical restoration not only immediately eradicated beachgrass, but effectively deer mice, as well. With herbicide treatment, changes in vegetation are more gradual due to slow decomposition of beachgrass and rhizomes: mice may nest even in dead beachgrass stands due to the persistence of dead rhizomes. Sufficiently large enough declines in rodent numbers to benefit reproductive success of Tidestrom's lupine populations threatened by excessive seed predation may not transpire unless there are drastic alterations in the landscape (e.g., mechanical removal) or some type of predator management.

For future coastal dune restoration projects, the Seashore may want to prioritize restoration of foredune and mid-dune areas using herbicide treatment, as sand overwash accelerates conversion of invaded areas to native dune habitat while the remnant European beachgrass and associated rhizomes limit large-scale remobilization of sands by high winds along the coast. While mechanical removal can also eliminate non-natives and reverse physical legacy effects of European beachgrass in foredune areas, the cost in terms of impacts to adjacent native habitats and ranchlands from sand remobilization is too high, and the approach itself too costly to be feasible for larger future restoration efforts. Success of creating native dune habitats using herbicide in heavily vegetated backdunes is more questionable however, jeopardized potentially by decomposition-related physical and soil legacy effects of beachgrass, as well as minimal deposition of new sand, which may favor establishment of weedy, non-native annual species over natives. More data is needed to accurately predict the evolutionary trajectory of these habitats. Clearly, though, both external and internal iceplant removal projects have

clearly documented that, regardless of method, removal of iceplant may not be enough, particularly in highly invaded areas due to legacy effects on soil chemistry. Removal of duff may improve “restorability” of these areas, but, ultimately, until measures can be found to counter these legacy effects, the park may want to direct its limited restoration resources to lightly to perhaps moderately invaded iceplant areas or areas that have only been invaded for a short while.

INTRODUCTION

Native dune habitat in Point Reyes National Seashore (Seashore) provides critical habitat for four federally listed and several additional rare and unique species of plants and animals and represents some of the largest expanses of rare native dune plant communities remaining in California. However, more than 60% (1,400 acres) of the park's roughly 2,200 acres of coastal dune, bluff, and scrub habitat has been invaded by European beachgrass (*Ammophila arenaria*) and iceplant (*Carpobrotus* spp.), which establish vast monocultures that crowd out native plant species (Figure 1).

The continued encroachment of these species not only resulted ultimately in dense monocultures that supported fewer native dune plant species, but it had potential repercussions on other species as well, including the federally endangered Myrtle's silverspot butterfly (*Speyeria zerene myrtleae*) that feeds on the nectar on many dune plants and the federally threatened Western snowy plover (*Charadrius alexandrinus nivosus*) that nests in unvegetated shorelines along the foredunes. Beachgrass can creep out into foreshore areas, and it also provides cover for potential predators of eggs, chicks, and adults. In addition, beachgrass indirectly impacted endangered dune plants such as Tidestrom's lupine (*Lupinus tidestromii*) by supporting much higher densities of native deer mice (*Peromyscus maniculatus*) that have been shown to eat as much as 82% of this species' seed, thereby threatening long-term viability of many of the Seashore's lupine populations (Dangremond et al. 2010).

These impacts have spurred the Seashore to embark on an ambitious coastal dune restoration program since 2000. The primary objectives related to dune restoration at the Seashore include:

- Remove non-native, invasive plant species from dune habitat where they interfere with natural physical processes such as sand movement and hydrology.
- Remove non-native, invasive plant species from dunes to create conditions under which native plant and wildlife species can flourish.
- Minimize potential for non-native species reinvasion of restored habitat.
- Increase potential coastal dune habitat for target threatened and endangered species affected by non-native, invasive plant species. The U.S. Fish and Wildlife Service (USFWS) Recovery Plan for endangered dune plant species calls for expanding population numbers and ranges through restoration and protection of dune habitats, with restoration to be focused on an "aggressive program to control non-native invasive plant species."
- Work with park ranchers to minimize impacts to beef and dairy cattle ranching operations on adjacent lands.

Monitoring of the effectiveness of restoration efforts was not necessarily identified as an objective, but several of the projects have attempted to monitor how successful projects have been in achieving at least some of these objectives. This report summarizes preliminary restoration results for vegetation-associated variables for three projects conducted by the Seashore as of 2018 – Abbotts Lagoon, which was conducted starting in 2011; B Ranch North, which was initiated in 2013, and AT&T, which was initiated in 2015.

Coastal Dune Restoration Environmental Assessment
Point Reyes National Seashore



National Park Service
Point Reyes National Seashore
Marin County, California

0 1 2 Miles



Figure 1. Beachgrass and Iceplant

European Beachgrass
Iceplant

The intent of the AT&T project is to link with the Abbotts Lagoon project to the north to create almost 400 acres of unfragmented, restored coastal dune habitat.

BACKGROUND ON COASTAL DUNE ECOSYSTEMS

Dune Processes

Coastal dunes are dynamic ecosystems formed by the combined action of wind and wave. Offshore sandbars and sediment deposited at the mouths of rivers are the most important sources of material for dune building. This sediment is carried by longshore currents down the coast and is deposited on the beach by wave action (CERES 2013). Dune formation begins when wind or so-called Aeolian transport blows dry sand particles landward from the beach (CERES 2013). Objects such as plants and logs can act as “drifts” that disrupt the wind flow and sediment transport, causing mounding of sand. Over time, the wind erodes sand particles from the windward side of these drifts or mounds and deposits them on the leeward side. Gradually, this process results in the dunes migrating inland, with more sand accumulating in these moving hills over time.

In some cases, dunes are formed as spits or long, narrow accumulations of sand. Spits establish from in-coming waves approaching the shoreline at an angle, generating a longshore current: this current transports sand that deposits and forms a bar in front of a bay or lagoon (Old Dominion University 2013). Limantour Spit is an example of a spit type of dune: waves refract around Point Reyes Headland, which leads to an angled wave approach along this section of coastline (Old Dominion University 2013).

As with many ecosystems, dunes often undergo periods of cyclic stabilization and rejuvenation (Pickart and Sawyer 1998). Rejuvenation events are the result of changes in relative sea level, which, in turn, have been attributed, at least in the past, to tectonic activity, including tsunamis (Vick 1988, Pacific Watershed Associates 1991, Clarke and Carver 1992, Komar and Shih 1993 *in* Pickart and Sawyer 1998). Both uplift and subsidence can trigger reactivation of dunes, with the former potentially building or expanding dunes through increased sediment supply, while the latter can destroy dunes through increased wave action or limit the expansion of new dunes (Pickart and Sawyer 1998).

During and in between these major dune-forming and dune-destruction periods, dune morphology continues to be shaped by other factors, including wave action and offshore winds. Offshore winds create “blowouts” or northwest-southeast trending swales or low areas that are parallel with the prevailing northwesterly winds. When dunes are sparsely vegetated, the strength of these winds is sufficient to mobilize sands and cause movement or creep of dunes inland over time. New foredunes and associated parabolic dunes are then created from new sediments supplied by the ocean.

In addition to tectonic activity, another factor that can stabilize or rejuvenate dune systems is climate change. Climate change can affect dune systems through a myriad of direct and indirect effects, including changes in temperature, wind, precipitation, freshwater hydrology, sediment supply and transport, sea level rise, and ocean circulation. Coastal environments are considered among the most vulnerable to changes from climate change, including direct changes (e.g.

changes of temperature and precipitation) and indirect changes (e.g. sea level rise, wind and water circulation, increasing storm events). As climate change study is a relatively young science, the exact magnitude and extent– and even the direction – of these changes on the northern California coast is still a matter of active debate.

Dune Structure

Coastal dune fields form characteristic patterns. Most dune fields consist of two or three sets of parallel dunes, with the most recently formed foredunes nearest the beach, and the older dunes farthest inland; the inland dunes may be as much as 18,000 years old (CERES 2013). A common pattern in some California dune systems is a series of parallel ridges perpendicular to the prevailing winds, called "transverse ridges" (CERES 2013, Pickart and Sawyer 1998). These dunes occurred in Pismo and Santa Maria further south, as well as historically in San Francisco and Dillon Beach (P. Baye, *pers. comm.*).

The other common type of dune system in California is the parabolic dune, which consists of a series of U-shaped dunes with the concave side facing the prevailing wind direction (CERES 2013, Pickart and Sawyer 1998). Most of the California parabolic dunes are attenuated, narrow parabolic or U-shaped dune fields, also called "tongue" dunes (Cooper 1958), with minimal or no dune slack or swale development except in a few cases (Point Reyes, Monterey, Humboldt, and Morro Bay sandspit; P. Baye, *pers. comm.*). At Point Reyes, most of the dune mass occurs in an attenuated lobe rather than on the flanking ridges of the "U," perhaps because these dunes are climbing up over a marine terrace with an erosion-resistant soil and vegetation community (P. Baye, *pers. comm.*).

In these systems, the primary foredune is a ridge of sand that forms parallel with the coast above the mean high tide line (Pickart and Sawyer 1998). It is buffeted by onshore winds and is vegetated by plants that are tolerant of sand burial (Pickart and Sawyer 1998). Under historical conditions (e.g., prior to invasion by non-native species), the foredune typically was a low-elevation, hummocky, non-continuous feature sparsely vegetated by species such as native American dunegrass (*Elymus mollis*), which allowed periodic blowouts of sand that leave valleys or openings within the foredune ridge (Pickart and Sawyer 1998). The wind pushes sand inland through these blowouts or valleys, forming crests and valleys or U-shaped dunes perpendicular to the coast.

The longitudinal ridges and valleys behind the foredune are also sparsely vegetated and move slowly inland by the forces of wind and slumping along the lee face as slopes exceed the angle of repose. In northern California, both parabolic and transverse dunes exist behind the foredune complex (Pickart and Sawyer 1998). Parabolic or U-shaped dunes move inland, sometimes merging into a large sand plain or sand sheet devoid of vegetation (Pickart and Sawyer 1998). As this highly mobile, unstabilized sheet moves, a large deflation plain or low elevation areas forms on which dune swale or dune slack wetlands can occur, fostered by a seasonal or perennial groundwater table exposed by sand loss (Pickart and Sawyer 1998). Wetland vegetation grows and stabilizes the dune swales. In some California dune systems, smaller transverse dunes or dunes oriented with crests perpendicular to the prevailing wind can

develop on the open sand surfaces of so-called moving dunes, which include parabolic dunes and sand sheets (Pickart and Sawyer 1998).

Both European beachgrass and iceplant have greatly altered the natural morphology of coastal dunes in many areas along the Pacific coast of North America, although European beachgrass more so. With colonization by European beachgrass, the primary foredune topography has changed to a steeply sloped, continuous ridge without blowouts, and the orientation of the ridges and troughs in areas behind the foredune complex are parallel rather than perpendicular to the coast (Cooper 1936; Barbour and Johnson 1988; Pickart and Sawyer 1998). Based on topographic analysis of dunes at Bodega, areas vegetated by European beachgrass can be 7- to 26 feet higher than unvegetated dunes (Cesmat et al. 2012). Rapidly accreting European beachgrass-dominated foredunes have an internal structure of interbedded unconsolidated sand (mostly horizontal or gently sloping beds) formed by pulses of rapid sand accretion (burial episodes) and shallow layers of persistent, dense beachgrass roots, rhizomes, and shoots (Baye 2008). European beachgrass roots and rhizomes concentrate in the upper 8- to 12 inches due to light exposure, but with frequent burial events, young, viable buds formed from rhizome breakage can be distributed throughout the foredune horizon (Baye 2008). This network of rhizomes, combined with the dense aboveground biomass, acts to trap sand and stabilizes the once active foredune, which prevents large-scale sand movement or fresh supplies of sand to dune areas inland of the primary foredune.

A second series of European beachgrass-dominated dunes occurs in the landward portion of the dune system, typically called the backdunes. Backdunes generally refer to dunes cut off from sand supply from the beach, and they are either stabilized or are internally remobilizing sand from blowouts or other mobile dunes inland from the beach (P. Baye, *pers. comm.*). At Point Reyes, most of the backdunes are heads or flanks of formerly migrating U-dune lobes that stabilized decades ago (Baye 2008). They are in early stages of Dune Scrub succession, as indicated by high frequency of coyote brush, wild cucumber, and mock heather (Baye 2008). Leading edges of U-dunes often stabilize by lateral, superficial colonization of European beachgrass, as the rate of dune migration inland slows. Because of the lower frequency of sand burial events in backdunes, rhizomes of European beachgrass do not extend as deeply down in the dune horizon in the backdunes as they do in the foredunes (Baye 2008).

At AT&T, just south of Abbott's Lagoon, both European beachgrass-dominated foredunes and backdunes exist, although the distance between foredune and backdunes is more compressed than at Abbott's, where foredunes and backdunes are separated by a large basin or deflation plain. Within the backdunes at AT&T, the ridges run in a perpendicular alignment to the beach, allowing for formation of moist meadow-drainage complexes between backdune ridges. A somewhat similar dune morphology exists at North Beach, although the size of foredune and backdune features is muted relative particularly to the northern portions of AT&T.

The dune systems at B Ranch, A Ranch, and Davis Property are different from those on the northern portions of the Great Beach. At B Ranch, there is a moderately steep drop from the dune "terrace" to the beach, with the oceanward portion of the dunes comprised of low-elevation European beachgrass-dominated foredunes. Behind the foredunes, the dunes slope gently upward in a series of concave basins bordered by marginal ridges to the elevated

European beachgrass-dominated backdunes directly adjacent to the pasture. These basins at B Ranch support a more native dune-open sand vegetation community with native plants and patches of iceplant, with ridges either comprised of native dune-open sand or colonized by European beachgrass or iceplant. At the Davis Property, the dune “terrace” is actually perched on a cliff.

Sand Movement

Presumably one of the primary differences between natural or native dunes and dunes heavily colonized by European beachgrass is the rate of sand movement, as species such as beachgrass and iceplant were planted to stabilize dunes and prevent encroachment into adjacent lands. In Humboldt, research found that parabolic dunes migrated on average 4.7 feet/year between 1939 and 1988, while some other parabolic dunes and transverse dunes moved as much as 15 to 21 feet/year (Wiedemann 1984; Pacific Watershed Associates 1991). Sand movement rates would be naturally expected to be much lower in invaded dune, as these species were plant to stabilize these systems. European beachgrass-dominated dunes at Bodega were estimated to accumulate on average as much as 5,232 cubic yards (CY) per year (Cesmat et al. 2012). In England, where European beachgrass is native, rates of inland movement of the dunes varied between 5-10 feet/year on the more stable dune sections and up to 22 feet/year on the most mobile dune sections (Ranwell 1958). The study at Bodega showed that, during peak wind conditions, sand transport was 450 times higher in unvegetated areas than in vegetated ones (Cesmat et al. 2012).

Some sediment transport monitoring has occurred in restored dunes. A series of plots at Little River State Beach in Humboldt County where excavation had been conducted to remove European beachgrass experienced sand movement at a rate of 3.9 cubic feet for every 10 square feet per year (Transou et al. 2007). During restoration, dunes moved, on average, 2.5 feet per month in a southeast or inland-trending direction, although the rate of inland dune movement may have been retarded by sand moving over existing European beachgrass stands (Vaughan and Fiori 2007). In the earlier restoration efforts at Abbott’s Lagoon conducted by the Seashore between 2001 and 2005, sand movement was also assessed, and, while sand height did change, most of this change appeared to involve redistribution of sand within the treated area rather than actual movement of sand (Peterson et al. 2003).

Monitoring of the recent Abbott’s mechanical removal project at Abbott’s Lagoon showed that acreage of open sand areas, specifically on the inland perimeter of the dune system, increased from 2011 to 2013 (Johnson 2013b.). By spring 2013, approximately 10.1 acres of grassland, scrub, and wetland had been buried by newly remobilized sand (Johnson 2013b), which is equivalent to about 3% of the entire 300-acre restoration project area. Of the 10.1 acres, 7.3 acres involved inland “creep” of dunes, with the remainder being internal to the dune system (Johnson 2013b).

While the restored dunes visibly looked different in 2012 and 2013 relative to immediate post-construction conditions, this difference was not evident from topographic monitoring that was conducted in 2012 (NPS, unpub. data). Real-Time Kinematic GPS was used to survey longitudinal and latitudinal cross-sections of the Phase I project area and then compared to pre-

restoration elevations estimated from LIDAR (2009) and immediate post-restoration elevations that were also collected using RTK GPS (2011). Some sample cross-sections show a considerable reduction in dune elevations immediately after restoration, with little change between 2011 and 2012: The specifications called for no change in post-construction elevations, so either elevations were lowered, or, more likely, LIDAR data is not directly comparable to the RTK GPS data. Sands also had appeared to bury some of the adjacent native dune areas, as observed from “mounding” or burial of the many existing native dune plants, but, again, RTK GPS results suggest little immediate post-restoration change in elevations in these areas.

Dune movement is not isolated to natural and restored dune systems. While European beachgrass and iceplant were planted to stop movement of the dunes, the dunes have continued to migrate inland over the past 60- 70 years despite rapid expansion of these species. Dune expansion rates during the period 1943-2007 have varied from 16% at G Ranch to 59% at AT&T, with B Ranch estimated at 27%, resulting in average annual dune expansion rates during this period of 0.25% to 0.91% (Johnson 2013b). Analysis of historical aerial imagery at 20-year intervals between 1943 and 2007 suggests that this expansion occurred fairly continuously over approximately 65 years rather than in disjunct stages (Johnson 2013b). Coastal dune represents varying percentages of the total leased lands for different ranches, ranging from as low as 2% at G Ranch – currently, most of the dune system is outside the lease boundary at this ranch – to 24% at AT&T, with B Ranch estimated at 22% (Johnson 2013b).

Based on review of historic aerial imagery, the proportion of barren, sandy areas within the pasture adjacent to the restored dunes at Abbott’s actually appears lower in 2013 – even with sand remobilization – than in 1963, around the time that the Seashore was established (Johnson 2013b). In general, sand movement since restoration at Abbott’s in 2011 has tended to follow established historical sand deposition paths as evidenced by the fact that most of the ungrazeable areas buried were pre-existing dune “fingers” that extended into the pasture, which were already very sandy and dominated by dune and scrub type species. In the 1960s, two decades after European beachgrass was planted at Abbott’s, these fingers are very distinguishable on aerial imagery as barren, sandy areas (Johnson 2013b). Over the next 30- to 40 years, these fingers remained within the pasture, but became more vegetated by dune and Dune Scrub plant species, as can be seen in 2010 aerial imagery (Johnson 2013b).

While there has been an inland creep from the perimeter of the Phase I mechanical removal area, this creep has largely been burying areas that have been repeatedly buried by sand in the past, with some very minor (<1.0 acre) exceptions. Inland sand migration rates were quite high during the first few years after Phase I implementation, averaging around 4 acres/year, but in 2014-2015, they dropped considerably to approximately 0.8 acre/year (NPS, preliminary unpub. data). Some of the movement during the first two years post-restoration may have been influenced by the two successive dry winters and the fact dry winters allow for reduced establishment by stabilizing vegetation and are accompanied by higher winds that discourage vegetation establishment (Baye 2008). Lower migration rates in 2014-2015 may be due to a reduction in maximum wind speed during at least 2014 (NPS, preliminary unpub. data). In the following two years (2015-2016 and 2016-2017), inland sand migration rates rose again, more than doubling to approximately 2.0 additional acres each of those years, but the wet winter also promoted re-colonization of some of these depositional areas, with the extent of colonized

areas jumping from 0.32 acre in 2015 to 4.98 acres in 2017 or approximately 29% of the 16.9 acres affected by sand remobilization (NPS, unpub. data).

In addition to sand movement being affected by weather-related factors, invasive species removal method also appears to influence sand migration rates. In the same year that mechanical removal was performed at Abbott's Lagoon, some of the backdune and foredune areas immediately adjacent to these areas were treated with herbicide. Although European beachgrass was almost completely killed during the initial treatment effort (>99%), the dead aboveground biomass and presumably the rhizomes did not decompose rapidly. This delayed decomposition may have reduced the potential for sands in these areas to remobilize, as they are being held in place by the dead rhizomes. No migration of dune had occurred inland of these herbicide-treated areas by 2013 (Johnson 2013b) or in subsequent years (2014-2017; NPS, unpub. data). In addition, monitoring of dune areas at AT&T in 2017 showed no inland migration of dune areas one year after treatment (NPS, unpub. data).

Dune Soils

Movement of sand not only affects dune topography but soil profiles and development. Much of the Seashore's coastal dune areas are mapped as the "Dune land" soil mapping unit, which is defined as consisting of hummocks, mounds, and hills of loose sand blown from nearby beaches (SCS 1985). Some areas are active and shifting, and other areas are partly stabilized by coastal sagebrush and grass (SCS 1985). As befitting its origins, Dune land "soils" exhibit no profile development (SCS 1985). Some small, low basins are surrounded by dunes that are wet most of the year: These areas are similar to Aquepts, seeped, but they commonly are sand or loamy sand throughout (SCS 1985).

Dune soils evolve as sands move inland. The landward edge of dunes is defined by unconsolidated sand that may have started to develop a soil profile, but does not show the red color of advanced chemical weathering characteristic of older sand deposits (Wiedemann 1984). Adjacent to the dunes are coastal marine terraces with paleodune soils, many of which have been mapped as Sirdrak Sand: Sirdrak Sand represents deposits of latest Pleistocene to Holocene-era Aeolian or wind-driven dune sand deposition approximately 30,000 years before present (Knudsen et al. 2000; Baye 2008).

Centuries of grassland establishment has led to establishment of a mollisol over most of these former dune sands. Mollisols develop through significant accumulation of humus or organic matter in the surface horizon, or uppermost layer, of soils: This organic matter always almost derives from grass vegetation. Mollisols have deep, high organic matter, nutrient-enriched surface soil (A horizon), typically between 60-80 cm thick, that is known as a mollic epipedon. Over time, these grassland-dominated terraces have been subject to natural dune transgression or migration, probably from specific episodes of climate-driven dune instability (Baye 2008). These are evident in the occurrence of the previously discussed dune "fingers" extending some distance sometimes into pastures or even in the presence of dune "islands" some distance from the beach, such as occurs at F Ranch.

Coastal dune sands tend to be nutrient limited, especially for the essential nutrients nitrogen, phosphorus, and potassium (Barbour et al. 1985). Sand deposition is a common means by

which plants acquire critically-limiting nutrients in dune ecosystems (Pickart 1997). In addition, dunes also rely on aerosol deposition of certain micronutrients: concentrations of sodium (Na), magnesium (Mg), calcium (Ca), and chloride (Cl) in salt spray and fog are high enough to meet or exceed plant needs (Clayton 1972, Art et al. 1974, van der Valk 1974, Barbour et al. 1985). Organic matter provides nearly all the cation exchange – or nutrient binding – capacity in the coarse-textured sands (Pickart and Sawyer 1998).

While soil nitrogen levels are low, nitrogen content within plants are actually similar to crop plants on fertile soils (Holton et al. 1991). Dune plants obtain nitrogen from episodic inputs of detritus (Ranwell 1972, Holton 1980), nitrogen fixation by symbiotic and associative bacteria (Wahab and Wareing 1980, Holton 1980, Robertson 1982, Rose 1988), and photosynthetic use (Barbour et al. 1985). A 1991 study by Holton and colleagues at a Point Reyes dune system characterized the total nitrogen pool at 348 lbs/acre, of which 78% was soil organic nitrogen; 4% was soil inorganic nitrogen; and 18% was tied up in plant biomass. Nitrogen contributions to this system principally came from fog condensation, bulk precipitation, and symbiotic nitrogen fixation, in decreasing order of importance (Holton et al. 1991). Levels of nitrogen in Humboldt area foredunes ranged from 4.9 ppm for ammonium and 5.9 ppm for nitrates, whereas phosphorous levels were triple those levels (17.7 ppm; Nature Conservancy, unpub. data in Pickart and Sawyer 1998).

Based on the Point Reyes study, nodulated legumes, including beach pea, provided only minor nitrogen inputs to dune systems and only at a very localized level (Holton et al. 1991). However, other members of the pea family have been demonstrated to have much more effect on soil nitrogen levels. Bush lupine (*Lupinus arboreus*) -- a species that is native to California, but possibly not Marin County – is a tremendous nitrogen-fixers that derives 60% to 70% of their total nitrogen from atmospheric sources (Bentley and Johnson 1991). Nitrogen-enriched litter accumulates under the plants, contributing as much as 72- 185 kg of nitrogen per hectare per year to some systems (Gadgil 1971, Palaniappan et al. 1979, Skeffington and Bradshaw 1980, Maron and Jeffries, unpub. data in Maron and Connors 1996). With leaf drop or lupine death, high levels of exchangeable ammonium and nitrate become available to plant colonists, particularly weedy introduced grasses and forbs (Maron and Connors 1996). Once established, these weedy annual exotics create a labile litter that completely turns over organic matter and nitrogen, favoring persistence of weedy annuals (Zink et al. 1995).

Iceplant also effects soil nutrient status, but in a slightly differ manner. Iceplant-invaded soils appear to have higher organic matter content and lower pH than uninvaded areas (D'Antonio and Mahall 1991, de la Peña et al. 2010, Santoro et al. 2011, Winsemius 2013) and, in some areas, organic carbon (Vilà et al. 2006): This may be partially due to the much higher amounts of detritus produced by this species (Vilà et al. 2006). Iceplant leaves have high contents of tannin and antibacterial compounds (Van der Watt & Pretorius 2001), which may reduce litter decomposition rates of detritus (Vilà et al. 2006). These compounds may subsequently leach into the soils and acidify them, inhibiting nitrification (conversion of ammonium to nitrate) and increasing leaching of calcium and magnesium (D'Antonio and Haubensak 1998, D'Antonio and Mahall, unpub. in Conser and Connor 2009). Magnesium levels in dune soils at the Seashore were found to be lower in iceplant-invaded areas than in native dune (Winsemius 2013). Soil acidification occurs naturally with dune succession, but iceplant invasion speeds up this process

(D'Antonio and Mahall, unpub. *in* Conser and Connor 2009). Changes in soil nutrient content with iceplant encroachment ultimately appeared more variable, with some studies showing an increase in total nitrogen (Santoro et al. 2011), some showing a decrease (D'Antonio and Mahall, unpub. data), and some showing no change at all (Vilà et al. 2006). The effect of European beachgrass on soil nutrient pools does not appear as well studied as that of iceplant or bush lupine. Chemical legacy effects can later lead to further changes in the soil microbial communities that may reinforce the competitive advantage for invasive species. A study in Spain documented that microbial communities changed substantially with invasion, which they attributed to iceplant creating higher levels of microbial carbon sources that resulted in an increase in total microbial biomass, a decrease in the bacteria/fungi ratio, and a shift in microbial species composition (Badalamenti et al. 2016).

Iceplant also appears to compete with native plant species for water (D'Antonio and Haubensak 1998). It could disrupt osmotic levels in the soil in a similar fashion to another member of the iceplant family, *Mesembryanthemum crystallinum*, which retains salts in its tissues that are then released into the soils after plant and leaf senescence (Vivrette 1973, Vivrette and Muller 1977). These salt infusions create an osmotic imbalance in dune soils that draw water out of neighboring plants, resulting in desiccation and death. Higher sodium levels – salts are sodium chloride – have been found in soils invaded by iceplant (D'Antonio and Mahall, unpub. data *in* Conser and Connor 2009), although some studies have found lower levels of sodium in iceplant-invaded soils (Winsemius 2013). Removal of iceplant from the vicinity of two dune shrub species, mock heather and Menzies' goldenbush (*Isocoma menziesii*), resulted in higher pre-dawn water pressure potentials in shrubs compared to shrubs still surrounded by iceplant, suggesting that iceplant was utilizing water that would have otherwise been available to shrubs (D'Antonio and Mahall 1991). The greenhouse study conducted by Conser and Connor (2009) found no differences in soil moisture content between iceplant-invaded soils and native dune soils, but this may point more to a lack of long-term changes in soil properties that could negatively affect soil moisture-holding potential in soils rather than differences in field moisture conditions. Ultimately, even if iceplant does not compete for water, it appears to fare better under drier conditions, with iceplant germinating in lower moisture soils than native dune plants in Spain (Novoa et al. 2012).

Sand transport and nutrients are not only the factors affected by non-native plant invasion. Despite its seemingly barren appearance, dune soils play host to a number of bacterial, fungal, and pathogen species that appear to at least partially drive the dynamics of invasion and succession within dune systems. In its home range, European beachgrass acts more as a foredune pioneer species that thrives under influxes of windblown sand, with vigor of stands declining over time (Hope-Simpson and Jeffries 1966, van der Putten et al. 1989). European beachgrass stabilizes sand, but stabilization may encourage microorganisms that infect beachgrass roots and lead to the decreased vigor and aboveground biomass production evident in backdune communities (van der Putten et al. 1988, de Rooij-van der Goes 1995). Within European dune systems, soil-borne pathogens, primarily endoparasitic nematodes (*Heterodera arenaria* and *Pratylenchus* spp.) and fungi, build up over time as sand features stabilize (van der Stoep et al. 2002), leading to eventual replacement of beachgrass by other, later successional species that are more tolerant of these pathogens (van der Putten et al. 1993).

In areas outside Europe, however, European beachgrass does not function strictly as a foredune pioneer, but often rapidly spreads to establish a persistent monoculture in both foredunes and backdunes. Some researchers have surmised that this may result from a release from beachgrass's natural soil enemies in its home range. For example, in South Africa, the absence of this negative soil feedback loop appears to be part of the mechanism that allows European beachgrass to expand and persist unchecked (Knevel et al. 2004). Sampling of soils within European beachgrass's introduced range suggested that invasives are not necessarily escaping from all soil enemies (e.g., generalist nematode taxa), but only from specific ones such as feeding-specialist natural root-feeding nematodes (van der Putten et al. 2005).

Beckstead and Parker (2003) documented only partial escape in California dune soils, with pathogenic nematodes absent, but fungal species similar to those of beachgrass' native regions (Netherlands) present. Early growth of European beachgrass in unsterilized California dune soils was depressed by pathogens at very similar levels to those seen in corresponding studies from The Netherlands (van der Putten and Peters 1997), and in some cases, fresh sand from California showed even more of suppressive effect (Beckstead and Parker 2003). Studies of soils from other portions of beachgrass's invaded ranges in the United States showed that, while root knot nematodes found in beachgrass's native range were not present, sedentary, endoparasitic cyst nematodes were (van der Putten et al. 2005).

Similarly, while relative production of European beachgrass was higher in South African dune soils than in European ones, a comparison of production rates from sterilized and unsterilized South African dune soils showed that local pathogens were inhibiting beachgrass growth to some degree (Knevel et al. 2004). A later study (Eppinga et al. 2006) argued on the basis of computer modelling incorporating plant competition and plant-soil community interactions that, rather than being released from its enemies, European beachgrass in California may accumulate local pathogens that eventually exclude native dune species. A similar dynamic may occur within the invaded ranges for iceplant. In some European systems, iceplant invasion is believed to be related more to the fact that iceplant may not be susceptible to existing soil pathogens: inoculation with rhizosphere soil from the invaded range in the Mediterranean region did not have a significant effect on plant biomass, while soils from the native range reduced iceplant biomass by 32% (van Grunsven et al. 2009).

Non-native invasives that are well-behaved within their home range may spread, then, either because their natural soil enemies are absent or that they are better able to tolerate accumulation of local pathogens than native neighbors. However, some researchers have argued introduced species are not always successful in invading native communities, because strong biotic interactions with native species can hinder their establishment and spread. A study on South African dunes found that one species, *Sporobolus virginicus*, might actually contribute to soil feedback resistance against European beachgrass and concluded that not only plant species diversity, but the type of plant species present, might be important in determining the potential for biotic resistance (Knevel et al. 2004). In a meta-analysis of 52 native-exotic pairwise feedback comparisons in 22 studies, researchers concluded that soil biotic resistance is somewhat frequency dependent: soils conditioned by native species could provide resistance when frequency of the invader is low, even if the invader exhibits a positive plant-soil feedback, but this resistance may be adversely impacted by disturbance events or that enable invaders to

spread quickly and establish high numbers in native ecosystems, particularly in situations in which non-native species have a dispersal or competitive advantage over native ones (Suding et al. 2013). Experiments have suggested that soil biotic community in at least some Mediterranean dune systems – where iceplant is not native – may initially resist the invader, but that, over time, the resistance to invasion breaks down (Vilà et al. 2006, de la Peña et al. 2010). High levels of a symbiotic fungal species, chytrids, were found in soils that had been invaded for some time by iceplant: This species may be detrimental to other native dune species, but not iceplant (de la Peña et al. 2010).

As noted earlier, legumes or members of the pea family have strong associations with bacteria in their root systems that help them fix nitrogen. Dune plants form relationships with arbuscular mycorrhizal (AM) fungi near roots that enhance phosphorous uptake and sand dune stabilization through aggregation of sand grains (Koske and Polson 1984). The percent colonization of vascular plants by AM appears to be higher in level and more stabilized dunes, but has not been found to vary that much by species, plant size, surrounding plant cover, or topography (Pickart et al. 1990). In Dutch dune systems, AM infection rates were positively associated with vigor of beachgrass stands (Kowalchuk et al. 2002), which may be linked to the fact that AM mycorrhizae may, in fact, be able to mediate adverse effects of root-feeding nematodes impacting roots of European beachgrass (de la Peña et al. 2006). However, this beneficial association does not necessarily exist in areas outside its native range, with AM infection rates showing no relationship with beachgrass biomass in Oregon dunes (David et al. 2016).

Invasive plants can either arrive with AMF already present in intact root systems, or they may be mycorrhizal generalists that can take advantage of AMF species already present in the new home (Pringle et al. 2009). Several studies have shown that invaders, once established, can increase abundance of AM or influence abundance of particular AM species that facilitates growth of the invader, but suppresses that of native plant species (Niu et al. 2007, Inderjit and van der Putten 2010, Zhang et al. 2010). While most of the evidence supporting this feedback mechanism comes from systems other than dunes, one study did show that germination of European beachgrass in iceplant-invaded soils resulted in lower aboveground biomass than in non-iceplant-invaded soils, which may relate to suppression of beneficial mycorrhizal interactions by iceplant (de la Peña et al. 2010). Invasives can also target symbiotic mutualistic organisms such as nitrogen-fixing bacteria associated with the root systems of legumes. Studies identified increased pathogenicity and possibly reduced symbiosis as negatively influencing vigor of the late successional nitrogen-fixing dune shrub species, *Hippophaë rhamnoides* (Oremus and Otten 1981).

Dune Vegetation

In general, species composition in dune vegetation along the Pacific Coast of North America does not vary greatly relative to latitude (Pickart and Sawyer 1998). The most significant ecological boundary for dunes on the Pacific coast occurs at 37 degrees North, where foredunes shift from being herb-dominated in the south and grass-dominated in the north (Barbour et al. 1976). Species composition, as well as species richness, appears more strongly correlated with

distance from the shoreline than geographic location (Barbour 1978, 1992) or physical dune forms (Parker 1974).

Dunes are a harsh environment for plants. In general, constant cycles of sand burial make available critically-limiting nutrients that normally would not be available, given the lack of organic matter in the soil, as well as the very poor nutrient-binding characteristics of sand (Pickart 1997). Specifically it appears that phosphorous, nitrogen and potassium are most limiting in sand dunes and appear to have the ability to restrict and/or bolster growth habits of native dune plants, as well as non-native species such as European beachgrass (Willis 1965).

The type of communities and assemblage of species within communities can be strongly driven by physical processes and structure within dunes. Initially, scientists felt that vegetation communities within dunes were determined by salt spray, but in later years, more have come to think that communities are shaped by species' ability to tolerate sand burial (Maun and Perumal 2002). Another study found that species composition in dunes was driven by exposure to wind (Lortie and Cushman 2007), which in itself may be indirectly related to the degree of sand movement and burial.

Very few coastal plant species have the ability to survive burial of greater than 1-2 feet at a given time. Non-native European beachgrass has a distinct advantage over many native dune plants in that it thrives in areas of increased sand burial and is adapted to survive single burial events of up to 3 feet (Ranwell 1958, Huiskies 1979). Only a few other plant species at the Seashore have shown resilience and vigor in response to sand burial of greater than 1-2 feet at a single time – salt rush (*Juncus lescurii*), Pacific wildrye (*Elymus pacificus*), and Tidestrom's lupine are surprisingly capable of surviving and growing through such large-scale sand deposition. Species that are not adapted to sand burial through physiological adaptation mechanisms such as adventitious roots (European beachgrass) or rapid vegetative growth rates cannot survive these burial events. In this way, sand burial has the ability to "reset" the structure of a plant community to favor burial-tolerant species.

Representative Communities

American Dunegrass: This plant community is mostly restricted to the foredune area of a coastal dune system. Prior to development and the introduction of non-native species, native American dunegrass was common along the U.S. west coast from Monterey northward. Currently it can be found only at Point Reyes (near Abbott's Lagoon) and the Humboldt Bay dunes (Pickart and Sawyer 1998). This community is ranked by the Nature Conservancy as critically imperiled, which means five or fewer global occurrences of fewer than 2,000 acres remain. Vegetation in the native dunegrass community is sparse (25%-75% cover) and is dominated by American dunegrass. Other associated species include beach bursage (*Ambrosia chamissonis*), Douglas's bluegrass (*Poa douglasii*), beach morning glory (*Calystegia soldanella*), yellow sand-verbena (*Abronia latifolia*), and beach evening primrose (*Camissonia cheiranthifolia*).

Dune Mat: Dune mat communities are comprised of mat-forming vegetation that is often variable and sparse. Species include beach bursage, dune sagebrush (*Artemisia pycnocephala*), coast buckwheat (*Eriogonum latifolium*), Douglas's bluegrass (*Poa douglasii*), yellow sand-

verbena, beach pea (*Lathyrus littoralis*), beach strawberry (*Fragaria chiloensis*) and beach evening primrose. Dune mat habitat is considered rare and has been severely impacted by European beachgrass (Pickart and Sawyer 1998). Many federal, state and locally listed species are restricted to Dune Mat communities, including the federally endangered beach layia (*Layia carnosa*; FE) and Tidestrom's lupine, as well as curlyleaf monardella (*Monardella sinuata* ssp. *nigrescens*; California Native Plant Society (CNPS) List 1B.2-Proposed) and dune gilia (*Gilia capitata* ssp. *chamissonis*; CNPS List 1B.1). In 2012, bluff wallflower (*Erysimum concinnum*) was also listed (CNPS List 1B.2): the Seashore reputedly has one of the largest remaining populations of this species. At Limantour, a few patches of pink sand-verbena (*Abronia umbellata* var. *breviflora*; CNPS List 1B.1) also occur along the beachward side of the dunes where European beachgrass and iceplant grow intermixed.

Native dune vegetation communities occur throughout the Seashore's dunes, but they are often isolated remnants of once large communities that are now fragmented by expanding monocultures of European beachgrass and iceplant. At the three general areas prioritized for near-term restoration, approximately 25 acres of Dune Mat have been mapped at B Ranch, with 8.5 acres mapped at AT&T. Dune Mat vegetation also exists at North Beach and Davis Property, with a very small amount in the northern portion of A Ranch. No Dune Mat has been mapped at Limantour specifically, although pockets of Dune Mat vegetation may exist within invaded areas.

Species within these Dune Mat communities seem to differ slightly depending on age or successional status of the dunes. Younger dunes with less differentiated soils (primary successional) support beach bursage, beach pea, beach morning glory, yellow sand-verbena, Tidestrom's lupine, and non-native species such as European searocket (*Cakile maritima*). Slightly more mature dune systems support vast numbers of Douglas's bluegrass, as well as beach evening primrose, although numbers of the latter are also often very high in stabilized dunes (mid-successional). As dune systems become even more stabilized – perhaps because they are surrounded by European beachgrass or iceplant stands that demobilize dunes or because they are on the inland edge of the inland-migrating dune systems – the composition of dominant species shifts again slightly towards species such as dune buckwheat, dune sagebrush, sea thrift (*Armeria maritima*), gumplant (*Grindelia stricta*), sand mat (*Cardionema ramosissimum*), and woolly lotus (*Lotus heermanii*; late-successional). A few species appear to occur across a range of successional stages, such as seaside daisy (*Erigeron glaucus*; mid- to late successional), beach layia (primary to mid-successional), and beach evening primrose (mid- to late successional).

Dune Scrub: In the inland portions of active dune fields, a more shrub-dominated vegetation community can occur. Dune scrub once established on stabilized dunes stretching from Bodega Head to Los Angeles, but, by the 1970s, only Point Reyes-Dillon Beach and Morro Bay-Purisima Point still supported sizeable Dune Scrub areas.

Most of the Dune Scrub establishes in the more stabilized backdune areas. A study from 1979 defined at least four (4) predominant Dune Scrub communities. On stabilized dunes, bush lupine (*Lupinus arboreus*) was dominant, with yarrow (*Achillea millefolium*), ripgut brome (*Bromus diandrus*), and sheep sorrell (*Rumex acetosella*) also present (Holton and Johnson

1979). A second association also occurred on stabilized slopes on inland sites and included bush lupine, mock heather (*Ericameria ericoides*), and coyote brush (*Baccharis pilularis*), with beach bursage and curlyleaf monardella as understory species (Holton and Johnson 1979). This tends to be a drier, more open type of site than in the first association (Holton and Johnson 1979). Deep, recently reworked sands on the top of ridges supports a community dominated by chamisso lupine (*Lupinus chamissonis*) and mock heather, with bush lupine much more infrequent (Holton and Johnson 1979). Even more exposed locations tend to be established only by chamisso lupine (Holton and Johnson 1979).

Intermediate between Dune Scrub and Dune Mat are two associations that are colonized by more herbs and sub-shrubs than the other associations. Bush lupine and lizard tail (*Eriophyllum staechadifolium*) occur on exposed slopes nearest the ocean (Holton and Johnson 1979). Even less stable sands play host to a mix of lizard tail and beach bursage, with Douglas's bluegrass and dune sagebrush also high in cover, particularly on semi-stable dunes immediately behind the foredunes (Holton and Johnson 1979). Bush lupine showed strong association with the highest nitrogen and organic matter content and smallest particle size of any of the scrub species (Holton and Johnson 1979). In terms of distance from the ocean, mock heather was restricted more to inland sites, while lizard tail occurred at more seaward locations: Dune sagebrush showed no relationship with distance to ocean, possibly because it colonizes recently disturbed sites, but is eventually replaced by other dune shrub species (Holton and Johnson 1979).

In general, most of these Dune Scrub areas have now been invaded by European beachgrass or iceplant, with Dune Scrub existing principally as "pockets" of native shrubs within dense European beachgrass stands or strongly intermixed within European beachgrass stands. Currently, most of these Dune Scrub pockets are dominated by coyote brush and mock heather, although other species may occur, including poison oak (*Toxicodendron diversilobum*), California blackberry (*Rubus ursinus*), bush lupine, bracken fern (*Pteridium aquilinum*), wild cucumber (*Marah fabaceus*), miner's lettuce (*Claytonia perfoliata*), and fairy mist (*Pterostegia drymarioides*). Some limited Dune Scrub exists at AT&T/North Beach, B Ranch/A Ranch/Davis Property, Abbotts Lagoon, and at Limantour, although its extent has not been conclusively mapped.

Dune Slack Ponds or Swale Wetlands: In addition to rare foredune communities, many dune systems also include "dune swale" or "dune slack" wetlands. Coastal interdunal wetlands are common components of larger active and stabilized coastal dune fields, ranging from Coos Bay, Oregon, and south to San Luis Obispo County, California. They can be referred to as "slack dune ponds" when associated with larger and deeper water or "coastal dune swales" when water is shallow. In general, most of the dunes swales and slacks within the Seashore tend to be large meadow or seasonal wetland features in low basins behind active foredunes or narrow drainages between dune ridges that are oriented perpendicular to the shoreline. Seasonal or even perennial ponding occurs when either sand supply is disrupted or sand is blown out, leaving certain areas closer to the groundwater table, or when dune movement interrupts and even "dams" surface water flow.

While obviously wetter than the adjacent dune communities, these seasonally wet areas may not be wet enough to be regulated as wetlands by regulatory agencies such as the U.S. Army Corps of Engineers (USACE). In some cases, only the lowest elevation portions of these distinct areas are wet enough to qualify as regulated wetlands. Most of these areas would be regulated by the California Coastal Commission (CCC) as species are typically listed as at least facultatively hydrophytic or occurring in wetlands at least 50% of the time, and the CCC only requires one parameter to be met to qualify as a wetland subject to oversight under the Coastal Act.

The vegetation at these sites typically falls into the sedge series and is dominated by a mix of sedge and rush species. Slough sedge (*Carex obnupta*), dune sedge (*Carex pansa*), and salt rush are the dominant species of the seasonally wet sedge series. Associated species include Pacific silverweed (*Potentilla anserina*), springbank clover, (*Trifolium wormskioldii*), marsh baccharis (*Baccharis douglasii*), beach starwort (*Stellaria littoralis*; CNPS List 4), and California blackberry. A few wetlands even support sparse stands of arroyo willow (*Salix lasiolepis*; Baye 2008). Cover commonly approaches 100% (Pickart and Sawyer 1998). Dune swale wetlands with a mix of meadow and riparian species represent an unusual plant association that is uncommon in central California coast dunes, although none of the component species are themselves that rare (Baye 2008). They appear to be intermediate between typical dune swale wetlands and freshwater fens, which are characterized by freshwater marsh and wetland scrub vegetation in low-flow systems that form on acidic sandy substrates (Baye 2008). One of these features occurs at Abbotts Dunes.

European beachgrass-dominated Foredunes and Backdunes: European beachgrass is a native perennial grass of Europe and North Africa introduced to California in the late 1800s to help stabilize blowing sand dunes. It is now recognized as an invasive species in North America, Australia, and New Zealand (Johnson 1982, Heyligers 1985, Wiedemann 1987). Throughout its range, beachgrass grows in sand dunes and adjacent coastal habitat (i.e., wetlands, grasslands, etc.). However, it grows most vigorously on open foredunes comprised of mobile and semi-fixed sand, where it tolerates extremes of substrate instability, water drainage, nutrients, organic matter content, and soil texture (Huiskes 1979). Beachgrass can actively grow in soil pH varying from 4.5 - 9.0 and soil temperatures varying from 10 - 40°C, although it appears less tolerant of higher salt concentrations, preferring levels between 1.0 and 1.5% (Slobodochikoff and Doyen 1977; Pickart 1997). In Britain, beachgrass tends to occur in two communities – mobile dune, where it is the dominant species, and semi-fixed dunes, where sand is less mobile, and European beachgrass becomes less competitive (British National Vegetation Classification System).

Because of this extreme adaptability, European beachgrass became a popular choice for stabilizing unstabilized dunes in the United States. European beachgrass was first planted on the west coast of North America at Golden Gate Park in San Francisco in the late 1800s (Lamson-Scribner 1895, Lamb 1898). The species was heralded as a desirable sand stabilizer and was eventually embraced by U. S. Soil Conservation Service and other agencies (Pickart 1997). Along the Pacific Coast, European beachgrass has spread to inhabit areas from Santa Barbara County, California, in the south to Canada in the north. Initially, beachgrass invasion appeared more severe in areas north of San Francisco more than in areas south, but, in recent years, beachgrass has been apparently rapidly expanding in some central and southern California

coastal dune systems, including Guadalupe-Nipomo Dunes and Vandenberg Air Force Base (Chestnut 1997).

While European beachgrass was present in San Francisco as early as the late 1800s, there is no reliable information as to when this species arrived in the Seashore, which is approximately 30- to 40 miles northwest of San Francisco Bay. Anecdotal information suggests that ranchers and other landowners probably planted both European beachgrass and iceplant to stabilize the dunes: The Seashore did not acquire most of these lands until 1963 or later. Historical aerial photographs from the 1940s show scattered pockets of European beachgrass at Abbotts Lagoon and other beach areas such as the dunes near B Ranch. Since then, this species has rapidly spread along the Great Beach and at Limantour Beach. Beachgrass tends to be more prevalent than iceplant along some of the northern sections of the Great Beach, stretching from North Beach northward to north of Abbotts Lagoon up to Kehoe, and at Limantour Beach. While European beachgrass is present south of North Beach, extending to the dunes near the Lighthouse, iceplant often either co-dominates certain areas or is actually even more dominant. The reason for this geographic pattern in extent of particular invasive species is unknown, but may be related to what species was originally planted in particular dune areas by landowners prior to park ownership and to the structure of the dune system present.

European beachgrass predominantly spreads vegetatively through rhizomes or belowground root structures. Past research has documented that, as plants are buried by blowing sand, their shoots can grow up to 3.3 feet per year (Ranwell 1972, Barbour et al. 1985), however, other studies have documented rates as high as 6.6 feet in six months (Bossard et al. 2000). If broken off, these rhizomes can re-establish plants from even the smallest rhizome fragment (<2 inches). Based on recent excavations in dunes at the Seashore, the depth of rhizomes can differ between foredune and backdune systems: In the foredune, rhizomes can extend anywhere from 6 to 12 feet, while in the backdune, rhizomes tend to be shallower, averaging between 3 and 6 feet. This agrees well depths from other studies, which showed a range between 3.3 feet (Salisbury 1952) to as much as 15 feet (Buchenau 1889).

The differences in backdune and foredune rooting depth may relate to a phenomenon known as the “Ammophila problem” (Marshall 1965) in which a lack of fresh sand accumulation results in reduced vigor of European beachgrass: Plants produce less roots (Wallén 1980, Gray 1985), which may potentially relate to the fact that roots of European beachgrass in stabilized dunes are infected by microorganisms that reduce formation of new tillers (van der Putten et al. 1988 *in* Pickart and Sawyer 1998). Fresh sand accumulation also provides nutrients (Pickart 1997) and inputs of arbuscular mycorrhizal fungi (symbiotic organisms that improve water uptake and nutrient acquisition) and prevents build-up of pathogens (van der Putten et al. 1988 *in* Pickart and Sawyer 1998). Apparently, fragments can even withstand prolonged seawater immersion, with viable fragments surviving in cold marine waters for as long as eight (8) tidal cycles (Baye 1990), although the plant itself can tolerate lower salt concentrations in dune soils than native plants (1.5%; Huiskes 1979).

Seed of European beachgrass is viable, but establishment by seedling is not common as vegetative establishment (Pickart and Sawyer 1998, Huiskes 1977) and may be more likely to occur only under certain climatic conditions or in certain types of microhabitat. A study in Great

Britain found that seed only germinated in places where sand was comparatively damp, specifically at the base of slopes where direct rainwater concentrations and leaching was highest (Huiskes 1977). Most critically, seedlings of beachgrass require some form of retained moisture, decreased sand burial, and an overlay of damp, clay-like soil (Huiskes 1977).

Once established, European beachgrass tends to dominate in both foredune and backdune areas, although community structure differs slightly. In foredune areas, cover of beachgrass tends to be dense, but other plant species that can establish and persist in “pockets” of open sand within the beachgrass stands include beach bursage, beach pea, sea rocket, yellow sand-verbena, beach morning glory, and Douglas’s bluegrass. This community occurs extensively in the northern portion of the Great Beach, near AT&T/North Beach, and in certain areas of the southern portion of the Great Beach such as B Ranch/A Ranch/Davis, as well as along Limantour Spit. European beachgrass-dominated backdunes are also characterized by high densities of European beachgrass, but occur further inland than 160 feet from the beachfront. In some backdune areas, the vegetation cover includes a sizeable percentage of native shrubs -- coyote brush, mock-heather, and bush lupine – and herbs. This community occurs in the central and northern portions of the Great Beach, including at AT&T, and more sporadically further south such as at B Ranch, A Ranch, and Davis Property.

Iceplant-dominated Communities: Iceplant, a succulent, prostrate member of the Aizoaceae family, was introduced to California in the early 20th century to stabilize dunes adjacent to railroad tracks and was later planted along highways and on coastal sand dunes (Albert 1995a). Its popularity as a planting along California’s highways and freeways has led this species (*Carpobrotus edulis*) to be also called, “freeway iceplant.” At one point, the California Department of Transportation was maintaining almost 6,000 acres of this species along roadways (Schmalzer and Hinkle 1987). *C. edulis* comes from South Africa, but the origins of another member of this family, *Carpobrotus chilensis*, is disputed (Albert 1995a). *C. chilensis* is not considered invasive, but hybrids of *C. edulis* and *C. chilensis* are (Albert 1995a). Unless otherwise noted, when this report refers to iceplant, it is denoting *C. edulis*.

Iceplant invades a number of coastal plant communities in California, including grasslands, dunes, and coastal scrub (D’Antonio 1993). Over time, in these habitats, this species forms dense, clonal mats up to 20 in thick that may overtop even native shrubs (D’Antonio and Mahall 1991). This succulent spreads both vegetatively and by seed. In one study, recruitment of seedlings accounted for only 1% to 2% of the change in cover measured over four years, suggesting that clonal growth often represents the primary means of expansion (D’Antonio 1993), although this is not always the case (de la Peña et al. 2010). Germination of iceplant seeds shows no effect from fire, but is enhanced by passage through animals’ digestive system, with animals being important dispersers of *C. edulis* seeds (D’Antonio 1993). Seeds can remain viable in soils for at least two years, and the species is both self-fertile and self-compatible, meaning that it can successfully fertilize itself (D’Antonio 1990, Suehs et al. 2004). In terms of vegetative reproduction, a single individual of iceplant, specifically *C. edulis*, can form dense, circular mats over 33 feet wide and up to 20 inches deep (D’Antonio and Mahall 1991). This species can spread radially at rates high as 3.3 feet per year by growing from multiple axes and rooting where nodes can contact the soil (Wisura and Glen 1993).

While iceplant can be sensitive to water stress (MacDonald et al. 1984), this species can employ a special type of mechanism called Crassulacean Acid Metabolism (CAM) that increases its ability to tolerate water-limited, nitrogen-limited, and saline environments (Schmalzer and Hinkle 1987). Other abiotic and biotic factors that affect iceplant viability include frost sensitivity (MacDonald et al. 1984), scale insects introduced to California from the species' native South Africa (Tassan et al. 1982, MacDonald et al. 1984, Washburn and Frankie 1985), and hybridization (Albert 1995b, Albert et al. 1997). In one study, only 25% of the plants retained the original parental genotype, leading researchers to conclude that the overall species complex in California appears characteristic of a so-called hybrid swarm (Albert et al. 1997).

Iceplant can obviously directly affect other plant species by displacing them and even overgrowing them, which reduces biomass, life span, and reproductive output of adjacent plants (Pickart and Sawyer 1998). Iceplant reduces native species diversity (Vilà et al. 2006, Carboni et al. 2010), the fitness of neighboring native plants (D'Antonio and Mahall 1991), and the pollination network in invaded communities (Bartomeus et al. 2008). As discussed previously, it also has indirect effects, as well, by lowering pH, reducing nitrogen content and available water, increasing organic matter and carbon, and even changing the soil biota (higher levels of chytrid fungi and pathogens; lower levels of mycorrhiza (D'Antonio and Mahall 1991, Vilà et al. 2006, Conser and Connor 2009, de la Peña et al. 2010, Santoro et al. 2011). The species' dense fibrous root system, which is concentrated in the upper 25 inches of the soil, may enable it to compete more effectively for water with native plant species (D'Antonio and Mahall 1991).

These direct and indirect effects work differently in different environments to promote iceplant establishment: In one study, iceplant established more readily in dunes than in coastal scrub due to lower abundance of brush rabbits and, therefore, reduced herbivory intensity (D'Antonio 1993). In grasslands, already established plants precluded establishment of iceplant seedlings, with disturbance and creation of gaps in the canopy cover required for successful seedling recruitment (D'Antonio 1993).

In dunes, resistance to invasion by iceplant in portions of its non-historic range may be mediated by soil biota (Vilà et al. 2006, de la Peña et al. 2010), with the resistance eventually broken down over time by the prolific seed rain (>1,000 seeds per fruit; de la Peña et al. 2010). As noted above, expansion by iceplant changes the soil by negatively affecting water availability for other plants, reducing soil pH, and altering nutrient and potentially soil biota dynamics (higher levels of chytrid fungi and pathogens; lower levels of mycorrhiza; D'Antonio and Mahall 1991, Vilà et al. 2006, Conser and Connor 2009, de la Peña et al. 2010, Santoro et al. 2011). These soil changes may cause iceplant to switch from sexual to asexual reproduction modes (de la Peña et al. 2010). In other regions, iceplant invasion appears to be successful, because the species is not affected by pathogenic species that are normally present in its native dune soils (Reinhart et al. 2005, van Grunseven et al. 2009 *in* de la Peña et al. 2010).

Over time, iceplant can develop into extensive carpets or mats that exclude almost all other species, but prior to this coalescence of patches, other species can often subsist within "pockets" of open, sandy areas within iceplant stands. The presence of iceplant appears to

drive the successional status of native dune “pockets” towards a more mature, late successional community supporting species such as dune buckwheat, dune sagebrush, sea thrift, sand mat, and woolly lotus. A few species appear to occur across a range of successional stages, including beach evening primrose (moderately to highly stabilized), which can be quite common in iceplant-dominated dunes. Iceplant becomes more dominant between North Beach and South Beach and further south near B Ranch, A Ranch, Davis Property, and at the Lighthouse. Most of the iceplant at Limantour Spit occurs intermixed with European beachgrass, although there are approximately 9 acres of pure iceplant patches.

Secondary Invaders: Another threat to successful re-establishment of native dune habitats comes from so-called “secondary invaders.” Secondary invaders are classified as species that are not the primary removal target, but that take advantage of decreases in cover of the primary target species and increases in available space to gain a foothold and potentially oust native species. Secondary invasion has been documented as issues for many invasives removal-related restoration projects (Pearson et al. 2016). At Abbotts Lagoon, the most abundant secondary invaders in mechanically restored and native dune areas were European searocket (*Cakile maritima*), along with New Zealand spinach (*Tetragonia tetragonoides*), European black nightshade (*Solanum nigrum*), and iceplant (*Carpobrotus* spp.). A different suite of secondary species impacted herbicide-treated backdune areas, including ripgut brome (*Bromus diandrus*), fescue brome (*Festuca bromoides*), common groundsel (*Senecio sylvestris*), and sow thistle (*Sonchus* spp.). These species have also encroached on other dune restoration areas, including at B Ranch North and AT&T, but the Seashore has actively countered these threats through both manual removal and herbicide treatment.

Of these secondary invaders, European searocket currently appears to pose the most threat. In terms of sheer acreage, European beachgrass and iceplant are definitely the “primary” invaders at the Seashore, but European searocket has definitely made inroads at some of the dune systems, Abbotts Lagoon in particular. By summer 2016, European searocket and these other species had established in relatively moderate to dense abundance in a large percentage of the mechanically restored areas at Abbotts Lagoon, as well as in native dunes that have been affected by restoration by inland movement of destabilized sands. In certain instances, European searocket was actually blanketing some of the extensive Tidestrom’s lupine colonies that had established in the mechanically restored areas.

European searocket reproduces by seed, which is distributed by ship ballasts, sand transport, tidal movement, and human activity (Cal-IPC 2012). In comparison to native dune plants, it produces more seed, disperses greater distances, and tolerates more disturbances (Cal-IPC 2012). It has a greatly extended flowering season that lasts almost year-round, at least in Point Reyes. Few sources of information could be found on seedbank longevity, but California Invasive Plant Council’s Plant Assessment Form characterized the seedbank as relatively short-lived in foredune environments (at least 2 years; Knapp and DiTomaso 2005). However, “if *C. maritima* can survive into a second or even a third year of reproduction, it can produce an 18-fold difference in seed output” (Knapp and DiTomaso 2005). Seed production is increased during colonization, with plants allocating more resources to seed production than plant survival, and the species has high fecundity, with seed set often being 90% (Knapp and DiTomaso 2005). Ironically, native deer mice – which have impacted Tidestrom’s lupine

populations in the Seashore – also acts as a predator of European searocket seeds, potentially removing as much as 95% of the seed in some instances and, thereby, resulting in only 20% survivorship (Knapp and DiTomaso 2005). However, burial of seeds by moving sands can help them escape mice predation (Knapp and DiTomaso 2005).

Laboratory experiments demonstrate that the plant may have allelopathic effects, but these effects have not been observed in field experiments (Cal-IPC 2012). Disturbance rather than allelopathy appears to be how European searocket and some of the other secondary invaders gain a foothold in restored dunes. Secondary invaders such as European searocket and New Zealand spinach often subsist in relatively low abundance until a disturbance event provides them an opportunity to dramatically expand in numbers. In a sense, they are “sleeper weeds” that often exhibit a “lag phase” between establishment and large-scale expansion, waiting until conditions become right for exponential growth (Hui and Richardson 2017, Crooks 2005).

At Abbotts, the disturbance trigger for searocket population expansion was, undoubtedly, the restoration project, either directly through restoration activities or indirectly through deposition of remobilized sands. Not only does restoration remove potential competitors (European beachgrass and iceplant), but both mechanical removal and herbicide treatment created open space that is more subject to wind and wave disturbance. Searocket also thrives on Camp Pendleton’s beaches and dunes, which are continually disturbed by the military conducting tank maneuvers (B. Klink, GSRC, *pers. comm.*). In fall 2015, contractors removed 10 tons of European searocket plant material from the 7-mile weed treatment area at Camp Pendleton (Klink, *pers. comm.*).

Other Invasive Species: Bush lupine is native to California, but its historic range apparently incorporated only central and southern California. In 1902, Davy reported that bush lupine did not occur north of Point Reyes, while the Jepson Manual (1993) listed the historic range as Sonoma County south to Ventura. Botanical experts still question whether this species is native to Marin County (Pickart and Sawyer 1998, Baye 2008). There are some indications that bush lupine was planted with European beachgrass for stabilization and reclamation of coastal dunes after 1870 (McLaren 1924) and was introduced to North Coast dunes where it is highly invasive (Baye 2008).

As with other coastal areas in Sonoma County, bush lupine establishes in both grasslands and dunes (Pickart and Sawyer 1998). Without grazing, numbers of this species can increase dramatically in the park’s coastal grasslands. In dunes at the Seashore, it can co-occur with European beachgrass in backdune areas, but it appears much less prevalent in these invaded communities than coyote brush and mock heather. At Tomales Dunes, it dominated the Dune Scrub community, however, and, after initial restoration at Abbotts, it established well in open sand conditions within the park’s dunes. In addition to altering soils and, therefore, potentially altering vegetation communities, this species is also known to hybridize with other native lupine species such as manycolored lupine (*Lupinus variicolor*) and seashore lupine (*Lupinus littoralis*), so it could threaten viability of other lupine species within the dunes, including Tidestrom’s lupine and chamisso lupine (Baye 2008).

COASTAL DUNE RESTORATION EFFORTS ALONG PACIFIC COAST

Land management agencies up and down the Pacific Coast have been experimenting with approaches for coastal dune restoration for decades. Some of the most well-known projects are in Humboldt, which focused primarily on manual removal using shovels or other hand-held tools. Early results from a Humboldt dune restoration project suggested this method showed considerable promise: the project successfully eliminated beachgrass, had relatively low re-treatment needs in the second year, and increased native plant cover from 2.7% to 38%, although it required re-treatment every week to every two (2) to four (4) weeks for approximately nine (9) months (Pickart and Sawyer 1998). The success of this project prompted larger projects to be conducted at Humboldt in 1992 and 1997, which also proved successful in terms of eliminating beachgrass and encouraging native species to colonize (Pickart and Sawyer 1998). Retreatment by hand required at least eight (8) visits during the first year and seven (7) visits the second year, followed by additional treatment during a third year (Pickart and Sawyer 1998).

A few agencies have used mechanical removal for treatment of European beachgrass. The U.S. Forest Service (USFS) first used mechanical removal at the Oregon Dunes National Recreation Area (Pickart and Sawyer 1998). Several projects conducted at this area between 1995 and 1996 excavated European beachgrass and its rhizomes to a depth of 3.3 feet and then “capped” these sands with another 3.3 feet of sand (U.S. Forest Service 1995 a,b in Pickart and Sawyer 1998). However, apparently, the depth of excavation and burial was inconsistent, and moderate to substantial resprouting occurred the next spring (Pickart and Sawyer 1998). Another technique used by the Bureau of Land Management (BLM) at Coos Bay Shorelands involved use of a ripper to sever rhizomes and break up soil compaction, followed by hand removal of re-sprouts. While this appeared successful, the program managers had previously applied saltwater to the beachgrass plants, which, while not successful, may have weakened them and affected treatment results (Pickart and Sawyer 1998).

In recent years, the USFS has at Oregon Dunes used a combination of mechanical techniques, including “scalp and bury,” which removes 2.5 feet of the upper European beachgrass sand layer and buries it in pits approximately 6 to 8 feet in depth, or a scalp-and-push technique, which scalps vegetation to a 6- to 8 feet depth and then pushes materials onto beach for redistribution by wind and tidal action (USFS 2010). Many of these areas have been re-treated several times apparently (USFS 2010), which underlines the issue that European beachgrass does recolonize following this type of mechanical treatment.

In 2004-2005, California Department of Parks and Recreation (CDPR) conducted a pilot project experiment on mechanical removal at Little River State Beach in California. The project compared three different approaches to mechanical removal, including one that primarily involved excavators (Excavator), one that involved excavation by D8 or D850 bulldozers to a depth of 3 feet and burial of beachgrass with a 2-foot cap similar to the “scalp and bury” technique at Oregon Dunes (Dozer-Grade), and one that involved bulldozers with a rake attachment capable of pulling up rhizomes from 2 feet deep (Dozer-Rake; Transou et al. 2007). Re-treatment effort was 30 percent lower in the Dozer-Grade areas than in the other two treatments (Transou et al. 2007). Treatment results, combined with cost, led CDPR to conclude

that mechanical removal was the most effective beachgrass removal method (Transou et al. 2007). The success of this pilot project encouraged development of an 80-acre dune restoration project at Little River that used mechanical grading to remove beachgrass.

A number of restoration efforts along the Pacific Coast have used chemical control to treat European beachgrass. Chemical control originally involved almost exclusively use of glyphosate (Pickart and Sawyer 1998). Indeed, some early studies suggested that glyphosate might not have been very effective for removal unless applied at higher concentration rates (10%; Aptekar, *in prep.* in Pickart and Sawyer 1998). In Monterey, a 7% solution of glyphosate was successfully used to treat 10.4 acres at Sunset State Beach (Hyland and Holloran 2005). Initial experiments with a grass-specific product, Fusillade®, did not adequately control the invasive, so, later, the project area was first burned to eliminate thatch and stimulate regrowth before herbicide was applied (Hyland and Holloran 2005). Subsequent herbicide treatments occurred at 6 and 12 weeks later, with a total of five (5) follow-up treatments required to achieve 100% control (Hyland and Holloran 2005). By the sixth treatment, total herbicide used had dropped from 100 gallons to 2.5 gallons (Hyland and Holloran 2005).

Over time, many agencies have switched to using imazapyr with glyphosate for treatment, which appears to be more effective than just glyphosate and to require less retreatment. Importantly, lower concentrations of these chemicals can be used apparently without sacrificing efficacy: Glyphosate concentrations have dropped from 10% in those early projects to 1% in more recent restoration efforts (Pickart and Sawyer 1998), whereas imazapyr concentrations have remained fairly constant (1-1.5%). In addition, as noted earlier, some agencies have experimented with use of pre-treatment measures such as prescribed burning or mowing to remove beachgrass thatch and stimulate re-growth and improve uptake of herbicide (Hyland and Holloran 2005). Some agencies have reported that use of this particular herbicide combination has controlled almost 80- to 90% of the European beachgrass present with as little as one (1) to three (3) treatments.

Based on information presented at a recent North American Pacific coastal dune restoration symposium in January 2016, most land managers are using a combination of manual removal, mechanical removal, and herbicide treatment to eradicate invasives, with prescribed burning often used as a pre-treatment method for herbicide. Most of these projects relied on chemical treatment using some combination of imazapyr and glyphosate as the primary tool for initial treatment, but sometimes used mechanical and manual for treatment in certain areas or for re-treatment.

COASTAL DUNE RESTORATION AT POINT REYES NATIONAL SEASHORE

Since 2001, the Seashore has removed approximately 269 acres of non-native invasive plants from approximately 524 acres of coastal dunes.

Restoration at Abbotts Lagoon

Because of concerns about impacts of invasive plants on native dune habitat and threatened and endangered species, the Seashore began a concerted dune and coastal bluff restoration

process in 2001. Early on, park staff worked on removal of European beachgrass from the mouth of Abbotts Lagoon, and, starting in 2002, iceplant was removed from 100 acres of rocky coastal bluff near the Lighthouse. During the first few years of restoration at Abbotts, removal efforts focused on manually digging up European beachgrass, which has deep rhizomes that can extend anywhere from 3 to 12 feet or more (Figure 2). Because it was difficult to dig deeper than 1.5- to 3 feet manually, the beachgrass rapidly regrew. In 2007, one park staff member estimated that 20% of the treated area re-grew within as little as six months, and some areas required as many 15- 20 repeat treatments before control appeared to be achieved (Charles 2007, Peterson 2004).

These issues encouraged the Seashore to explore other approaches for removing at least European beachgrass. In 2004-2005, a 20-acre mechanical project was conducted in the same vicinity as the hand removal ones, but with more successful results: It appeared to be more practical – and perhaps more effective -- than hand removal ones, particularly for larger areas. Excavators were able to dig deeper and then bury and “cap” the rhizome-contaminated materials with clean sand. Because of this success, the park submitted a proposal to the federal government to fund a much more ambitious restoration project that would mechanically remove 132 net acres in an approximately 300 gross acre project area just south of Abbotts Lagoon (Figure 3).

The first phase of the Abbotts Lagoon Coastal Dune Restoration Project moved forward in early 2011. Unfortunately, bids for the project came back extremely high, so the project area was reduced to 80 net acres in an approximately 190-acre gross project area. The park focused the Phase I restoration efforts around the approximately 95-acre native dune area that was, at this point, completely surrounded by extensive stands of European beachgrass in both the foredunes and backdunes (Figure 3): the native dune supported the park’s largest population of Tidestrom’s lupine, as well as large populations of beach layia, curlyleaf monardella, and other rare plants. Hand removal of European beachgrass and iceplant was performed in the native dune in native dune and sensitive habitats such as wetlands and areas adjacent to wetlands.

While mechanical removal again appeared, at least initially, to be highly successful in eradicating European beachgrass similar to the earlier project, it was also highly expensive (\$25,000 to \$30,000/acre) due to the depth of excavation required (from 6- to 9 feet) to bury rhizome and biomass-contaminated material with a sufficient “cap” of clean sand (3 feet), so that buried materials could not re-sprout readily.

The Seashore wanted to build upon these seemingly very ecologically successful dune restoration efforts by expanding restoration southward, but it realized that it needed to shift its removal strategy to more cost-effective ones, such as those utilized by restoration projects conducted in some other California coastal state parks. Many of these state parks have used a combination of fire and herbicide to successfully treat these invasive species. As discussed earlier, the mixture of glyphosate with imazapyr has been particularly successful in treating almost 80- to 90% of the European beachgrass with as little as one to three treatments (Bill Maslach, CDPR, *pers. comm.*; Tim Hyland, CDPR, *pers. comm.*, Brendan O’Neill, CDPR, *pers. comm.*).

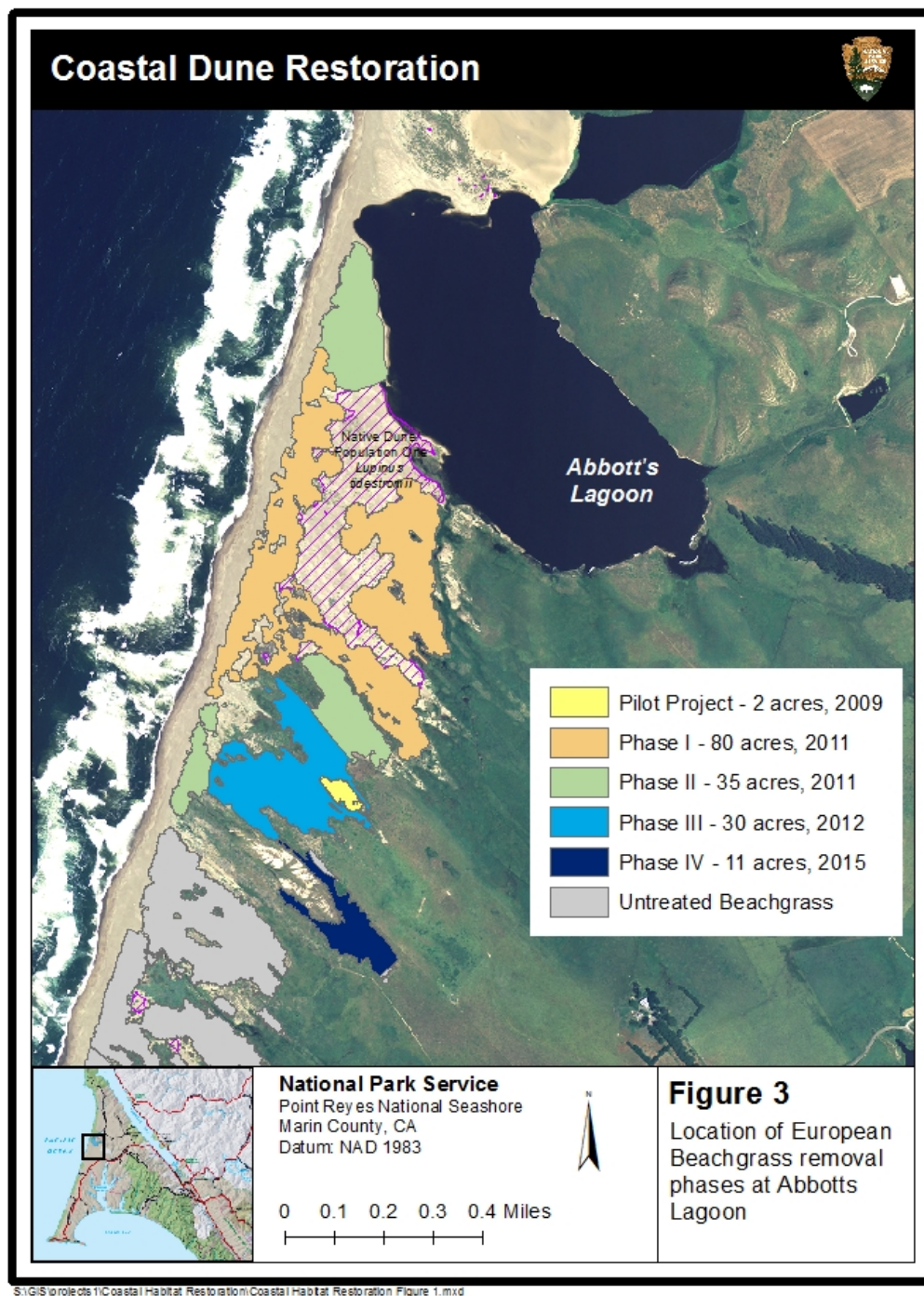


Figure 3. Location of initial restoration efforts between 2011 and 2015 at Abbotts Lagoon

To help develop the best methods for cost-effectively treating European beachgrass in the park, the Seashore initiated Phase II, which treated another approximately 31 acres using an experimental approach (Figure 3). The 31 acres included a portion of the area near the mouth of Abbotts Lagoon treated between 2001 and 2005 in which beachgrass had re-established due to insufficient re-treatment after 2005. Because various state park units have used different concentrations of glyphosate to successfully treat European beachgrass, two different concentrations of glyphosate (2% and 7%) were used with a uniform imazapyr concentration (1%), with treatments assigned randomly to plots in backdune and foredune stands to determine whether effectiveness varied based on physical location or age of stand. This pilot project was implemented in fall 2011, and the Seashore's early monitoring efforts showed that more than 95% of the beachgrass appeared to have been eradicated with just one treatment and that there appeared to be no differences due to glyphosate concentration. In addition, as part of this project, the dead aboveground biomass was physically removed from half of at least the southernmost Phase II plots in fall 2012 to determine whether biomass removal through mowing or shallow bulldozing improved and/or sped up colonization by native dune plant species: European beachgrass decomposes very slowly. Approximately 5.95 acres of backdune area were either mowed using a skid-steer loader with a rotary mower attachment or scraped with a bulldozer, while approximately 2.95 acres of foredune were mowed. Other methods used by state parks for reducing beachgrass biomass – and improving herbicide efficacy – such as controlled burning were not included as part of this experimental project due to the fact that more recent restoration efforts showed that treatment could be just as effective without burning (B. Maslach, *pers. comm.*), and burns can be difficult to conduct in the Point Reyes area due to highly variable climatic conditions.

Restoration efforts continued south of Abbotts Lagoon in the original project area in fall 2012 with herbicide treatment of another 30 acres of European beachgrass, with mechanical removal used in 25-foot buffers adjacent to ranchlands managed as organic pasture and hand removal used in wetlands (Figure 3). Based on the results from the Phase II project, which showed no noticeable differences between use of 2% versus 1% glyphosate, the Seashore elected to use a lower concentration of glyphosate (2%, AquaMaster®, currently marketed as Round-Up Pro®), as well as 1.5% imazapyr (Habitat®), 1% non-ionic vegetable oil surfactant (Competitor®), and 1.5% blue dye. Follow-up treatment of Phases I-III continued between 2012 and 2018. In addition, in 2015, the final 11 acres on Ammophila Mountain (Phase IV) were sprayed, finally completing at least one round of initial treatment of all European beachgrass (and iceplant) within the 255-acre Abbotts Lagoon Project Area. Follow-up treatment has been conducted annually since 2015.

Restoration at B Ranch North

In fall 2013, the park also initiated restoration efforts at B Ranch near the B Ranch North population, where Tidestrom's lupine numbers have decreased dramatically over the last decade. Due to this rapid decline, the Seashore requested funding from the USFWS to conduct an emergency project to halt the expansion of iceplant and remove plants that were directly displacing lupine from its habitat.

The first phase of the project involved hand removal of approximately 1.5 net acres of iceplant in 15.1 gross acres located in the B Ranch North population (Figure 4). Iceplant removal was not conducted at B Ranch South, even though it was included in compliance efforts. Follow-up treatment of iceplant removal at B Ranch North occurred in 2014, 2015, and 2016, with more opportunistic re-treatment in 2017 and 2018.

In September 2014, the Seashore conducted an initial treatment using herbicide of 14.9 net acres of European beachgrass in a 20.9 gross-acre project on the perimeter of the B Ranch North population (Figure 4). PRNS focused treatment efforts in a horseshoe-shaped project area around the Tidestrom's lupine, avoiding areas marked as "downwind" beachgrass areas (Figure 4). Herbicide treatment involved application of 2% glyphosate (Round-Up Custom®, formerly marketed as AquaMaster®), 1.0% imazapyr (Habitat®), 1.0% non-ionic vegetable oil surfactant (Competitor®), and 1.0% blue dye. Removal of European beachgrass along the perimeter should help to reduce seed predation pressure, as previously discussed studies by Washington University in St. Louis, Missouri, have demonstrated that seed predation is reduced dramatically at least 75 m from the nearest European beachgrass stand. Follow-up treatment was conducted annually between 2015 and 2018.

Restoration at ATT

In fall 2015, the park initiated restoration efforts at a third site, AT&T, which is directly south of the Abbotts Lagoon Coastal Dune Restoration Project Area. The project not only benefits federally listed species such as Tidestrom's lupine, beach layia, western snowy plover, and Myrtle's silverspot butterfly, but it expands the restoration envelope southward from Abbotts Lagoon, potentially creating almost 400 acres of unfragmented restored coastal dune habitat. The Seashore received funding from the National Park Service (Park Service), with additional funding raised by its non-profit organization, Point Reyes National Seashore Association (PRNSA) from National Fish and Wildlife Foundation for this project, which expanded treatment areas southward.

The first phase of the project in 2015 involved mechanical removal of approximately 2.4 acres of European beachgrass along a dune swale wetland within the AT&T project area where herbicide treatment could not occur (Figure 5). Another 13.5 gross acres of European beachgrass in the AT&T foredunes was mowed using a front-loader with a mower attachment to reduce the amount of biomass that would need to be treated the subsequent year (Figure 5).

In August-September 2016, contractors hired by the Seashore conducted an initial treatment using herbicide of 37.7 net acres of European beachgrass in a 68.0 gross-acre project focused on AT&T North or the north side dune swale wetland that roughly divides the project area in two (Figure 5). As with other projects, herbicide treatment involved application of 2% glyphosate (Round-Up Custom®, formerly marketed as AquaMaster®), 1.0% imazapyr (Habitat®), 1.0% non-ionic vegetable oil surfactant (Competitor®), and 1.0% blue dye. In 2016 and 2017, 4.79 net acres of dense iceplant in AT&T North was also treated with herbicide.

Initial treatment of European beachgrass areas south of the dune swale wetland occurred in summer 2017 and 2018, along with re-treatment of sprayed areas in AT&T-North. AT&T South Phase I in 2017 treated 15.75 net acres of European beachgrass intermixed with iceplant and

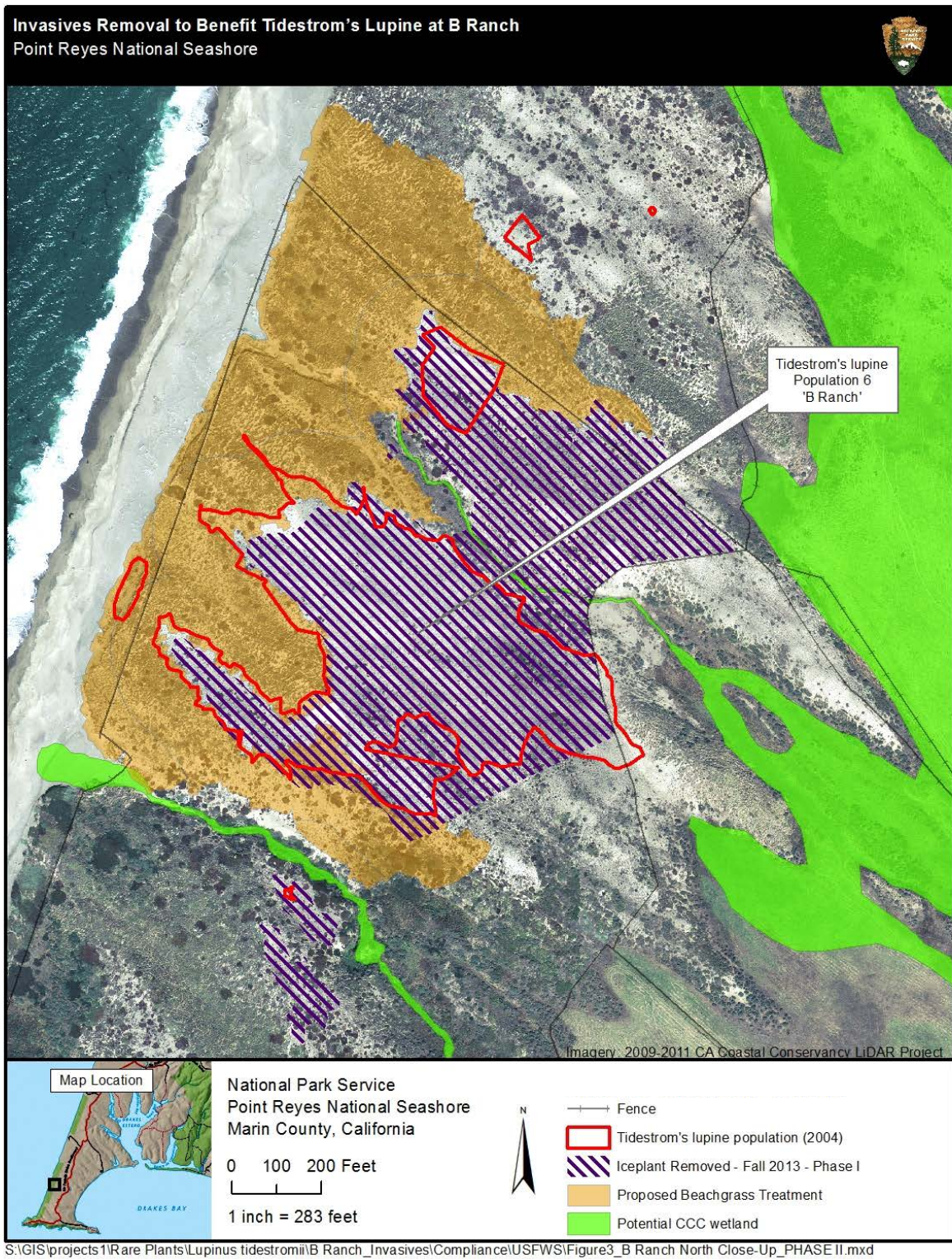


Figure 4. Location of initial restoration efforts at B Ranch North between 2013 and 2014

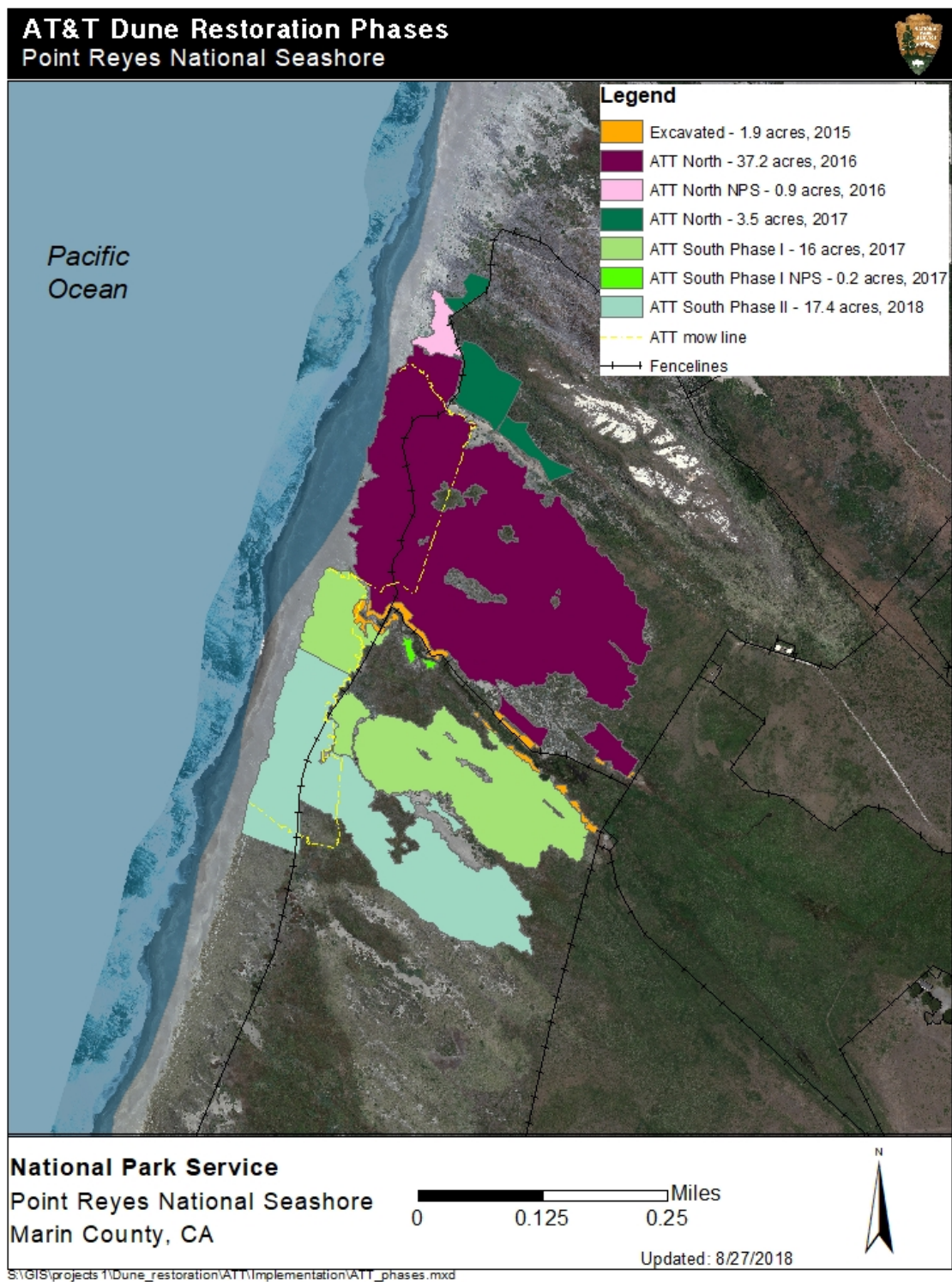


Figure 5. Location of restoration efforts at AT&T Coastal Dune Restoration Project Area (2015-2018)

0.3 net acres of standalone iceplant (Figure 5). Removal of beachgrass and iceplant was expanded further southward in 2018 with implementation of Phase II, which involved 16.6 acres of European beachgrass/iceplant treatment and 0.2 acres of scattered iceplant treatment (Figure 5). Most of the funding for AT&T South initial treatment and re-treatment came from National Fish and Wildlife Foundation through the grant awarded to PRNSA, with Park Service funding paying for AT&T North and a small portion of AT&T South (3.25 acres). Total acreage for area restored at AT&T North and AT&T South using both sources of funding was 77.2 acres of non-invasives in a 134-gross-acre coastal dune area. Re-treatment has been conducted annually.

MONITORING SUCCESS OF RESTORATION EFFORTS

A key, but often unperformed, component of restoration projects is monitoring the success of restoration efforts. Funding is often not available to perform monitoring other than effectiveness monitoring aimed at determining whether re-treatment is necessary. Equally important, however, to the future of these restoration efforts is determining how successful the restoration project has been in changing and, hopefully, improving physical and biological conditions and ecological health of park resources.

At the Seashore, most of the monitoring associated with dune restoration has involved assessment of vegetation cover, number or areal extent of endangered plants, and Western snowy plover nesting attempts. In addition, some projects have attempted to evaluate changes in sand movement topography related to removal of invasive plant species, most of which alter natural physical processes and conditions in coastal dunes.

Early Monitoring Efforts at Abbotts Lagoon

Vegetation and sand movement monitoring was performed as part of the 2001-2005 Abbotts Lagoon project. Twenty-two 10X10-m macroplots were installed in a number of the areas treated both manually and mechanically (15) and in native dune areas (7) to assess changes in vegetation cover, European beachgrass stem density, and bareground (Peterson et al. 2003). The number of beach layia was counted, and the perimeter of the Tidestrom's lupine areas was mapped (Peterson et al. 2003). This monitoring continued after the restoration project was completed by researchers at Washington University until 2008, when monitoring efforts were discontinued. Sand movement was monitored in at least the first two years by measuring the height of sand along 1 m tall posts marking a corner of each permanent vegetation plot.

While results were never formally analyzed, some preliminary analyses suggested that percent cover of native vegetation in treated areas had reached equivalency with native dune areas by three years post-restoration, but that native species richness and evenness was considerably lower (Kroiss, unpub. data). Sand moved as part of restoration efforts, but, in general, this movement appeared more related to redistribution of sand within the restoration area, creating more natural topography and contours (Peterson et al. 2003).

Monitoring Efforts at Abbotts Lagoon

In 2009, the Seashore initiated a pre- and post-restoration monitoring program for the Abbotts Lagoon Coastal Dune Restoration Project. The monitoring program ultimately incorporated vegetation monitoring of both dunes and adjacent grassland, rare plant censuses and sampling, and inland dune movement or “creep.” This report specifically focuses on dune vegetation monitoring.

Early on, staff decided against using the same vegetation monitoring protocol as was used for the earlier Abbotts Lagoon (and the Paleodunes) restoration projects. Subsampling larger macro-plots ultimately reduces the power of vegetation monitoring efforts as all the subsamples must be averaged for each macroplot, ultimately decreasing sample size while increasing sampling effort.

Staff reviewed the literature on monitoring for other coastal dune restoration projects throughout the world, but these projects have not been as extensively monitored as other types of vegetation systems. One approach used by several studies involved establishment of line transects perpendicular to the coastline, but this was considered too logistically complicated given the very steep and undulating pre-restoration terrain at Abbotts.

Similar to many other restoration projects, the monitoring program relies on a modified BACI sampling approach (Stewart-Oaten et al. 1986, Underwood 1991). “BACI” refers to monitoring of an “impact” (I) area both “before” (B) and “after” (A) an activity is implemented, with concurrent monitoring of “control” (C) areas. Based on this sampling design, we have been evaluating the Project Area (Abbotts Lagoon Project Area) before and after restoration was implemented and have been using native Dune Mat communities within the Project Area boundaries to better discern the effects of the restoration “impact” relative to natural variability.

Vegetation monitoring at Abbotts is being conducted using a different protocol based on establishment of 2X1 m plots throughout the Project Area, including both Phase I (mechanical removal) and Phase II/III (Herbicide Removal). The Project Area was divided into four sampling areas for the purpose of plot selection. Areas with European beachgrass were classified as “Invaded,” and areas still supporting native dune vegetation were classified as “Native Dune” or “Reference.” The Native/Reference and Non-Native/Treatment areas were then further divided into Foredune and Backdune strata for the purpose of random stratification. (Statistical analyses incorporated distance to the ocean as a covariate.) Within each sampling area, individual sampling plots were selected by placing a grid over a map of the Project Area. Each 50m by 50m grid square housed a single, computer-generated, random sampling plot location. A subset of the plot locations was randomly selected by hand within each sampling area. There are now 62 2X1-m monitoring plots in the 131-net-acre Project Area (Appendix A-1).

Selected UTMs were located in the field using a Trimble GeoXH handheld GPS unit with beacon. Plots were oriented in a north-south direction centered on the GPS coordinate. Marking consists of two pieces of rebar – one inside the southwest corner of the 2X1-meter sampling quadrat plot, and one inside the northeast corner – each with a copper tag indicating plot name and corner.

Cover of vegetation and other functional groups is assessed using a 2X1-m PVC quadrat separated into 10 X 10 cm grids that allows for estimation of cover using the point-intercept method, resulting in a maximum of 171 sampling points. The quadrat is positioned near the height of vegetation such that the grid plane is parallel to the general slope of the ground surface. A notepad is used to mark down the plot number, and then the quadrat and notepad are photographed from the south end. At each sampling point, plant species present (both alive and dead) are recorded, along with functional groups such as bareground and detritus. The presence of plant species at each sampling point is only recorded once, even if there are multiple point-intercept “hits” at that point. Standing dead biomass is considered a dead plant, but if the biomass falls to the ground as thatch or breaks down further into smaller organic matter, it is recorded as “detritus.” Monitoring is conducted between May and July to capture both annual and perennial species. If spring annuals are present that have senesced already, park staff make every effort to conduct “forensic” plant identification. Once cover assessment is finished, all species rooted within or crossing into the plot that were not detected during point-intercept sampling are recorded, and the number of individuals for rare plant species is counted.

For Native Dune, the full suite of 171 points is “read,” but after the first few years, the number of sampling points within Invaded Dunes was reduced by half, because of the length of time it was taking to conduct the monitoring effort: preliminary data analyses indicated that reducing the number of points did not alter data quality.

The 2X1 monitoring effort started in 2010, although some plots were added later in 2011 or 2012. A large percentage of the plots were installed prior to restoration, although a few Mechanical plots were added subsequently. The Seashore has been assessing Mechanical, Herbicide, and Native Dune Plots every year since installation: 2016 represented the five (5)-year mark since Phase I and Phase II efforts. Full monitoring was conducted again in 2017, but, in 2018, only herbicide-treatment plots were monitored.

As noted earlier, in 2015, the park initiated restoration of the final area designated for invasives removal under the original Abbotts Lagoon plan – *Ammophila* Mountain. Vegetation in this area had not been incorporated into the 2X1 m monitoring effort being conducted elsewhere in the Project Area, so, for this 13-acre area, photopoints were established within the treatment area and are being assessed annually.

Sand movement inland from Abbotts Lagoon has been assessed by walking the eastern edge of the dunes where it borders grassland/pasture. Sand burial was mapped using a hand-held GPS where there was 50% or greater cover by open, barren sand or a combination of sand and widely spaced Dune Mat- or Dune Scrub-characteristic vegetation compared to grasslands or wetlands (i.e., where there was <50% cover by grassland or wetland vegetation, with a relatively visible organic component on the surface). Starting in 2014, surveys focused specifically on inland sand migration into adjacent non-dune habitats and included mapping of the dune edge not only adjacent to mechanical removal areas, but adjacent to herbicide treatment areas, as well. In 2015, park staff started to notice that some “sand sheets” or large areas where sand had blown into former grassland or wetland areas were being colonized by vegetation: as vegetation could help to stabilize these sand sheets or “fingers,” they were also

mapped when vegetation cover exceeded 20%. Sand movement at Abbotts has been mapped through 2017; no monitoring was conducted in 2018. These results are discussed under Chapter 1 in “Dune Processes.”

Recent Monitoring Efforts at Abbotts Triangle

After monitoring efforts were developed and implemented at Abbotts Lagoon, the Seashore recognized that additional treatment would be necessary within the 2001-2005 Project Area at Abbotts Lagoon if the park wanted to create an expanse of unfragmented, restored coastal dune that stretched from the mouth of Abbotts Lagoon to Ammophila Mountain or the boundary with AT&T. Treatment at Abbotts Triangle – as this area became known – involved exclusively herbicide treatment. Park staff wanted to monitor vegetation community changes and success in eradicating European beachgrass, but, for reasons discussed earlier, did not want to use the macroplot methodology originally established to assess success of the earlier project.

However, park staff also had concerns about the applicability of the 2X1-m quadrat method for Dune Mat or sparsely vegetated native dune communities. The point intercept method is one of the least biased and most objective measures of cover. However, species with low cover values such as Dune Mat species are often not sampled efficiently because points rarely intersect the species of interest. Also, detecting small changes, especially at low cover values, is problematic. Finally, the commonly used method of lowering a pin into the vegetation is subject to the bias with which the observer angles and moves the pin. The pin method may also overestimate cover – especially for narrow-leaved species – due to the diameter of the pin (Elzinga et al. 1998).

For these reasons, park staff experimented with a different vegetation monitoring method for this area. This method involves using sampling vegetation species frequency at uniformly spaced distances along uniformly spaced “line” transects. Frequency measurements can be more sensitive to small changes at low cover values than other types of sampling. This makes frequency sampling good for measuring changes in small, sparsely-distributed species, as well as for detecting the recruitment of new species, including undesirable invaders (Elzinga et al. 1998).

Transect locations are established in GIS, and for sampling, a GPS unit is used to walk along each transect. These transect locations are temporary, have no markers, and are re-randomized each year. Transects run from northeast to southwest (314°), in the direction of prevailing winds, across the entire sampling area. The transect angle is meant to encompass any gradient that may occur due to wind and proximity to ocean. Transect locations are designed so that there is adequate interspersions between points along all transects (e.g., roughly 200 points within each sampling area) and between transects (with transects being located not more than twice or less than half of distance between sampling points positioned along the transect).

For this method, at each sampling point, both cover and frequency is assessed. Cover estimates are obtained using the point intercept method. Frequency sampling is conducted using nested quadrats of four sizes (10 cm, 50 cm, 100 cm, 200 cm). For frequency sampling, each quadrat is considered an individual sampling unit, assuming good interspersions. Sampling is scheduled to

take place after germination of all species has occurred, but before annuals, such as beach layia, have senesced and lost root-hold. The presence of any piece of vegetation that is attached to a rooted plant is considered present in the plot. For species that may not have fully leafed-out for the year, presence of leafless branches (possibly last-year's growth) is considered presence of that species in the plot.

At Abbotts Triangle, pre-restoration monitoring in summer 2011 involved sampling of 25 transects that had 15 m between transects and 20 m between quadrats along each transect for a total of 223 quadrats and 1,137 points. Frequency and cover monitoring took place between 2011 and 2015, except for 2014. The number of transects and quadrats/points was reduced slightly after 2011, but still averaged around 200.

In 2017, Park Service staff refined its monitoring methodology again. While adept at capturing species diversity within dunes, the frequency-transect method can be very time-consuming, particularly when repeated annually within a number of restoration areas. Also, the resulting data can be more cumbersome to analyze, as frequency of species cannot be easily collapsed into native/non-native or successional guilds as can be done with vegetation cover data. In 2017, park staff experimented with a new approach, transect-cover, that had been used already at B Ranch North. A similar grid of GIS transects is overlaid on the restoration area, but rather than collecting cover and frequency at each point, only vegetation cover is assessed. To improve the ability of this approach to discern changes in vegetation cover, the number of cover "points" per GPS transect line was increased relative to the frequency transects: every 15 feet versus every 60 feet. At each cover point, all plant species intercepted by the pointed toe of the boot were recorded, along with bareground and detritus. The sampling "point" or area on the toe of the boot was estimated to be approximately 2.5 cm wide.

Monitoring Efforts at B Ranch North

In conjunction with restoration efforts at B Ranch North, park staff developed a vegetation monitoring program to estimate percent cover and frequency of European beachgrass and iceplant and other plant species within treatment areas and, thereby, assess the effectiveness of restoration in eradicating these invasives and creating more habitat for native dune and rare plant species. Separate sampling areas were established for European beachgrass and iceplant treatment areas. In addition, due to rancher concerns about sand movement, park staff are also attempting to assess qualitatively change in vertical elevations within the Project Area and, thereby, infer how much sand movement may have occurred since restoration was implemented.

As B Ranch North contains a substantial amount of Dune Mat habitat and relatively moderate terrain conditions, the Seashore implemented frequency transects at this site, very similar to those developed for Abbotts Triangle. Sampling of transects first occurred for the iceplant area in 2013, which represents pre-restoration conditions. Sampling for the iceplant treatment area took place again in the summers between 2014 and 2016. Photopoints have been established within iceplant removal areas to track success of this project component in eliminating iceplant from within Population #6 (Appendix A-2).

Sampling for the European beachgrass area was initiated in 2014 and repeated in fall 2015, one year following initial treatment. For the first two years, the Seashore utilized the transect-frequency approach described earlier. In late summer 2016 (August), park staff moved to the transect-cover approach described under Abbotts Triangle monitoring above. To keep transect lengths reasonably approximate, the northern group of transects was divided into East and West.

To monitor potential inland movement of sand at B Ranch North in response to dune restoration efforts, park staff installed transects of rebar posts throughout the restoration area. Sand movement transects are 100 m apart and are aligned in the direction of the prevailing wind, approximately 315 degrees, in the areas iceplant and beachgrass treatment areas. The transects stretch from the westernmost edge of the beachgrass to the westernmost portions of the pasture areas. Seven (7) transects were installed in 2013, and sampling points along the transects are designated by letters, with “A” closest to the ocean. Sampling points with dune areas are rebar posts with yellow caps, while those in grazed areas were rebar that had white PVC pipes installed on top. Measurements are taken to the nearest 1/8th inch: measurements can easily vary by 1/2” depending on technique and side measured, even when the post is relatively straight and undisturbed. During monitoring, lengths of all “sides” of the rebar or PVC pipe are measured, and an average distance to ground is calculated. Following measurements, photographs are taken standing 2 m back from the post in the oceanward direction along the entire transect. Monitoring has these rebar posts have not been conducted in the past few years, because rebar posts either knocked down by cows (in areas open to cattle) or blown sideways by winds, greatly complicating measurements of sand deposition and any meaningful interpretation of results. A better system would need to be developed to more accurately determine inland movement of sands from dune restoration efforts.

Monitoring Efforts at AT&T

Restoration efforts at AT&T North – located directly south of the Abbotts Lagoon Project Area – started in 2015.

Based on past experiences with 2X1 m plots and frequency/cover transects, park staff elected to go with 2X1 m plots at AT&T, using sample size analyses derived from results at Abbotts Lagoon to determine how many plots might be needed to adequately characterize vegetation communities. Pre-restoration surveys indicated that most of the native vegetation stands at AT&T were primarily Dune Scrub with slightly varying species composition, as well as some Moist Meadow communities composed of rush (*Juncus*) spp.: Native Dune Mat only occurred in a few areas in the Project Area, and most of these had been invaded to some degree by iceplant, and, to a lesser extent, beachgrass. Based on results from Abbotts, the 2X1 m plots appear to function effectively in shrub- or large grass-dominated stands, but are less effective in more barren dune communities characterized by sparse vegetation and large expanses of sand. Also, the AT&T area is quite hilly, and the transect method is ill-suited to steep or sharply undulating topography, because it is difficult to maintain uniform distances between sampling points along the transects.

Separate sampling areas were established in AT&T for European beachgrass/iceplant treatment areas and native Dune Scrub communities. Pre-restoration plots were established the summer prior to treatment for each of the three phases (AT&T North/2016; AT&T South Phase I/2017; and AT&T South Phase II/2018). Using a random stratified sampling scheme within these two communities, eighteen (18) 2X1 m plots were established within Dune Scrub at AT&T North, and 20 plots were installed in European beachgrass/iceplant areas (Appendix A-3). Sampling at AT&T North in 2016 established pre-restoration conditions in both invaded European beachgrass communities and in adjacent stands of relatively intact native Dune Scrub, although a few plots were installed in areas that were mechanically restored along the *Alopecurus* Swale in 2015. Due to the low number of plots, these were not included in the pre-restoration data analysis. An additional 14 plots were set up in invaded areas in AT&T South Phase I, along with four (4) plots in native Dune Scrub (Appendix A-4). Similar to the vegetation monitoring protocol for Abbotts Lagoon, the plots were monumented with rebar in the opposite corners and marked with flagging tape: locations were recording using handheld GPS with beacon (Trimble GeoXH). Two (2) photos were taken of each plot – one directly of the plot and one including background landscape.

The AT&T South Phase I area included lightly invaded Dune Mat community that supports rare plant species such as Tidestrom's lupine and beach layia. This area was actually monitored using the GPS-transect method discussed earlier for Abbotts Triangle and B Ranch North.

In addition, 11 photopoints have been established within European beachgrass removal areas at AT&T North at high-elevation locations that provide a panoramic perspective of the restoration project, as well as eight (8) locations in AT&T South Phase I (Appendix A-5).

Starting in 2017, the eastern or inland sand edge along AT&T was GPSed using the same technique as described earlier for Abbotts Lagoon. These results are discussed under Chapter 1 in "Dune Processes."

Statistical Analysis

For 2X1-m plot vegetation, data analysis focused on comparison of cover and species richness-associated variables using General Linear Model (GLM) analyses (Minitab® Statistical Package v. 17 and 18). The analyses evaluated differences between 1) pre-restoration invaded areas (pre-treatment) vs. native dune areas (either Dune Mat or Dune Scrub) reference areas; 2) pre-restoration invaded areas (pre-treatment) vs. post-restoration invaded areas (post-treatment); and 3) post-restoration invaded areas (post-treatment) vs. post-restoration reference areas. Dependent variables included percent cover of native species, non-native species, primary invaders (European beachgrass/iceplant), dead primary invaders, detritus, bareground, secondary invaders, as well as non-cover data such as total species richness, native and non-native species richness, species diversity measures, and rare plant density. Summary statistics are presented with means or averages and standard errors ($\pm$ S.E.). Some rare plant population data is presented with estimated total number of plants with a 90% confidence interval (C.E.). In this document, data from GIS-based line transects is presented simply as means or averages.

Data were evaluated for normality and heterogeneity of variance. If these assumptions were not met, data were either transformed for analysis, or a non-parametric statistical technique

was used, such as Mood Median test (Minitab®) or Permutation testing (R Statistical Package 2013). In addition to comparative analysis of cover, diversity, and density-associated data, the park also evaluated changes in species composition among treatment types (treatment vs. control) between pre- and post-restoration using multivariate analysis procedures such as Non-metric multidimensional scaling (NMS), Multiple Response Permutation Procedure (MRPP), and Indicator Species Analysis (PcOrd® Statistical Package v. 6).

RESULTS

Pre-Restoration Conditions

Abbotts Lagoon: Prior to restoration, most of the restoration areas south of Abbotts supported largely non-native plant species, with non-native cover averaging $81.9 (\pm 4.1)\%$ and $98.1 (\pm 4.2)\%$ for mechanical removal and herbicide-treated areas, respectively (Figure 6; Table 1). These areas both differed significantly from the “reference” native dune areas, which supported only $4.8 (\pm 4.3)\%$ non-native cover, almost all of which was European sea-rocket (GLM, $df=2$, $F=116.4$, $P<0.0001$; Figure 6; Table 1). The number of non-native species in these native dune areas was extremely low, averaging only $0.13 (\pm 0.07)$ species, significantly less than invaded dunes (GLM, $df=2$, $F=101.0$, $P<0.0001$, all adj. $P<0.0001$; log-transformed; Figure 7; Table 1).

It is unclear why these native dune areas, which, in many cases, were bordered on all sides by European beachgrass, did not also become invaded (Figure 8). Historical aerial photographs taken since the 1940s show European beachgrass cover slowly increasing over the decades in the foredunes and backdunes areas adjacent to Abbotts Lagoon. Ultimately, build-up of the foredunes and backdunes due to European beachgrass stabilization led to native dunes being almost like a basin or valley in between the high-elevation beachgrass-dominated hills. Iceplant occurred in isolated patches throughout the native dunes, but these patches had not coalesced to create the dense, carpet-like mats that they had in other portions of the Seashore further south toward the Lighthouse. The opportunity for invasion was certainly there, as bareground accounted for $54.5 (\pm 5.4)\%$ of the cover in native dune areas (Figure 6; Table 1). It is possible that some type of soil biota acted to protect these areas from invasion, making conditions inhospitable for large-scale colonization or expansion of invasive species and, thereby, increasing resistance of the seemingly vulnerable sparsely vegetated dunes.



Figure 8: Aerial view of Abbotts Lagoon Project Area in 2002 prior to any restoration. Source: californiacoastline.org

Within invaded areas at Abbotts, European beachgrass accounted for most of the non-native cover, with mean values ranging from $78.5 (\pm 4.2)\%$ in herbicide-treated areas to $79.5 (\pm 3.9)\%$ in mechanical removal areas (LM Permutation, $df=2$, $P<0.0001$; Figure 6; Table 1). In mechanical removal areas, European beachgrass represented almost the only non-native species, as reflected by the fact that non-native species richness averaged only $1.8 (\pm 0.2)$ species/ $2m^2$ (Figure 7; Table 1). While beachgrass was also dominant in herbicide-treated areas, other non-native species occurred, as well, leading to significantly higher means

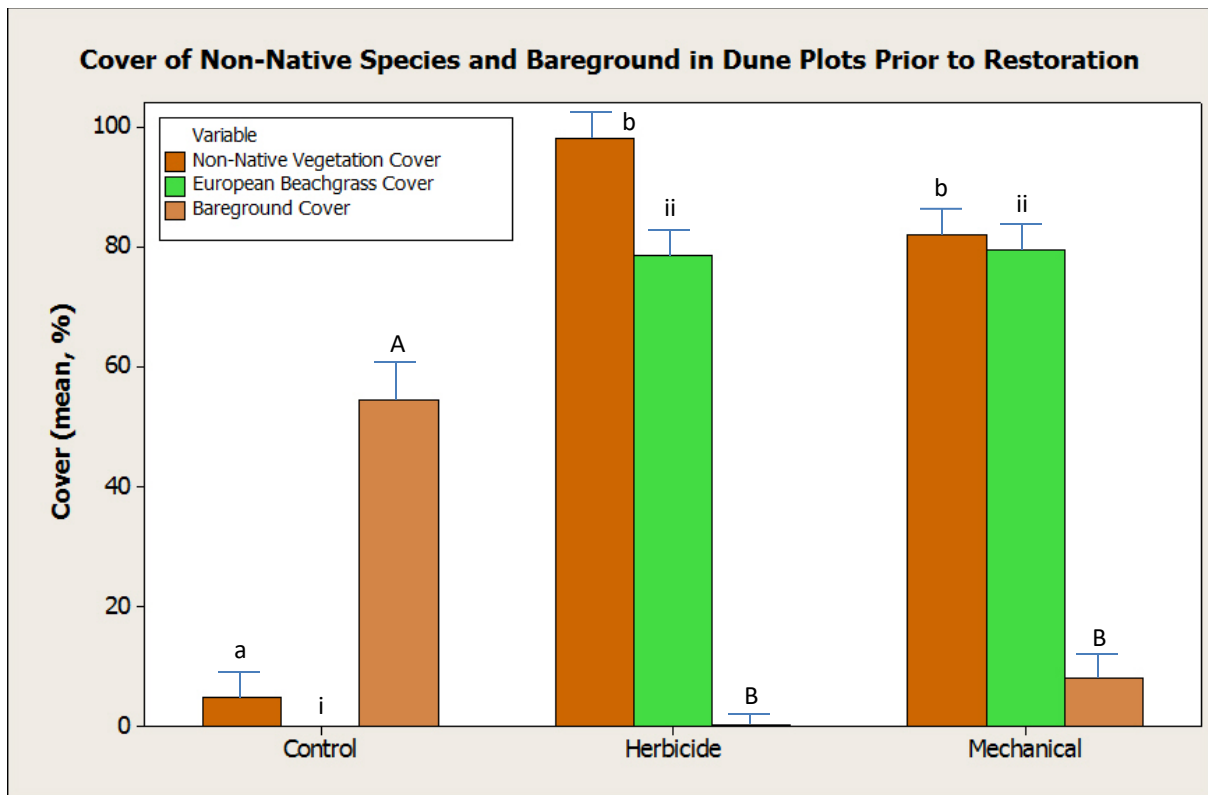


Figure 6. Mean percent cover of non-native species and bareground in pre-restoration plots at Abbotts Lagoon. Bars indicate standard error (S.E.) of the mean. Different letters/numerals indicate statistical differences between means.

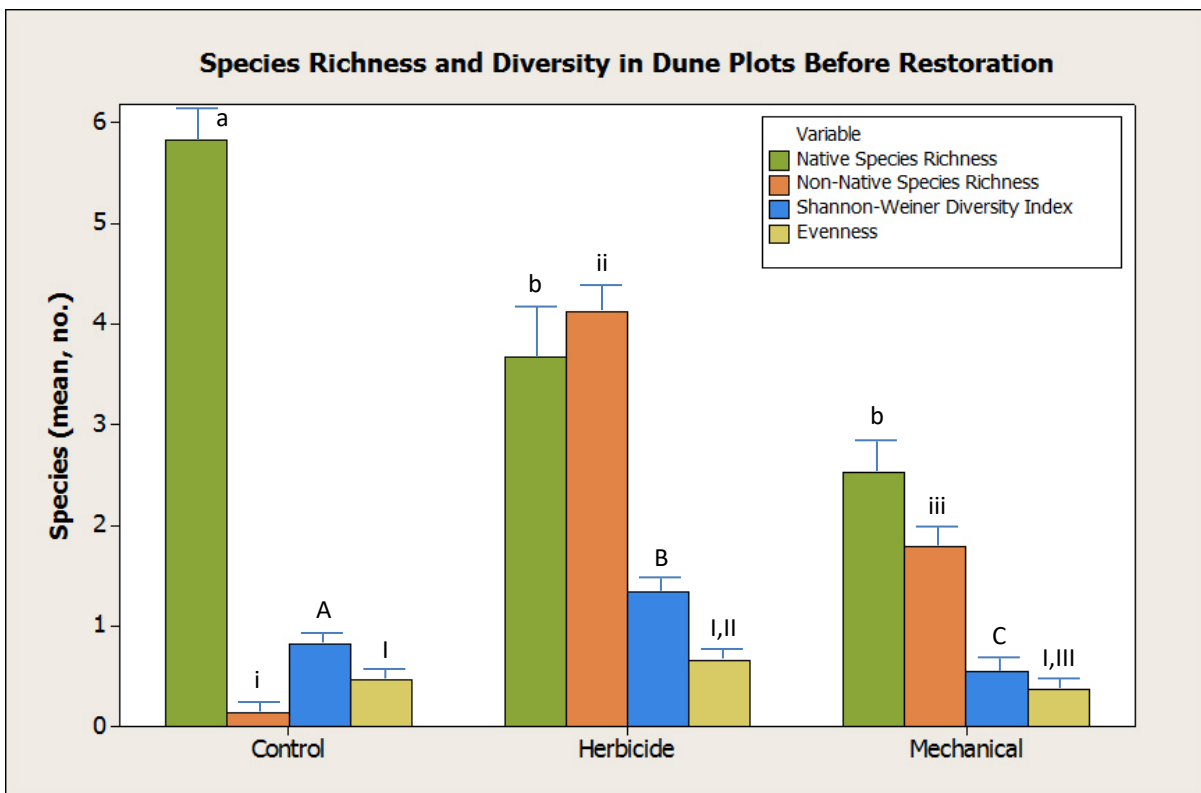


Figure 7. Mean percent cover of non-native species and bareground in pre-restoration plots at Abbotts Lagoon. Bars indicate standard error (S.E.) of the mean. Different letters/numbers indicate statistical differences between means.

for non-native species richness (4.1 ± 0.3 species/2m²; GLM, adj. $P < 0.0001$; log-transformed; Figure 7; Table 1).

Interestingly, despite the prevalence of non-native plants, native plants did occur in high numbers within all of the monitored areas, not just within the native dune plots. In fact, native plant species cover was higher in herbicide-treated areas (72.9 ± 1.5 %) than in mechanical removal areas (31.0 ± 7.9 %; GLM, $df=2$, $F=3.89$, $P=0.027$; adj. $P=0.02$; log-transformed) and, seemingly, native dune areas (39.9 ± 5.1 %), although this difference was not statistically significant ($P > 0.18$; Figure 9; Table 1). Native species richness was statistically equivalent between herbicide-treated (3.7 ± 0.4 species/2m²) and mechanical (2.5 ± 0.3 species/2m²) areas (GLM, $df=3$, $F=19.6$, $P < 0.0001$; adj. $P=0.83$; Figure 7; Table 1). This equivalence takes into account the fact that native species richness increased with distance from the ocean ($P=0.005$) and that a much higher percentage of the herbicide treatment areas were located in the backdunes. These numbers suggest that a few native plant species accounted for considerable native plant cover in herbicide treatment areas prior to restoration such as coyote brush and mock heather. Within restoration areas, most of the native plant species probably established in open or bareground patches within beachgrass stands -- particularly in mechanical removal areas where bareground averaged $8.1 (\pm 2.9)$ % cover -- or as an understory to the taller beachgrass plants -- particularly in herbicide-treated areas where bareground averaged only $0.3 (\pm 0.2)$ % (LM Permutation, $P < 0.0001$; Figure 6; Table 1).

Not surprisingly, the number of native plant species was highest in the native dune areas (5.8 ± 0.6 species; GLM, all adj. $P < 0.002$; Figure 7; Table 1). However, in terms of the Shannon-Weiner Diversity Index, herbicide treatment areas supported a higher plant diversity (1.3 ± 0.1) than either mechanical treatment (0.6 ± 0.1) or native dune areas (0.8 ± 0.1), probably due to the higher number of both native and non-native species (GLM, $df=2$, $F=8.89$, $P < 0.0001$; all adj. $P < 0.04$; Figure 7; Table 1). Evenness also appeared to differ, at least between mechanical and herbicide-treated areas (LM Permutation, $P=0.05$), with the latter having a more balanced distribution of species and cover (0.65 ± 0.03) than the former (0.36 ± 0.06), probably due to the fact that non-beachgrass species within mechanical removal areas were more patchy in distribution, while those in the herbicide-treated areas were interspersed within the beachgrass canopy (Figure 7; Table 1). Native dune plots fell intermediate in terms of evenness between herbicide-treated and mechanical removal areas -- $0.46 (\pm 0.05)$; Figure 7; Table 1).

Based on species cover, most of the native plants in restoration areas were species more characteristic of Dune Scrub (herbicide-treated areas) or Dune Scrub/late-successional dune communities (mechanical areas), while those in the native dune plots represented species considered more typical of mid- to primary successional dune communities (Chi-Square = 2372, $df = 6$, $P\text{-Value} < 0.0001$; Figure 9). Decades of establishment by European beachgrass undoubtedly altered the sand structure, increasing organic matter and nutrient content, which improved abiotic conditions for plants that would typically not be adapted to the harsh dune environment. As discussed earlier, research on bush lupine, a species native to California that occurred historically only in central and southern portions of the state, has shown that this species creates high levels of exchangeable ammonium and nitrate with leaf drop or shrub that

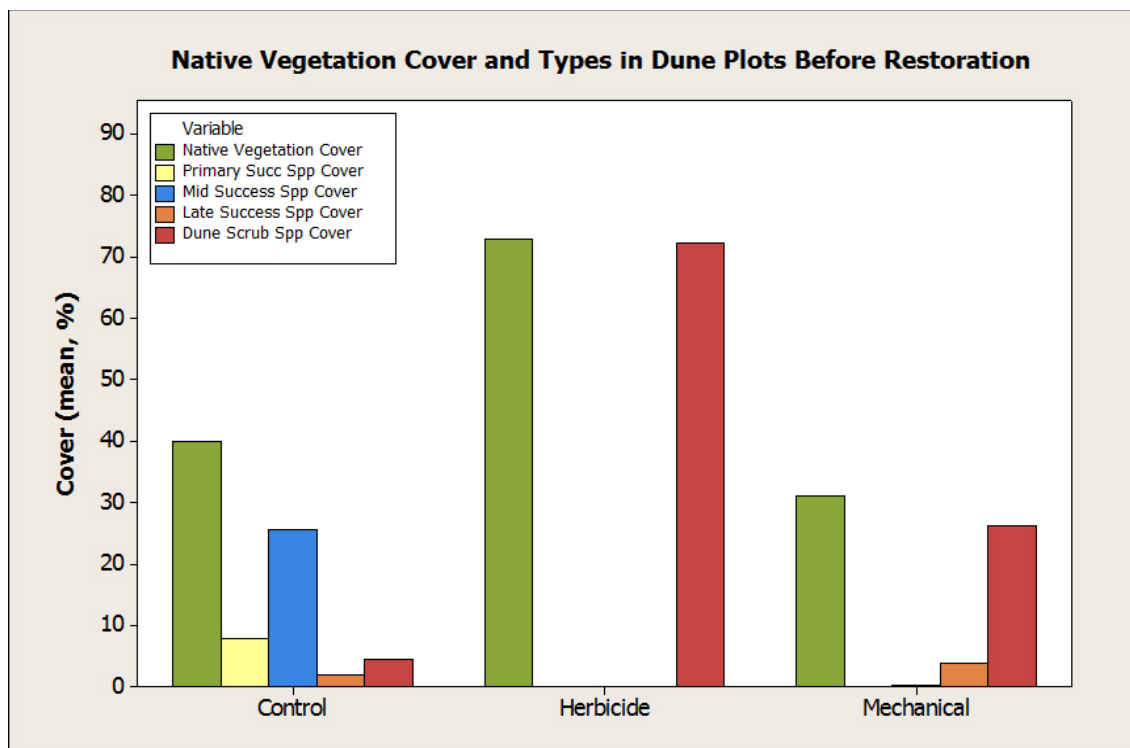


Figure 9. Mean percent cover of native dune species grouped by successional status in different treatment and control areas prior to restoration at Abbotts Lagoon.

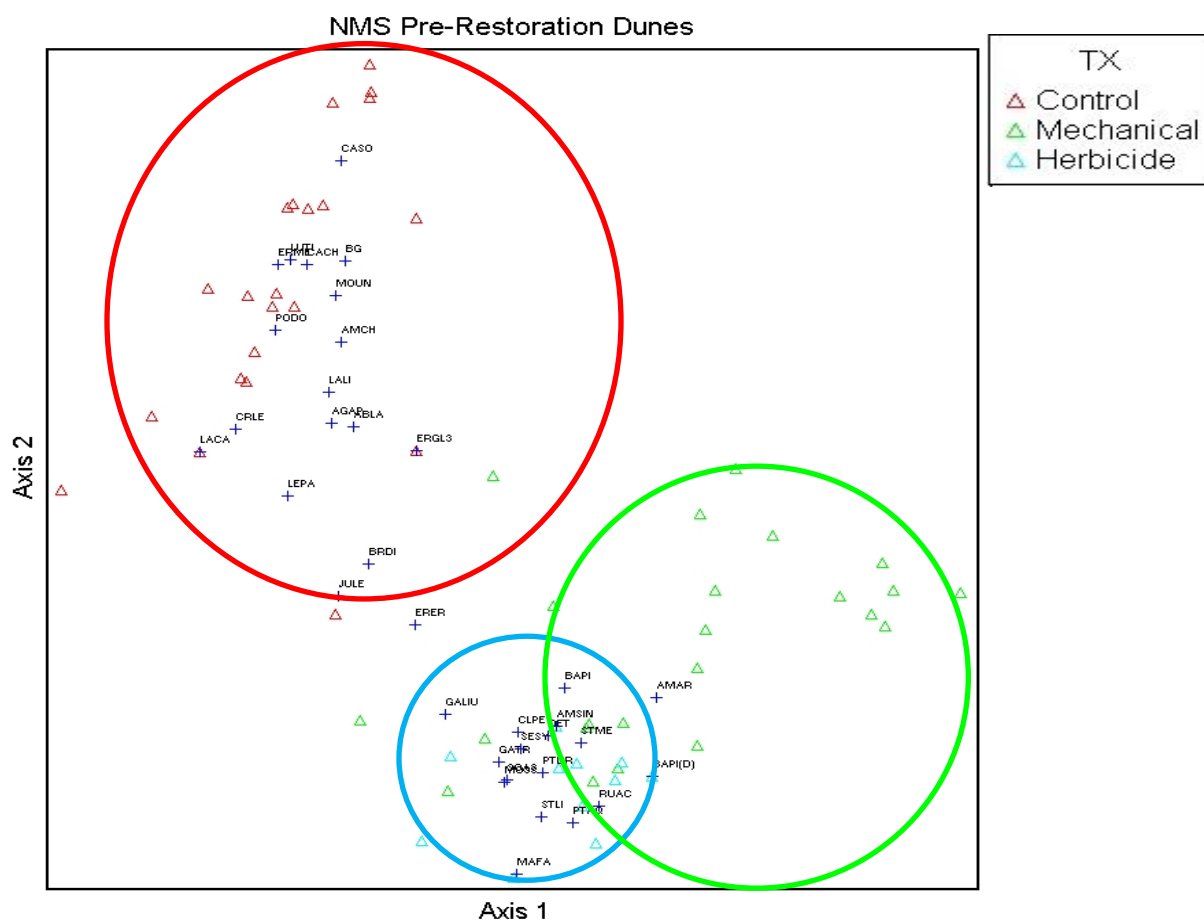


Figure 10. Non-metric multidimensional scaling multivariate analysis of Abbotts Lagoon pre-restoration vegetation communities

then becomes available to weedy introduced grasses and forbs (Maron and Connors 1996). Once established, these weedy annual exotics created a labile litter layer that completely turns over organic matter and nitrogen, favoring persistence of weedy annuals (Zink et al. 1995). While no similar studies have been conducted on beachgrass facilitating invasion by non-natives, it seems possible that a similar phenomenon occurs in beachgrass-dominated dunes.

In retrospect, then, it is difficult at Abbotts to compare change between treatment methods, because, inadvertently, due to construction and funding logistics, the treatment approach was unevenly allocated among dune areas that were dissimilar even prior to restoration, despite sharing the common characteristic of being labeled, “invaded.” Foredune and backdune areas near the lagoon that were treated mechanically were characterized by a more monotypic European beachgrass community with scattered pockets of bareground that provided habitat for a few other native and non-native plant species, most of which were characteristic of either Dune Scrub or, to a lesser extent, late successional plant associations. In contrast, herbicide treatment areas, which were largely located in the southern, more interior backdunes area (although some foredune areas were also treated), supported a more dense and diverse European beachgrass community that had a considerable number of other native and non-native plant species intermixed within the canopy, all of which would be considered Dune Scrub such as coyote brush and mock heather.

In between these invaded treatment areas occurred a relatively pristine native dune community characterized almost exclusively by large expanses of open sand and a diverse mix of native mid- and primary successional plant species, although late successional and Dune Scrub species were more prevalent in portions of the native dune further from the ocean. By 2011, scattered iceplant and, to a lesser extent, beachgrass plants had established, but had not gained sufficient foothold to rapidly expand throughout this seemingly very vulnerable area. Multivariate analyses using Non-Metric Multidimensional Scaling (NMS) clearly show the strong distinction between native dune and restoration areas along Axis 2, while Axis 1 points to perhaps a less dramatic difference between restoration areas that would be primarily treated using either mechanical or herbicide approaches (Figure 10).

B Ranch North: Little information of comparable detail exists on other dune systems in the Seashore. The Seashore has conducted some pre-restoration monitoring efforts in conjunction with some of its other dune restoration projects, including B Ranch North, Abbotts Triangle, and AT&T, although monitoring methods were not always equivalent to those used at Abbotts. These sites do not support anywhere near as extensive a native dune or Dune Mat community as can be found at Abbotts Lagoon, even if some of these areas would have evolved into Dune Mat had they not been invaded.

At B Ranch North, dunes invaded by European beachgrass and, to a lesser extent, iceplant ring a large Dune Mat area that also supports one of the Seashore’s Tidestrom’s lupine populations, as does the one at Abbotts (Figure 11). The large native dune area at B Ranch North, however, has been invaded by iceplant, although cover of this species was relatively sparse compared to other invaded dune areas: In 2-m square quadrats, frequency of iceplant within the B Ranch North Dune Mat area reached 61.9% of quadrats prior to restoration (2013), with occurrences typically composed of small patches of iceplant scattered throughout the Tidestrom’s lupine



Figure 11: Aerial view of B Ranch population surrounded by European beachgrass (Source: californiacoastline.org)

population. In contrast, frequency of European beachgrass in the native dune area was only 12.5%. There were very few individuals of European beachgrass: most of the European beachgrass colonization came from encroachment along the perimeter.

Indeed, the relatively uninvaded nature of this Dune Mat area is particularly surprising, considering that it is surrounded on almost all sides by European beachgrass (Figure 11). Again, as with the Abbotts native dune area, some type of soil biota may have

enabled this isolated native dune patch to resist invasion, particularly from European beachgrass, although, left unchecked, the scattered iceplant patches may have eventually coalesced and taken over this area. While not firmly entrenched in the Dune Mat area, iceplant, in fact, may have already been beginning subtly alter successional dynamics of the dunes through its well-documented ability to change soil conditions, tipping the successional dynamic toward later successional communities. Most of the Dune Mat area supported mid- to late-successional type native dune species (combined 70% frequency), while Tidestrom's lupine appears to prosper best under more primary successional conditions: primary successional species occurred in the Dune Mat area with only approximately 14% frequency.

The surrounding European beachgrass areas – which are largely foredunes with some backdune features running perpendicular to the shoreline – were characterized by moderate cover of European beachgrass (41%), with a considerable amount of bareground (22%) and, to a lesser degree, iceplant (8%). European beachgrass occurred in 95.6% of the 2-m frequency quadrats. Twenty-five (25) other, largely native species did co-exist with beachgrass and iceplant, with most of the species being either late successional (20% frequency) or Dune Scrub (13% frequency) ones that had colonized open bareground patches or establish underneath the beachgrass canopy. Primary successional species occurred at frequency rate of less than 4%: occurrence of these species in heavily invaded dunes may directly relate to the fact that most of the treatment area occurred in foredunes directly adjacent to the beach.

Abbotts Triangle: At the Abbotts Triangle foredunes -- located north of the larger, more recent Abbotts project and just south of the mouth of Abbotts Lagoon (Figure 12) -- European beachgrass cover was considerably lower (30%) than other invaded areas in the Seashore, probably because this area had been treated (and re-treated) between 2001 and 2005, but not



Figure 12: Aerial view of Abbotts Triangle in 2002 prior to any restoration. Source: californiacoastline.org

retreated subsequently. By 2011, beachgrass had reinvaded much of the area: the frequency of European beachgrass prior to re-treatment in 2011 was 73.4% of the 2-m frequency quadrats. However, there was a considerable amount of bareground remaining (54%), and native plants present appeared to be tilted more heavily toward primary successional species (21% frequency) and mid-successional (11%), probably because this area represents an active foredune area sandwiched between the ocean and Abbotts Lagoon.

AT&T North: Unlike the other restoration project areas, AT&T North primarily supported invaded dunes and pockets of native or lightly invaded Dune Scrub habitat: there was no remnant native Dune Mat in this project area. Average percent native and non-native cover, native species richness, and Shannon diversity indices all differed significantly between the invaded and the reference or native Dune Scrub plots (GLM, $df=1$, all $F>14.7$, all $p<0.001$; Table 2). Similar to other dune systems being restored at the Seashore, average percent non-native cover averaged $83.1 (\pm 5.5)\%$ in invaded plots, with European beachgrass accounting for 82.4% of the non-native species in invaded areas. In contrast, non-native cover in reference plots only averaged $17.3 (\pm 5.1)\%$ (Table 2). Despite differences in percent non-native cover, non-native species richness did not differ significantly between study areas, with medians being 2.5 in Dune Scrub and 2.0 in invaded plots (Mood Median, $df=1$, Chi-Square=2.1, $p=0.15$; Table 2).

Percent native cover ranged from an average of $120.9 (\pm 10.1)\%$ in native Dune Scrub to $55.3 (\pm 9.1)\%$ in European beachgrass stands (GLM, $df=1$, $F=21.87$, $P<0.0001$; Table 2). Average native species richness in Dune Scrub stands was also more than double that of beachgrass plots – $9.9 (\pm 0.7)$ versus $4.3 (\pm 0.5)$, respectively (GLM, $df=1$, $F=44.87$, $P<0.0001$; Table 2). Species abundance was relatively evenly distributed within both study areas (GLM, $df=1$, $F=2.6$, $p=0.12$), but, due probably to higher number of natives in reference plots, species diversity in general was higher in Dune Scrub areas than beachgrass ones (GLM, $df=1$, $F=14.7$, $p=0.001$), with indices averaging $1.6 (\pm 0.1)$ and $1.0 (\pm 0.1)$, respectively (Table 2).

Based on NMS results, plots that had already been restored mechanically showed strong differences from the Dune Scrub and Invaded plots (NMS, Final Stress=11.3, $R^2=0.75$, Figure 13). Almost all of the separation occurred along the first axis ($R^2=0.57$, GLM, $df=2$, $F=177.5$, $p<0.0001$), with mechanical plots on the left and Dune Scrub and invaded plots on the right: Dune Scrub and invaded plots also grouped pretty distinctly based on treatment type (Figure

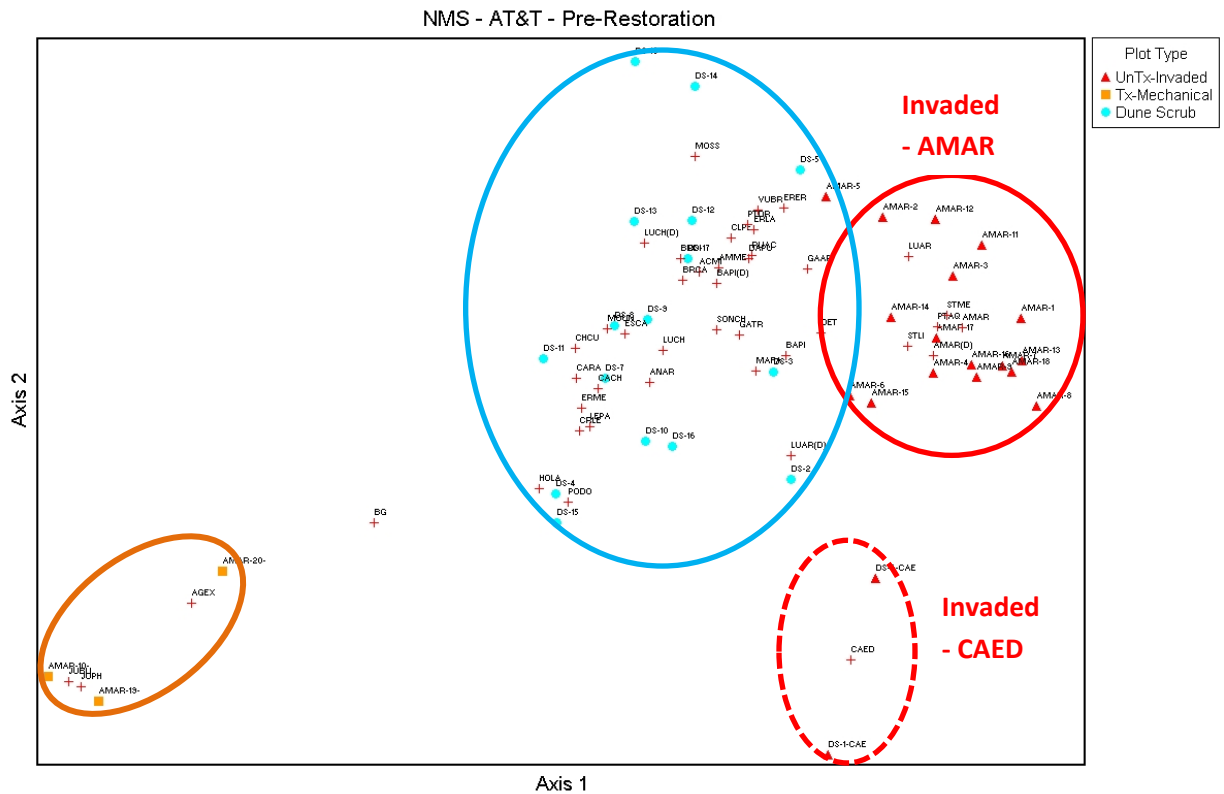


Figure 13. Non-metric multidimensional scaling multivariate analysis of AT&T North reference communities (Dune Scrub), invaded European beachgrass (AMAR) and iceplant (CAED) communities prior to treatment in 2016, as well as invaded communities following mechanical treatment in 2015.

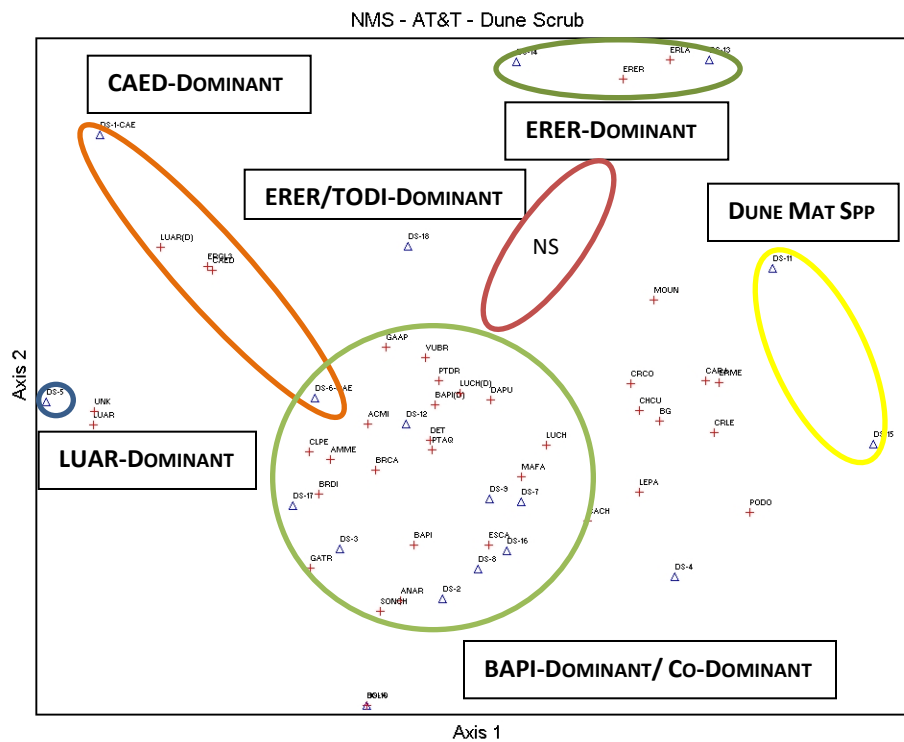


Figure 14. Non-metric multidimensional scaling multivariate analysis of AT&T North reference communities (Dune Scrub) in 2016, with communities dominated or co-dominated by particular shrub or herb species such as coyotebrush (BAPI), bush lupine (LUAR), mock heather (ERER), iceplant (CAED), poison oak (TODI), iceplant (CAED), or dune mat species. ERER/TODI community was not directly sampled (NS), but position estimated from species dominance.

13). European beachgrass represented the one species considered an indicator for invaded plots (Indicator Species Analysis, $p=0.0002$), while chamisso lupine, yarrow, California brome, riggut brome, miner's lettuce, sand mat, and seaside fiddleneck all distinguished Dune Scrub from other plots (all $p<0.05$). The mechanical plots differed strongly from the other types in terms of species assemblage largely because they primarily supported a number of seasonal wetland species, probably due to their proximity to the *Alopecurus* Swale, as well having more bareground or open sand. Very little bareground existed in either the Dune Scrub (10.1%) or the Invaded (1.1%) plots (Table 2).

The natural evolutionary trajectory for most of the areas at AT&T, with the possible exception of the foredunes, would probably be native Dune Scrub. From a geologic standpoint, Dune Scrub may represent another step in the evolutionary pathway between primary successional dune and terrestrial vegetation communities such coastal prairie, much of which at the Seashore is formed on historic dune soils. It could also represent a separate evolutionary pathway for higher elevation areas along the coast. Unlike its neighbor to the north (Abbotts), which has more low-lying dune areas, most of the elevations in AT&T fall towards the higher end of the elevation spectrum, so, therefore, it is likely that most of AT&T would naturally have been Dune Scrub if these areas had not been invaded by European beachgrass and iceplant.

Dune Scrub could be considered a variant of Coastal Scrub, although it tends to support more dune plant species. While NMS results would potentially suggest that Dune Scrub stands at AT&T North were a relatively homogenous vegetation community, distinct differences between stands were apparent in terms of shrub and understory herb/forb composition. Most of the stands were characterized by having coyote brush as a dominant or co-dominant shrub species with a number of common understory species, while a very few plots were dominated either by mock heather/coast buckwheat or bush lupine (Figure 14; NMS, Final Stress=9.6, $R^2=0.68$). At least one type of stand supported a dense thicket of mock heather and poison oak (*Toxicodendron diversilobum*), with coyote brush and wild cucumber as sub-dominants. (Cover of this plot was not directly sampled, but estimated visually). A few plots leaned slightly more toward Dune Mat in terms of species composition, supporting sand mat, coast cryptantha, curlyleaf monardella, bluff wallflower, spineflower (*Chorizanthe cuspidata*; CNPS List 1B), Douglas's bluegrass, Pacific wildrye, and beach primrose (Figure 14). Two plots were largely dominated by iceplant, with seaside daisy and dead bush lupine also present (Figure 14).

AT&T South: While topography in AT&T South is a little less extreme than in AT&T North, which has some very tall dune features and highly undulating topography, vegetation-related characteristics of these areas remained pretty similar. Percent cover of European beachgrass was significantly higher in the treatment group ($83.6 \pm 8.3\%$) than in the reference or native Dune Scrub areas in AT&T South ($0 \pm 0\%$; $t=-8.05$, $df=7.00$, $p<0.0001$; tangent transformed; Figure 15; Enders and Parsons 2018). Iceplant existed at low cover in both reference ($0.6 \pm 0.6\%$) and treatment plots ($3.6 \pm 3.6\%$; Figure 15). There was no significant difference in percent cover of iceplant between the two treatment groups ($t=-0.83$, $df=7.35$, $p=0.433$).

Native Dune Scrub communities in AT&T South supported high levels of both native ($120.9 \pm 33.1\%$) and non-native ($95.1 \pm 36.5\%$) plant cover with very little bareground ($2.0 \pm 2.0\%$) and relatively little rare plant cover ($7.0 \pm 4.3\%$; Figure 16). The high abundance of non-native

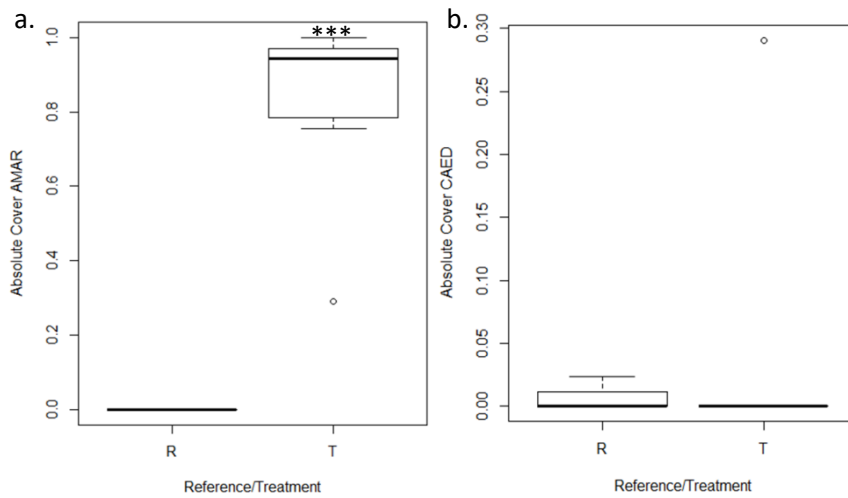


Figure 15. Box and whisker plots of AMAR (a) and CAED (b) cover in reference (R) and treatment (T) plots representing the median, interquartile range, data extremes, and outliers of the data. Asterisks above boxes indicate significant differences between means: '***' $p < .0001$; '**' $p < .01$; '*' $p < .05$; '.' $p < 0.1$.

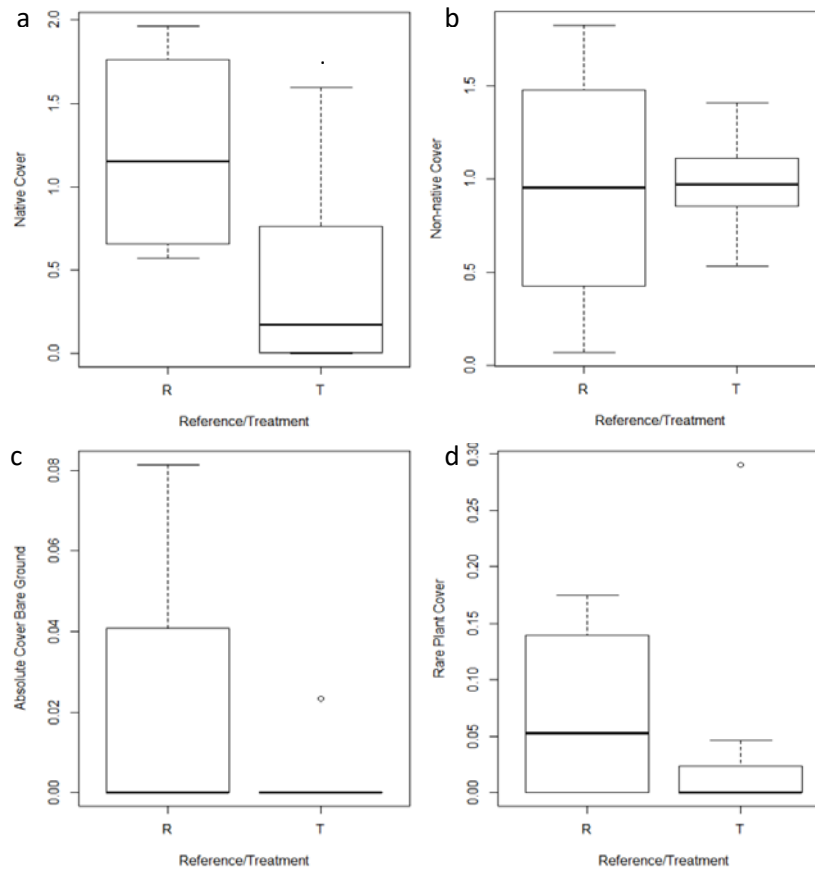


Figure 16. Box and whisker plots of native (a), non-native (b), bare ground (c) and rare plant (c) cover in reference (R) and treatment (T) plots representing the median, interquartile range, data extremes, and outliers of the data. Asterisks above boxes indicate significant differences between means: '***' $p < .0001$; '**' $p < .01$; '*' $p < .05$; '.' $p < 0.1$.

species in AT&T South native Dune Scrub plots would seemingly differentiate these communities to some degree from those in AT&T North, which had low non-native species cover. Prior to restoration, the treatment area supported less native cover ($43.5 \pm 20.7\%$; $t=2.07$, $df=10$, $p=0.065$), but not significantly different levels of non-native cover ($97.7 \pm 9.4\%$; $t=-0.07$, $df=3.40$, $p=0.949$), bareground ($0.3 \pm 0.3\%$; $t=0.85$, $df=3.12$, $p=0.456$), or rare plant cover ($4.2 \pm 3.6\%$; $t=0.49$, $df=7.21$, $p=0.636$; Figure 16).

Total species richness, native species richness, and non-native species richness were all significantly higher in the native Dune Scrub plots than in the treatment plots (total, $t=4.82$, $df=10$, $p=0.0001$; native, $t=3.95$, $df=10$, $p=0.003$; non-native, $t=2.62$, $df=10$, $p=0.026$). Total species richness in reference plots averaged $11.8 (\pm 1.0)$ species while treatment plots contained an average of $4.8 (\pm 0.9)$ species. Breaking down species richness into native and non-native richness, reference plots contained an average of $7.3 (\pm 1.0)$ native species and 4.5 ± 0.6 (S.E.) non-native species where treatment plots supported only $3.6 (\pm 0.7)$ native species and $2.1 (\pm 0.5)$ non-native species on average.

The two Study Areas were also significantly different in diversity and evenness. The reference plots supported a significantly higher Shannon-Weiner Diversity Index (1.9 ± 0.1) compared to the invaded community (0.7 ± 0.2 ; $t=5.35$, $df=8.45$, $p=0.001$). Likely driven by the high percent cover of European beachgrass in treatment plots, the invaded community also had significantly lower evenness (0.5 ± 0.1) than the native Dune Scrub one (0.8 ± 0.0 ; $t=2.51$, $df=7.61$, $p=0.038$).

The multivariate NMS analysis of differences in species composition between Study Areas shows separation between the reference and treatment plots along the first NMS axis ($t=-6.23$, $df=7.32$, $p=0.0004$), but no significant distinction between groups along the second axis ($t=-0.98$, $df=10$, $p=0.349$). The invaded plots aligned along the upper end of the first NMS axis correspond with the two target invasive species, European beachgrass and iceplant, and also a native species, bracken fern. The reference plots cluster on the lower end of the NMS Axis 1 along with native beach starwort, common bedstraw (*Galium aparine*), and non-native or questionably native species such as ripgut brome and bush lupine.

Post Restoration Conditions

Changes in Control or Reference Areas

Abbotts Lagoon: In theory, reference or control areas are ones that are not affected by the restoration action. However, as noted earlier, the native dune areas at Abbotts Lagoon were surrounded on almost all sides by invaded dunes, so the restoration did, inevitably, affect “control” areas. The almost instantaneous removal of European beachgrass, particularly on the foredune side, resulted in large-scale sand remobilization that buried portions of the native dune, with burial more evident in areas immediately to the foredunes and adjacent to distinct “lobes” or ridges of actively moving sand. In many instances, burial of both common and rare plant species was quite evident.

However, based on data from plots, there were almost no statistically significant changes in any of the cover or diversity-related variables between pre-restoration (2010) and post-restoration (2017; Paired t-test or Permutation test, all $P>0.20$; Table 1). In 2015, towards the tail end of

the drought, non-native cover did show a change relative to pre-restoration conditions, albeit a weakly significant one from a statistical standpoint (LM Permutation, $df=1$, $P=0.06$). However, differences between pre- and post-restoration in non-native cover were not evident the following year, and the winter of 2015-2016 was the first wet one since the mechanical restoration project was implemented in 2011, so potential shifts in vegetation cover might have been expected with the sharp increase in abiotic resources. By 2017, the only significant change between pre- and post-restoration in native dune areas was an increase in non-native species richness, with non-native species climbing from $0.13 (\pm 0.07)$ species/ $2m^2$ in 2010 to $0.39 \pm (0.10)$ species/ $2m^2$ in 2017 (Lm Permutation, $df=1$, $P=0.04$). Increases appeared to be largely due to occasional, sparse establishment of plots by European searocket, sow thistle (*Sonchus* spp.), and fescue brome (*Festuca bromoides*).

Vegetation monitoring may not have accurately captured change in native Dune Mat communities, however. Post-hoc analyses conducted on 2015 data from the reference or native Dune Mat plots suggested that power of these pre- and post-restoration comparisons may have been relatively weak, with β ranging from 0.35 to 0.58 where $\alpha=0.10$, with low number of replicates ($n=23$) and high variability between plots potentially obscuring changes. This conclusion seems to be borne out by the fact that, despite data suggesting that no change had occurred in control plots in 2016, park staff had by this time noticed an alarmingly explosion in the abundance of European searocket and, to a lesser extent, other invasive, non-native species such as New Zealand spinach, iceplant, and European beachgrass in fall 2016, which probably resulted from higher rainfall levels during the preceding winter. Certain areas supported almost 75% cover of European searocket, with native Dune Mat areas closest to the mechanically restored foredunes appearing to be more vulnerable (Figure 17). These areas are more subject to frequent disturbance from moving foredune sands and high winds than more inland areas. Of biggest concern was that monitoring conducted in June 2016 only documented four (4) plots with European searocket, all of which had cover totals below 2%: this includes mechanically restored areas, as well. The number of reference plots with searocket climbed to nine (9) species in 2017, however, in general, cover of non-native species remained low in 2017,



Figure 17: European searocket in Native Dunes at Abbotts Lagoon

as well. This may be at least partially due to the fact that the Seashore and PRNSA mounted an extensive searocket removal effort starting in late summer 2016, with more than 3,000 hours of manual removal of secondary invaders by staff and volunteers invested through 2018.

Despite being buffeted by deposition of remobilized sands and encroachment of secondary invaders, successional communities in native Dune Mat communities appeared to remain relatively stable following restoration (Figure 18). The percentage of species

assemblages reflecting primary successional habitats had not changed literally at all, accounting for 7.8% both pre- and post-restoration. However, mid-successional habitats had seemingly dropped slightly from 25.5% in 2010 to 19.4% in 2016. Changes in late successional and Dune Scrub communities were quite small, averaging almost less than a percentage point difference. By 2017, primary and mid-successional dune species most strongly characterized native Dune Mat areas. These species included yellow sand-verbena, coast dandelion (*Agoseris aparagoides*), beach evening primrose, beach morning glory, bluff wallflower, beach pea, curlyleaf monardella, Douglas' bluegrass, as well as bareground (Indicator Species Analysis, all $P < 0.05$).

A key factor in many of these communities is the amount of bareground, as native Dune Mat is characterized by high amounts of bareground, and many rare plant species tend to cluster in areas with either higher or lower amounts of bareground or open space. Cover of bareground changed imperceptibly over the 7-year period, ranging from 54.5% in 2010 to 57.8% in 2017 (Paired t-test, $df=22$, $t=-1.14$, $P=0.265$). While bareground did not necessarily increase, rare plants still appeared to benefit within native dune areas: In 2010, the vegetation monitoring plots supported an average of $2.4 (\pm 1.1)$ FE plants/ $2m^2$, but, by 2017, that average had seemingly increased to $18.0 (\pm 10.9)$ FE plants/ $2m^2$, although changes in median density were statistically insignificant (Mood Median test, $df=1$, Chi-Square=0.79, $P=0.38$), which suggests that density increases were largely focused in a few plots.

The existing population of Tidestrom's lupine in the native dune area has thrived since the restoration and has seemingly countered any of the initial adverse effects that remobilization of sands from adjacent mechanical removal areas and subsequent wind-driven re-deposition into lupine areas had on plant survival and reproduction. Population numbers at Abbotts Lagoon have remained fairly consistent before and after restoration, given statistical confidence intervals, with abundance estimated at 183,726 ($\pm 24,290$) in 2011, the year restoration was implemented. Since then, plant numbers (with 90% confidence intervals (C.I.) have ranged from 186,756 ($\pm 35,797$) in 2017 to as high as 283,378 ($\pm 65,534$) in 2015.

Given the wide confidence intervals, the change in estimated number of plants is not substantial enough to be characterized as an increase in population size, but certainly plant burial and other impacts from restoration do not appear to have effected a decrease in plant numbers. Indeed, it would appear that Tidestrom's lupine responds extremely well to disturbance activities, quickly re-establishing in newly scoured -- or deposited -- sands. Established populations may even be rejuvenated, to some degree, by moving sand. Given the species' predilection for disturbance, it is perhaps not surprising that European searocket also appears to respond positively to sand disturbance and quickly colonizes available open space (bareground), which may place in direct competition with other primary successional species such as Tidestrom's lupine. European searocket is often found co-occurring with Tidestrom's lupine, probably taking advantage of the nitrogen-fixing and microhabitat-building characteristics of this ecosystem pioneer (Figure 19).

In addition to improving habitat quality for Tidestrom's lupine, the restoration also appeared to succeed in one of its other objectives – reducing European beachgrass habitat for native deer mice that consume Tidestrom's lupine seed and, thereby, reduce reproductive success. Predation pressure plummeted almost immediately after the project was completed, most likely due to the instantaneous removal of 80 acres of beachgrass and other vegetation that directly bordered the existing Tidestrom's lupine population, as well as removal of a large number of the mice living in these areas. Prior to implementation, the mean percentage of racemes or flowering stalks of the Abbott's Lagoon Tidestrom's lupine population that suffered predation ranged from 44.8% to 93.9% across a six-year observation period (Pardini and Knight 2013; Figure 20). Predation rates were significantly lower in the first two full years following the restoration (7.8% in 2012 and 5.1% in 2013 (Pardini et al. 2018; Figure 20). Since then,



Figure 19. European searocket establishing on mounds created by Tidestrom's lupine

predation rates have ranged from 3.7% (2015) to 9.4% (2014; Pardini et al. 2018, Figure 20). Predations rates averaged 4.1% in 2018 (WU, unpub. data). It should be pointed out there was a dip in predation pressure in 2009-2010 (44.8 – 57.6%) before restoration relative to pre-2009 rates (66.5% to 93.9%), therefore, not all of the decline in predation rates may not have been necessarily related to restoration activities. Lower numbers of deer mice due to reductions in European beachgrass, however, may ultimately also benefit eradication of European searocket, the seeds of which are apparently also vulnerable to mouse predation (Knapp and DiTomaso 2005).

Changes in Mechanical Treatment Areas

Abbotts Lagoon: Restoration activities at Abbotts dramatically changed successional patterns of invaded dunes and seemingly in different ways. In mechanical removal areas, non-native and European beachgrass cover dropped dramatically following restoration, but, then, so did cover of all species (Table 1). Non-native plant cover plummeted from 81.9 ($\pm$ 4.1)% pre-restoration to 0.5 ($\pm$ 0.5)% in 2017 (LM Permutation, df=1, P=0.001; Appendix B-1, Table 1). Non-native species richness dropped from 1.8 species ($\pm$ 0.2) species/2m² pre-restoration to 0.08 in 2014, 0.04 in 2015, and 0.00 in 2016. In 2017, the number of non-native species actually climbed dramatically to 0.2 ($\pm$ 0.1) species in 2017, although this was still lower than pre-restoration conditions (LM Permutation, df=1, P=0.001; Table 1). Both non-native species richness (LM Permutation, df=1, P=0.02) and native species richness (GLM, df=1, F=3.95, P=0.05) values were strongly influenced by distance to the ocean. With the loss of all this non-native plant cover came a dramatic increase in the amount of bareground. The mean percent cover of bareground jumped from 8.2 ($\pm$ 2.9)% pre-restoration to 87.7 ($\pm$ 5.2)% in 2017 (LM Permutation, df=1, P=0.001; Appendix B-1, Table 1).

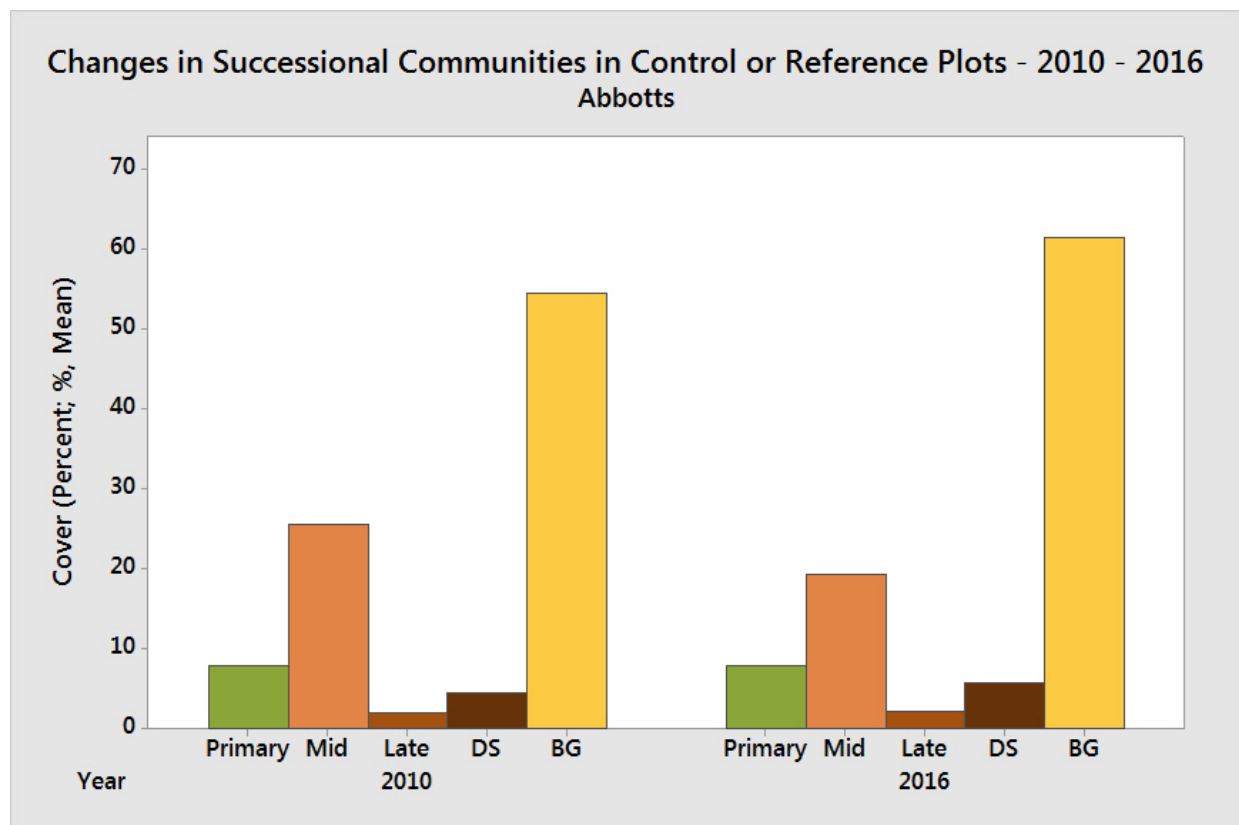


Figure 18. Changes in mean percent cover of native dune species grouped by successional status between pre- (2010) and post- (2016) restoration in control or reference plots at Abbotts Lagoon.

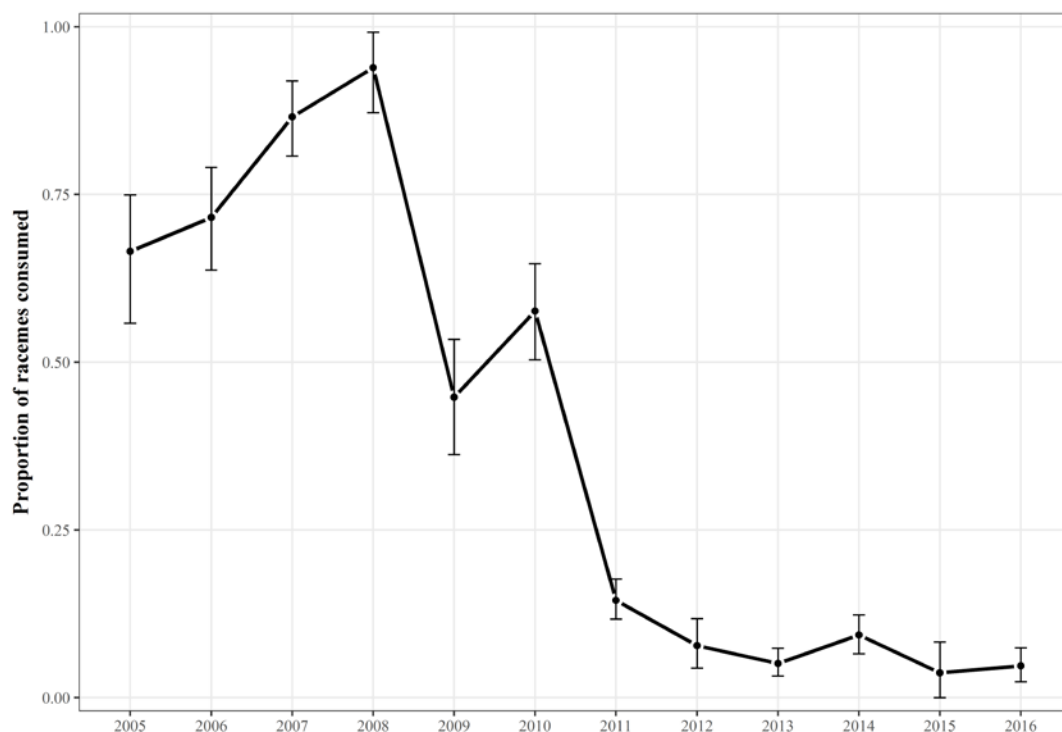


Figure 20. Proportion of racemes consumed of Tidestrom's lupine plants at Abbotts Lagoon Population #1 before restoration (2005 – 2010) and after restoration (2011-2016). Graph courtesy of Washington University.



Figure 21. Oblique aerial photograph of Phase I Abbotts immediately after restoration in 2009 (Photographer: Robert Campbell).

However, the increase in bareground did not immediately translate into more opportunities for native plants, which also dropped dramatically directly after restoration. Immediate post-construction aerial photography shows large expanses of open sand with relatively little vegetation cover except for the remnant native dune areas (Figure 21). A full three years after restoration, percent

native cover averaged $0.7 (\pm 0.5)\%$ in 2014 compared to $31.0 (\pm 8.0)\%$ during pre-restoration, when native plants were relegated to open areas between beachgrass stands or growing under the taller beachgrass canopy (Appendix B-1). Between 2015 and 2016, native plant cover climbed to between an average of 2 and 3%, but still remained pretty low. However, in 2017, native plant cover jumped to $13.1 (\pm 6.3)\%$, which, while still much lower than pre-restoration conditions (GLM, $df=1$, $F=3.17$, $P=0.08$), was a definitive improvement that was also evident on-site (Figure 22, Appendix B-1, Table 1).

In keeping with the drop in native plant cover immediately after mechanically flipping the dunes, native species richness also plummeted from $2.5 (\pm 0.3)$ species/ $2m^2$ pre-restoration to $0.04 (\pm 0.04)$ species/ $2m^2$ two years after restoration. By 2015 and 2016, the number of native species had climbed incrementally to average around 0.5 species. In 2017, the number almost doubled to $0.9 (\pm 0.3)$ species/ $2m^2$,



Figure 22. Diversity of native dune plant species in mechanically restored areas at Abbotts Lagoon in 2017.

which was still significantly lower than the number of natives in invaded areas prior to restoration, but only weakly so (GLM, $df=1$, $F=19.7$, $P<0.0001$; Table 1).

The Shannon Diversity Index strongly reflects the abrupt change in vegetation conditions, with indices declining from 0.55 (± 0.10) pre-restoration to between 0.03 and 0.07 through 2016. In 2017, the Diversity Index more than tripled to 0.18 (± 0.07), although it was still lower than values reported prior to mechanical removal (GLM, $df=1$, $F=13.71$, $P=0.001$; Table 1). Evenness showed a similar pattern, dropping from 0.36 (± 0.06) in 2010 to between 0.03 and 0.07 during 2014-2016. In 2017, evenness appeared to almost double to 0.12 (± 0.05), which, again, was still significantly lower than during pre-restoration (GLM, $df=1$, $F=9.49$, $P=0.003$).

The failure for native dune vegetation to re-colonize mechanically restored areas for many years immediately after restoration may have been due to many factors, including low soil nutrients or low soil biota in “flipped” soils or even presence of some parasitic microbe. A study conducted by a U.C. Berkeley student interning at the park in 2013 looked at plant establishment relative to several different soil conditions – native dune soils, mechanically treated or “flipped” soils, and iceplant-dominated soils with duff remaining and with duff largely removed (Winsemius 2013). Two rare plant species – beach layia (FE) and curlyleaf monardella (CNPS List 1B) – showed the highest germination rates in native dune soils (57.5-72.5%), followed by flipped soils (30- 52.5%), with no germination in either of the two iceplant soils (Winsemius 2013). However, flipped soils had lowest average biomass per species – more than 100% less than native dune soils, which led Winsemius to conclude that it was possible that flipped soils play host to some type of parasitic microbe that inhibits germination and, to a greater degree, plant growth (Winsemius 2013).

Certainly, the very dry winters and springs between 2011 and 2015 undoubtedly played a large role in discouraging vegetation establishment in mechanical treatment areas during the early years following project implementation. During these years, California was subjected to some very dry winters, with rainfall in many cases 6 inches below normal. While rainfall totals for 2012 and 2013 appeared average, both years were considered much drier than normal from a statewide perspective. Conditions became even drier in 2014 and 2015. By summer 2015, California was considered to be in the fourth – and seemingly driest – year of a four-year drought. While rainfall totals seemingly increased in 2015 relative to 2014, most of this increase came from torrential rains in December 2014, and the late winter/early spring was one of the driest periods on record, with the state of California declaring a Drought State of Emergency in January 2015.

Dry winters are often accompanied by high spring winds in coastal dune systems (Baye 2008). These winds compound the prohibitive lack of winter moisture by destabilizing dune systems through large-scale sand movement, where no vegetation currently exists to bind sands in-place, and by increasing moisture loss of both sand and the few hardy plants able to colonize to evaporation/evapotranspiration. Maximum wind speed exceeded 30 mph anywhere between 3.6% (Water Year (WY) 2013) and 7.9 % (WY 2011) of the time during and immediately after restoration. Conditions calmed considerably in 2014, with winds exceeding 30 mph only 1.6 % (WY 2014), only to pick up again somewhat in 2015 (WY 2015; 3.0%).

The effects of these strong spring winds were quite evident. Sand now free of stabilizing root systems began to move *en masse* across the restored landscape, burying portions of wetland and native dune features within the dune system. It also continued to move inland, extending in discrete, finger-like projections into the adjacent coastal grassland and wetlands. As discussed earlier in this document, an analysis of historic aerial imagery showed that, even with expansion of European beachgrass in the adjacent dunes after being planted probably in the 1940s, sand and dunes continued to encroach episodically into the adjacent grassland and that the spatial pattern of inland migration was identical to that occurring since 2011, with the prevailing northwesterly winds creating long, narrow ridge-like features aligned parallel with the wind direction (Johnson et al. 2013b). This large-scale movement of sand following dune restoration in 2011 essentially flattened the foredune and backdune features – in fact, so much so that the so-called native dune “basin” became seemingly higher in elevation than the surrounding restored areas. However, no topographic cross-section surveys have been conducted of the Abbotts area since 2012, immediately after restoration, so there is no concrete evidence of this large-scale morphological change.

Ironically, one of the few species that flourished in this highly disturbed environment was Tidestrom’s lupine. A study conducted by Pardini and Knight in 2012 in the existing dune habitat at Abbotts showed that Tidestrom’s lupine (and beach layia) were more frequently found in early successional microhabitats within the site, which are characterized high amounts of bare ground (84.4%; Pardini and Knight 2013). In addition, early successional habitats were more likely to contain newly recruited individuals or seedlings of Tidestrom’s lupine (Pardini and Knight 2013). However, within existing non-restored habitats, early successional habitats only accounted for 15% of the 145 plots in the existing native dune area (Pardini and Knight 2013). The remaining 85% of plots were either mid-successional (60% bare ground) or late successional (18.8% bare ground), which was characterized by much higher vegetation cover (Pardini and Knight 2013). Interestingly, these results are in marked contrast to earlier hypotheses on habitat preferences of Tidestrom’s lupine: The USFWS Recovery Plan characterized Tidestrom’s lupine as a later successional species that was intolerant of sand burial (USFWS 1998).

Restoration converted the restoration area from heavily invaded Dune Scrub and late successional plant communities to one dominated by primary successional plants and, of course, extensive amounts of bareground (Chi-Square, $df=4$, $P<0.0001$). By flipping the dunes, mechanical removal essentially re-started the successional “clock,” creating a very large, unfragmented expanse of primary (or even pre-primary) successional habitat. This primordial state appeared to dominate until possibly 2015, when data showed a small increase in the percentage of primary, mid-, and late successional dune vegetation communities (Appendix B-1). This trend was even more evident by 2017 (Appendix B-1). By that year, mechanical restoration areas supported a very different mix of species that had moved much closer since even 2016 to native dune areas in terms of species composition (NMS, final Stress = 14.4, 250 iter.; $R^2=0.857$, Figure 23), with treatment areas not being significantly different from native Dune Mat areas on Axis 2 of the NMS, which accounted for 67% of the variation between sites (Tukey multi-comparison, $P>0.05$; NMS GLM, $df=2$, $F=63.57$, $P<0.0001$).

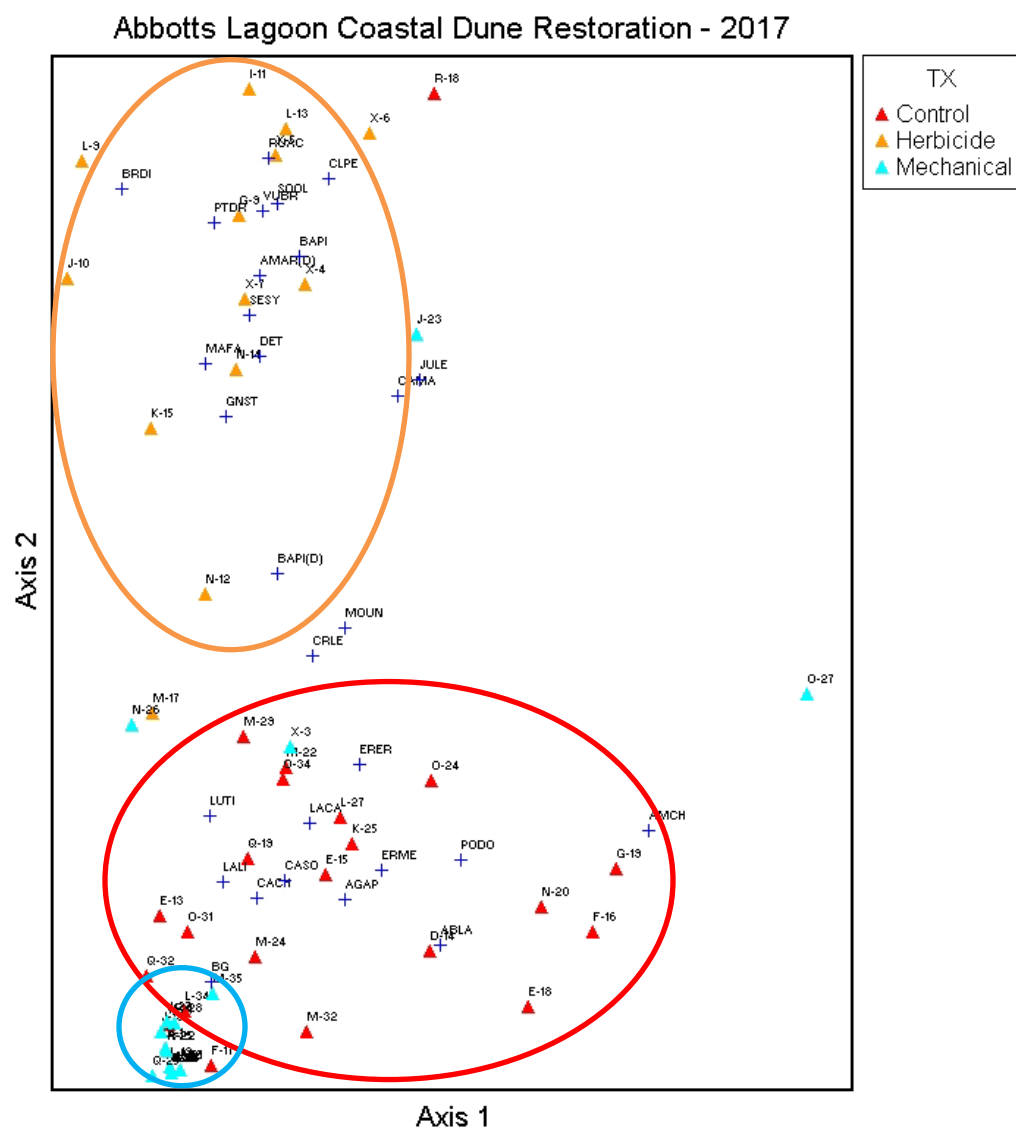


Figure 23. Non-metric multidimensional scaling multivariate analysis of Abbotts Lagoon post-restoration vegetation communities in 2017.

Species Analysis, all $P < 0.05$).

Tidestrom's lupine responded almost immediately once construction was completed. Thousands of Tidestrom's lupine started germinated even in fall 2011 (Johnson et al. 2012). These plants persisted through a relatively dry winter and scouring winds in the spring, and, by February 2012, approximately 15,884 plants had established in 15.8 of the 80 acres that were restored (Johnson et al. 2012). At least 25% of these plants actually flowered and set seed, setting the stage for potentially even more plants to establish in the second year after restoration (Johnson et al. 2012). In 2013, numbers of the perennial rare plant were estimated at approximately 20,552 individuals: because populations were estimated using census, it is unclear whether population increased or whether the increase is due to differences in counting technique (NPS, unpub. data). While pre-restoration topsoils ended up almost 3 to 5 feet or more below the new soil surface -- which represented dune sands that had probably been buried for decades -- Tidestrom's lupine seed apparently persisted even in these deeper dune soils, as the almost immediate germination following restoration suggests that germinants must come from buried seed bank, not blowing or movement of seed from adjacent native dune areas.

Despite the drought, numbers of Tidestrom's lupine continued to increase in the newly restored Abbotts Lagoon area. By 2014, Tidestrom's lupine had established in most portions of the mechanically restored dunes and in several of the areas treated with herbicide, and the population appeared to have skyrocketed in terms of plant establishment, with an estimated 74,111 individuals counted (NPS, unpub. data). In 2015, use of sampling rather than census methods estimated the Tidestrom's lupine population in the restored dunes at 118,340 ($\pm 31,706$) plants in a 45.1-acre area, with another 5,924 plants counted outside the sampling area (NPS, unpub. data). In some areas, Tidestrom's lupine has become so dense that it has created almost a "carpet," with little bareground between plants, and this vegetative mat appears to be naturally stabilizing some of these mechanically restored dune areas (Figure 24). As referenced earlier, lupine also is an ecosystem engineer, building mounds with sand accretion that appear to create hospitable micro-environments or micro-sites for other less hardy species.



Figure 24. Tidestrom's lupine creates dense carpets within mechanically restored backdunes at Abbotts.

Beach evening-primrose and, less frequently, beach pea, Douglas' bluegrass, bluff wallflower, and chamisso lupine often established in the backdunes either behind or on top of mounds created by Tidestrom's lupine plants, as well as European searocket.

Tidestrom's lupine was not necessarily the only species that took advantage of these early successional habitats. Some other native species did colonize the mechanically restored

foredunes, including beach pea, sand-verbena, beach-bur, and beach morning glory. In the backdunes, bush lupine was fairly common, with approximately 250 shrubs having established in the northern backdunes in 2012 (PRNS, unpub. data).

Tidestrom's lupine, beach pea, and bush lupine – all early colonizers -- are members of the pea family, which are nitrogen-fixing and can, therefore, act as pioneering species to increase nitrogen resource within soils for other non-nitrogen-fixing species. In fact, as discussed earlier, past research has shown that bush lupine – another member of the Fabaceae family -- fertilize sandy soil with nitrogen-rich litter, perhaps excessively so, such that areas with bush lupine had twice the vegetative biomass as those without bush lupine, but only half the plant diversity (Maron and Connors 1996). In addition, the small to moderate- sized mounds created by Tidestrom's lupine may also shelter plants from high coastal winds: most of the other plant species appear to establish in the leeward side of the mounds (L. Parsons, NPS, *pers. obs.*).

Another rare species also characterized as favoring early successional habitats is beach layia. Interestingly, beach layia did not respond as strongly to this surfeit of primary successional dune, at least immediately after restoration. In the early years, the plant was only observed on the margins of the mechanical removal area. However, in 2014, 902 plants were counted within the restoration area, along with approximately 1,492 curlyleaf monardella. In 2015, monitoring shifted primarily to sampling: based on those results, beach layia numbers rose to 4,172 ($\pm 3,104$) within the 45-acre sampling area, with an additional 2,095 plants in census areas. Based on observations on its occurrence, curlyleaf monardella might be categorized more as a mid- to late-successional species: certainly, there did not appear to be strong overlap in 2015 with either Tidestrom's lupine or beach layia, because only 32 individuals of curlyleaf monardella were counted in the Tidestrom's lupine transects.

Some of the early trends in vegetation establishment – or lack of vegetation establishment -- were reversed in 2016 due, in large part, to the wetter winter conditions in 2015-2016. Rainfall in Marin County ended up being only slightly below average after several years of drought. While this change was not necessarily reflected in the 2X1 m plot data, vegetation staff definitely observed an increase in colonization of mechanically restored areas by both native and non-native species (Figure 22). In areas already supporting dense colonies of Tidestrom's lupine, more native plant species took advantage of lupine mounds and established on the mounds or on the more-sheltered leeward side of these mounds. Native dune plant species abundance also appeared higher in other areas such as the foredunes, where establishment was not necessarily associated with the presence of lupine mounds. This trend continued in 2016-2017 and 2017-2018, which were even wetter than 2015-2016.

Unfortunately, as with the reference or native Dune Mat areas, the most abundant colonizer in 2016 appeared to be European searocket, which spread rapidly in the more heavily wind-disturbed areas such as the foredunes and so-called "Sand Alley," a linear feature oriented perpendicular to the shoreline where winds and sand movement are often highest. Searocket individuals also favored establishment in or around lupine mounds, often becoming so dense that they literally completely blanketed the lupine plants, although, in general, the smothered lupines did not appear to die, at least immediately, as a result (Figure 25). Searocket abundance decreased in the mechanically restored backdunes just south of Abbotts Lagoon

where Tidestrom's lupine abundance is actually highest, so this would suggest that disturbance is the key factor to establishment of searocket, with presence of nutrient-enhanced lupine mounds more of an ancillary or opportunistic one.

Tidestrom's lupine did not appear to respond as favorably, at least initially, to the wet conditions. Transect sampling and census yielded estimates of 118,186 ($\pm 30,721$), roughly comparable numbers given the confidence intervals to the year prior (2015). In stark contrast, beach layia abundance seemingly skyrocketed in 2016, with transect sampling estimating numbers within most of the mechanically restored areas at 54,516 ($\pm 27,608$), a 90% increase in abundance. Numbers of curlyleaf monardella also appeared to increase, climbing to an estimated 19,337 individuals. As both beach layia and curlyleaf monardella are annuals, they may be better primed to take advantage of sudden changes in climatic conditions. The success of many rare annual species is closely tied to climate patterns (Higgins et al. 2000, Levine and Rees 2004, Levine et al. 2008).

While the two winters following 2015-2016 were seemingly drier, the temporary stasis in lupine population growth did not persist, so it is possible that lupines do better under drier conditions or simply have delayed response to the surfeit of abiotic resources associated with higher rainfall years. By 2017, the estimated numbers of lupines had jumped to 297,637 in a 52.0-acre area, although transect sampling estimates included some backdune areas that had been previously censused, so there was a shift in methodology. While, then, this increase may be an artifact of a change in sampling approach, abundance of lupine did jump dramatically again in 2018 to 575,450. The increase in beach layia that year was even more striking, skyrocketing to an estimated 4,795,310 ($\pm 1,977,670$). Only curlyleaf monardella numbers remained seemingly static during this period, remaining essentially flat with an estimated 16,975 plants. The failure of curlyleaf monardella to leverage additional rainfall into population growth may be due more to the fact that much of the mechanically restored area at Abbots may still not provide optimal habitat for this mid- to late-successional species.



Figure 25. European searocket establishing on Tidestrom's lupine mound.

Changes in Large-Scale Manual Treatment Areas

B Ranch North: Based on vegetation monitoring, the frequency of iceplant in the lightly invaded Dune Mat area supporting Tidestrom's lupine dropped from 61.9 to 17.3 between 2013 and 2015 (Chi-Square, $p < 0.0001$). The frequency of dead iceplant, not surprisingly, increased during this period from 0.6 to 13.3 (Chi-Square, $p < 0.0001$). While these transects did not necessarily incorporate most of the European beachgrass treatment areas, there must have

been some overlap, as the cover of this invasive grass declined significantly, as well, from a frequency of 20.8 to 11.0 (Chi-Square, $p=0.02$). In 2016, park staff noticed a resurgence in the abundance of iceplant, primarily seedlings, probably due to the wetter winter conditions, with frequency increasing to 37.9, which was still a significant decrease (Chi-Square, $p<0.0001$). The scattered stands of beachgrass within this sampling area, however, continued to decrease in frequency to 2.3 (Chi-Square, $p<0.0001$) despite changes in climatic conditions, possibly because of annual re-treatment efforts.

In terms of native dune plant species, three (3) native dune species appeared to respond to restoration. At least one late successional species, Heerman's lotus, increased in abundance, climbing in frequency from 11.9 in 2013 to 36.2 in 2016 (Chi-Square, $p<0.0001$). Beach primrose, a mid- to late-successional species, also climbed slightly in terms of frequency, increasing from 76.8 in 2013 to 96.1 in 2016. Both beach layia and curlyleaf monardella -- two rare dune plant species -- dropped in frequency the first year after restoration (2014; all $p < 0.006$), but appeared to rebound somewhat in Year 2 (2015). Beach layia declined from a frequency of 14.3 in 2013 to 4.8 in 2014. In 2015, frequency doubled to 8.7 and then jumped considerably in 2016 to 23.6, a significant increase from its pre-restoration abundance (Chi-Square, $p=0.02$).

In 2017, the park decided to shift monitoring efforts to focus more on cover than frequency, because frequency results were hard to interpret. Results from GPS transect monitoring in 2017 was compared with cover results from earlier monitoring efforts, which included both frequency and cover, although the number of sampling points for cover was considerably less in prior years (2013-2016). The graphic in Appendix B-2 shows five years of cover data for the once lightly invaded Dune Mat area. Results from 2017 would suggest much lower primary invasives cover (iceplant), as well as lower cover of bareground and detritus relative to earlier years with a strong increase in native plant species cover, but results may be strongly influenced by the much higher number of sampling points in 2017 (517 points) than between 2013 and 2016 (168-174 points; Appendix B-2).

Most of the increase in native plant species cover in 2017 appears to come from species considered characteristic of mid- and late-successional communities, while those representative of primary successional communities remained low in abundance (Appendix B-2). Even if iceplant is fully removed, native dune vegetation communities may not be able to be fully restored. As was discussed earlier in this document, iceplant not only alters soil properties (D'Antonio and Mahall 1991, de la Peña et al. 2010, Santoro 2011, Vilà et al. 2006, Novoa et al. 2012), but water availability (D'Antonio and Haubensak 1998). Iceplant leaves have been found to have high levels of tannin and antibacterial compounds (van der Watt and Pretorius 2001) that reduce litter decomposition rates (Vilà et al. 2006) and leach into soils, acidifying them and thereby reducing nitrate, calcium, and magnesium availability (D'Antonio and Haubensak 1998, D'Antonio and Mahall, unpub. *in* Conser and Connor 2009). Soil acidification occurs naturally with dune succession, but iceplant invasion speeds up this process (D'Antonio and Mahall, unpub. *in* Conser and Connor 2009).

These soil changes may preclude re-establishment of certain types of native dune vegetation communities, such as primary or mid-successional ones, and ultimately favor later successional

ones or even Dune Scrub. Studies conducted in Spain suggested that success of iceplant removal efforts could be compromised by lingering legacy effects (Novoa et al. 2012). While some legacy effects reverted within a year to pre-restoration conditions, others did not, with establishment of diverse native vegetation communities constrained in the interim by competition from opportunistic ruderal species (Novoa et al. 2013). In another study of restored dunes in Spain, removal of iceplant caused a significant increase in species richness, particularly of annual plants, but not necessarily in native species cover or diversity (Andreu et al. 2010). Similar preliminary results were found with an iceplant removal project near the Seashore's Lighthouse initially conducted in 2008, with manual removal seemingly increasing percent cover of exotic forbs and grasses (Hamingson and Voeller 2011). As was shown in Winsemius's 2013 study, even when iceplant duff was removed from the soil surface, neither beach layia or curlyleaf monardella germinated in the former iceplant soils, although the greenhouse component indicated higher growth in soils without duff. One factor that may bode well for this site is the fact that manual removal was conducted before iceplant had completely taken over the area and potentially irreversibly changed the soil abiotic and biotic conditions.

One of the primary objectives of this project focused on improving habitat conditions for a very imperiled population of Tidestrom's lupine. The frequency of Tidestrom's lupine within the B Ranch North population boundaries seemingly did not change significantly in the period 2013 - 2016 (Chi-Square, $p=0.25$), although the trend in frequency appeared to be one of decline. Based on census data collected between 2012 and 2018 and population sampling estimates from 2001-2011, plant numbers within Population 6 have declined precipitously over the last 10 years. Plant numbers have dropped from an estimated high of 12,050 in 2004 to less than 2,000 plants between 2011 and 2013. Numbers climbed slightly to an estimated 2,186 in 2014 due partly to seedling recruitment. However, numbers plummeted to only 706 plants in 2015 and slid even lower to 583 in 2016 despite a wetter winter. As numbers were low in many lupine populations in 2015, the precipitous decline in 2015 appeared linked more to severe spring drought conditions and related biotic impacts such as increased herbivory on remaining plants than to any direct or indirect impact from restoration. However, the fact that the population did not rebound with higher rainfall conditions in 2016 suggests that factors influencing viability of this population may be more complicated than for rare annual species.

As was discussed in relation to the Abbotts Lagoon population, it may also be that the benefits of increased rainfall and more abiotic resources simply take longer to materialize in terms of boosting population numbers and viability. Estimated plant numbers did double to 1,086 in 2017 (WU, unpub. data) and remained roughly equivalent in 2018 (1,146; NPS, unpub. data). Some of the annual variation in numbers for this short-lived perennial species must be ascribed to sampling error as different people are typically conducting the counts each year, with some consistency built in through oversight by park or WU staff. However, this cannot account for all the variation. This population more so than many of the others appears heavily impacted by a combination of factors, including drought stress, herbivory, low recruitment, and, perhaps most importantly, flower/seed/raceme predation by mice, rabbits, and insects.

Another more indirect measure of overall project success involves changes in predation rates on Tidestrom's lupine seed. While iceplant may not have provided much cover for mice, the larger project also included treatment of European beachgrass surrounding the Tidestrom's

lupine population, which, undoubtedly, supported high numbers of this native rodent. Prior to 2014, Washington University researchers found that the proportion of racemes consumed at B Ranch North averaged 70% (WU, unpub. data). Since 2014, predation rates have dropped minimally, averaging 58%, although rates were lower in 2018 (33.2%; WU, unpub. data).

As treatment efforts would take time to yield results in terms of decreasing mice densities, park staff attempted some emergency management measures such as caging plants: cages did reduce predation rates, but did not eliminate predation (Parsons and Ryan 2015, NPS, unpub.



Figure 26. Deer mouse caught scaling side of Tidestrom's lupine enclosure at B Ranch North

data). Mice were able to scale cages, no matter what remedial measures park staff took to thwart predators, including placing smooth metal flashing around cage tops and covering tops with mesh netting (Parsons and Ryan 2015, NPS, unpub. data; Figure 26). Despite the fact that cages were not wholly successful in excluding mice, predation rates were much higher on control plants with no protection, averaging from 35% in 2016 to 68% in 2015 (Parsons and Ryan 2015; NPS, unpub. data). In contrast, predation rates on caged plants averaged between 9% (2014) and 30% (2015).

So far, removal of European beachgrass (and iceplant) does not appear to have impacted deer mice – or Tidestrom's lupine -- populations substantially. European beachgrass treatment was initiated in 2014, so changes in mice densities may only occur after beachgrass plants decompose, which can take five years or more. However, it is also possible that mice have simply found alternate habitat in native shrubs or other persistent vegetation or that they remained, at least initially in the dense stands of standing dead beachgrass biomass. European beachgrass decomposes very slowly (>5 years), as does its rhizomes: persistence of the rhizomes even with eventual decomposition of aboveground biomass may maintain suitable nesting habitat for deer mice. Pitts and Barbour (1979) speculated that deer mice preferred European beachgrass and rush (*Juncus* spp.) for nesting habitat because of the surficially dense rooting patterns of these species and the fact that the dense aboveground cover kept the soil substrate cooler and more moist than native Dune Mat habitat. Large enough declines in rodent numbers to benefit Tidestrom's lupine reproductive success may not transpire unless there are drastic alterations in the landscape (such as the mechanical removal at Abbots that left little to no vegetation and probably little to no mice either) or some type of predator management. As noted earlier, this population appears impacted by a number of factors: even if mice populations decline, the Tidestrom's lupine population may still not be able to recover. Population Viability analyses conducted on 2008-2009 data by WU showed that, even if seed predation was removed, increases in population growth rate did not appear sufficient to ensure that population would become stable.

Changes in Herbicide Treatment Areas

Abbotts Lagoon: Herbicide treatment resulted in a somewhat different restored environment at the Abbotts Lagoon Project Area. Die-back of European beachgrass was every bit as dramatic in backdune treated areas as in mechanical ones. Percent cover of live beachgrass in restored Abbotts backdunes dropped to $0.1 (\pm 0.1)\%$ immediately after restoration in 2013, which represented more than a 100% drop from pre-restoration beachgrass invasion levels ($90.2 \pm 3.2\%$). These levels have remained extremely low since then, ranging from 0.1 to 1.0% and averaging 0.00 (± 0.00)% in 2017, a significant difference from percent cover of beachgrass prior to restoration (LM Permutation, $df=1$, $P=0.001$; Appendix B-3; Table 1). While herbicide application was as effective as mechanical in almost immediately eliminating beachgrass after treatment, one key difference between the two restoration approaches was that herbicide treatment left



Figure 27. Undecomposed European beachgrass at Abbotts Lagoon Project Area. Recently treated individuals are yellower.

large amounts of standing, dead biomass that did not decompose quickly (Figure 27). One to two years after treatment (2013), standing, dead biomass of beachgrass represented 62.2% of the cover in herbicide treatment plots. In 2015, three to four years after treatment, standing, dead biomass still accounted for 22.7% of the plot cover. However, by 2017 (5- to 6 years post-treatment), standing dead beachgrass only averaged $4.7 (\pm 3.0)\%$, which was still seemingly significantly different than pre-restoration conditions (Lm Perm, $df=1$, $P=0.07$). Similar patterns of slow decomposition have been observed in other systems such as at Bodega.

Once beachgrass started to decompose, the stalks often fell to the ground, leaving a thick thatch on the soil surface. Over time, this thatch would further break down into smaller pieces of dead organic material or “detritus.” Monitoring combined both the larger thatch and smaller, organic material into the detritus category. Because plants are continually dying and decomposing, even without treatment, detritus levels in thickly vegetated stands can be quite high: they averaged $97.2 (\pm 0.9)\%$ prior to restoration in herbicide treatment plots (Appendix B-3). Based on data collected, detritus cover peaked in 2013 (152.1%) and has been declining since, averaging $69.5 (\pm 5.9)\%$ in 2017 (Lm Perm, $df=1$, $P=0.001$; Appendix B-3).

What this means is that, while live European beachgrass was largely gone, the eradication of beachgrass did not immediately translate into new areas for native dune species to colonize

because of the persistence of standing dead biomass, thatch, and detritus. In stark contrast to mechanical removal, herbicide treatment did not dramatically increase the amount of bareground or open space. Prior to restoration, bareground in backdunes was at a premium, averaging only 0.2 ($\pm$ 0.2)% (Appendix B-3). Over the first few years after restoration, the percent cover of bareground slowly climbed to between 6.5 and 7.8% (2013-2015; Appendix B-3). In 2016, bareground more than doubled to 15.1% and averaged 11.3 ($\pm$ 0.4) % in 2017, which was dramatically more than existed when this area was heavily invaded (Lm Perm, df=1, P=0.002; Appendix B-3), but still less than adjacent mechanical restoration areas (average=87.7%).

In 2012, the park experimented with some post-treatment measures (mowing, scraping with bulldozer) in certain herbicide treatment areas to try and accelerate decomposition of European beachgrass. By 2016, there were visible differences between control areas (herbicide treatment, but no post-treatment) and post-treatment areas. Both post-herbicide treatments met one of the treatment goals – reduced dead European beachgrass cover – with the doze treatment being the more extreme of the two treatments, resulting in the lowest dead European beachgrass cover, non-native species cover, and the highest amount of bareground (Ender and Parsons 2016; Appendix C). However, the doze treatment also yielded the lowest species richness, evenness, and species diversity. The mow treatment also reduced dead standing European beachgrass, but was more similar to the control in terms of having higher non-native species cover, species richness, evenness, and diversity than doze plots. Native species cover did not differ among the three treatments.

Cover of both native and non-native species decreased significantly after restoration, even though native and other non-native species were not targeted for removal (Appendix B-3; Table 1). Non-native cover (European beachgrass plus other non-natives) dropped from 137.5 ($\pm$ 9.6)% pre-restoration to 12.8% in 2013, increasing somewhat over the next few years to 19.2% in 2016. The largest jump in non-native cover occurred in 2017, when it accounted for approximately 58.3 ($\pm$ 10.6)% of cover in the plots, which is still considerably lower than pre-restoration levels (GLM, df=1, F=28.17, P<0.0001; Appendix B-3). This sudden increase in non-native species other than beachgrass may be attributable to several factors, including a high rainfall year in 2015-2016 combined with faster decomposition of standing dead beachgrass and thatch following that, leaving more ground available for non-native colonization. European searocket – which quickly moved in on mechanical removal and native dune areas in 2016 – was not recorded in any of the herbicide-treated plots in 2017, however, backdune areas did seem to support increasing numbers over the years of weedy, non-native, annual species such as common groundsel (*Senecio sylvestris*), riggut brome, fescue brome, and sow thistle, as well as perennial species such as iceplant and New Zealand spinach. Cover of these secondary species accounted for 50.8% of the 58.3% cover of non-native species in 2017 (Appendix B-3).

This explosion in the abundance of secondary species within restored backdunes is supported by non-native species richness data. The number of non-native species dropped relative to pre-restoration conditions initially, averaging between 2.0 and 2.9 species/2m², but then climbed back up in 2017 (3.9 $\pm$ 0.5 species/2m²), reaching equivalence with pre-restoration levels (4.3 $\pm$ 0.2 species/2m²; GLM, df=1, F=0.38, P=0.55; Table 1). Interestingly, richness of non-native

species showed a weakly significant, negative relationship with distance from the ocean in herbicide treatment plots (GLM, $df=1$, $F=3.45$, $P=0.08$).

The same decrease in vegetation cover occurred with native species. Immediately after restoration, native species cover plummeted by more than 50% to 26.1% compared to pre-restoration levels of $70.8 (\pm 12.4)\%$ (Appendix B-3; Table 1). Native species cover has slowly climbed over the years, averaging $41.6 (\pm 8.9)\%$ in 2017, which is still lower than pre-restoration levels (GLM, $df=1$, $F=4.06$, $P=0.06$; Appendix B-3; Table 1). Despite the decreases in cover, native species richness between pre- and post-restoration conditions was actually statistically equivalent for the first time since restoration was conducted in 2015 (GLM, $df=1$, $F=2.46$, $P=0.13$, $\beta=0.98$; Table 1). The number of native species rose from 1.9 species/ $2m^2$ in 2014 to 3.0 species/ $2m^2$ in 2015 and then jumped to $5.2 (\pm 0.7)$ species/ $2m^2$ in 2017 (Appendix B-3; Table 1). While 2017 native species richness appears to exceed that of pre-restoration levels, statistically, these averages were equivalent (GLM, $df=1$, $F=2.45$, $P=0.13$).

Diversity indices also initially dropped after restoration, but, by 2017, they were equivalent between restoration periods, with Shannon-Weiner values averaging $1.4 (\pm 0.1)$ pre-restoration to $1.3 (\pm 0.1)$ in 2017 (GLM, $df=1$, $F=1.3$, $P=0.27$; Table 1). Evenness values had already reached statistical equivalence earlier, but showed statistically significant differences with pre-restoration values in 2017, even though pre- and post-restoration values actually appeared to move closer together. Evenness averaged $0.68 (\pm 0.02)$ prior to treatment and $0.60 (\pm 0.03)$ approximately five- to six years following treatment (GLM, $df=1$, $F=4.82$, $p=0.04$). Variability in this dataset probably decreased, thereby increasing statistical power and enabling a more accurate statistical comparison relative to previous years' results.

This initial decline in native and other non-native plant cover and richness and diversity indices in herbicide-treated plots could be ascribed to adverse impacts on non-target species either directly from drift or indirectly from absorption of released chemicals from decaying rhizomes. Treatment with imazapyr in certain types of vegetation communities has sometimes led to issues with long-term adverse impacts on non-target plant species. Treatment of perennial pepperweed (*Lepidium latifolium*) in seasonal wetlands in the upper margins of San Francisco Bay resulted in "severe" and "persistent" non-target effects, with species richness and percent cover of non-target plant species significantly reduced even two years after treatment (Whitcraft and Grewell 2011). However, other dune projects that used imazapyr for control of European beachgrass have not encountered adverse impacts on non-target plant establishment or survival. In Mendocino County, no evident lasting effects were observed either with respect to germination or establishment of native dune plants, including rare plants (CDPR 2012). A restoration project at Año Nuevo State Reserve near Monterey that used imazapyr resulted in colonization "by a diverse suite of natives in a relatively short time" (Hyland, unpub. data).

While the vegetation monitoring data summarized above would suggest that treatment activities had a large initial negative impact on non-target vegetation, visual observations immediately after initial treatment in 2011 or 2012 indicated otherwise. A number of species established in the first year post-treatment, including miner's lettuce, wild cucumber, woodland



Figure 28. Beach layia grew in treated European beachgrass stands.

groundsel, galium (*Galium trifidum*), fairy mist (*Pterostegia drymarioides*), coast cryptantha, bitter-cress (*Cardamine oligosperma*), tarweed (*Hemizonia* sp.), seaside fiddleneck (*Amsinckia spectabilis*), fescue brome, sheep sorrel (*Rumex acetosella*), and California poppy (*Eschscholzia californica*; Johnson 2013a). Additional species included sand mat, beach morning-glory, beach primrose, and yarrow. Some individuals of beach layia, Tidestrom's lupine, and curlyleaf monardella were also observed (Figure 28). These results are somewhat similar to what Hyland and Holloran (2005) found after treatment of Monterey dunes, with initial colonization by native annual plant species, early successional woody

shrubs such as bush lupine, and European searocket. These examples would suggest that herbicide treatment is not having "severe" or "persistent" effects on either the soils or emerging vegetation in this particular dune environment.

In general, most of the plant species present in plots continue to be strongly representative of Dune Scrub vegetation communities (33.7% in 2017) relative to 86% prior to restoration (Appendix B-3). As all of the herbicide-treated plots are in the backdunes, this is not surprising, as the backdunes are more likely to evolve into a Dune Scrub rather than Dune Mat vegetation communities due to distance from the ocean and the infrequency of sand deposition events that would encourage colonization of more primary successional species. Less sand movement makes it more unlikely that early (or even mid-) successional dune species will establish in these areas (Zhang and Maun 1990, Maun 2004). Multivariate statistical analysis continues to depict the backdune herbicide treatment areas as distinctly separate from both mechanical removal and control (native Dune Mat) areas (Figure 21-NMS, Axis 2, $R^2=0.67\%$, GLM, $df=2$, $F=63.57$, $P<0.0001$, Tukey multi-comparison, $P<0.05$; MRPP, all $P<0.0001$), although species composition appears to be moving closer to native dune areas when compared with earlier NMS graphic results. Species assemblages in 2017 supported the premise that backdune areas will probably eventually evolve into Dune Scrub, with characteristic indicators being coyote brush, miner's lettuce, wild cucumber, fairy mist, and sheep sorrel, as well as secondary invaders (ripgut brome, fescue brome, common groundsel, sow thistle), dead beachgrass, and detritus (Indicator Species Analysis, all $P<0.05$).

No sizeable stands of Dune Scrub exist within the Abbotts Project Area boundaries that would allow for comparison to determine whether restoration areas are heading on a trajectory toward eventual succession into Dune Scrub, however, the AT&T Project Area supports a number of relative intact Dune Scrub stands. Figure 29 shows a NMS analysis between Abbotts restored backdunes and AT&T Dune Scrub stands: the results suggest that the non-restored backdunes were quite distinct from native Dune Scrub, but that the restored areas are moving closer over time to vegetation communities characteristic of native Dune Scrub (NMS, Final Stress=16.55, $R^2=0.65$), with statistically significant differences along both Axis 3 ($R^2=0.45$; Mood Median, $df=4$, Chi-Square=28.85, $P<0.0001$) and Axis 2 ($R^2=0.20$, GLM, $df=4$, $F=3.02$,

P=0.02). One site stands out in this NMS analysis: this is a herbicide treatment plot at Abbotts Lagoon from 2013 that was treated in 2012 through bulldozer scraping, which left it largely unvegetated for a few years (M-17; Figure 29). Convergence with native Dune Scrub has not happened yet, however (MRPP, $T=-8.93$, $P<0.0001$). Indicator species analysis points to higher abundance of chamisso lupine, sand mat, miner's lettuce, California brome (*Bromus carinatus*), California poppy, yarrow, galium, and Pacific wildrye (*Elymus pacificus*; all $p<0.05$).

Tidestrom's lupine and beach layia were less frequent colonizers in herbicide treatment areas due probably due to the lack of open space or bareground for colonizing. In addition, herbicide treatment generally resulted in less early successional habitats with a few exceptions. Where mechanical removal had been performed in herbicide treatment buffer areas, Tidestrom's lupine again readily recruited without seeding being even necessary. Tidestrom's lupine also moved into more open, bare sand areas within herbicide treatment areas where scraping and, to a lesser degree, mowing had been performed in 2012 to accelerate decomposition of European beachgrass or where sand overwash from the beach occurred (Abbotts Lagoon foredune treatment areas). Many of these sandy areas in the backdunes received infusions of sand from the mechanical removal area, secondary access roads nearby, or the beach. Beach layia and curly-leaf monardella established in the early successional areas, but they were also present in herbicide treated areas with more late successional vegetation communities, particularly in small, open areas between dense stands of standing dead beachgrass. Similar to the mechanical restoration areas, Tidestrom's lupine typically colonized these areas first, followed a few years later by beach layia and curlyleaf monardella. Numbers of these species were combined with mechanical restoration results and presented earlier.

Abbotts Triangle: Dramatic declines in European beachgrass were also documented at herbicide-treated foredunes at Abbotts Triangle, with the frequency of beachgrass dropping from 73.4% in 2011 to 2.0% in 2015 (Chi-Square, $P<0.0001$). Similar to B Ranch North, the Seashore switched monitoring methods in 2017 from frequency to cover using GPS line transects. Cover data from 2017 was compared with more limited cover data from previous years. Immediately after treatment, cover of primary invasives such as European beachgrass and iceplant dropped from 30.3% in 2011 to between 1.6 and 1.9% during the period 2012 – 2015 (Appendix B-4). In 2017, cover of primary invasives seemingly tripled to 6.0% cover, but this may have been partly due to the fact that the monitoring approach changed that year (Appendix B-4).

In stark contrast to the herbicide treated backdunes a little further south in the Abbotts Lagoon Project Area, standing dead biomass of European beachgrass appeared to rapidly decompose at Abbotts Triangle, except for areas on the most leeward side of the foredune system adjacent to the lagoon itself (L. Parsons, *pers. obs.*). Detritus also remained a relatively minor component of this system, actually peaking at 18.4% cover in 2015 (Appendix B-4). Another factor that varied considerably from the herbicide-treated backdunes was bareground. The percent cover of bareground in the Abbotts Triangle jumped dramatically immediately after restoration, increasing from 55.3% in 2011 to 76.7% in 2012 (Appendix B-4). The absence of standing dead beachgrass biomass and detritus along with considerable increase in bareground, was largely generated by sand overwash, particularly in windward portions of the foredune ridge (Figure 30). Bareground declined from its peak in 2012 to around 60% cover in 2015 and 2017



Figure 30. Sand overwash in treatment area in foredunes at Abbotts Triangle just south of the lagoon

(Appendix B-4).

This decline in bareground appears to be associated an increase in cover of other species, primarily native dune plant species, although there was a small surge in a second primary invasive species in 2017 (iceplant; 6%; Appendix B-4). In the first year following treatment, native dune plant species such as beach evening primrose and beach morning-glory dropped significantly in frequency (Chi-Square, all $P < 0.003$), with estimated native cover that year being 10.2% or roughly equivalent to pre-

restoration conditions (9.5%; Appendix B-4). Over the years, native cover has slowly, but steadily grown, with the biggest increase coming in 2015 (21.6%; Appendix B-4). In 2017, native cover averaged 25.2% (Appendix B-4). Most of this increase appeared to come from species characterized as mid-successional species (Appendix B-4). The total number of species jumped from four (4) species in 2011 (pre-restoration) to seven (7) in 2013, 13 in 2015, and 20 in 2017 (Appendix B-4). At least some of this increase in species richness came from secondary invasives such as European searocket (5.4%) and common groundsel (<1%).

Tidestrom's lupine and, to a lesser extent, beach layia had been documented in the Abbotts Triangle area prior to the 2011 herbicide treatment. Following initial mechanical removal efforts at Abbotts Triangle in 2003-2004, park staff found 182 Tidestrom's lupine and 18 beach layia seedlings (Rodgers 2006). Most of these rare plant establishment efforts were later threatened, however, by re-growth of European beachgrass between 2005 and 2011. In monitoring efforts in subsequent years, abundance of these species in Abbotts Triangle was not differentiated relative to the adjacent historic population in the native Dune Mat area at Abbotts Lagoon, with some of the northernmost transects directly bordering the leeward portions of the Abbotts Triangle area. Since restoration in 2011, no species-specific sampling or census has been attempted, but data generated by the GPS line transects suggests that cover of rare plant species may have increased somewhat, climbing from 0% or nearly 0% in 2011-2012 to 2.5% in 2017 (Appendix B-4).

B Ranch North: The frequency of European beachgrass decreased dramatically after the first year restoration, dropping from 95.6% frequency in 2014 to 4.1% frequency in 2015 (Chi-Square, $P < 0.0001$, Figure 31). The frequency of iceplant within beachgrass stands also dropped significantly (from 64.6 to 20.7%). As noted earlier, park staff switched monitoring methods of beachgrass-treated areas at B Ranch North in 2016. While it is not possible to directly assess changes between years due to the change in approach, more limited cover data was collected pre- and immediate post-restoration. Cover of beachgrass averaged 50.9% in 2014 prior to treatment, dropped to 0.5% in 2015, but then climbed back up to 6.4- to 6.9% in 2016-2017

(Appendix B-5). However, in 2018, cover of beachgrass dropped to 0% (Appendix B-5).

As with Abbotts' backdune areas, most of the sprayed areas remained an extensive monoculture of standing, dead European beachgrass biomass the first few years, although plant decomposition appeared to occur more quickly at B Ranch North than at Abbotts (Figure 29). Standing dead biomass peaked at 37% cover the year immediately after restoration, dropping to



Figure 31. Treated European beachgrass stands with intermixed iceplant at B Ranch North with untreated iceplant in the background.

34.1% in 2016, and 21.8% in 2017, although it rose slightly to 27.0% in 2018. Treated areas at B Ranch North are exposed more to the harsh maritime influences of high winds and moisture than many of the herbicide-treated areas at Abbotts, which may accelerate breakdown of dead plants (Figure 27). Dead beachgrass showed visible signs of shredding or tearing apart of aboveground stems. Detritus jumped from 19.0% pre-restoration to 86.2% in 2015 – probably because of breakdown of standing dead beachgrass -- and then dropped to 61.8% in 2016, 49.5% in 2017, and 11.8% in 2018 (Appendix B-5). Interestingly, the cover of bareground also appeared to decrease from 21.6% in 2014 to 9.0% in 2015, with cover of open space increasing slightly to 10- 13% in 2016-2017 and then jumping to 21.7% in 2018 (Appendix B-5).

As with the other restoration projects, cover of native plant species also declined immediately after restoration. The only species that appeared to possibly increase in frequency immediately following treatment was coast cryptantha (*Cryptantha leiocarpa*; Chi-Square, $P=0.08$). Based on visual observations by park staff, Heerman's lotus also appeared to be a frequent colonizer in the first year post-restoration, although data seemingly showed no difference in frequency (Chi-Square, $P>0.1$). In terms of cover, native vegetation dropped from 8.0% pre-restoration to 4.0% in 2015, but quickly jumped to 16.0% in 2016, 27.1% in 2017, and 22.5% in 2018 (Appendix B-5). Total number of plant species or total species richness was halved immediately after restoration, dropping from 8 species to only 3 in 2015, but has skyrocketed in subsequent years, climbing to between 25 and 30 species between 2016 and 2018. Some of this may be artifact of the change in monitoring approach, but it does appear to reflect an actual increase in the number of colonizers. In addition to dead beachgrass, bareground, and detritus, species with highest percent cover included: dead and live iceplant (15.1%), coyote brush (6.0%),

Heerman's lotus (5.4%), fescue brome (2.6%), cudweed (*Pseudognaphalium stramineum*; 2.5%), fairy mist (2.2%), miner's lettuce (1.7%), dune sagebrush (1.6%), and beach evening primrose (1.6%). Similar to pre-restoration conditions, most of the native species that have persisted within treated areas are late successional or Dune Scrub in nature (Appendix B-5), which means that, while restoration finally appears to be increasing native species cover and richness, restoration has not effectively re-set or rejuvenated the successional timeline at B Ranch North.

Similar to other restoration projects, elimination of the primary invasive (European beachgrass) has opened the door to new invasives. Secondary invaders have climbed from 0% cover in 2015 to 0.5% cover in 2016, 3.0% in 2017, and 5.3% in 2018 (Appendix B-5). The biggest re-invasion threat at B Ranch North has been iceplant, which existed within beachgrass stands at low



Figure 32. Treated iceplant at B Ranch North still partially alive following year.

frequency prior to treatment. Iceplant stands have seemingly showed more resiliency to treatment efforts than have those near Abbotts and AT&T (Figure 32). As the treatment method is the same, it may be that the foggy conditions at night and early mornings increase condensation on the waxy surface of iceplant leaves to the point that herbicide treatment is not allowed to fully absorb, even with use of surfactants. While not as common as at Abbotts, European searocket has also made inroads, particularly closer to the foredunes and beach area. Other secondary invaders at B Ranch North include New Zealand spinach, sow thistle, and common groundsel.

Most of the rare plant monitoring at B Ranch North has focused on the lightly invaded Dune Mat areas that support existing populations of Tidestrom's lupine, beach layia, and other species. Some rare plant species have been observed in the beachgrass treatment areas, but they have not been censused. Rare plant cover averaged 0% between 2014 and 2017 and 0.2% in 2018.

AT&T North: Removal of primary invaders such as European beachgrass and, to a lesser extent, iceplant appears to be initially very successful at AT&T North and other NPS-funded treatment areas at AT&T (Parsons and Ender 2019). As discussed earlier, prior to restoration, AT&T North largely supported beachgrass, coyote brush, and detritus, as determined from multivariate analysis (Indicator Species Analysis; $p < 0.05$). Cover of live beachgrass and iceplant dropped from an average of $77.7 (\pm 5.5)\%$ in 2016 before restoration to $1.9 (\pm 0.9)\%$ in 2017 and $0.8 (\pm 0.5)\%$ in 2018 (lm Permutation, $P < 0.0001$; Appendix B-6; Table 2). It should be noted that data presented represents only the herbicide treatment plots, although three plots were also established in mechanical removal areas: the low number of plots made statistical comparisons infeasible. However, based on review of the data, regrowth in mechanical removal areas was also very low.

Standing dead biomass of primary invaders jumped dramatically from $2.9 (\pm 1.1)\%$ prior to treatment to $79.3 (\pm 5.9)\%$ in 2017, dropping to $55.2 (\pm 7.8)\%$ in 2018 (Lm Perm, $df=2$, $P=0.001$; Appendix B-6; Table 2). Detritus or thatch remained roughly equivalent during this time period, ranging from a low of $77.6 (\pm 6.0)\%$ in 2017 to a high of $89.1 (\pm 2.6)\%$ in 2018, compared to $88.5 (\pm 4.1)\%$ in 2018 (GLM, $df=2$, $F=2.03$, $P=0.142$). Herbicide treatment also did not increase the percent cover of bareground, which ranged very tightly from 1.1% (2016-2017) to 1.9% (2019; Lm Perm, $df=2$, $P=0.57$; Appendix B-6; Table 2). This may result from the fact that most of the treated beachgrass or iceplant remains as standing dead biomass and has not had time to break down into detritus, which would then decompose in turn and create more bareground or open space for new plant colonization. Figure 33 illustrates how treated beachgrass persists as large standing dead biomass even years after treatment.



Figure 33. Treated European beachgrass stands at AT&T North in 2018.

As discussed earlier in this document, dieback of primary invaders can set the stage for invasion of other non-native species or so-called secondary invaders. Absolute cover of primary invaders and other non-native species at AT&T North plummeted from $83.1 (\pm 5.5)\%$ in 2016 (pre-restoration) to $7.4 (\pm 2.5)\%$ in 2017 or the first year after treatment (GLM, $F=84.2$, $P<0.0001$; Appendix B-6; Table 2). However, average non-native cover seemingly more than doubled in the second year (2018) post-restoration to $16.1 (\pm 5.1)\%$, however, these differences were not statistically significant (Tukey pairwise comparison, $P>0.05$; Appendix B-6, Table 2). Potential increases in non-native cover appear attributable more to expanded cover of a few species than establishment of many new non-native species following treatment, as non-native species richness remained equivalent pre- (average = 2.1 ± 0.3 spp/ $2m^2$) and post-restoration (average = 2.1 ± 0.4 spp/ $2m^2$; GLM, $F=0.0$; $P=1.0$; Appendix B-6; Table 2). Year 1 post-restoration plots were characterized primarily by dead beachgrass and coyote brush, as well as large amounts of detritus (Indicator Species Analysis; all $p<0.05$), but, by Year 2 post-restoration, groundsel and brome fescue also contributed significantly to plant cover (Indicator Species Analysis; all $p<0.10$). These backdune secondary invaders probably accounted for weakly statistically significant difference in secondary invader species pre- and post-restoration in AT&T North plots (lm Permutation, $df=2$, $P=0.06$; Appendix B-6; Table 2).

In addition to non-native species cover, cover of native species also appeared to significantly decrease after restoration, dropping from $55.3 (\pm 9.1)\%$ in 2016 to $32.6 (\pm 8.3)\%$ in 2017 and $33.9 (\pm 8.5)\%$ in 2018 (GLM, $df=2$, $F=2.63$; $P=0.08$; Appendix B-6; Table 2). Cover of native species also showed a relationship with distance from the ocean, with the percentage of native species increasing somewhat further inland (GLM, $df=1$, $F=7.9$, $P=0.007$, $r^2=2.2\%$). While

herbicide treatment involves use of backpack sprayers with wands and nozzles to try and avoid native species as much as possible, native species often grow amidst beachgrass and iceplant, so indirect effects on the native community are unavoidable. However, the objective is to minimize those indirect effects and leave as much native species as possible to allow for future expansion and regeneration of native species into treated areas.

Native species richness dropped slightly between pre- and post-restoration, averaging $4.3 (\pm 0.5)$ species/ 2m^2 in 2016 and $3.2 (\pm 0.6)$ species/ 2m^2 in 2018, however, differences were statistically insignificant (GLM, $F=0.98$, $P=0.38$; Table 2). In contrast, the Shannon Diversity Index remained roughly comparable before and after treatment, averaging $0.95 (\pm 0.12)$ in 2016 compared to $0.80 (\pm 0.15)$ in 2017 and $0.84 (\pm 0.16)$ in 2018 (GLM, $F=0.34$, $P=0.72$; Table 2). Species evenness also stayed fairly equivalent between restoration periods, averaging $0.52 (\pm 0.05)$ in 2016 and $0.45 (\pm 0.08)$ in 2018 (GLM, $F=0.52$, $P=0.60$; Table 2).

Multivariate analyses of AT&T North plots pre- and post-restoration showed that pre-restoration invaded areas differed substantially in species composition than the same areas post-restoration, with considerable overlap shown between Year 1 and Year 2 post-treatment plots (NMS, Final Stress=17.65, $R^2=0.81$; Figure 34). The only exception were two plots where iceplant had been the primary invader (Figure 32). Axis 2 of the NMS, which accounted for 42% of the variation, showed statistically significant differences in species assemblages between each of the Dune Scrub monitoring events (2016 and 2017), as well as between reference plots and all of the invaded plots (2016-2018), regardless of whether they were pre- or post-restoration (NMS GLM, $df=4$, $F=23.06$, $P<0001$, Tukey pairwise). In terms of the type of native plant species present in treatment plots, most were characteristic of Dune Scrub vegetation communities (Appendix B-6). Interestingly, native Dune Scrub plots displayed distinct differences in species composition between 2016 and 2017, despite the fact that they were unaffected by treatment (Figure 32). The 2017 plots supported higher abundances of weedy species such as ripgut brome and chickweed, perhaps because the winter of 2017 was a very wet one precipitation-wise after a number of dry years in California. This is reflected in striking variation in non-native plant cover in Dune Scrub plots between years, with non-native cover jumping from $17.3 (\pm 5.1)\%$ in 2016 to $67.1 (\pm 16.7)\%$ in 2017 (GLM, $F=7.72$, $P=0.009$; Table 2). As with treatment plots, non-native species richness remained roughly equivalent between 2016 (2.6 ± 0.4 (S.E.) spp/ 2m^2) and 2017 (3.1 ± 0.6 species/ 2m^2 ; GLM, $F=0.4$, $P=0.47$; Table 2), suggesting that a few non-native species rapidly expanded in cover. Wet winters often prompt an explosion in weedy, non-native species that take advantage of the boom in abiotic resources such as water and nitrogen to spread quickly (Hobbs & Mooney 1991; Dukes & Mooney 1999; Milchunas & Lauenroth 1995; White et al. 1997 in Alpert et al. 2000.)

In keeping with the fact that most of AT&T North is topographically elevated and would probably have naturally evolved into Dune Scrub vegetation communities, the rare plant community is generally different from what would be found in Dune Mat vegetation communities. Rare plant species within Native Dune Scrub at AT&T North generally included San Francisco Bay spineflower, beach starwort, curlyleaf monardella, and wallflower. At least two of these species co-occur in mid to late successional Dune Mat communities (curlyleaf monardella, wallflower), and San Francisco Bay spineflower can be found in late successional dune habitats, as well. Density of rare plants within native Dune Scrub plots varied

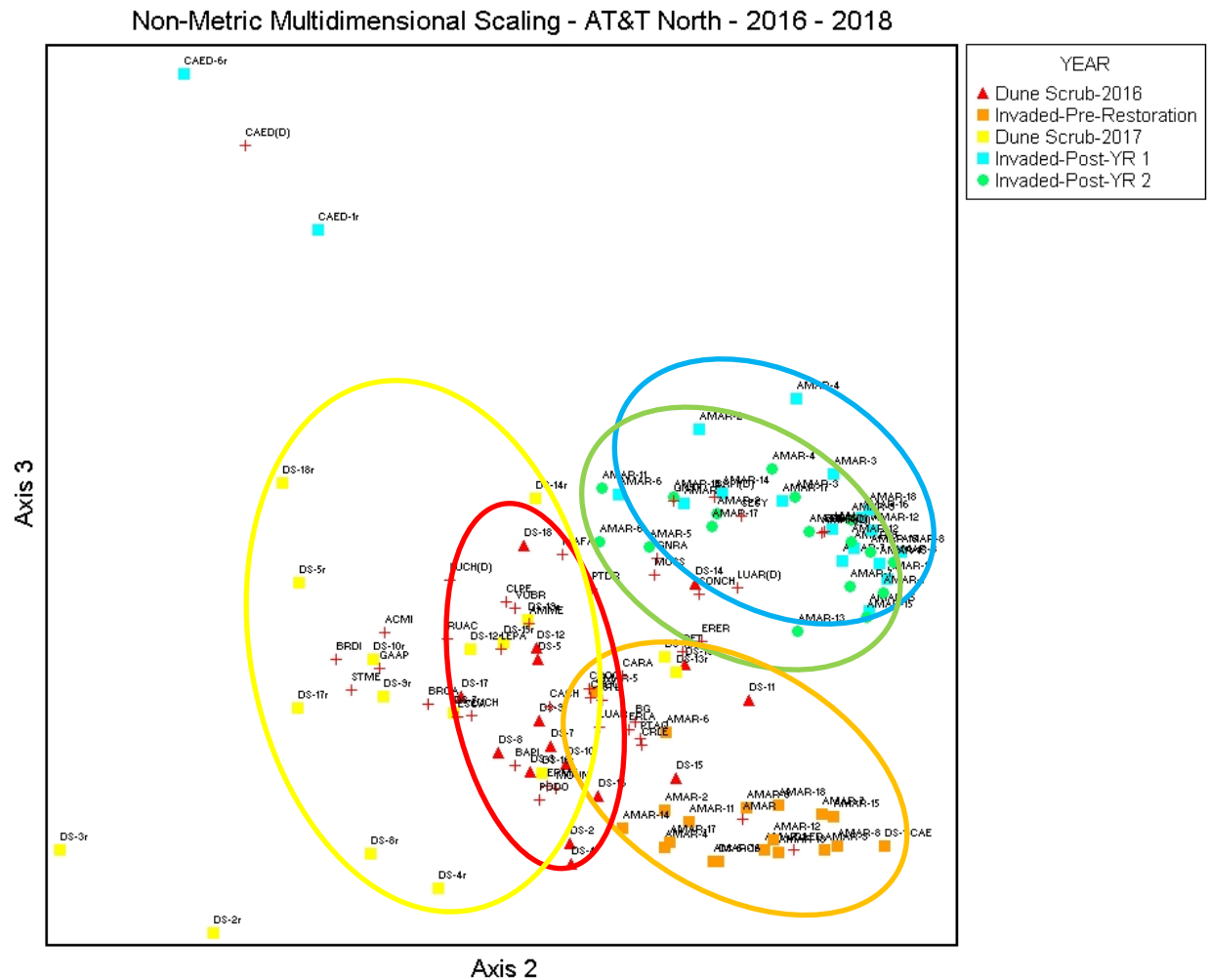


Figure 34. Non-Metric Multidimensional Scaling Analysis for pre-restoration (2016) and post-restoration (2017-2018) treatment and control (2017 only) plots at AT&T North. NMS indicates a distinct shift in post-treatment invaded communities, with little change immediately following restoration. Native Dune Scrub plots also showed some changes between years, but remained distinct from invaded communities.

tremendously between years, again probably because of the difference in precipitation totals and the rapid response of many annual rare plants to changes in rainfall volume. Studies of other rare coastal annuals have found climatic variables to be an important determinant of plant establishment success (Parsons and Whelchel 2000, Fox et al. 2006, Levine et al. 2008). In 2016, density averaged 12.4 (± 7.20) plts/2m², but numbers jumped in 2017 (52.3 $\pm$ 32.4 plts/2m²), largely because of high numbers with certain plots (Table 2). No rare plants were found in treatment plots in any of the monitoring years, but due to the persistent standing dead biomass of beachgrass and lack of bareground, this is not an unexpected result.

As discussed earlier, the AT&T Project Area does support sizeable populations of several federally endangered plant species such as Tidestrom's lupine, beach layia, and Sonoma alopecurus. Sonoma alopecurus is a freshwater marsh species that occurs in the Alopecurus Swale in between AT&T North and AT&T South. As discussed earlier in this document, Tidestrom's lupine and beach layia tend to preferentially colonize in areas with more bareground or open sand, with Tidestrom's lupine being very disturbance-tolerant and establishing in primary to mid-successional habitats. Primary to mid-successional dune habitats exist now only as pockets of open sand areas within beachgrass stands, although it is possible that foredune areas will develop into these habitats over time. Most of the existing primary to mid-successional dune habitats occur in AT&T South, which is where most of the established Tidestrom's lupine and beach layia populations are located.

There are sporadic colonies of these species in AT&T North, with many patches actually establishing in the UTV secondary access roads created to implement the project. These secondary access roads are open and sandy, which is perfect habitat for these species. One "hotspot" for both species is the UTV road on the northwestern corner of AT&T North. In 2016, this area supported 45 Tidestrom's lupine adults and 10 seedlings, as well as beach layia. Other colonies have established within bareground or open areas in less dense native Dune Scrub, specifically on the south side of AT&T North near the Alopecurus Swale. This patch had 12 Tidestrom's lupine adults and 3 seedlings in 2016. No census of rare plants in AT&T North was conducted in either 2017 or 2018.

AT&T South: AT&T South was treated initially in 2017 and 2018 during two separate phases. This summary discusses only the Phase I results from treatment conducted in 2017 (Ender and Parsons 2019). Percent cover of primary invaders such as European beachgrass and iceplant was significantly lower the first year after treatment (Chi squared LR = 111.83, Df = 1, 12, $p < 0.0001$,) decreasing from an average of 86.0 (± 11.0)% cover to 2.7 (± 0.1)% cover (Figure 35a). Similarly, non-native cover as a whole, which included other non-native species such as ripgut brome, common velvetgrass (*Holcus lanatus*) and brome, decreased significantly from an average of 98.0 (± 10.8)% before treatment to 3.0 (± 0.8)% immediately after treatment (Chi squared LR = 158.95, Df = 1, 12, $p < 0.0001$; Figure 35b). Bareground increased significantly (Chi squared LR = 8.01, Df = 1, 12, $p = 0.0047$), but was very low – under 2% cover on average – in both years (Figure 35c). Similar to AT&T North and other Point Reyes dune restoration projects, native cover appeared to decrease the first year after treatment from 49.7 (± 22.9)% to 15.1 (± 9.8)%, but this change was not statistically significant (Chi squared LR = 2.30, Df = 1, 12, $p = 0.13$; Figure 35d).

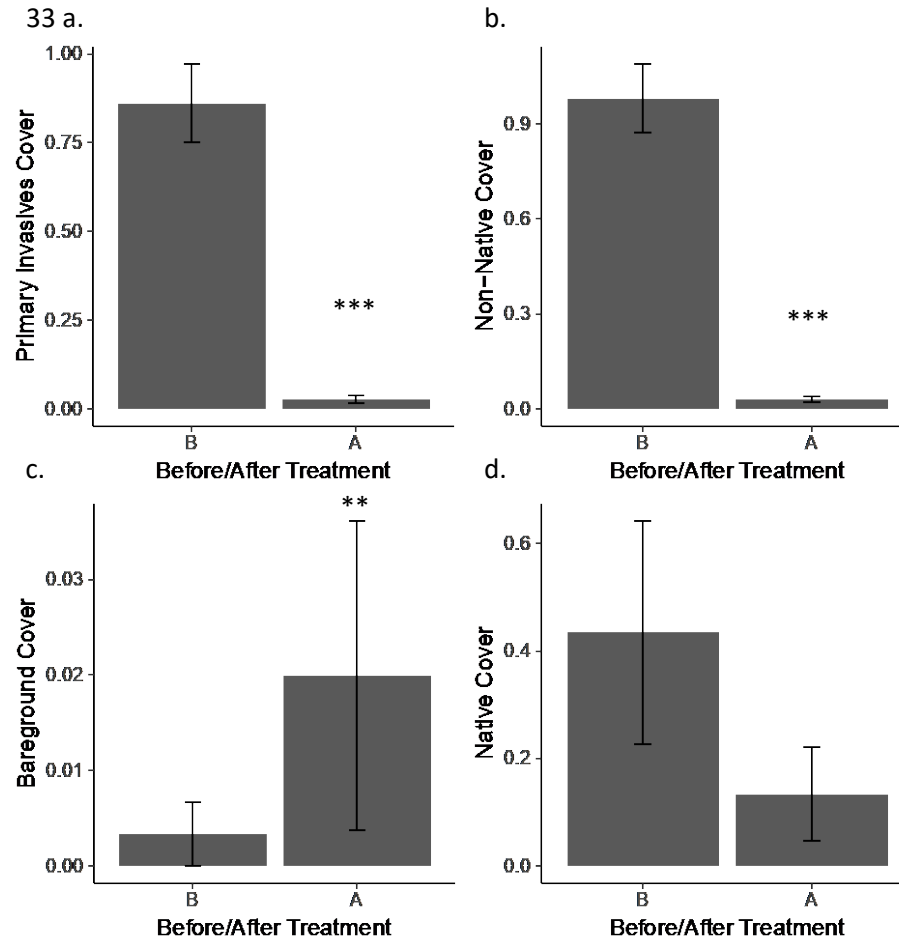


Figure 35. Mean ($\pm$ S.E.) cover for AMAR (a), non-native species (b), bare ground (c) and native species (d) in plots before (B) and after (A) treatment. Asterisks indicate a significant effect of period in generalized linear models: '****' $p < .0001$; '***' $p < .01$; '**' $p < 0.05$; '*' $p < 0.1$.

Overall, species richness and native species richness both were significantly higher before treatment (Chi squared LR = 7.89, df = 1, 12, p = 0.0050; Chi squared LR = 5.31, df = 1, 12, p = 0.021 respectively). Total species richness varied from 2 to 9 species per plot before treatment (5.14 species on average) to 0 to 6 species after treatment (2.29 species on average). The decrease in non-native species richness (2.29 ± 0.61 species before and 1.14 ± 0.34 species after) was weakly significant (Chi squared LR = 2.72, df = 1, 12, p = 0.10). Beachgrass, wild cucumber, and coyote brush were the most common live plants found in invaded plots immediately after treatment.

Species evenness started at 0.55 (± 0.10), and, while evenness appeared to decline to 0.39 (± 0.18) following herbicide treatment, this change was not statistically significant (Chi squared LR = 0.61, df = 1, 12, p = 0.44; Ender and Parsons 2019). Diversity did appear to be halved immediately post-restoration from 0.83 (± 0.20) to 0.38 ± 0.17 (S.E.): this change was weakly significant (Chi squared LR = 2.98, df = 1, 12, p = 0.08).

The NMS analysis shows a significant treatment by time period interaction (Stress = 0.15; Chi squared LR = 3.97, df = 1, 51, p = 0.046; Figure 36). Graphically, the reference plots or native Dune Scrub plots (Control) have substantial overlap in species composition before and after treatment, while the invaded plots (Impact) clump separately. Additionally, the invaded plots have very little overlap with reference plots prior to treatment, but shift to a species composition more similar to the reference plots even just one year after treatment. We also found significant effects of treatment (control/impact; Chi squared LR = 33.56, df = 1, 51, p < 0.0001) and time period (before/after; Chi squared LR = 8.01, df = 1, 51, p = 0.005) on their own.

Rare plant monitoring was not specifically conducted in Phase I, but a sizeable, lightly invaded native Dune Mat area occurs at the boundary of Phase I and Phase II in AT&T South. The southern portion of AT&T incorporates the third largest population of Tidestrom's lupine (Population #8) and one of the largest remaining populations of beach layia (Population #5). The center of these populations is the 0.37-acre native Dune Mat area that has been invaded to some degree by iceplant (~30% gross cover). Interestingly, the Tidestrom's lupine population at AT&T remained relatively small (234 – 413 individuals) until 2012, when it began to rapidly growing, more than doubling in numbers by 2014 (1,013) and peaking in 2018 (2,144). This is almost a 500% increase since 2011, and none of it would seemingly be attributable to restoration, as no invasives removal in AT&T South occurred until 2016. Interestingly, seed predation rates have remained roughly comparable before and after 2012, averaging 46% and 36% (WU, unpub. data), so population growth does not appear to be a result of reduced predation. The only other growing lupine population – except for Abbots – is located just south of AT&T, and it has shown very similar trends in population growth, as well as in predation rates (average=44%, WU, unpub. data). These results suggest that these populations are perhaps benefitting from some localized factor or factors that have yet to be identified. It may not be a coincidence that the only unrestored population of beach layia that has been stable since monitoring was established in 2004 occurs at AT&T South. Layia estimates totaled 4,331 in 2004 and 4,216 in 2018. Iceplant was directly treated in 2018, and surrounding beachgrass stands were treated in 2017 and 2018, but it is still too soon to see if invasives

removal of iceplant and surrounding beachgrass will provide additional benefits to these already very successful populations. A Population Viability Analysis conducted by Washington University on lupine data collected between 2005 and 2008 suggested that, based on reproduction rates, this population might be the only population with a stable or slightly increasing population growth rate and that removal of seed predation pressure by deer mice would be sufficient to induce a strongly positive population growth rate ($\gg 1$; Dangremond et al. 2010). However, data collected one year later (2008-2009) showed more equivocal results (WU, unpub. data). Beach layia is not affected predation of seeds by mice (or seemingly any other mammals), so any benefits realized from the recent restoration would revolve more around an increase in available habitat once the abundance of adjacent beachgrass and iceplant declines, and the abundance of bareground expands as a result of eventual decomposition of dead invasives biomass and detritus.

SUMMARY DISCUSSION ON CHANGES IN RESTORATION AREAS

The objective of dune restoration generally is to eradicate non-native, invasive species that are altering native dune ecosystems and threatening habitat quality and functions. The visualized end-point for this restoration process is not only a system free of non-native, invasive plants, but a system that looks like -- and functions like -- nearby native dune systems. In many instances, invasives have either eliminated any resemblance to native dunes in either species or function, or they have forced these systems to age prematurely, transitioning them into later phases or successional stages that don't support the same type of special status plants and wildlife that younger successional communities do.

In a sense, determining whether our restoration efforts have been successful is really a two-part question, starting with the question of whether we can successfully eliminate non-native invasive species. Early on, the answer to this seemed in doubt, as our efforts at manual removal at Abbotts Triangle seemed almost a Sisyphus-like endeavor, with endless re-treatment rewarded only with endless re-growth. However, once the park switched to mechanical removal and herbicide treatment, the success of eradication efforts improved exponentially, with both methods reducing abundance of primary invaders to less than 3- 5% within the first year after treatment, and cover continued to decrease in subsequent years with minimal re-treatment.

However, the park's challenges were not over yet in terms of invasive species. While the battle against the primary invaders such as European beachgrass and iceplant appeared largely won, a new battle ensued against a much larger suite of secondary invader species, many of whom subsisted at low levels of abundance prior to restoration. These "sleeper weeds" wait until conditions are optimal and then rapidly explode out of this "lag phase" into a period of active growth, fueled by landscape disturbance or a change in biotic or abiotic resources. One of the most prevalent secondary invaders at Abbotts, European searocket, was only infrequently observed in 2012, and the park considered targeting it for removal, but, after consulting with online resources and other biologists, dismissed it as inconsequential. Searocket still remained in relatively low and almost unnoticeable abundance in 2015, but, in 2016, searocket appeared to be everywhere, expanding exponentially to the point that searocket plants were actually smothering newly established Tidestrom's lupine in some portions of the mechanically restored dunes. This seemingly overnight explosion in numbers is why several invasives experts have argued that all weeds should be considered "guilty before being proven innocent" in terms of removal prioritization (Simberloff 2006, Edelaar and Tella 2012). This is by no means a unique problem to this project: as was discussed earlier in this report, a recent meta-analysis of research on invasives removal projects pointed to them often being very successful at eliminating the target species, but later these successes being significantly compromised -- and sometimes even overshadowed -- by invasion of new, secondary invader species (Pearson et al. 2016).

The more difficult question to answer about our restoration efforts is whether they have been successful in creating native dune ecosystems. To date, the results of our restoration efforts have underscored the maxim that restoration is an inexact science -- or a science its practitioners do not understand well enough yet. Yet, monitoring of dune restoration efforts in

the Seashore since 2011 has revealed some commonalities among results for these different types of dune restoration approaches. All of the methods – manual, mechanical, and herbicide – resulted in an immediate decrease in primary invader species, whether it be European beachgrass or iceplant. Mechanical removal and herbicide treatment projects also reduced abundance, at least temporarily, of other non-native species. Unfortunately, percent cover and richness of native plant species also plummeted immediately after restoration, which did not bode well for efforts to restore a native dune ecosystem. In some senses, mechanical removal almost proved too extreme in terms of re-setting the successional clock, while herbicide treatment and manual removal have not been extreme enough. With mechanical removal, literally all non-native – and native – species were eliminated almost instantaneously, leaving vast expanses of open sand that remained largely unvegetated for years due to the vagaries of drought and high winds. Soil conditions and dune processes proved too primordial except for the hardiest of pioneering species such as Tidestrom’s lupine. Meanwhile, in stark contrast, herbicide treatment left vast forests of standing dead European beachgrass biomass that seemed impervious to decomposition: between the dead beachgrass and detritus, there was little opportunity for open space or bareground to develop that could support colonization of native plant species. The only exception was in herbicide-treated foredune areas where sand overwash occurred, blanketing the stands of dead beachgrass within a few years with new sand that provided substrate for natives (and secondary invaders) to more quickly gain hold.

For years following implementation, the restoration success picture remained pretty stark. Standing dead European beach biomass and detritus continued to dominate herbicide treatment plots that weren’t directly adjacent to the beach, and native plant cover and species richness remained much lower than pre-restoration levels. However, in recent years, there has been some renewed hope in that project areas initially treated either mechanically or chemically between 2011 and 2014 have started to show an increase in abundance and richness of native plant species (Appendix B). While multivariate analyses indicate that species composition is not comparable yet with reference systems, whether Dune Mat or Dune Scrub, results suggest that assemblages are moving closer towards native dune system conditions. With these changes in species composition has come some movement in terms of the type of successional stage represented by colonizing native dune plant species, with some areas exhibiting a resurgence in primary- or mid-successional species, but more time will be needed to fully realize the ecosystem-level changes associated with restoration.

Until this happens, the restored dunes may not fully function as native, unimpaired ones do – if they ever do so at all. The issue is that removing the primary target species and even the secondary target species may not be enough to successfully restore a native dune ecosystem, because invasives removal addresses only part of the changes brought about by non-natives (Suding et al. 2004, Suding and Hobbs 2009, Yelenik and Levine 2010, Yelenik and D’Antonio 2013). Long-term alterations or “legacy” effects to the physical and biological ecosystem must be expected after decades of dominance by non-native species, with the most likely including changes in the nutrient status, physical characteristics, and biota of soils, as well as in the structure of dunes. Changes in soil conditions following long-term establishment by iceplant is very well-documented, and so have been the issues with trying to restore iceplant-invaded systems, with legacy effects seemingly hampering with re-establishment of native plant species

and favoring non-native, weedy ones (Andreu et al. 2010, Hamingson and Voeller 2011, Novoa et al. 2012, Novoa et al. 2013, Winsemius 2013). Legacy effects of European beachgrass are less well-understood, but it definitely can change foredune structure (Pickart and Wiedemann 1997) and slow, if not possibly preclude, efforts to re-establish native vegetation due to its slow decomposition and the abundant of thatch and detritus produced. We do not have enough information yet to predict how persistent these decomposition-legacy effects will be, but bush lupine, again believed to be non-native in Marin County, has well-documented, long-term legacy effects on soils from decomposition of nitrogen-enriched litter that promotes colonization of weedy, annuals such as we have observed in herbicide-treated backdune areas (Zink et al. 1995, Maron and Connors 1996). And most of the work done to-date does not even address some of the secondary relationships or mutualisms that may have been lost during the period of non-native dominance. We simply don't know enough about these systems to know what has changed and what is missing.

Based on our restoration efforts to date, it is clear that we need to better understand the big picture of dune restoration, if these and future efforts are to be successful. We need to understand how soils, dune structure, and other ecological relationships have changed with invasion and how they change with restoration and what might be done to help tip the soil scale in restoration's favor. To this partial end, the park reached out to researchers at University of Arizona to study the soil biota of lightly, moderately, and highly invaded dunes, as well as areas restored either through herbicide treatment or mechanical removal. The hope is that this information can be used to better plan or implement dune restoration projects in the future so that we are not just removing invasives, but restoring physical and ecological processes that will better help us truly restore native ecosystems.

However, despite these native ecosystem-creation challenges, it is clear that restoration efforts have already at least partially restored some ecosystem functionality. The sheer numbers of Tidestrom's lupine and beach layia plants that have established within the mechanically restored and, to a lesser degree, herbicide treatment areas at Abbotts are in and of itself a



Figure 34. Myrtle's silverspot butterfly on curlyleaf monardella in restored area

tremendous success story, giving recovery efforts of these endangered species a substantial boost. Restoration has also benefitted the existing Tidestrom's lupine population at Abbotts by reducing seed predation and possibly by rejuvenating the existing native Dune Mat habitat. Restoration has also increased the abundance of other rare plant species such as curlyleaf monardella, which services as one of the preferred nectar species for the Myrtle's silverspot butterfly (Figure 34). While Myrtle's silverspot butterfly numbers are not formally monitored, a number of butterflies have been observed on curlyleaf monardella and other species in the herbicide-treatment areas (Figure 34).

In addition to providing new habitat for establishment of Tidestrom's lupine and other rare plant species, restoration at Abbotts also benefitted the existing Tidestrom's lupine population in the adjacent native Dune Mat habitat by decreasing predation of seed by native deer mice. Predation rates dropped from between 44.8% and 93.9% prior to restoration to between 3.7% and 9.4% immediately after restoration (Pardini and Knight 2013; Pardini et al. 2018). The same benefits, however, have not been realized from herbicide treatment of European beachgrass stands encircling a native Dune Mat habitat area that supports a critically imperiled Tidestrom's lupine population at B Ranch North. Prior to restoration, predation rates averaged 70%; since restoration, predation rates have dropped minimally, averaging 58%, although there was a drop in 2018 to 33.2% (WU, unpub. data). Differences in benefits to existing lupine populations may come down to the fact that mechanical restoration not only immediately eradicated beachgrass, but deer mice, as well. With herbicide treatment, changes in vegetation are more gradual, as beachgrass and its rhizomes decompose very slowly (>5 years): mice may continue to find habitat even in dead beachgrass stands due to the persistence of dead rhizomes for nesting. Sufficiently large enough declines in rodent numbers to benefit reproductive success of Tidestrom's lupine populations threatened by excessive seed predation may not transpire unless there are drastic alterations in the landscape (e.g., mechanical removal) or some type of predator management.

The story for Western snowy plover, a federally threatened shorebird that nests on the Seashore's beaches and dune areas, has also been a little more complicated. In the first few years after restoration, plovers immediately began using the restored Abbotts Dune Project Area for nesting and rearing. In 2013, the number of nests in the dunes jumped dramatically, with plovers nesting in backdune areas and even in Tidestrom's lupine (Figure 35). However, the number of nests and rearing attempts have dropped off in recent years considerably, probably largely because ravens cued in on exposed nests in the largely unvegetated restoration areas, preying particularly on young plover chicks that had not fledged. Nesting efforts might have been more successful in producing fledged chicks if more vegetation had been present in mechanically restored foredunes to better enable nests and fledglings to avoid predator detection (D. Press, NPS, Wildlife Biologist, *pers. comm.*). As with the deer mice-Tidestrom's lupine situation at B Ranch North, habitat restoration in these instances may not be enough to fully restore ecosystem functionality or to fully recover federally listed species. Other types of management may be necessary.



Figure 35. Snowy plover eggs in Tidestrom's lupine plant – 2013

As of now, the restoration projects appear to hold strong promise of re-creating systems that may eventually resemble native dunes, at least in terms of plant species composition. Key to success of these efforts will be vigilant re-treatment of these areas to eliminate resurgence of primary invaders and to keep out new secondary invaders. Similar to many other disturbance-prone ecosystems such as wetlands, disturbance definitely drives the threat of perpetual non-native re-invasion. As Suding et al. (2013) postulated, many native ecosystems can resist

invaders, with native soils favoring natives over non-natives, until some type of disturbance event allows non-natives to establish at high enough numbers to award them a competitive advantage. At the Seashore, deliberate planting of beachgrass and iceplant in the early to mid 1900's probably overwhelmed the innate capacity of the native dune ecosystems to resist these invaders, which inexorably spread and monopolized adjacent areas through superior dispersal and reproductive advantages. Subsequent disturbance events have allowed new invaders to arrive and persist. By definition, dunes are a disturbance-prone environment, whether it be wind or waves or sand movement. Restoration adds yet another disturbance element to this naturally unstable environment and allows secondary invaders such as European searocket, which were present initially in low abundance, to rapidly expand in numbers and areal extent. For the Abbotts project, the disturbance associated with mechanical removal and, to a lesser extent, herbicide treatment was compounded by climatic factors, specifically a prolonged drought that started immediately after implementation in 2011.

Given all this, the reality is that restoration is not a one-year or even a five-year prospect. It is likely to be an ongoing, long-term maintenance responsibility. This may include not only invasives removal, but management of predators to improve plover nesting and fledging success. Restoration of natural processes and biotic and abiotic conditions may help to minimize the degree of management required to maintain restoration success. However, even if all the non-native species can be ousted from an area, and all the other pertinent factors can be restored, the unfortunate reality is that new invaders will arrive via water or wind or bird or someone's foot, only to re-start the invasion process. One of the most successful dune restoration projects along the West Coast – Lanphere/Ma-le'l – openly acknowledges that ongoing invasives removal maintenance is required to maintain the success of this project. Projects in Oregon routinely scalp dunes to create favorable habitat for snowy plovers. In southern California, invasives removal and predator control are considered long-term obligations of many mitigation and restoration projects, probably due to the high and consistent level of disturbance in such an urbanized environment.

CURRENT RECOMMENDATIONS FOR COASTAL DUNE RESTORATION IN THE SEASHORE

- If funds are limited, priority should be on restoring foredune and mid-dune areas using herbicide treatment approach, as sand overwash accelerates conversion of invaded areas to native dune habitat while the remnant European beachgrass and associated rhizomes limit large-scale remobilization of sands by high winds along the coast. When possible, restored areas should be discrete units of invasive plants to limit potential for re-invasion by adjacent stands of primary target species. While this approach does not alter the landscape as dramatically as mechanical treatment, and therefore may not directly promote as extensive colonization by listed plant species, these species should colonize over time as the habitat transitions from invasive plant stands to native dunes.
- As discussed, European beachgrass decomposes very slowly in herbicide-treated backdunes. Correspondingly, there has been a very slow rate of conversion from European beachgrass-dominated stands to what would probably develop under optimal conditions into Dune Scrub habitat. Not only do backdune areas receive less in the way of sand deposition from adjacent beaches, which could potentially jumpstart native community development, the

very slow decomposition rates of beachgrass, coupled with high amounts of thatch and detritus produced, may create somewhat of a legacy effect by leading to excessively high inputs of organic matter into the soils, which may favor establishment of weedy, non-native annual plant species over native dune and Dune Scrub ones, even if ample seed and vegetative propagule sources of native plant species are available. Therefore, the potential for success of restoring native vegetation communities in the backdune areas using herbicide is more questionable at this point than that in foredune and mid-dune areas, although restoration using mechanical removal poses the risk of sand remobilization into adjacent ranchlands and wetlands. Restored backdune areas will continue to be evaluated over the coming years to determine the trajectory of evolution – and success of restoration -- for these areas.

- While mechanical treatment undeniably has yield huge benefits for listed species at Abbotts, the indirect impacts to adjacent native habitats and ranchlands makes use of this method highly problematic. Any use of mechanical restoration in the future should be reduced in scale and carefully planned to avoid large-scale remobilization of decades of accumulated sands. Mechanical restoration could be used in mid-dune areas, for example, in conjunction with herbicide treatment of foredunes: decomposing European beachgrass and associated rhizomes would minimize wind-associated erosion of foredunes and transport of those sands inland, while allowing sand overwash to rebuild native dune communities on top of decomposing biomass. In the meantime, mechanical restoration would rejuvenate dune habitats where sand deposition rates are much lower and less likely to effect large-scale habitat conversion. Mechanical removal should NOT be used adjacent to wetlands, organic pastures, or existing native dune habitat. The Seashore had originally proposed to use mechanical removal in herbicide-treatment buffers, but the risk to these habitats and listed species using them is too great.
- Manual removal is most appropriate for initial treatment of sparse iceplant and re-treatment of European beachgrass in mechanically restored areas and within buffers. Manual removal is ineffective for initial treatment of European beachgrass due to the extensive depth of the rhizome system.
- Both external and internal iceplant removal projects have clearly documented that, regardless of method, removal of iceplant may not be enough, particularly in highly invaded areas due to legacy effects on soil chemistry. Removal of duff may improve “restorability” of these areas, but, ultimately, until measures can be found to counter these legacy effects, the park may want to direct its limited restoration resources to lightly to perhaps moderately invaded iceplant areas or areas that have only been invaded for a short while.

SUMMARY DISCUSSION ON CHANGES IN MONITORING APPROACHES

Monitoring of dune vegetation communities is challenging due to the wide disparity in percent cover of bareground relative to plants and large diversity in species present. Dune mat communities are characterized by large expanses of bare sand, colonized seemingly intermittently by a wide variety of small- to medium-sized herbs along with the occasional shrubs. In contrast, Dune Scrub communities have less to sometimes no bareground and are

largely colonized by medium- to large- sized shrubs, with herbs and grasses growing within the bare patches or underneath the shrub canopy.

The early Abbotts project emphasized monitoring of subplots within a larger macroplot, however, some of the statistical power for analyzing change was lost, because subplots technically represented replicates for the particular macroplot, not independent replicates. For the later Abbotts project, the Seashore staff wanted to increase the number of replicates and, thereby, the power of our statistical analyses, so we switched to 2X1 m plots. However, methods such as the 2X1 m plots – which have been used fairly successfully in other vegetative communities – did not appear to capture the vegetative character of Dune Mat communities very well, overemphasizing bareground and considerably underemphasizing certain, less common plant species. Staff was concerned that changes in Dune Mat communities could occur without this type of monitoring approach being able to detect these changes. Certainly, the large-scale invasion by European searocket of mechanically restored and native dune areas completely escaped detection despite these monitoring efforts, which raises some definite red-flags regarding the adequacy of the current protocol, at least within certain types of dune ecosystems. Results from data collected within the 44-acre native dunes at Abbotts between 2010 and 2015 suggested that anywhere from 30 (bareground) to 121 (percent native cover) permanent quadrats would be needed to detect at least a 20% change between monitoring years. There are currently 23 plots.

To remedy this, a different type of monitoring approach was used for the Abbotts Triangle and B Ranch North areas. This method relied on “line” transects actually established using a GPS and systematically collecting point cover and species frequency data along these GPS transects, which were systematically replicated throughout the Project Areas. Based on results, this method appeared to reflect more accurately the diversity of species within Dune Mat areas, but frequency data does not lend itself as well as cover data to summary calculations of total percent cover of vegetation and bareground, as well as cover of species within different successional communities. In addition, this method is dependent on systematic interspersions of points/frequency measurements for statistical power. It is difficult to maintain the required interspersions when the terrain within dunes becomes much more steeply undulating, with large dune hills and steep slopes, because staff cannot adequately maintain consistent pacing between points. In 2016, park staff explored a variant of this in the European beachgrass treatment areas at B Ranch North, which systematically collected point-intercept vegetation cover across the entire treatment area using GIS-generated “transects” that were navigated in the field with a hand-held GPS. The long-term value of this approach is not yet known, but it was quicker than the frequency method and seemed to provide more usable data.

Ultimately, it may be that dune monitoring may require a combination of approaches. For very sparsely vegetated Dune Mat communities within less steeply undulating terrain, GPS line transects collecting cover data may continue to represent the best choice – at least until another method can be developed. However, for more densely vegetated communities such as invaded, Dune Scrub, or later successional native dune communities – particularly in more steep or undulating terrain – the 2X1 m plot may remain the better solution, as it’s easier to navigate to randomized points, and it appears to capture both bareground and vegetative diversity quite well.

A sample size analysis for the 2X1 m plots established in the 50-acre herbicide treatment plots in the backdunes at Abbotts suggested that as few as 3 to 12 permanent plots would be needed to assess percent non-native cover, native cover, and some of the diversity indices with a 20% chance of detecting change between years: more plots would be needed, however, to adequately assess the percent cover of bareground (23). The backdune areas at Abbotts are likely to transition to Dune Scrub, so the 2X1 m plot method should be able to accommodate this vegetative shift. In addition, almost all of the native dune vegetation communities in AT&T are some variant of Dune Scrub, which is also the likely native plant target community for the invaded dunes, which were treated starting in 2015. This, combined with the steep and highly rugged terrain, suggested that the 2X1 m plot might be a better option for monitoring in this area, so park staff opted for this monitoring method for all of AT&T.

The optimal method – or methods – for monitoring vegetation in dunes may still yet to be developed. Seashore staff will keep exploring for ways to improving monitoring changes in these unique vegetation communities.

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**APPENDIX A. VEGETATION MONITORING AND PHOTOPOINT PLOTS AT ABBOTTS LAGOON
COASTAL DUNE RESTORATION AREA, B RANCH NORTH, AND AT&T COASTAL DUNE
RESTORATION PROJECT**

Restore Critical Dune Habitat to Protect Threatened and Endangered Species

Vegetation Monitoring - All Plots



Point Reyes National Seashore
Marin County, CA

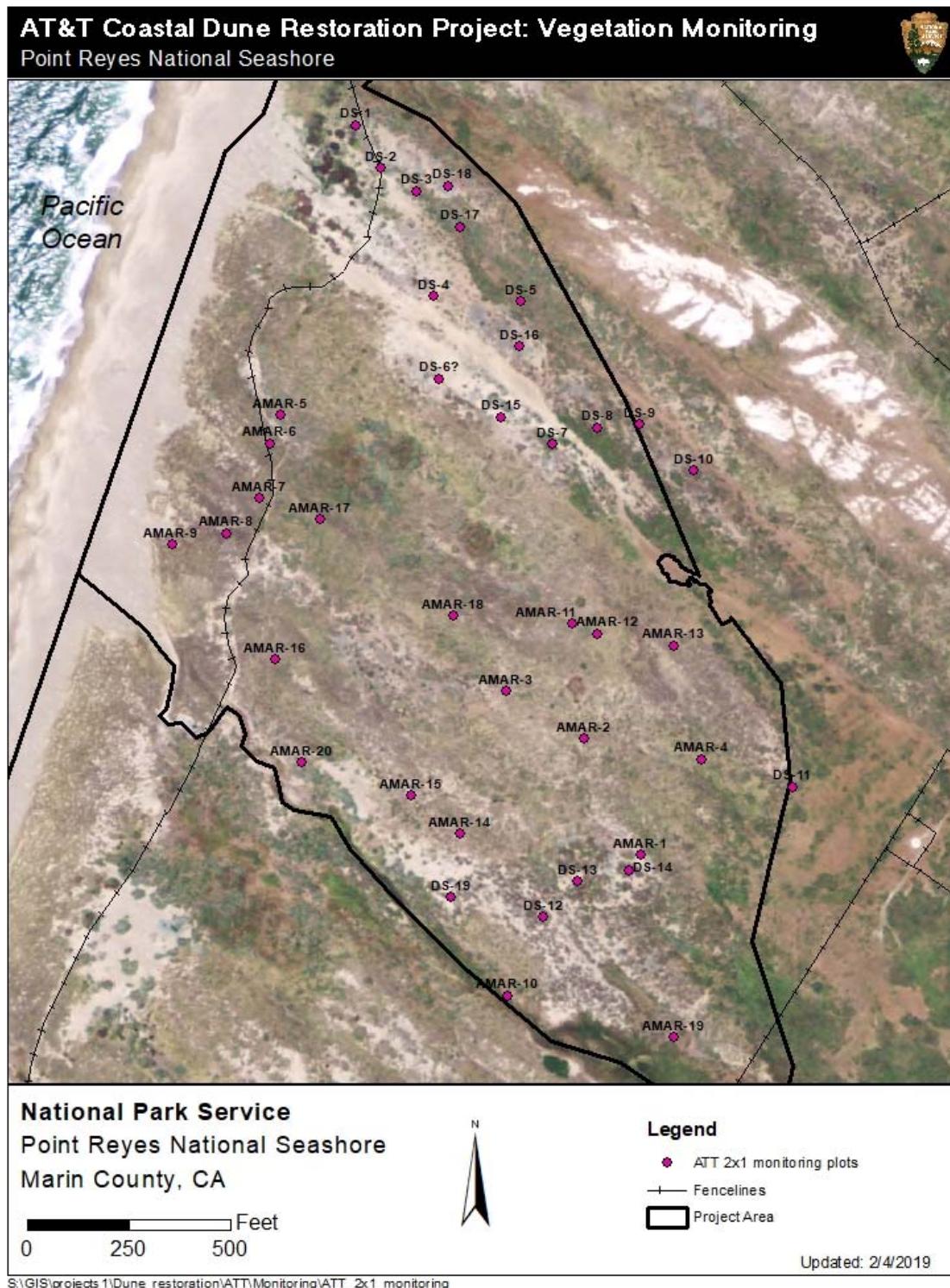
0 0.1 0.2
Miles



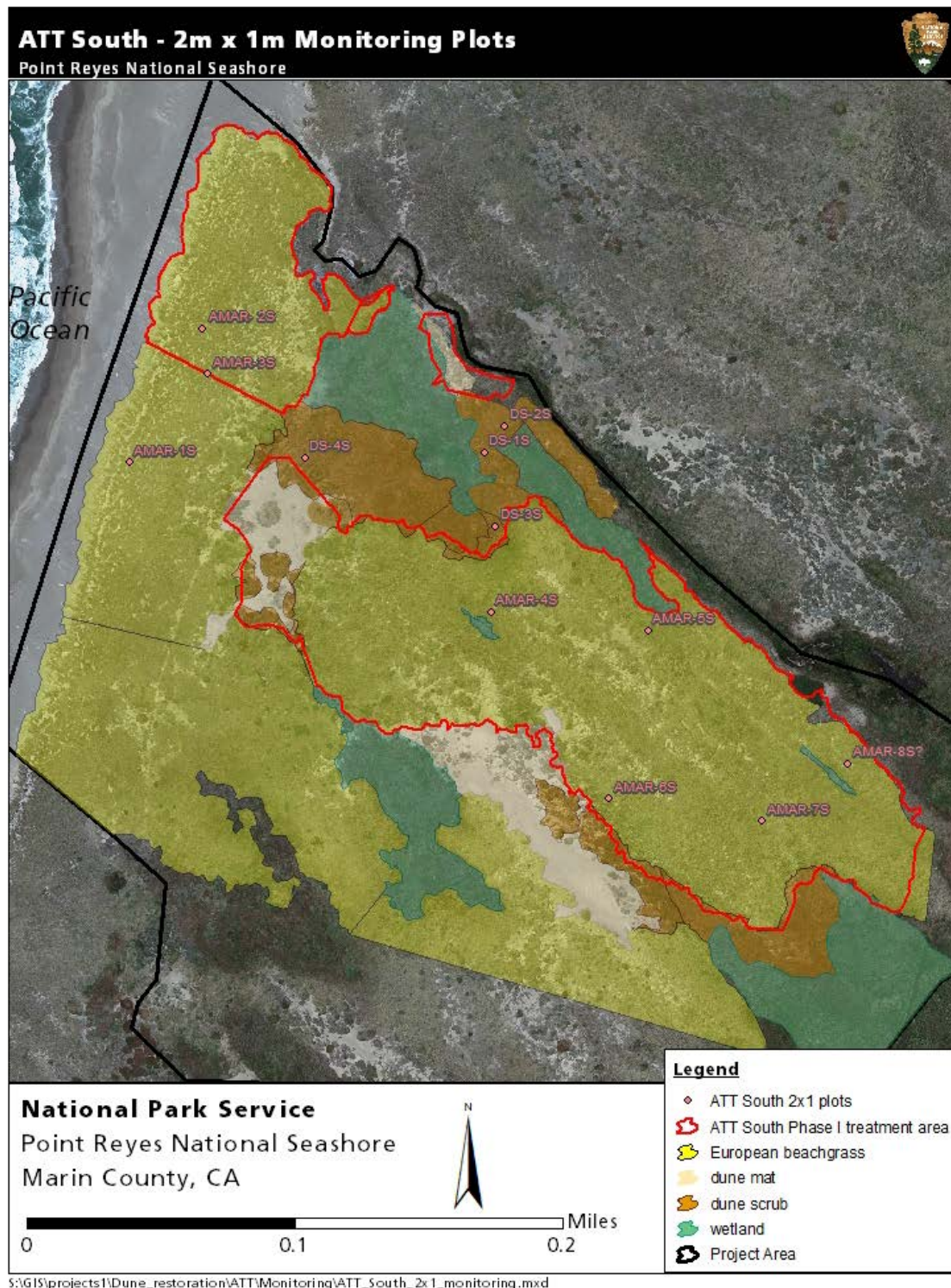
Legend

- Fence
- REFERENCE, BACKDUNE
- TREATMENT, FOREDUNE
- REFERENCE, FOREDUNE
- TREATMENT, BACKDUNE
- DELETED
- project area
- Mowed Fall 2012
- Access Road
- Phase I - mechanical 2011
- Phase II - herbicide 2011
- Phase III - herbicide 2012

Appendix A-2: ATT North - 2X1 Vegetation Monitoring Plots

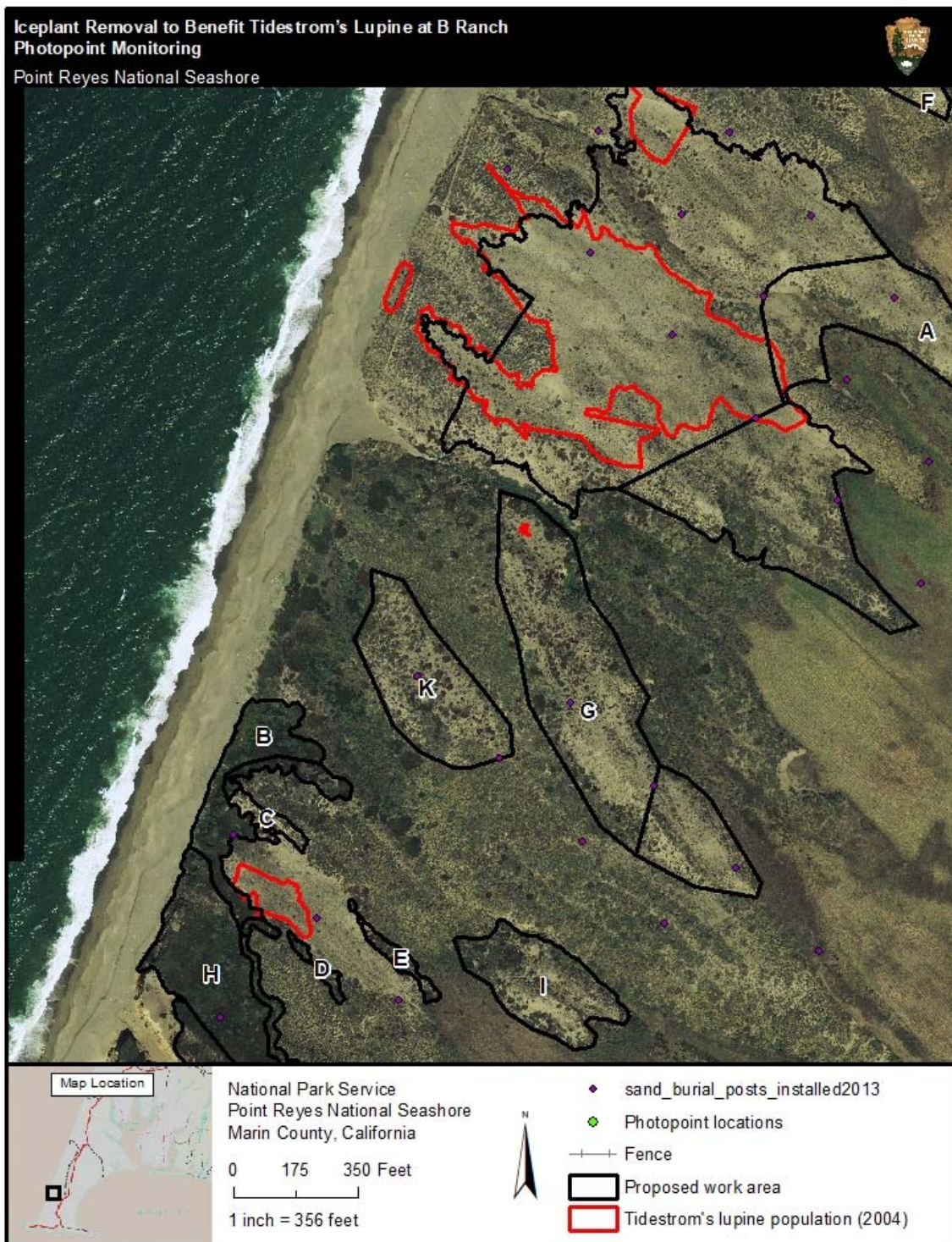


Appendix A-3: ATT South - 2X1 Vegetation Monitoring Plots

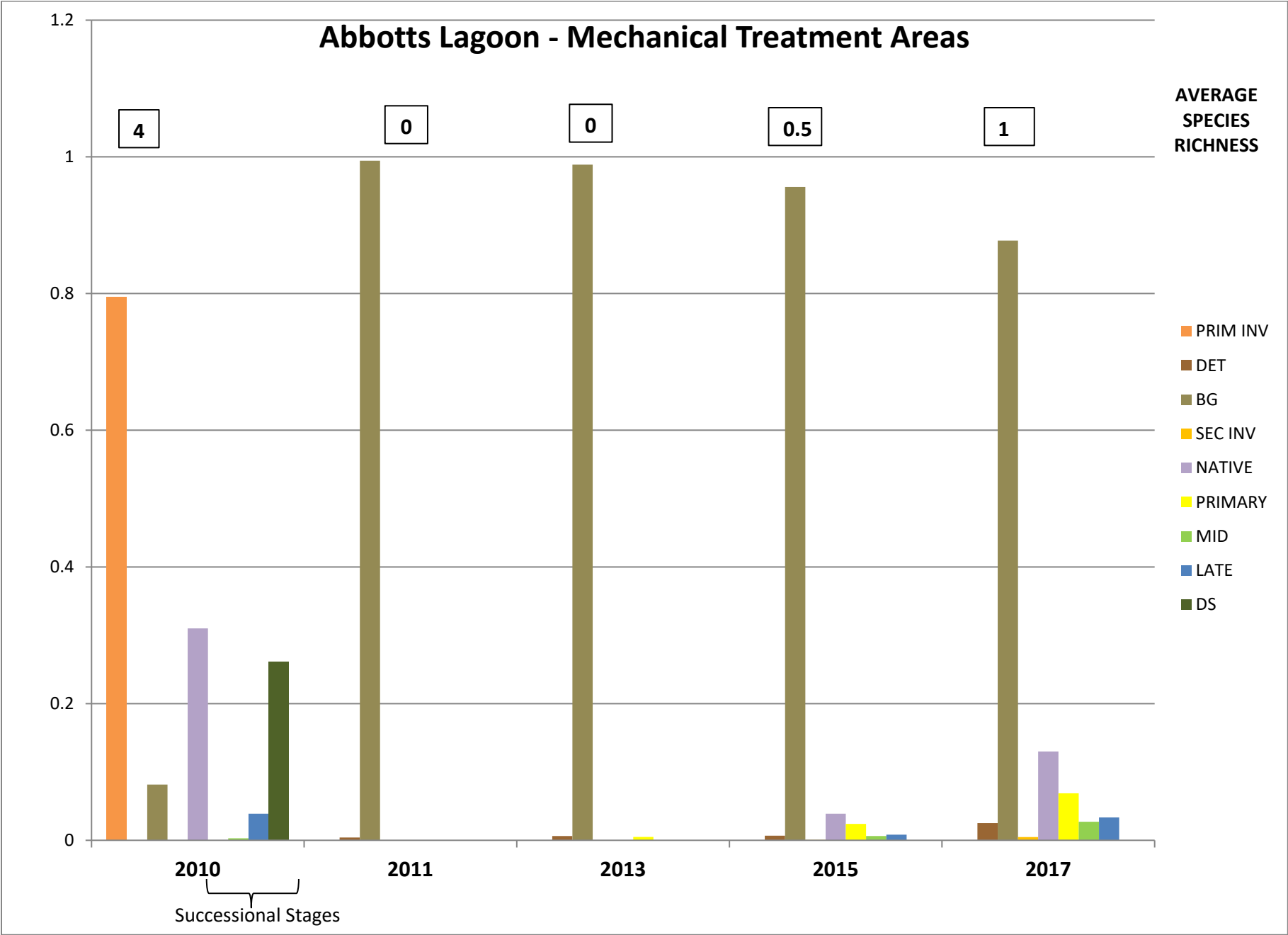


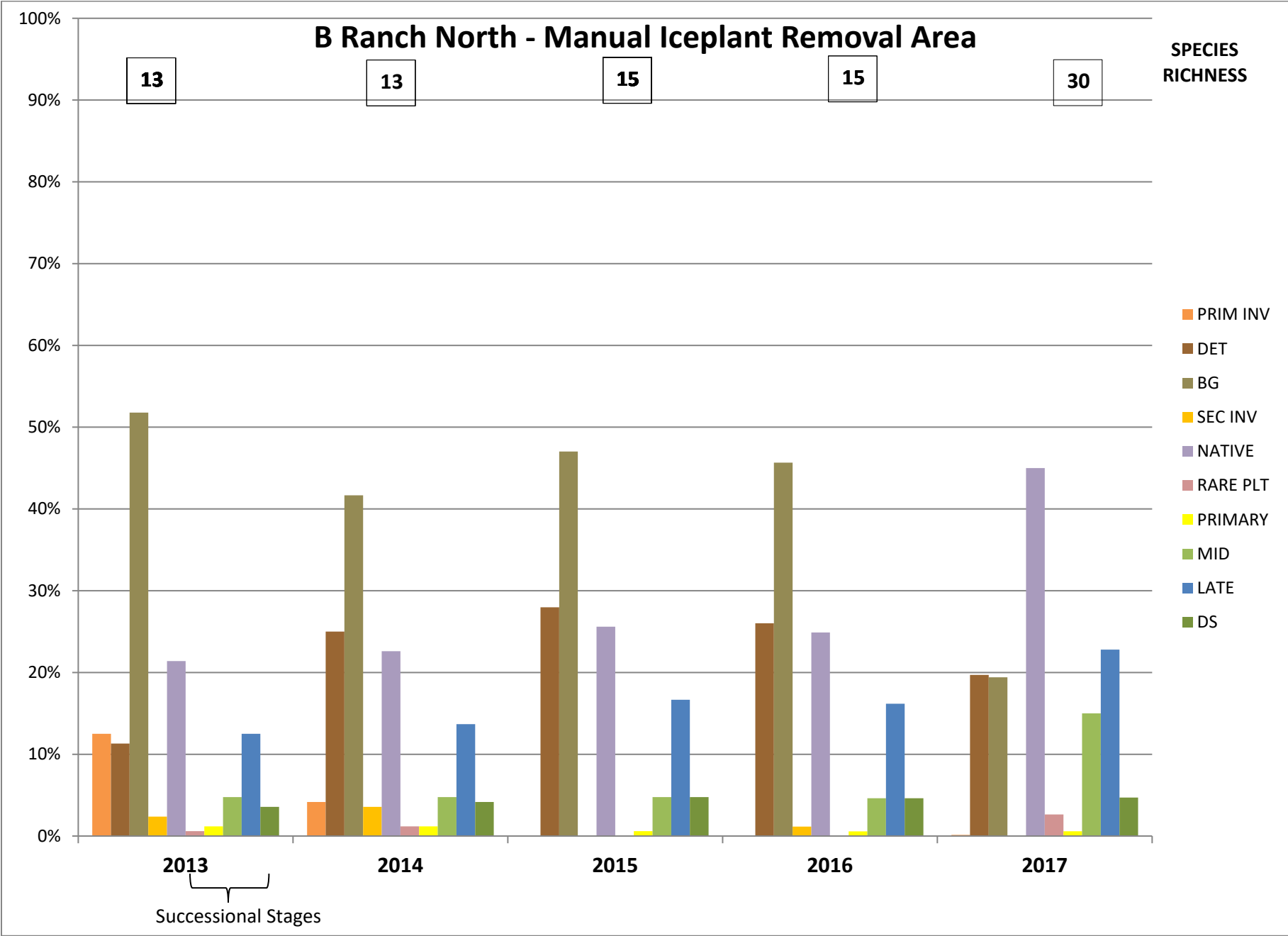
Appendix A-4. ATT South Phase I permanent monitoring plot locations. “DS” or Dune Scrub plots were chosen in native dune scrub habitat as reference sites, and “AMAR” plots were chosen in European beachgrass/iceplant-invaded areas. Note that plot AMAR-1S is outside of the Phase I treatment area – consequently, this plot was removed from statistical analyses.

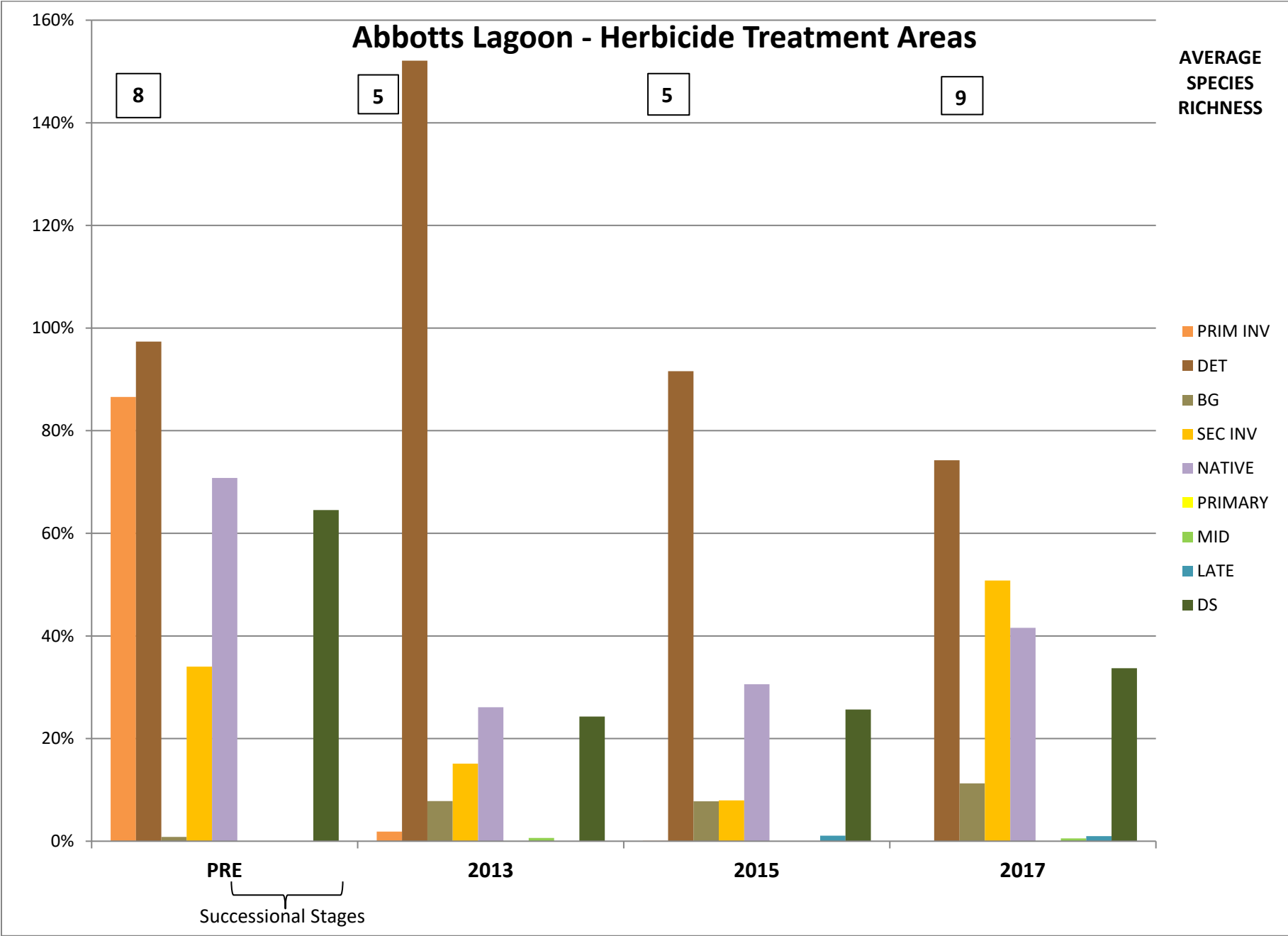
Appendix A-4: B Ranch North Photopoints

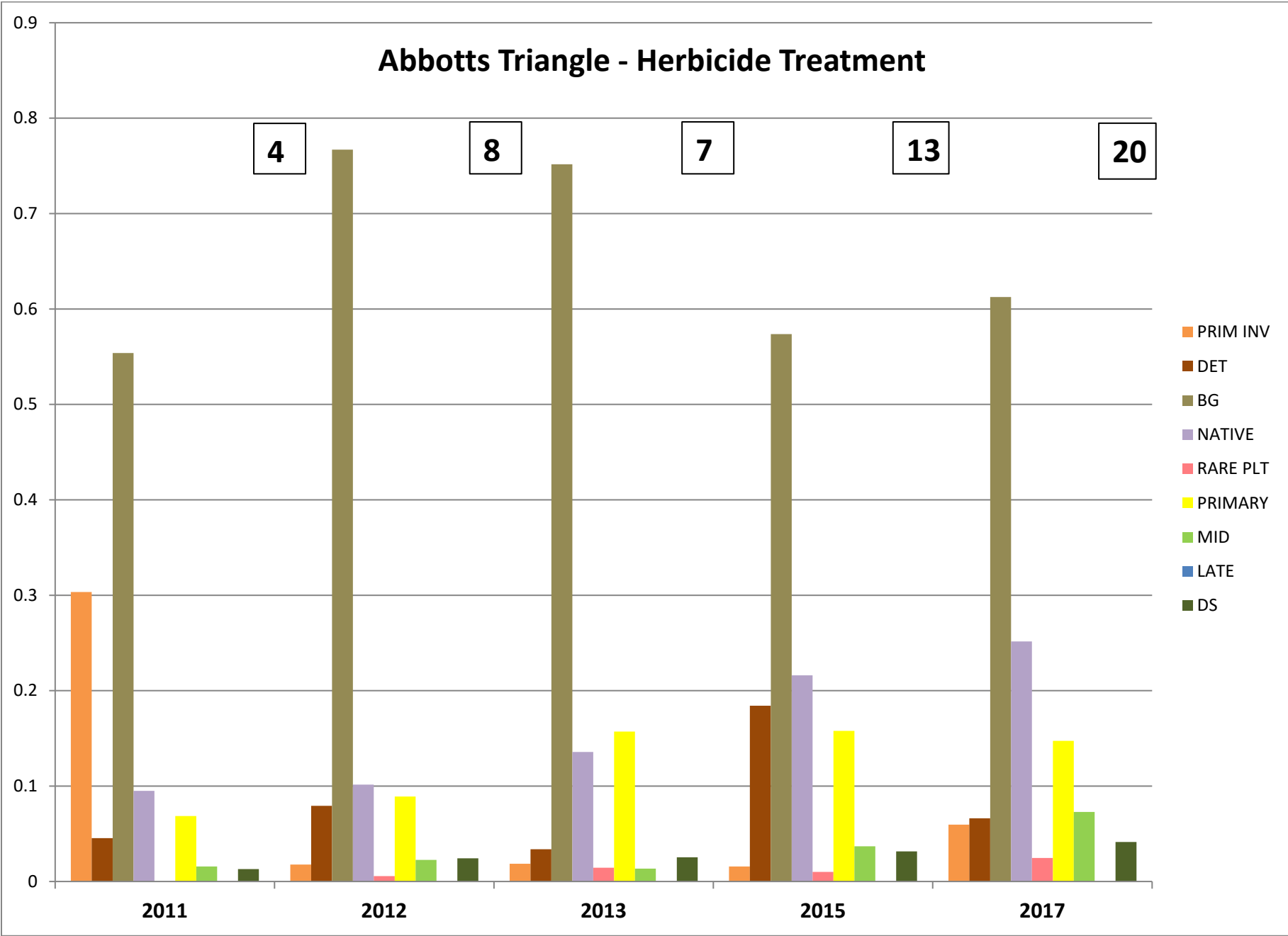


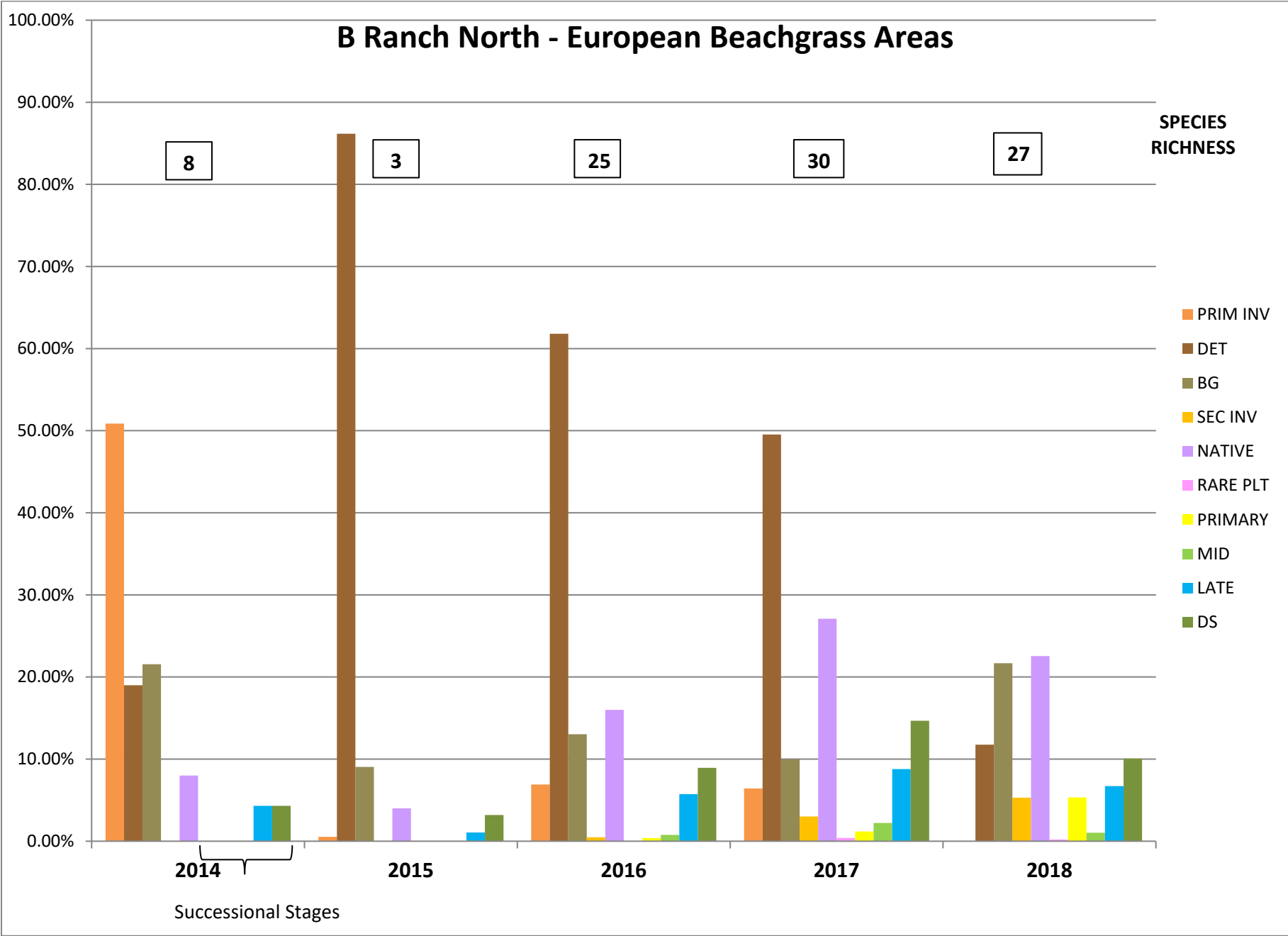
APPENDIX B. GRAPHS DISPLAYING VEGETATION COMMUNITY DATA FOR MULTIPLE DUNE RESTORATION PROJECTS AT POINT REYES NATIONAL SEASHORE



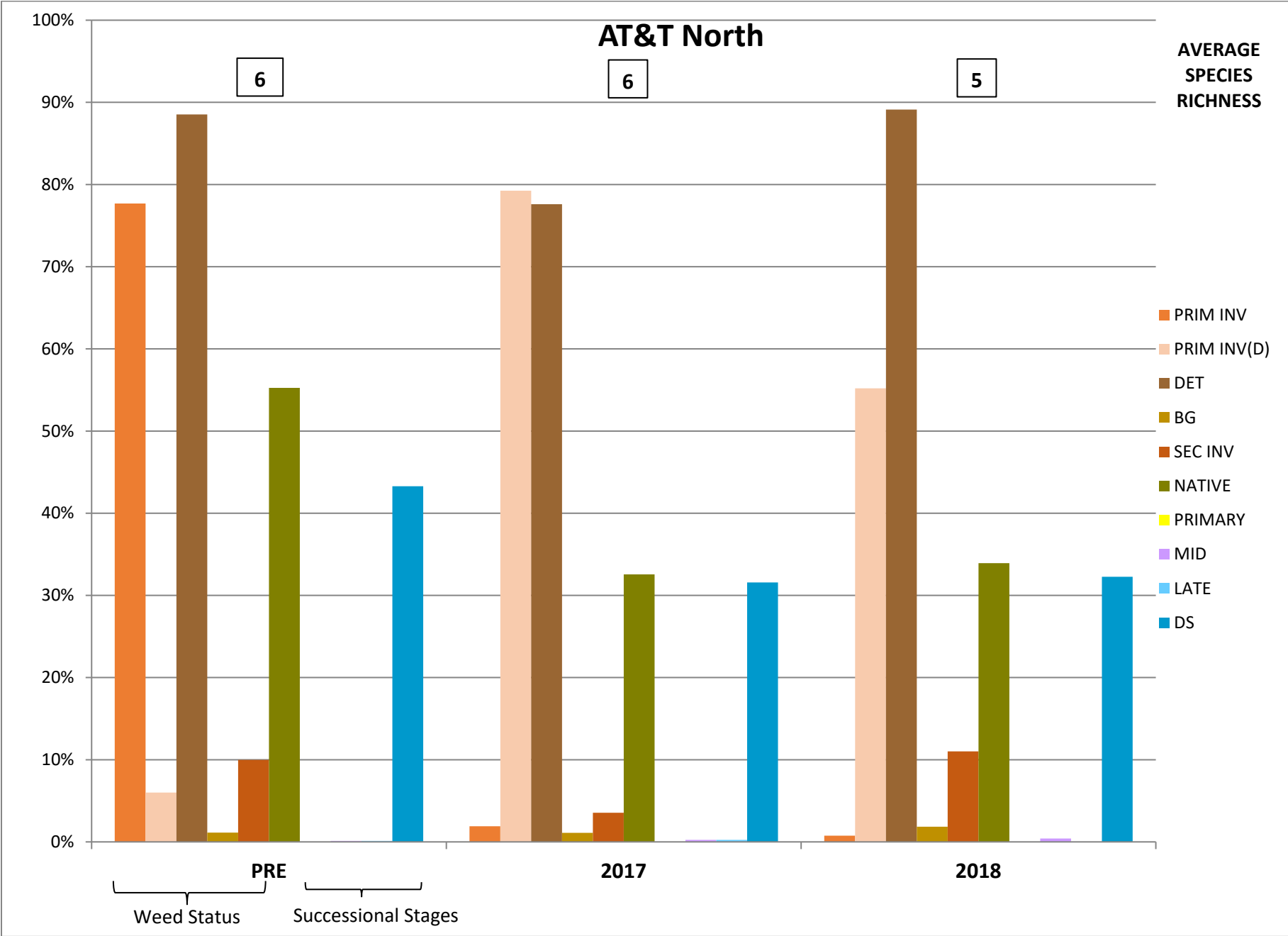








APPENDIX B-6



**APPENDIX C. POST-HERBICIDE TREATMENT OF EUROPEAN BEACHGRASS (*AMMOPHILA ARENARIA*)
AT ABBOTTS LAGOON**

Post-herbicide treatment of European beachgrass (*Ammophila arenaria*) at Abbotts Lagoon

Cody Ender and Lorraine Parsons
Point Reyes National Seashore
Point Reyes, CA 94956

January 27, 2017

Background

In fall 2012, a portion of the Phase II herbicide treated area (treated the previous year in 2011) received additional post-herbicide treatment with the intent of speeding up decomposition of dead European beachgrass (*Ammophila arenaria*; AMAR) and improving colonization by native dune plant species. Lingering biomass above- and below-ground poses an obstacle to successful dune restoration by reducing bare ground characteristic of native dune communities, occupying space that could be colonized by native species, and preventing sand remobilization. On the other hand, standing dead biomass may help to prevent excessive sand movement into nearby pastures or low-lying swales and existing native dune habitats and may help prevent colonization by secondary invaders or other non-native species originally in much lower abundances prior to restoration. European beachgrass decomposes very slowly: dead aboveground biomass -- as well as rhizomes belowground -- have been found at the Abbotts Lagoon restoration area up to five years post herbicide treatment (Figure 1). This pattern of persistent dead biomass has also been observed by California Department of Parks and Recreation (CDPR) staff at Bodega (Chris Heintzelman, CDPR, *pers. comm.*).

In order to accelerate decomposition of European beachgrass, the Seashore experimented with mowing or scraping with a bulldozer one year after herbicide treatment. The mowing treatment (mow) removed vegetation approximately 4- 5 inches on average

above ground level leaving much of the basal portion of plants and their root systems intact. The



Figure 1. Standing dead European beachgrass (above) and dead rhizomes (below) in 2016.

scraping treatment (doze) removed vegetation just below ground level with minimal root disturbance. For comparison, some areas received no post-treatment (control). The post-herbicide treatments were meant to decrease standing biomass without substantial soil disturbance and loss of native species that occurred with mechanical removal. These three post-treatments (mow, doze, and control) were all located in the Phase II herbicide treatment area (Figure 2).

Other European beachgrass removal projects using herbicide have experimented with use of pre-treatment measures such as prescribed burning or mowing to remove beachgrass thatch, stimulate re-growth, and improve uptake of herbicide (Hyland and Holloran 2005). At Bodega dunes, staff and volunteers have actually pulled by hand individual dead European beachgrass clumps from the ground and stockpiled clumps in central locations to reduce standing dead biomass (C. Heintzelman, *pers. comm.*). However, we are not aware of other attempts at post-treatment measures to remove dead biomass.

A study by Pearson et al. (2016) showed that, after control of the primary target invasive species, it is very common for those target species to be replaced by secondary invaders instead of native species. We wanted to encourage native species establishment without also acquiring more invasive plant species. We were especially concerned about bush lupine (*Lupinus arboreus*; LUAR) becoming dominant in restored areas because bush lupine fixes nitrogen in the soil, produces abundant nitrogen-rich leaf litter, and provides shade, all characteristics that could encourage weedy annuals (Zink et al. 1995, Maron and Connors 1996). Bush lupine became prolific enough in the mechanically restored dunes by 2016 that efforts were made to remove seedlings and adult plants. We hypothesized that the doze treatment -- the treatment with more soil disturbance and more similar to the mechanically treated dunes -- might result in increased bush lupine cover.

Methods

To compare post-herbicide treatments with no post-treatment, we monitored 26 randomly chosen and permanent plots for species composition and cover in the summer of 2016 (five years after herbicide treatment, four years after post-treatment). From a larger set of randomly generated points, we hand-selected a subset of points to monitor within each treatment type (Figure 2). Existing permanent 2x1 plots that landed in each of the treatment areas were also included in data analysis (M-17 in doze, N-12, N-14, X-6 and X-7 in mow and L-13 and K-15 in control). In total, we sampled 10 plots from the doze treatment, nine (9) plots from the mow treatment, and seven (7) plots that received no post-herbicide treatment.

Seashore staff used the same sampling methods as the permanent monitoring plots at Abbotts Lagoon, using a 2x1 m quadrat to quantify species cover using point-intercept sampling and identifying all species present in the plot (Parsons et al. 2016). All plots were located in restored dunes so we used the "reduced" sampling protocol, only reading 68 points on the quadrat instead of the full 171 points.

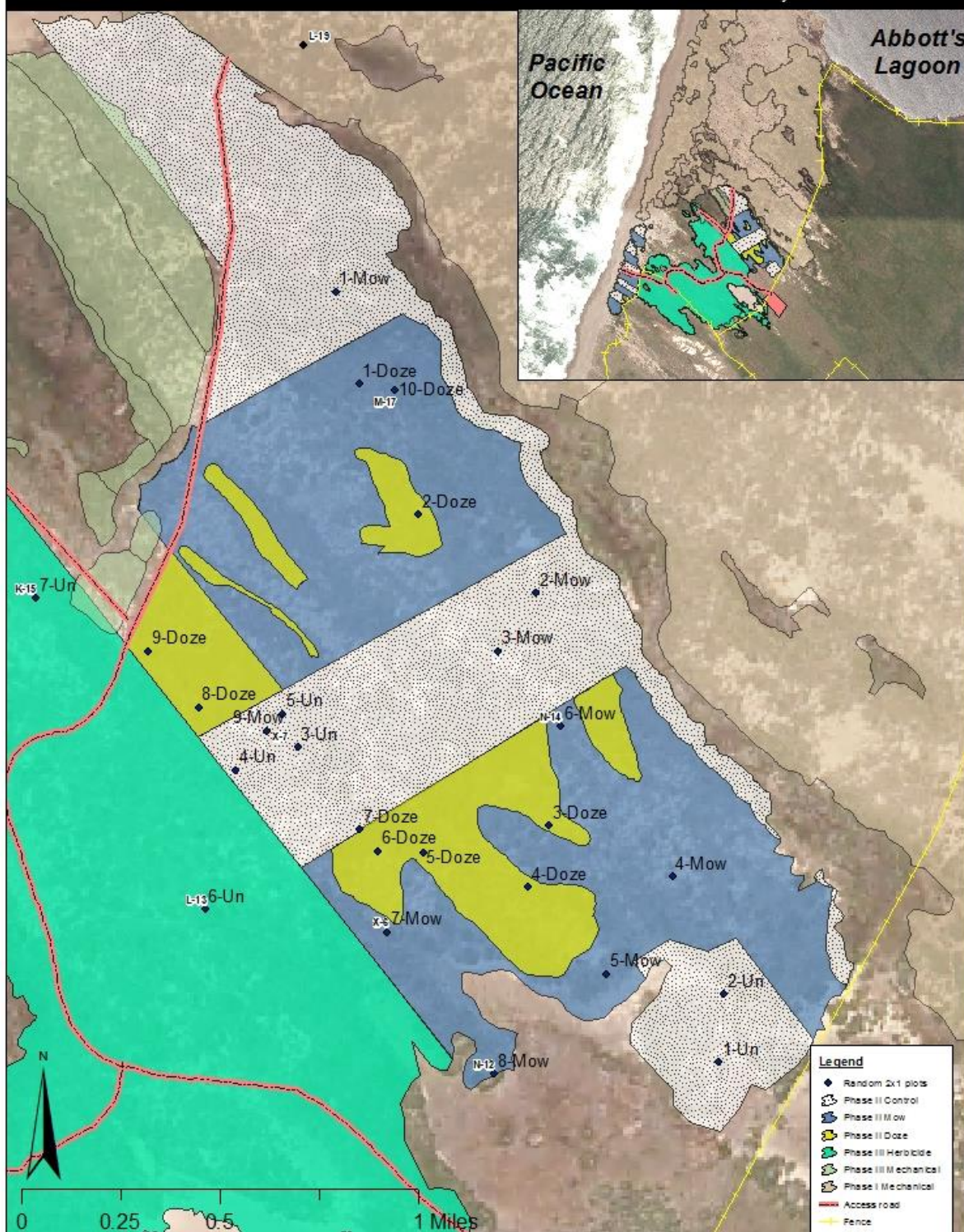
Because one of the main goals of post-herbicide treatment was to reduce standing dead European beachgrass, we analyzed overall dead European beachgrass cover (dead AMAR) as well as dead cover separated into three height classes (dead AMAR short, medium, and tall). We also analyzed bush lupine

Abbott's Lagoon Dune Restoration - Vegetation Monitoring

Random 2x1 Plots Mow vs. Doze



Point Reyes National Seashore



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Figure 2. Locations of post-herbicide treatments and subsequent 2x1 monitoring plots.

cover separately to see if any of the treatments had an effect on the colonization of this secondary invader. We grouped all other species (including European beachgrass and bush lupine) into native and non-native functional groups to test for differences in native cover, non-native cover, native species richness, and non-native species richness across the three treatment groups. Additionally, we tested for differences in overall species richness, Shannon-Weiner diversity, evenness, detritus, and bareground cover.

For all response variables except the dead AMAR height classes, we fitted generalized linear models (GLM) with post-herbicide treatment (mow, doze, or control) as an independent factor. All cover responses (native, non-native, dead AMAR, bush lupine, detritus, and bareground cover) were fitted using a binomial distribution. Species richness, native species richness, and non-native species richness were fitted using a Poisson distribution. Diversity and evenness were fitted using a Gaussian distribution. Whenever treatment had a significant effect, Tukey multiple comparisons tests were used to evaluate differences among the means. Contingency table analysis was used to assess changes in the proportion of dead European beachgrass in the short, medium, and tall height classes. We performed non-metric multidimensional scaling (NMDS) analysis on individual species cover in the 2x1 m plots and then tested for differences in species composition among the three post-herbicide treatments using plot ordination scores from NMDS as dependent factors (NMDS1 and NMDS2) and post-herbicide treatment as an independent factor.

Results

Both mow and doze post-herbicide treatments significantly reduced standing dead European beachgrass compared to control plots (mow, $df=23$, $z=-9.53$, $p<0.0001$; doze, $df=23$, $z=-13.16$, $p<0.0001$), from 35.5 ± 11.5 (S.E.)% in control plots to $5.1 \pm 2.9\%$ in doze and $13.0 \pm 5.4\%$ in mow plots (Figure 3a). The proportion of dead European beachgrass in each of the height categories (short, medium, or tall) varied

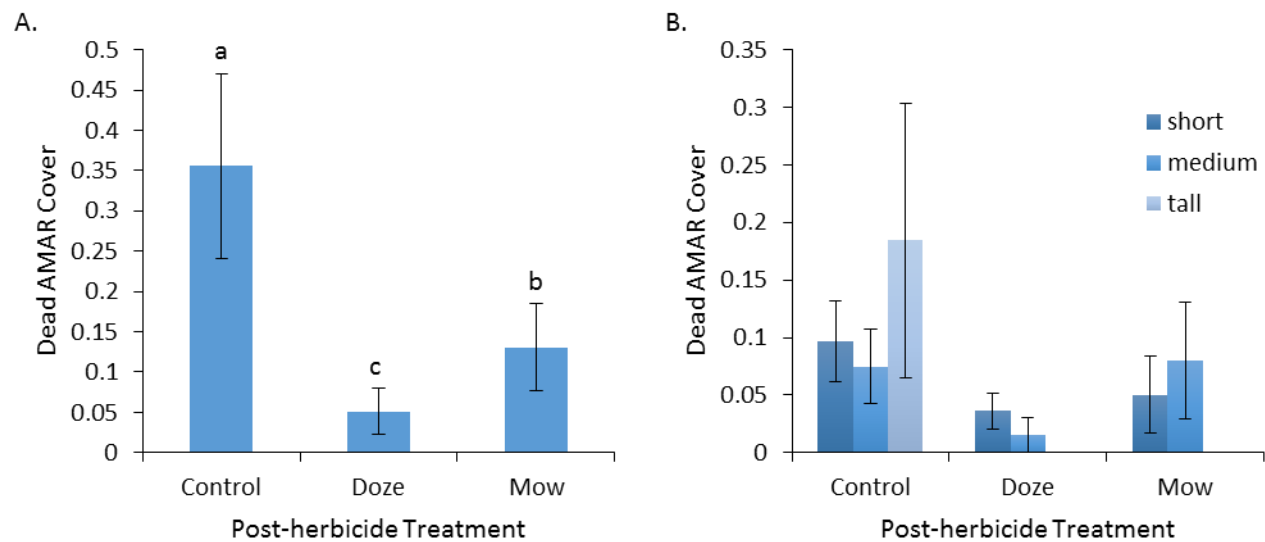


Figure 3. Mean ($\pm$ S.E.) cover of all dead European beachgrass (a) and dead European beachgrass by size class (b) as a function of post-herbicide treatment (control, doze or mow). Letters above bars correspond to the results from Tukey multiple comparison tests.

significantly with post-herbicide treatment ($X^2 = 15.49$, $df=4$, $p=0.004$, Figure 3b).

Post-herbicide treatment had a significant effect on non-native species cover, with lowest cover in the doze treatment ($3.5 \pm 1.7\%$, $df=23$, $z=-5.66$, $p<0.0001$) and with no difference between the mow treatment ($15.1 \pm 6.6\%$) and the control ($11.5 \pm 2.6\%$; $df=23$, $z=1.96$, $p=0.05$; Figure 4a). Native cover was unaffected by post-herbicide treatment (control, $24.3 \pm 8.9\%$; mow, $21.6 \pm 10.7\%$, $df=23$, $z=-1.17$, $p=0.24$; doze, $22.8 \pm 11.0\%$, $df=23$, $z=-0.64$, $p=0.52$; Figure 4b). Another focal species with invasive tendencies, bush lupine (LUAR), was only found in doze plots, yet we failed to find any significant effect of treatment on LUAR cover (mow, $df=23$, $z=0$, $p=1$; doze, $df=23$, $z=0.01$, $p=0.99$). One of the most visible effects of post-herbicide treatment was on bareground cover. Treatment had a significant effect on bareground cover, with the doze treatment increasing bareground from $11.8 \pm 4.2\%$ in control plots to $60.1 \pm 6.4\%$ in doze plots ($df=23$, $z=16.79$, $p<0.0001$). The mow treatment increased bareground to a

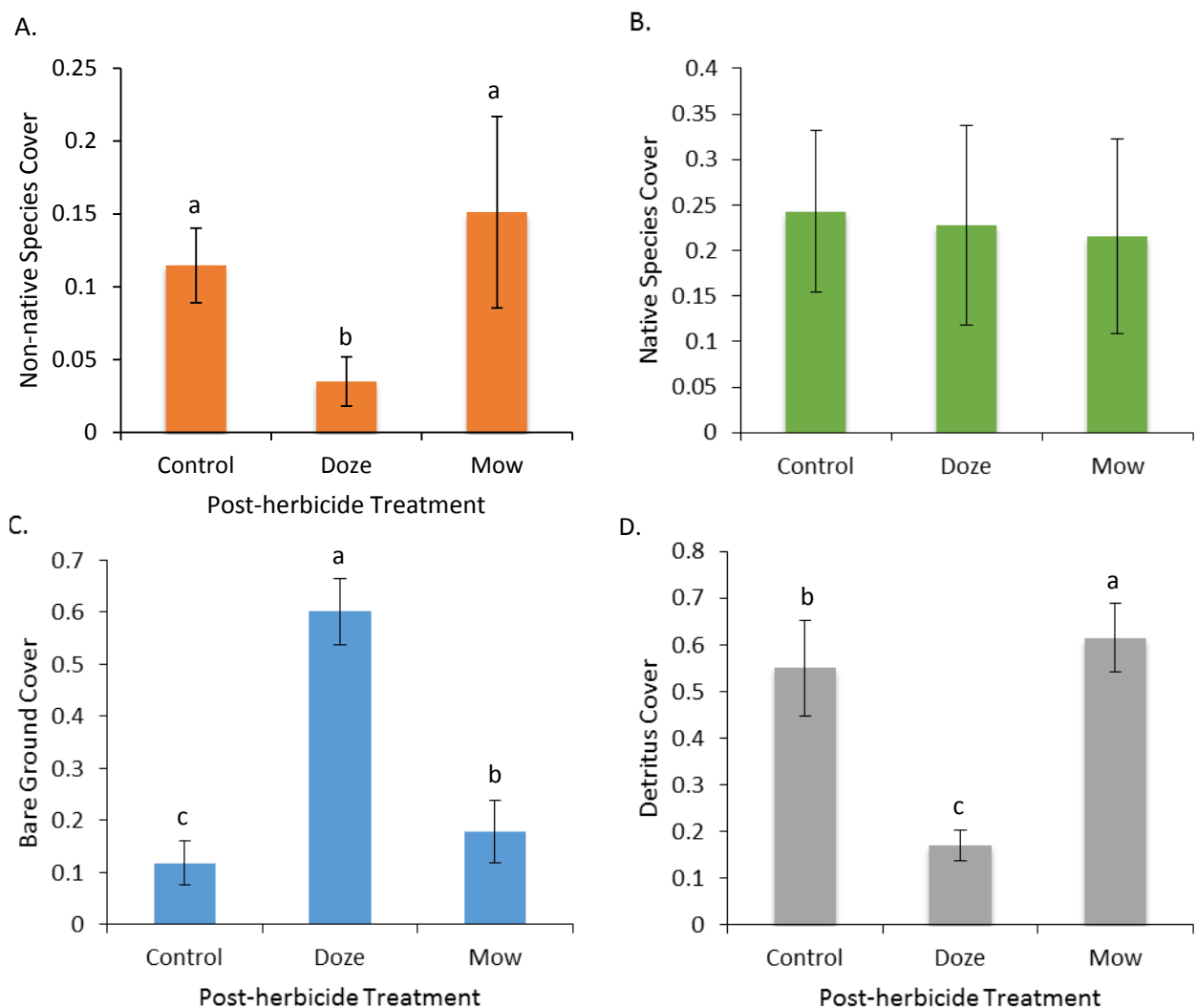


Figure 4. Mean (± S.E.) cover of non-native species (a), native species (b), bare ground (c), and detritus (d) as a function of post-herbicide treatment (control, doze or mow). Letters above bars correspond to the results from Tukey multiple comparison tests.

lesser degree with $17.8 \pm 6.1\%$ bareground cover in those plots ($df=23$, $z=3.08$, $p=0.002$; Figure 4c). Detritus was significantly increased compared to the control in mow plots (control, $55.0 \pm 10.3\%$; mow $61.5 \pm 7.3\%$ $df=23$, $z=2.43$, $p=0.02$). Conversely, the doze treatment decreased detritus compared to the control (doze $17.0 \pm 3.3\%$, $df=23$, $z=-14.61$, $p<0.0001$; Figure 4d).

Species richness varied significantly among post-herbicide treatments, with decreased richness in the doze treatment compared to the control (control, 6.0 ± 0.8 ; doze, 2.2 ± 0.6 ; $df=23$, $z=-3.81$, $p=0.0002$) but no significant difference between mow treated areas and the control (4.6 ± 0.4 ; $df=23$, $z=-1.25$, $p=0.21$; Figure 5a). Non-native species richness of doze plots was significantly lower than control plots (control, 2.7 ± 0.5 ; doze, 0.8 ± 0.2 ; $df=23$, $z=-2.90$, $p=0.004$), while non-native richness of mow plots was not significantly different from either the control or doze treatments (mow, 1.8 ± 0.3 ; $df=23$, $z=-1.25$, $p=0.21$; Figure 6a). Native species richness showed a similar pattern with significantly lower values in doze plots (control, 3.3 ± 0.8 ; doze, 1.4 ± 0.5 ; $df=23$, $z=-2.52$, $p=0.01$) and no significant difference in mow plots when compared to doze plots or the control (mow, 2.8 ± 0.5 ; $df=23$, $z=-0.58$, $p=0.56$; Figure

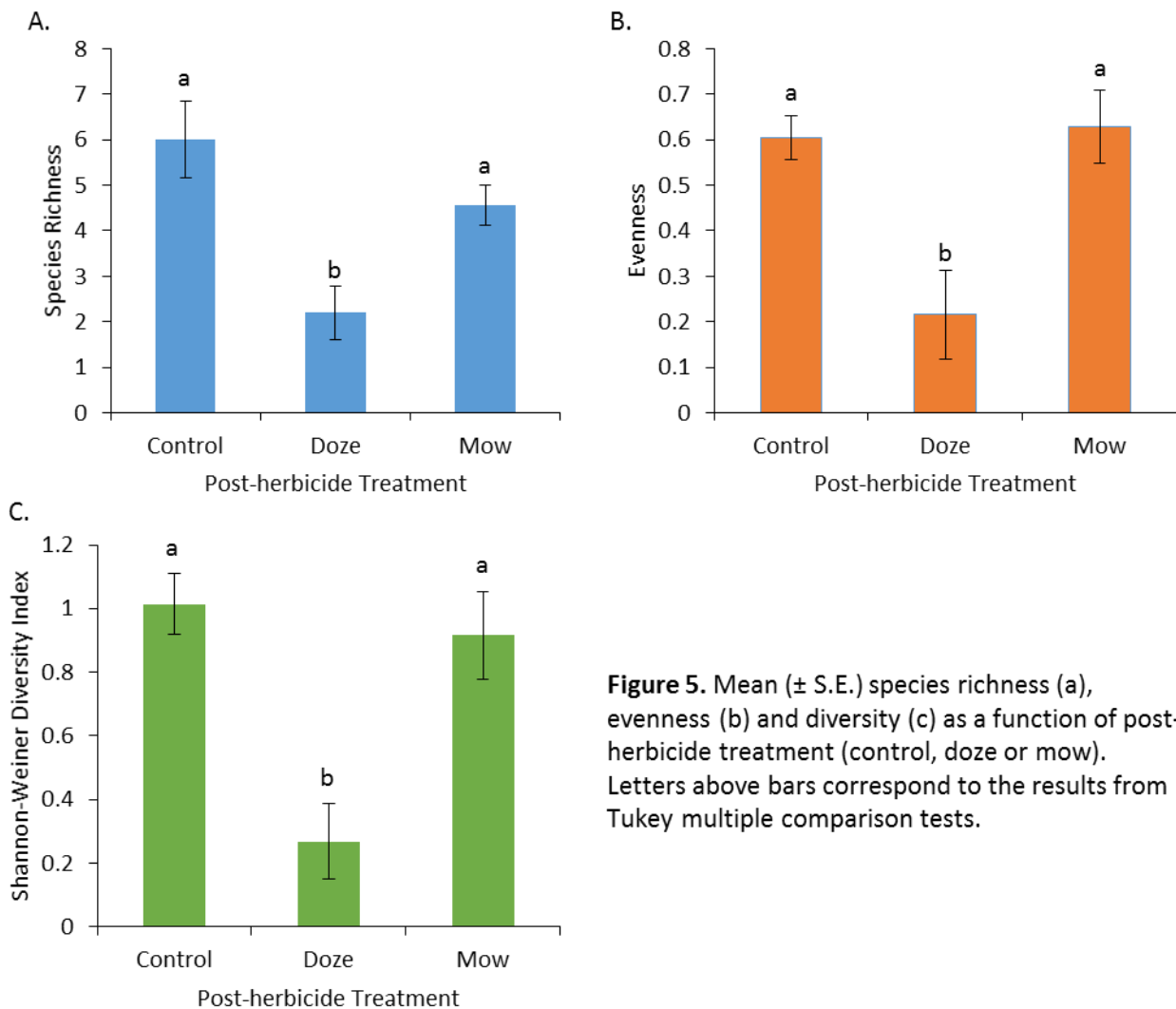


Figure 5. Mean ($\pm$ S.E.) species richness (a), evenness (b) and diversity (c) as a function of post-herbicide treatment (control, doze or mow). Letters above bars correspond to the results from Tukey multiple comparison tests.

6b). Doze plots had significantly lower species evenness than control plots (control, 0.61 ± 0.05 ; doze, 0.22 ± 0.10 ; $df=23$, $z=-3.18$, $p=0.004$), but we failed to find any significant difference in mow plots with either doze or control plots (mow, 0.63 ± 0.08 ; $df=23$, $z=0.19$, $p=0.85$; Figure 5b). Similarly, we found significantly lower diversity (Shannon-Weiner diversity index) in doze plots compared to controls (control, 1.01 ± 0.09 ; doze, 0.27 ± 0.12 ; $df=23$, $z=-4.17$, $p=0.0004$), but failed to detect a difference in mow plots with either control or doze plots (mow, 0.92 ± 0.14 ; $df=23$, $z=-0.54$, $p=0.60$, Figure 5c).

We found significant differences in species composition among post-herbicide treatment groups along NMDS axis 1 ($F_{2,23}=34.9$, $p<0.0001$), but no significant difference along NMDS axis 2 ($F_{2,23}=0.38$, $p=0.69$; Figure 7). The NMDS analysis showed species overlap between mow and control plots and a separation along the first axis with doze plots (Figure 7). Doze plots clustered lower on the first NMDS axis along with bareground, rhizomes from dead European beachgrass, woolly lotus (*Lotus heermanii*), curly leaf monardella (*Monardella sinuata* ssp. *nigrescens*), bush lupine, and gumplant (*Grindelia stricta*). Control and mow plots clustered higher along the first NMDS axis along with detritus, ripgut grass (*Bromus diandrus*), dead European beachgrass, coyote brush (*Baccharis pilularis*), woodland groundsel (*Senecio sylvaticus*), fairy mist (*Pterostegia drymarioides*), miner's lettuce (*Claytonia perfoliata*), sheep sorrel (*Rumex acetocella*), bracken fern (*Pteridium aquilinum*), and sand mat (*Cardionema ramosissimum*).

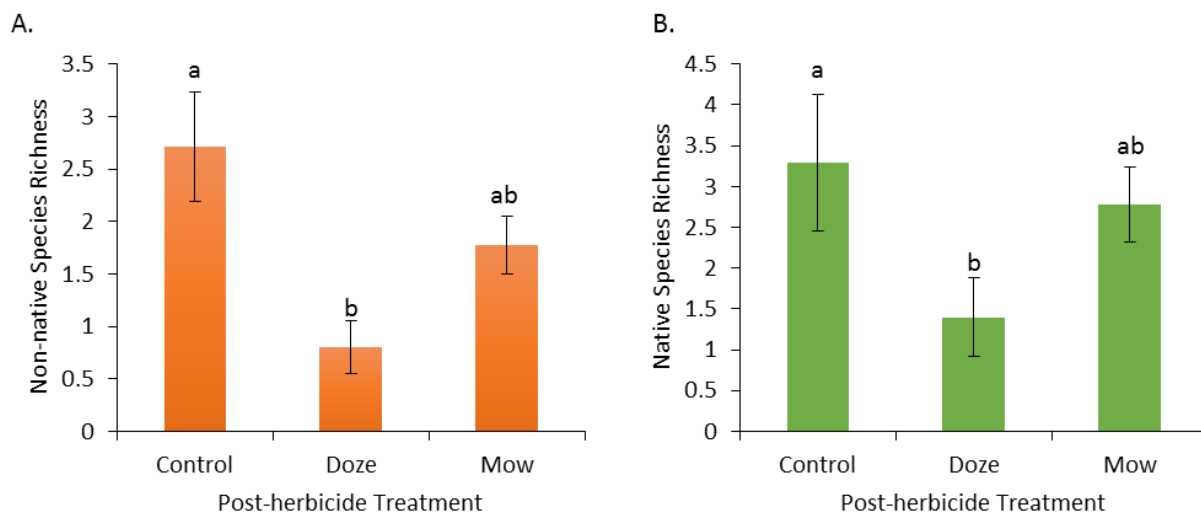


Figure 6. Mean ($\pm$ S.E.) non-native species richness (A) and native species richness (B) as a function of post-herbicide treatment (control, doze or mow). Letters above bars correspond to the results from Tukey multiple comparison tests.

Conclusions

Overall, both post-herbicide treatments met one of the treatment goals -- reduced dead European beachgrass cover -- but have not yet shown an increase in native species establishment. The doze treatment was the more extreme of the two treatments resulting in the lowest dead European beachgrass cover, non-native species cover, and the highest amount of bareground, but also the lowest species richness, evenness, and diversity. The mow treatment also reduced dead standing European

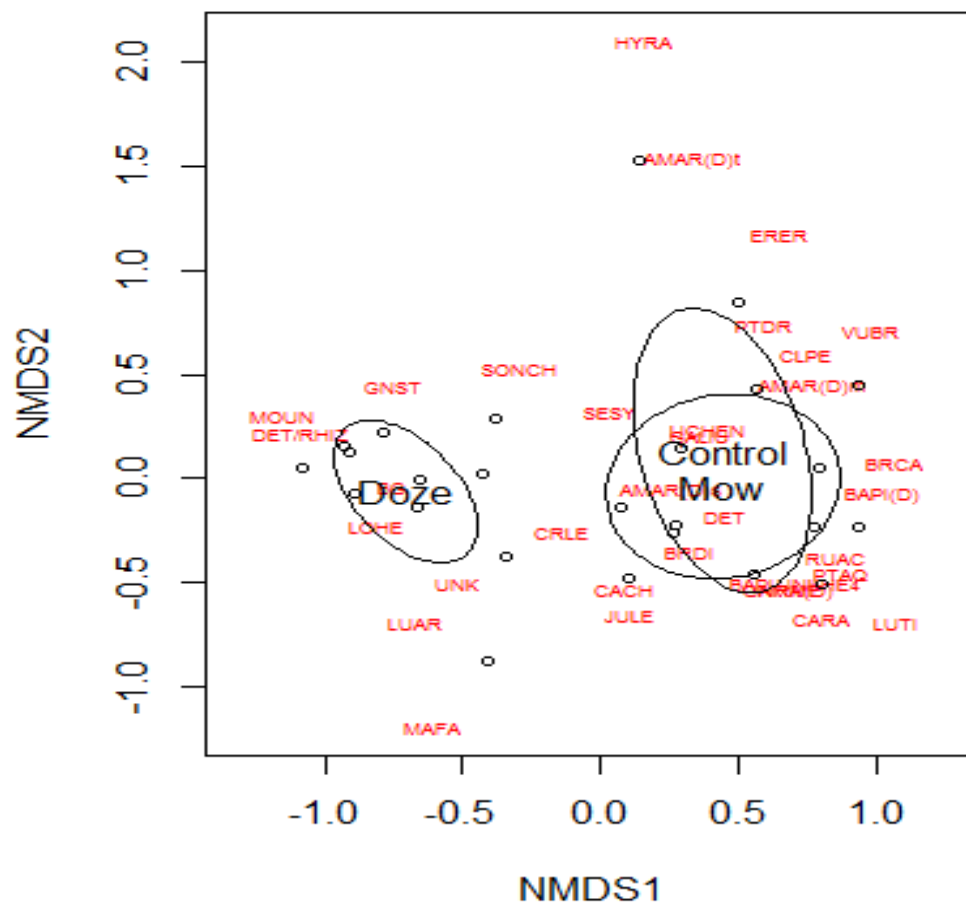


Figure 7. Non-metric multidimensional scaling multivariate analysis of plots from control, doze or mow post-herbicide treatments.

beachgrass, but was more similar to the control in terms of having higher non-native species cover, species richness, evenness, and diversity than doze plots. Native species cover did not differ among the three treatments.

While bush lupine definitely appears to be a more prominent species in the doze and, to a slightly lesser extent, mow treatments than control ones, our results were unable to discern any statistically significant difference, possibly because statistical power was too low. This issue underscores some of the difficulties with use of the 2X1 plots for evaluating vegetation changes in the restored dunes: due to the vagaries of random plot placement and heterogeneity in spatial patterns of dune species cover, the plots do not always capture the full complexity of what is going on vegetation-wise in the dunes. However, we have yet to find a time-efficient and yet biologically powerful vegetation monitoring method to effectively replace this one, as vegetation sampling in the dunes can be quite complicated due to extreme topographic variability in foredunes and backdunes.

Post-herbicide treatment reduced standing dead European beachgrass cover to 13% for the mow treatment and 5% for the doze treatment. Considering that the control with no post-treatment had an average cover of 35%, treatments were successful from the standpoint of reducing standing dead biomass. Of the remaining dead plants, there was a lower proportion of tall plants in mow and doze treated plots as well as a lower proportion of medium-sized plants in doze treated plots. Plots with the doze treatment also had a higher proportion of short dead European beachgrass. An overall decrease in the volume and height of standing dead biomass should increase light available for new plant establishment. Although standing aboveground biomass of dead AMAR was also reduced in the mow treatment, detritus was higher in these treatment areas than doze ones, which could reduce germination of both native and non-native species (Xiong and Nilsson 1999) or possibly tip the establishment success scales toward non-native species.

While our monitoring efforts did not evaluate belowground variables, post-herbicide treatment did not appear to affect belowground rhizomes, which continued to stabilize the soil. Less sand movement might make it less likely that early (or even mid-) successional dune species will establish in these areas (Zhang and Maun 1990, Maun 2004). Interestingly, NMDS results suggest that doze plots have supported colonization of mid- to late-successional dune species such as woolly lotus, curly leaf monardella, and gumplant, while the species assemblage in the mowed and control plots (coyote brush, fairy mist, bracken fern, miner's lettuce) are more characteristic of dune scrub communities, with the possible exception of sand mat, a species common in late-successional dune communities. As Phase II is largely a backdune area, somewhat removed from the shore and ocean- and wind-associated movement of sand, we have assumed that the evolutionary trajectory in terms for Phase II in terms of native vegetation communities would be a dune scrub assemblage, however, these results suggest that disturbance – natural or anthropogenic – could shift evolution on a different trajectory.

There was no increase in native species cover for either post-herbicide treatment, and the doze treatment actually had the opposite from the desired effect, decreasing native species richness (there was no change in native species richness in mow plots compared to control plots). Additional monitoring in the future may show a different story, but as of now, the post-herbicide treatments show no advantage compared to the control in terms of native species colonization.

In addition to decreasing standing dead European beachgrass and increasing native plant establishment, we wanted to avoid colonization by other exotic and/or invasive species. The more destructive doze treatment decreased both non-native species cover and richness, while the mow plots showed no significant difference compared to the control. Fewer non-native species in the doze plots can likely be attributed to the destruction caused by scraping that decreased cover and richness of all species. Bush lupine was found at very low cover only in doze plots. Although this finding suggests that the doze treatment might have increased bush lupine establishment, statistically, our results failed to show that post-herbicide treatments had any significant effects on bush lupine cover.

Implications for Future Management

Dead European beachgrass was reduced in both the mow and doze treatments compared to no post-herbicide treatment. However, removing standing dead biomass has not produced the desired ancillary effects of increasing native species. The mow treated plots were not different from control plots in non-native species cover or richness, native cover or richness, bareground cover, or overall species richness, evenness, or diversity. In this case, the results of post-herbicide treatment do not seem to warrant the cost of bringing heavy machinery to the dunes for treatment. The doze treatment did reduce non-native species cover and richness, but these results came with undesired effects like reducing native species richness and overall species richness, evenness, and diversity. From this point of view, the doze treatment also does not seem to be a viable option for post-herbicide treatment. It is important to keep in mind that over a longer period of time post-herbicide treatment might prove to be valuable in recruiting native dune species, and additional monitoring may be needed.

Finally, additional post-treatment efforts such as direct seeding or outplanting may be necessary to attain the desired native community in herbicide treated plots. Revegetation plants were established in winter 2015-2016 to set up seeding with coyotebrush and mock heather within different types of herbicide-treatment areas: formal monitoring was not conducted in 2016, but staff observations of plots indicated that, despite an increase in rainfall last winter, there was not abundant germination of sown seed (L. Parsons, NPS, *pers. obs.*). However, these seeds may germinate and flourish more in future years.

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