3819

UNIVERSITY OF HAWAII LIBRARY

RESEARCH ON NATIVE PLANTS FOR COASTAL WETLAND RESTORATION ON O`AHU

A THESIS SUBMITTED TO THE GRADUATE DIVISION OF THE UNIVERSITY OF HAWAI`I IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE OF

MASTER OF SCIENCE

IN

BOTANY
(ECOLOGY, EVOLUTION AND CONSERVATION BIOLOGY)

DECEMBER 2003

By

Karen A. Brimacombe

Thesis Committee:

David Duffy, Chairperson
Donald Drake
Joan Canfield

© Copyright 2003

by

Karen A. Brimacombe

ACKNOWLEDGEMENTS

I would like to thank the United States Fish and Wildlife Service, Ecological Services in Honolulu, Hawaii for funding this research. I would also like to thank the United States Fish and Wildlife Service – O`ahu National Wildlife Refuge Complex for the use of Pearl Harbor National Wildlife Refuge as a field site. I would especially like to thank Mike Silbernagle, Wildlife Biologist, U.S. Fish and Wildlife Service, for all his support, sharing of knowledge, and encouragement for the duration of this study.

I would also like to thank the members of my thesis committee, David Duffy, Donald Drake and Joan Canfield for their support, suggestions and timely help. Thanks to my labmates, especially Stephanie Joe and Chuck Chimera, for their help with fieldwork. Lastly, I'd like to thank Mike Leech for his continual support and encouragement and his assistance in the field.

ABSTRACT

Outplanting field trials were conducted to identify native wetland plant species that would be useful to coastal wetland restoration on O`ahu. The effects of soil salinity and soil temperature, initial planting density, and removal of competition on survival and growth of outplanted species were examined. Germination response of four native wetland sedges to different water, salinity and temperature regimes was also investigated. Final survival exceeded 70 percent in four of the seven outplanted species. Soil salinity and soil temperature were negatively correlated with survival in only one species. Planting density and removal of competition had little effect on final survival and growth of outplanted species. Germination under experimental conditions exceeded 25 percent in only two of the four sedge species studied. Overall, seeds germinated to a higher percentage under lower temperatures and salinities than under higher temperatures and salinity levels. Water level had little effect on final germination percentage.

TABLE OF CONTENTS

Acknowledgements iv

Abstract v

List of Tables ix

List of Figures x

Chapter 1: General Introduction and Proposed Research 1

Introduction 1

Proposed Research 4

Purpose of Study 4

Objectives 4

Justification 5

Research Questions and Hypotheses 6

Methods 8

Species Background 10

Chapter 2: Restoration of Native Wetland Plant Species: Outplanting Field Trials 13

Abstract 13

Introduction 14

Methods 17

Study Site 17

Preparation of Plant Material 18

Experimental Design 19

Monitoring 21

Results 23

Survival and Physical Appearance 23

Growth 29

Reproduction 32

Percent Cover 35

Outplanting Treatments 42

Environmental Parameters 48

Individual Native Species Results 50

Discussion 63

Survival and Growth 63

Soil Salinity and Soil Temperature 67

Competition and Density 68

Discussion and Management Recommendations for Individual Native Species 72

Chapter 3: Germination of Native Wetland Sedge Species Under Different Moisture, Temperature and Salinity Levels 79

Abstract 79

Introduction 80

Methods 82

Water Level and Temperature 83

Salinity and Temperature 84

Results 85

Trial 1: August 2002 – October 2002 85

Trial 2: October 2002 – December 2002 86

Trial 3: February 2003 – April 2003 89

Discussion 95

Literature Cited 99

Appendix A: Map of Outplanting Field Trials Study Site 107

Appendix B: Composition of Non-planted Species in Outplanting Treatment Plots, April 1, 2003 108

Appendix C: Results of Density and Competition Treatments 110

LIST OF TABLES

Table		Page
1.	Initial outplanting densities of native wetland plant species	20
2.	Quarterly survival rates of outplanted native wetland plant species	26
3.	Species composition in study plots prior to outplanting (March 2002). Relative cover of dominant species in outplanting and control plots prior to outplanting of native species	38
4.	Dry biomass (g) (mean ± 1 SD) of non-planted species removed from no-competition planting plots on April 1, 2003	47
5.	Final percent germination (mean ± 1 SD) for seeds of native wetland sedges after eight weeks under different water-levels, salinity levels, and temperature regimes	90
6.	Summary of two-way analysis of variance (ANOVA) of the effects of temperature and water level on final germination of native wetland sedge species	91
7.	Summary of *t*-test results comparing final germination percentage of *Cyperus polystachyos* in 0% and 0.85% salinity treatments under alternating temperature regimes	94

LIST OF FIGURES

Figure		Page
1.	Monthly percent survival of outplanted sedge species and outplanted groundcover species from April 1, 2002 to April 1, 2003	25
2.	Monthly physical appearance of outplanted sedges and outplanted groundcovers	28
3.	Average maximum height (cm) of outplanted individuals	30
4.	Monthly average number of shoots/tillers per live individual	31
5.	Monthly percentage of outplanted individuals with at least one reproductive stem/tiller	32
6.	Monthly average number of reproductive stems/individual for *B. maritimus, C. javanicus, C. polystachyos,* and *S. virginicus*	34
7.	Monthly average number of flowers per 0.25m^2 outplanting plot for *S. portulacastrum* and *J. ovalifolia*	35
8.	Monthly average percent cover (mean of absolute cover ± 1 SD) of outplanted native sedge species	36
9.	Monthly average percent cover (mean of absolute cover ± 1SD) of outplanted native groundcover species	37
10.	Overall relative cover of dominant species in sedge planting and control plots ("competition" plots only). Columns indicate relative cover prior to outplanting of native species and one year later	39
11.	Overall relative cover of dominant species in groundcover planting and control plots ("competition" plots only). Columns indicate relative cover prior to outplanting of native species and one year later	41
12.	Final average percent cover (mean ± 1 SD) of outplanted native sedge species in "density" and "competition" treatment plots	42
13.	Final average percent cover (mean ± 1 SD) of outplanted native groundcover species in "density" and "competition" treatment plots	43

14 Absolute cover (mean ± 1 SD) of non-planted species in sedge planting plots on April 1,2003 45

15. Absolute cover (mean ± 1 SD) of non-planted species in groundcover planting plots on April 1, 2003. 46

16. Monthly totals of all reproductive stems and newly emerging reproductive stems in *C. javanicus* outplanting plots 53

17. Final percent germination (mean ± 1 SD) of native wetland sedge seeds after eight weeks in four water-level treatments under alternating light (12-hout light/dark) and temperature (29/23 °C) regimes 86

18. Final percent germination (mean ± 1 SD) of native wetland sedge seeds after eight weeks in four water-level treatments and four salinity treatments under alternating light (12-hout light/dark) and temperature (24.3/18.2 °C) regimes 88

CHAPTER 1. GENERAL INTRODUCTION AND PROPOSED RESEARCH

Introduction

The landscape of the Hawaiian Islands has been drastically altered since the arrival of Polynesian and European settlers (Athens *et al.* 1992; Loope 1998). Lowland plains and coastal areas have been particularly susceptible to human alteration (Kirch 1982). Following Cuddihy and Stone (1990), Athens (1997) asserted that: "the vegetation in the few remaining "wild" lowland areas of O`ahu is absolutely different from what would have been in these areas in pre-human times." In Hawai`i, as in the continental United States, large tracts of natural wetland and coastal regions have been lost or highly modified because of human action (Elliott 1981; Meier *et al.* 1993).

Since the last century, lowland wetland and coastal areas have not only been filled for urban development but they have also been drained or dredged and filled for agricultural purposes, such as sugar cane production (Elliott 1981; Shallenberger 1981; Meier *et al.* 1993; Loope 1998; Ducks Unlimited 2000). Thomas Dahl (1990) estimated that in the 1780's approximately 8,990 hectares (ha) (22,475 acres) of wetlands existed within Hawai`i's coastal plains. By 1990 the U.S. Fish and Wildlife Service (USFWS) estimated that only 6,190 ha (15,474 acres) of lowland wetlands remained, a loss of 30 % (Dahl 1990; Ducks Unlimited 2000). Coastal wetland losses have been greatest on Oahu due to extensive draining and filling of wetlands for industrial, resort, and residential development (Meier *et al* 1993; U.S. Geological Survey 1997).

Loss of wetland habitat impacts many ecosystem services such as flood control, groundwater recharge, nutrient cycling, and water purification that are provided by these areas (Mitsch and Gosselink 2000; Zedler 2000). In Hawai`i, one of the most detrimental

effects of wetland loss and alteration has been the impact on native waterbird populations, which require wetland habitat for survival (Elliott 1981; Chang 1990; Meier *et al.* 1993). Five of the six endemic Hawaiian waterbirds are currently listed as endangered: the Hawaiian moorhen or `alae `ula (*Gallinula chloropus sandvicensis*), the Hawaiian stilt or ae`o (*Himantopus mexicanus knudseni*), the Hawaiian duck or koloa maoli (*Anas wyvilliana*), the Hawaiian coot or `alae ke`oke`o (*Fulica americana alai*), and the Laysan duck *(Anas laysanensis*) (Shallenberger 1981). All of these species, except the Laysan duck, are presently found on the main Hawaiian Islands.

In the 1970's the USFWS established several National Wildlife Refuges (NWR) in wetland areas in the Hawaiian Islands. The primary goal of these refuges is to preserve and enhance habitat necessary for protection and recovery of Hawai`i's endangered waterbirds (USFWS 2003). Two of these refuges, James Campbell NWR and Pearl Harbor NWR, are located on the island of O`ahu and encompass approximately 80 hectares (200 acres) of wetland habitat (USFWS 2001).

Unfortunately, loss of habitat is not the only threat to the survival of Hawai`i's waterbirds. Introduced predators, such as the mongoose (*Herpestes auropunctatus)* and feral cat (*Felis domesticus*), are a continual threat to waterbird populations (Meier *et al.* 1993). Invasive alien plants also pose a threat to native waterbirds and degrade remaining wetland habitat (Ducks Unlimited 2000; U.S. Department of Agriculture 2001). Introduced species such as California grass (*Brachiaria mutica*), Indian fleabane (*Pluchea indica*), cattail (*Typha latifolia*), saltwort or pickleweed (*Batis maritima*), and California bulrush (*Schoenoplectus californicus*) are aggressive, quick-growing species

that often form dense, monotypic stands that can choke out open water and cover mudflats, areas necessary for native waterbirds' existence (Chang 1990; USFWS 2001).

These introduced plant species also displace native Hawaiian wetland plants. Many wetland areas around the state are currently overrun by alien and weedy plant species. By 1981 over half (55%) of the species of plants known to occur in Hawaiian wetlands were non-native (Stemmermann 1981, cited in Chang 1990; Meier *et al.* 1993). Control of these species is necessary to maintain the integrity of Hawai`i's remaining wetland habitats.

In order to restore degraded wetland habitat, it is necessary not only to control invasive species (Meier 1993; Turner *et al.* 1998), but also to return native vegetation to these sites (Streever and Zedler 2000; Seabloom and van der Valk 2003). A great deal of attention and research has been focused on the life history and habitat requirements of Hawai`i's endangered waterbirds, but scant attention has been given to native wetland plants. Little is known about the life history or ecological requirements of many native wetland plant species. However, interest in restoration of native wetland vegetation in Hawai`i appears to be increasing. Unfortunately, although many state, federal, and non-profit agencies and organizations have begun to implement wetland restoration projects in Hawai`i, rarely has information regarding design and results of these efforts been printed or published. Gathering information concerning current and past wetland projects in the state usually requires personal communication with individuals involved in these efforts. Additionally, although native wetland and coastal plants have been outplanted in a few areas around the state, rarely have they been monitored for survival and reproduction; records of success or failure are largely anecdotal. The main goal of my

research was to add to the limited information on native wetland plants and provide recommendations regarding use of native wetland plant species in future wetland restoration projects in Hawai`i.

Proposed Research

Purpose of Study

The goals of this research project were to identify native plants that may be useful for wetland restoration, evaluate outplanting techniques that may aid survival of outplanted species, and determine environmental conditions that support germination, survival, growth, and reproduction of these native plants. The results from this project are intended to not only help identify native plants that may be useful for coastal wetland restoration due to their ability to survive outplanting and ability to compete with invasive species, but also to provide information on methods that will aid in establishment and recruitment of these native species. This project will also contribute to the scant information regarding ecology and life history of native wetland plants. Restoration of native vegetation to Hawaiian wetlands will not only increase native biotic diversity, but could also improve native waterbird habitat and reduce the need to control invasive plants.

Objectives

This study has eight objectives: 1) identify native plants that are able to survive and reproduce over time with a minimum of human intervention; 2) identify native plants that are able to resist and/or out-compete invasive plant species; 3) determine whether removal of non-planted species aids survival and growth of outplanted native species; 4)

determine whether initial planting density affects establishment of native wetland plant species and re-colonization of non-planted species; 5) investigate effects of environmental parameters on survival, growth and reproduction of native wetland plant species; 6) investigate the reproductive phenology of outplanted native wetland species; 7) investigate the relationship between water depth and salinity on seed germination response of wetland sedges; and 8) provide native plant restoration recommendations to natural resource managers.

Justification

Outplanting of native species is potentially one of the quickest ways to restore native vegetation to an area. Unfortunately, transplanted individuals do not always survive, especially without continual maintenance. Further, aggressive alien weeds may invade a restoration site and replace native vegetation. Experimental outplanting trials can help assess which native wetland species are best able to survive, grow and reproduce with minimum maintenance. Additionally, varying the initial planting densities may help determine if planting density limits re-colonization by invasive weeds. Presumably, planting native species in dense stands should help minimize encroachment by alien species. Experimentally testing this hypothesis may help establish whether, in the long run, outplanting native species in dense stands is worth the initial cost.

Plant vegetative growth, flowering and reproductive output all shape plant population dynamics. Choosing plants that readily reproduce may improve the outcome of a wetland restoration effort by increasing the likelihood that populations of outplanted species will persist. Monitoring the temporal occurrence of phenological events such as vegetative growth, flowering and seed production can provide land managers with vital

information regarding timing of vegetative production and seed production. This knowledge can help guide management decisions, such as when to conduct vegetation control and how abiotic conditions impact survival, growth and reproduction of native wetland plant species.

Understanding specific germination requirements of native species is important for any restoration effort. Outplanting seedlings of native species is time consuming and expensive. Sowing seeds of native plants would provide a cheaper alternative for restoration of native wetland plants. Seeds of many wetland species germinate in response to changes in hydrology, such as continued drawdown or flooding (LaDeau and Ellison 1999). In fact, many emergent wetland species require bare mudflats or shallow water for germination and seedling establishment (ter Heerdt and Drost 1993). Salinity is another factor that may influence germination of emergent wetland species (Martinez *et al.* 1992). Seeds of many wetland species are unable to germinate under conditions of high salinity and/or high water (Baskin and Baskin 2001). Determining the influence of abiotic conditions, such as water depth and salinity, on germination rates, is vital not only for restoration efforts but also for those attempting to manage for native plant communities.

Research Questions and Hypotheses

1. Which outplanted native wetland species are able to survive and reproduce over time with a minimum of human intervention?

Seven native wetland plant species will be outplanted and monitored for survival and reproduction to determine which species have the highest survival, establishment and reproductive rates.

2. Which native plants are able to resist and/or out-compete invasive plant species?

Percent cover for all species, outplanted as well as non-planted species, will be measured to assess not only the growth or increase in density of each species, but also to determine which outplanted species are able to survive and grow despite competition from non-planted species. Measuring percent cover of all species will also help determine which outplanted species limit the establishment of non-planted species.

3. Does removal of competition (i.e., removal of non-planted species) aid survival and growth of outplanted native wetland plant species?

H1: Survival, vigor and/or growth of outplanted individuals will be greater when non-planted species are periodically removed from outplanting plots.

4. Does initial planting density affect establishment of native wetland plant species and re-colonization of non-planted species?

H1: Higher initial planting density will increase the probability of establishment and persistence of outplanted species.

H2: Higher initial planting density will reduce re-colonization of non-planted species.

5. Are environmental parameters of rainfall, air temperature, soil moisture, soil salinity, soil temperature, and pond water depth correlated with survival, growth and reproduction of native wetland plant species?

H1: Low rainfall, soil moisture and pond water depth will be correlated with reduced survival, growth and reproduction of outplanted wetland species.

H2: High air and soil temperature and high soil salinity will be correlated with reduced survival, growth and reproduction of outplanted wetland species.

6. What are the duration and temporal occurrence of reproductive phenological events of outplanted wetland species? Do the plant species under investigation demonstrate seasonality in their reproduction?

H1: Reproductive events of outplanted native wetland species will exhibit no seasonal pattern. Reproduction will occur equally throughout the year.

7. Do soil moisture and salinity affect the seed germination response of wetland sedges?

H1: Germination percentages will decrease with increasing water depth.

H2: Germination percentages will decrease with increasing salinity.

Methods

Outplanting trials were conducted at the Honouliuli Unit of Pearl Harbor (NWR). The Honouliuli Unit of Pearl Harbor NWR consists of two brackish-water ponds. Seedlings of seven native wetland plant species were outplanted on the margins of Pond 2. Prior to any manipulation, baseline data on current vegetation and soil characteristics (salinity, moisture and temperature) was taken. Outplanting plots were manually cleared of all species prior to outplanting.

Four sedge species, *Cyperus javanicus* (Houtt.) Merr. & Metcalfe, *Cyperus laevigatus* L., *Bolboschoenus maritimus* Palla, and *Cyperus polystachyos* (Rottb.) P. Beauv., were planted near waterline around the pond edges. One grass, *Sporobolus*

virginicus L., and two perennial groundcovers, *Sesuvium portulacastrum* (L.) L. and *Jacquemontia ovalifolia* (Choisy), were planted higher up on the banks above the waterline.

Subsequent to outplanting, each seedling was subjected to one of four treatments: high density / no competition, low density / no competition, high density / competition or low density / competition. Density treatments consisted of initially planting seedlings in one of two different densities: high density (9, 10 or 13 seedlings per 0.25 m^2 plot depending on species) or low density (4, 5 or 6 seedlings per 0.25 m^2 plot). Competition treatments consisted of either removing non-planted species from outplanting plots once a month (no-competition treatment) or not removing non-planted species from planting plots (competition treatment). Outplanted individuals in each plot were monitored twice monthly for one year to evaluate percent survival, vegetative growth, and reproductive phenology of each species. Percent cover was monitored twice monthly for one year to document growth of outplanted individuals and to assess colonization by non-planted species. Plots were also monitored for seedlings of outplanted species. Salinity, soil moisture and soil temperature were measured in each plot twice a month. Monthly data on rainfall, air temperature and pond water level was also collected.

Germination of native sedge seeds was assessed in the laboratory. Seeds of native sedges were collected periodically from outplanted individuals. Controlled laboratory experiments using growth chambers were performed to determine germination rates under different conditions of water depth and salinity.

Species Background

Seven species of native plants were used during this project. All species are indigenous to Hawaiian wetland areas. Six of the seven species, *Bolboschoenus maritimus, Jacquemontia ovalifolia, Cyperus javanicus, C. polystachyos, Sesuvium portulacastrum,* and *Sporobolus virginicus*, either currently occur or have been documented as previously occurring at Pearl Harbor NWR. Currently, only *B. maritimus* and *C. javanicus* grow there in populations of any abundance (pers. obs.).

Distributed nearly worldwide, *Bolboschoenus maritimus* occurs in fresh or saline water, on mudflats and in marshes at low elevations in temperate and tropical regions (Wagner *et al.* 1999). Natural populations of *B. maritimus* at Honouliuli form dense stands, primarily in shallow water and extending slightly in from the edge of the water. This perennial sedge rapidly reproduces and can produce as many as 45 tillers from a single corm (Kantrud 1996). Further, endangered Hawaiian coots and Hawaiian moorhens have been observed nesting in stands of *B. maritimus* (Mike Silbernagle, USFWS, pers. comm. 2001). Seeds of this sedge may also provide a food source for waterbirds.

Cyperus javanicus, a perennial sedge, is apparently a re-colonization within the Honouliuli Unit refuge ponds and has only recently been observed (Silbernagle, pers. comm. 2001). *Cyperus javanicus* is often found in coastal marshes exposed to salt or brackish water, as well as in freshwater marshes and taro paddies (Stemmermann 1981). Previous studies have found *C. javanicus* to be a fast-growing and hardy plant that can tolerate drought as well as standing water (Koob 1999). Endangered Hawaiian moorhens have been observed nesting in *C. javanicus* (pers. obs.).

Cyperus laevigatus is not known from Pearl Harbor NWR, but is found in other coastal wetlands around the state (Wagner 1999; pers. obs.). *Cyperus laevigatus* is found in warm temperate and subtropical regions of the globe. In Hawai`i, *C. laevigatus* occurs on the edges of fresh to saltwater ponds, on mudflats, and in sandy coastal sites (Stemmermann 1981; Wagner 1999). This perennial sedge survives equally well in shade house or full sun, but may grow faster when grown in standing water or kept very wet (Koob 1999). This sedge may provide a food source and nesting material for native waterbirds.

Cyperus polystachyos is currently represented by just a handful of plants at Honouliuli Unit of Pearl Harbor NWR. *Cyperus polystachyos* is a slender annual to short-lived perennial (Wagner 1999). According to Stemmermann (1981), in Hawai`i, *C. polystachyos* is found in cultivated wetlands and ruderal wetlands as well as areas under the occasional influence of brackish water. *Cyperus polystachyos* is also found in open or grassy, often disturbed areas, from coastal sites to wet forests (Wagner 1999). Coots have been observed feeding on seed heads and young leaves of these plants at James Campbell NWR located near Kahuku, O`ahu (Silbernagle, pers. comm. 2001).

Jacquemontia ovalifolia is a perennial groundcover currently found in wetland and coastal areas of O`ahu. It is a sprawling vine that is tolerant of salt and mild drought (White 1999). The plant is fast growing and requires minimum water and maintenance and its dense cover may be useful in shading out emergent invasive species (White 1999).

Sesuvium portulacastrum is a perennial, succulent herb common in wetland areas. It is salt-tolerant and often grows in coastal wetlands and mudflats. It is a facultative wetland species found in varying coastal habitats including edges of fresh, brackish and

saltwater ponds (Koob 1999). It generally occupies habitat above the waterline, but is tolerant of periodic inundation.

Sporobolus virginicus is a halophytic perennial grass with buried rhizomes and erect branches to 30 cm tall (Blits and Gallagher 1991; Stemmermann 1981). It may be attractive to waterbirds as a food source and, if grown in dense stands, may help shade out invasive species. *Sporobolus virginicus,* often found along the coast rooted in sand, is also found both in coastal marshes as well as along lowland streambeds (Koob 1999; Stemmermann 1981). This species may survive equally well when kept constantly moist or allowed to dry out between watering (Koob 1999).

CHAPTER 2: RESTORATION OF NATIVE WETLAND PLANT SPECIES: OUTPLANTING FIELD TRIALS

Abstract

To determine native plant restoration potential, seven native wetland plant species were outplanted at Pearl Harbor National Wildlife Refuge, O`ahu, Hawaii. From April 2002 to April 2003, outplanted species were monitored for survival, physical appearance, growth, reproduction, and percent cover. Effects of initial planting density and impacts of competition from non-planted species on survival, growth, reproduction, and percent cover of native species were examined. Influence of soil salinity and soil temperature on these parameters was also investigated. *Cyperus javanicus, Sporobolus virginicus,* and *Jacquemontia ovalifolia* maintained high survival, percent cover, and reproductive rates for the duration of the study. *Bolboschoenus maritimus, Cyperus polystachyos, C. laevigatus,* and *Sesuvium portulacastrum* suffered higher mortality and showed the most noticeable decline in physical appearance, growth, and percent cover. Initial planting density did not significantly affect final survival, physical appearance, growth, or reproduction in any outplanted species. Final percent cover in density treatments was only significantly different in *S. virginicus.* Initial planting density resulted in a significant difference in average biomass of non-planted species removed from plots of *C. javanicus, C. laevigatus, C. polystachyos,* and *S. virginicus.* Competition from non-planted species had little effect on final survival, physical appearance, growth, or reproduction of outplanted species. Final percent cover of *B. maritimus, S. portulacastrum,* and *S. virginicus* was greater in no-competition than in competition

plots. In general, soil salinity and soil temperature had little effect on final survival, health, growth and reproduction for most of the outplanted species over the range of conditions examined. Results suggest that restoration of *C. javanicus, S. virginicus,* and *J. ovalifolia* in other coastal wetlands in Hawai`i has a high likelihood of success. Restoration of *C. polystachyos* may be more successful in freshwater than in brackish wetlands. Outplantings of *B. maritimus* should be focused in or near shallow water. Further study of environmental conditions hindering survival and health of *C. laevigatus* and *S. portulacastrum* is recommended prior to large scale restoration projects utilizing these species.

Introduction

Wetland ecosystems provide extremely important "services" such as flood control, water quality improvement, groundwater recharge, minimizing erosion and decreasing sedimentation of coastal waters (Mitsch and Gosselink 2000; Zedler 2000). Unique recreational and educational opportunities are also provided by these ecosystems and, in the main Hawaiian Islands, wetland areas are vital to the preservation of four endangered endemic waterbirds (Elliott 1981; Chang 1990; Meier *et al.* 1993). Until the mid 1970's destruction of wetlands was a common and accepted practice in the United States (Mitsch and Gosselink 2000; National Research Council 1995). Wetlands were drained, dredged, ditched, and filled for development of agricultural fields and urban expansion. Hawai`i did not escape these practices and by 1990 approximately 31% of Hawai`i's coastal wetlands had been lost or altered for agriculture and urban development (Dahl 1990).

Most of Hawai`i's remaining coastal wetland habitat is degraded due to invasion by non-native plant species (Elliott 1981; Ducks Unlimited 2000). Invasive alien species impair habitat for native and migratory waterbirds, alter ecosystem structure and decrease native wetland plant biodiversity (Walker and Smith 1997; Turner *et al.* 1998; Wilcox and Whillans 1999; Ducks Unlimited 2000; USDA 2001). Removal and control of alien plant species is an ongoing, expensive, and time-consuming process.

Restoration of degraded wetland ecosystems has become commonplace in the continental United States (Callaway *et al.* 1997; Keddy 1999). However, wetland restoration in the Hawaiian Islands has, until recently, received little attention. The primary goal of most wetland restoration projects has been to establish or enhance habitat for Hawai`i's endangered waterbirds (Meier *et al.* 1993; Ducks Unlimited 2000). Thus, restoration efforts have focused on pond creation, reinstating previous hydrological conditions, and control and removal of non-native invasive vegetation. Natural recolonization has often been relied upon for the re-establishment of native vegetation. However, studies in the continental United States have demonstrated that many important wetland species that typically occur in natural wetlands do not readily colonize restored wetlands (Reinartz and Warne 1993; Keddy 1999; Budelsky and Galatowitsch 2000; Streever and Zedler 2000). In Hawai`i, reintroduction of native vegetation has occurred infrequently and the successes (and failures) of these reintroductions have rarely been documented. Consequently, little is known about restoring native plants to wetland habitat in Hawai`i.

Outplanting of native species is one of the quickest, albeit not the least inexpensive, ways to restore native vegetation (Wilcox and Whillans 1999).

Transplanted seedlings or rhizomes show higher percentages of establishment and require less time to reach reproductive maturity than seeds (Falk 1996; van der Valk *et al.* 1999; Yetka and Galatowitsch 1999; Fattorini 2001). Unfortunately, transplanted individuals do not always survive, especially without continual maintenance (Wilcox and Whillans 1999; Budelsky and Galatowitsch 2000). Abiotic conditions of a restoration site also influence the success of wetland plant restoration efforts (Keddy 1999; Yetka and Galatowitsch 1999; Zedler 2000). Each target species may thrive under specific environmental conditions. Understanding the environmental parameters that affect growth, survival and reproduction of native wetland plants can help determine suitable locations for reintroduction (Bradshaw 1987; Callaway *et al.* 1997; Middleton 1999; Zedler *et al.* 2001). Field trials were initiated in the spring of 2002 to investigate the establishment, growth, and reproductive capabilities of native wetland species outplanted in a brackish coastal wetland on the island of O`ahu, Hawai`i.

These field trials were also designed to test alternative revegetation methods. Experimentally testing different methods of outplanting not only helps determine which species are able to persist after outplanting, but helps evaluate which techniques may increase survival of outplanted species (Jordan *et al.* 1987; Callaway *et al.* 1997; Streever and Zedler 2000). Plants initially outplanted in dense stands may have higher rates of establishment. Densely planted stands of native species may also reduce or limit the re-colonization of alien species (Keddy 1998; Streever and Zedler 2000; Sustainable Resources Group International 2001). Further, aggressive alien weeds may invade or re-colonize a restoration site and may hinder survival of or replace the native outplanted

vegetation (Yates and Hobbs 1997; Clark and Wilson 2001). Thus, regular control and removal of non-native species may be necessary for the persistence of restored species.

The primary objectives of this study were to 1) identify native plants that may be useful to wetland restoration efforts due to their ability to survive, grow, and reproduce over time; 2) determine whether removal of non-planted species, prior to outplanting and periodically after outplanting, aids survival and growth of outplanted native species; 3) determine whether initial planting density affects establishment of native wetland plant species and re-colonization of non-planted species; and 4) investigate effects of environmental parameters on survival, growth and reproduction of native wetland plant species. Results from this study will provide native plant restoration recommendations to natural resource managers.

Methods

Study Site

This study was conducted within the Honouliuli Unit of Pearl Harbor National Wildlife Refuge (NWR). Established in 1972, Pearl Harbor NWR is comprised of three units: Honouliuli, Waiawa and Kalaeloa. The Honouliuli and Waiawa Units of Pearl Harbor NWR were created as replacement habitat for native and migratory waterbirds when expansion at the Honolulu International Airport destroyed mudflats and other areas used by these waterbirds (USFWS 2001). Honouliuli Unit is located on the West Loch of Pearl Harbor on the southern shore of the Island of O`ahu (Appendix A).

Covering 14.6 ha (36 acres), Honouliuli Unit encompasses two slightly brackish man-made ponds. These ponds contain water year round, although the depth of the water

fluctuates slightly seasonally and through manual water level control via freshwater pumps located in each pond. The salinity of the ponds averages between two to five parts per thousand (ppt). Outplanting field trials were conducted solely around the perimeter of Pond 2 within the Honouliuli Unit.

Located in the dry Ewa plains, Honouliuli Unit receives approximately 510 mm of rainfall per year and mean monthly temperature ranges from approximately 22° C in January to 27° C in August (Juvik and Juvik 1998). The soil at Honouliuli is classified as part of the Lualualei Fill Land Ewa soil association (USDA 1972). This association is characterized as having deep, nearly level to moderately sloping well-drained soil with fine or moderately fine textured subsoil. Soils in this association are also characterized as fill land. Honouliuli soil is classified as "fill land – mixed"; with areas filled with material dredged from the ocean or brought in from nearby areas (USDA 1972).

The area is dominated by non-native coastal and wetland vegetation including *Batis maritima, Pluchea indica, Typha latifolia, Prosopis pallida,* and *Leucaena leucocephala.* Management of these non-native species is currently primarily through mechanical methods and rapid flooding and/or drawdown of pond water levels.

Preparation of plant material

Seven indigenous wetland plant species, four sedges (*Bolboschoenus maritimus, Cyperus javanicus, C. laevigatus,* and *C. polystachyos*) and three groundcovers (*Jacquemontia ovalifolia, Sesuvium portulacastrum,* and *Sporobolus virginicus*) were chosen for the experimental outplanting trials. They were chosen based not only on their ease of propagation and tolerance of harsh (saline, hot, and dry) environmental conditions, but also because all but one of the species have been documented as currently

or previously occurring at Honouliuli. Seedlings of two of the sedge species (*C. javanicus* and *C. polystachyos*) were grown from seeds collected from Pearl Harbor and James Campbell NWR's in August and September of 2001. Seeds of *B. maritimus* were collected from Pearl Harbor NWR, but they failed to germinate, so transplants from Pearl Harbor NWR were used for outplanting. Seedlings of the remaining species were propagated from cuttings of local plant stock. Seedlings and propagules were grown in ambient conditions, at Hui Ku Maoli Ola Native Plant Nursery in Waimanalo, Hawai`i. Initial seedling sizes were, on average, between 20 to 25 cm for *B. maritimus, C. laevigatus, J. ovalifolia, S. portulacastrum,* and *S. virginicus,* and 50 to 60 cm for *C. javanicus* and *C. polystachyos.*

Experimental Design

Approximately 300 seedlings and transplants of each of the seven study species were tagged and outplanted in single species blocks in March of 2002. A randomized block design was utilized with ten 1.25 m x 6.5 m blocks for each of the four sedge species and ten 2.5 m x 3.5 m blocks for each of the three groundcover species. Each block contained six 0.25 m^2 square plots representing one replicate of each treatment for a total of 420 0.25 m^2 plots. A 0.5 m buffer was left between each plot and from the edge of each block to each treatment plot. A 2 x 2 factorial design was used with treatments of high density/low density and competition/no competition. Thus, each plot was planted in one of six different arrangements: high density/no competition, low density/no competition, high density/competition, low density/competition, control (no seedlings planted)/competition, control (no seedlings planted)/no competition.

Based on the size of the seedlings, species were initially planted at six different densities (Table 1). Prior to outplanting, percent cover of existing vegetation was recorded for each permanent plot. This existing vegetation was then manually removed from each outplanting block and planting of native species commenced. "Competition" treatments were subjected to competition from recolonization of non-planted species, whereas in each "no competition" plot all non-planted vegetation was removed once a month. To promote establishment of outplanted seedlings, each plot was given 3.8 liters of supplemental water once a week for the first month after outplanting. After the first month, plots were given 3.8 liters of water every two weeks for the next month. At this point no further supplemental water was provided to the outplanting plots.

Table 1. Initial outplanting densities of native wetland plant species. Densities reflect number of seedlings per 0.25 m^2 planting plot.

Species	***Density*** **(number of seedlings)**	
	Low	**High**
Bolboschoenus maritimus *Cyperus javanicus*	4	9
Cyperus laevigatus *Jacquemontia ovalifolia* *Cyperus polystachyos* *Sesuvium portulacastrum*	5	10
Sporobolus virginicus	6	13

Monitoring

Twice monthly from April 2002 to April 2003, each outplanted seedling was monitored to assess survival, physical appearance, growth, and reproductive status. Percent cover was also recorded twice monthly for all species in each treatment plot. Percent cover was estimated using a 0.5 m x 0.5 m frame divided into 25 10 cm x 10 cm squares, with each square equaling four percent cover (following Elzinga *et al.* 1998).

Survival in the sedge species and *Sporobolus virginicus* was assessed by monitoring the presence or absence of above-ground live shoots. An outplanted individual was considered "alive" only if above-ground live shoots were present. Individuals of *Jacquemontia ovalifolia* and *Sesuvium portulacastrum* were considered alive if at least one green leaf or stem was present.

Physical appearance, which was assessed by a visual estimate of the percent of dark-green leaves and shoots versus the percent of yellowing or dead and dying leaf/shoot parts, was rated on a scale of 1 to 5 with 1 being very poor health and 5 being very healthy. The following is the rating scale that was used: 0 = plant dead, 1 = 81-99% of plant dead or dying, 2 = 61-80% dead or dying, 3 = 41-60% dead or dying, 4 = 21-40% dead or dying, 5 = 0-20% dead or dying. These ratings will hereafter be referred to as "health-code" values. Vegetative growth of outplanted species was assessed by measuring the number of shoots per individual and/or the maximum height of each individual. Maximum height was recorded as the length from tip to base of the tallest leaf, shoot or reproductive stem of an individual. Due to their prostrate nature and stems which root at the nodes, heights and number of shoots of *Sesuvium portulacastrum* and *Jacquemontia ovalifolia* seedlings were not recorded. The number of shoots/tillers was

counted for each outplanted individual of *Bolboschoenus maritimus* and *Cyperus javanicus*, and estimated for *Sporobolus virginicus* individuals. *Cyperus laevigatus* and *C. polystachyos* produce too many shoots to accurately count, thus, the number of shoots/tillers of these species was not recorded. Percent cover data were also used to assess growth of all outplanted species.

The number of reproductive shoots/tillers for each tagged seedling was recorded during each monitoring period for *Bolboschoenus maritimus, Cyperus javanicus, C. polystachyos* and *Sporobolus virginicus*. The number of reproductive shoots per tagged seedling was estimated for *Cyperus laevigatus*. The number of flowers per plot (vs. per tagged individual) was recorded twice monthly for *Jacquemontia ovalifolia* and *Sesuvium portulacastrum*.

Plots were also monitored twice monthly for the appearance of non-planted species. Percent cover for all non-planted species was recorded for each plot. Once a month, after percent cover for each non-planted species was recorded, these species were removed from each no-competition treatment plot. Plants were clipped at ground level and separated by species. Clippings were placed in paper bags and dried at 45°C and then weighed. Dry weights were used to assess differences in colonization and production of non-planted species in low density, high density, and control treatments. Colonization by non-planted individuals of *Bolboschoenus maritimus* and *Cyperus javanicus* were observed in a few outplanting plots of these species. These volunteer individuals were considered in percent cover estimates in outplanting plots of these species. Although this could potentially confound results, colonization of volunteer individuals, when it occurred, tended to occur across entire planting blocks. Thus, the

influence on percent cover estimates was relatively similar across all treatments in planting blocks where volunteers were observed.

Measurements of soil salinity and soil temperature were taken in each plot twice a month from July 14, 2002, through April 1, 2003, using the Aquaterr Ec-200 soil moisture/soil temperature/soil salinity meter (Aquaterr Instruments, Freemont, CA). Measurements of soil salinity and soil temperature, prior to July 14, 2002, were sporadic due to equipment failure. Measurements of soil moisture were taken in March 2002 and again from late-July of 2002 through October 2002. At this point the accuracy of the soil moisture readings became questionable, i.e., measurements displayed by the meter would rarely exceed 10 percent soil moisture even in water-logged soil, so soil moisture measurements ceased in November 2002 and soil moisture measurements were not used for any statistical analysis. Pond water level was recorded twice monthly using a permanent water level gauge located at the west end of Pond 2. Monthly rainfall and air temperature data were provided by the USFWS O`ahu Refuge Complex's automated weather station located at the west end of Pond 2 at Honouliuli.

One additional monitoring session was conducted in early May 2003. During this monitoring period, only sedge species were evaluated and only a subset of variables (survival, physical appearance and reproduction) was measured.

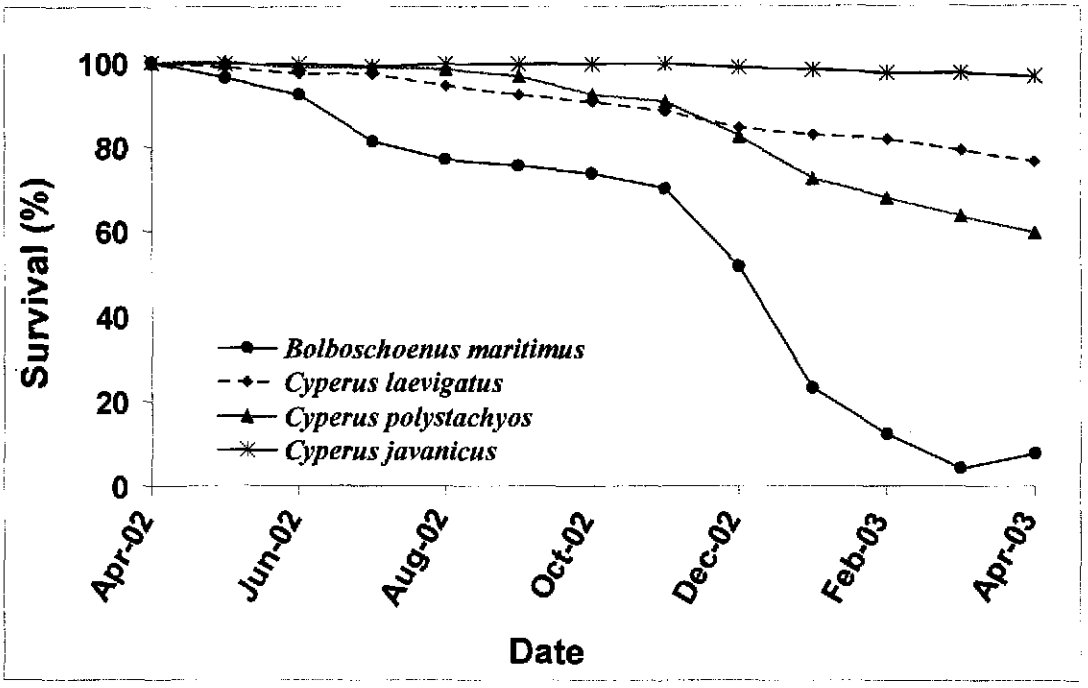
Results

Survival and Physical Appearance

More than 90 percent of outplanted individuals of all species (except *Bolboschoenus maritimus*) survived through September 2002 (Figure 1). No marked

decline in survival for any species, except *B. maritimus*, was noticeable until December 1, 2002, at which point survival dropped below 90% for several species.

a)



b)

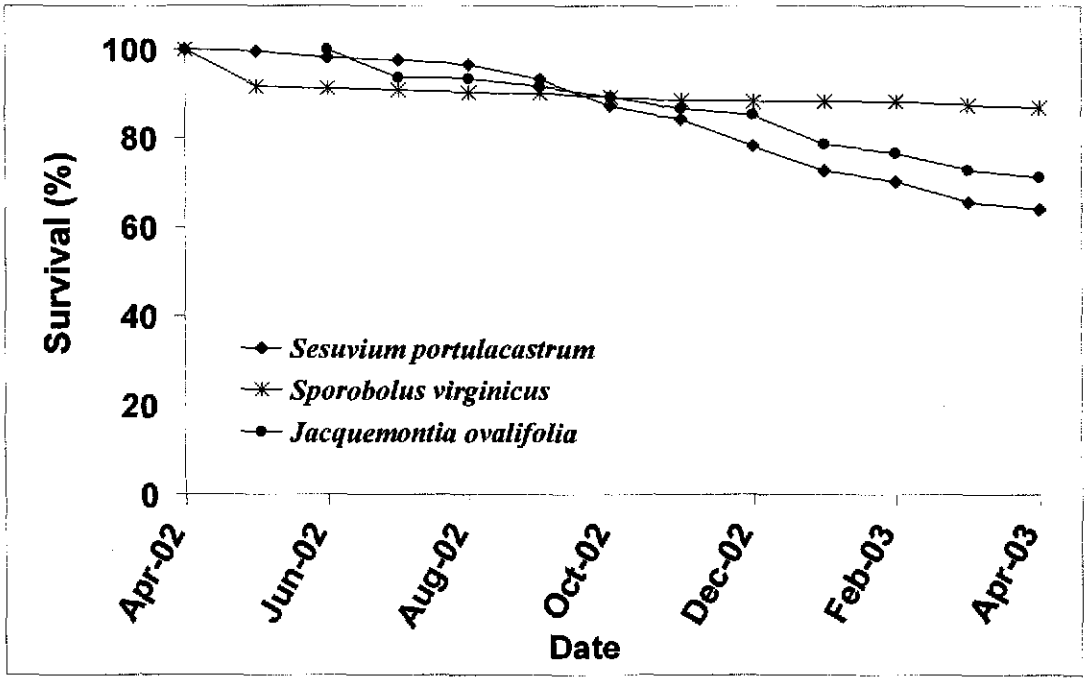


Figure 1. Monthly percent survival of (a) outplanted sedge species and (b) outplanted groundcover species from April 1, 2002, to April 1, 2003. Outplanting occurred in late-March 2002 for all species except *J. ovalifolia*, which was outplanted in early May 2002.

One year after outplanting, 67.3 percent of the original 2,100 plants were still alive (Table 2). *Cyperus javanicus* had the highest survival rate (96.5%) with 251 of the original 260 plants possessing live shoots one year post-outplanting. Despite being one of the most common native species found growing naturally at Honouliuli, *Bolboschoenus maritimus* had the lowest survival rate, with only 7.7 percent of transplanted individuals possessing live shoots in April 2003.

Table 2. Quarterly survival rates of outplanted native wetland plant species.

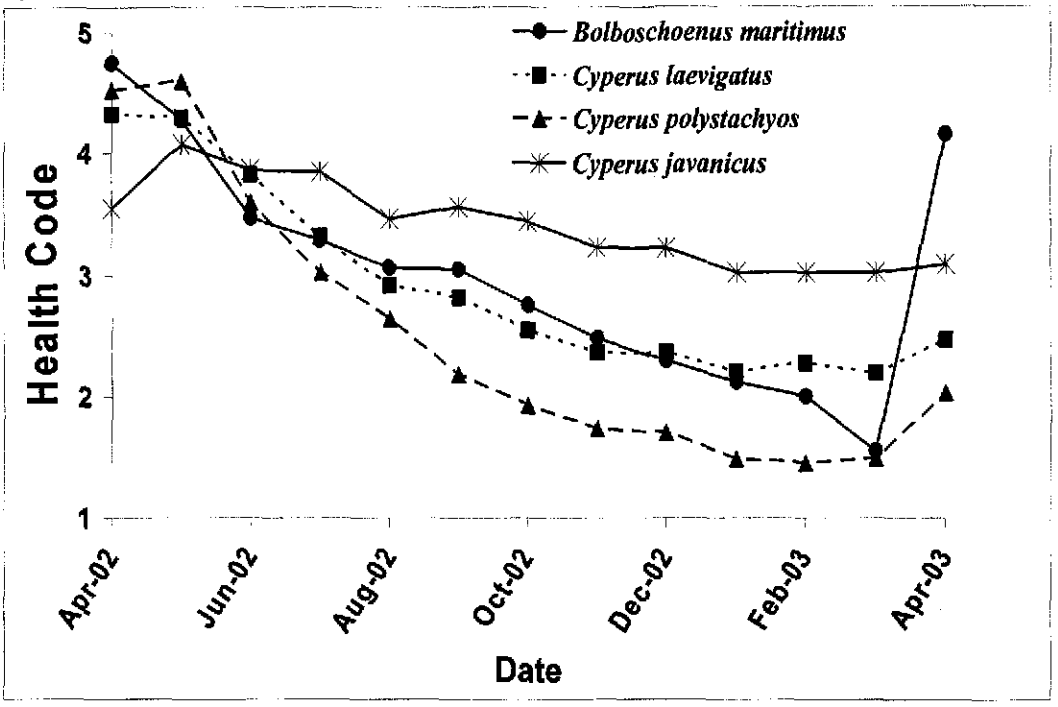
Species	Percent Survival			
	April 2002	**August 2002**	**December 2002**	**April 2003**
Sedges				
Bolboschoenus maritimus	100	76.9	51.5	7.7
Cyperus laevigatus	100	94.3	84.3	76.3
Cyperus javanicus	100	99.6	98.5	96.5
Cyperus polystachyos	100	98.3	82.3	59.3
Groundcovers				
Jacquemontia ovalifolia	100*	93.7	85.3	71.3
Sesuvium portulacastrum	100	97.7	78.3	64.0
Sporobolus virginicus	100	90.3	88.2	86.8

* Numbers reflect totals on May 1st, 2002.

Despite high overall survival rates, physical appearance of outplanted individuals started declining by June 2002 (Figure 2). This decline in physical appearance was most noticeable in *Bolboschoenus maritimus, Cyperus laevigatus, C. polystachyos,* and *Sesuvium portulacastrum.* Individuals of *C. polystachyos* and *S. portulacastrum*

exhibited the poorest physical appearance of any species in February 2003 with average health-codes of 1.46 ± 0.78 (SD) and 1.58 ± 0.85 (SD) respectively.

a)



b)

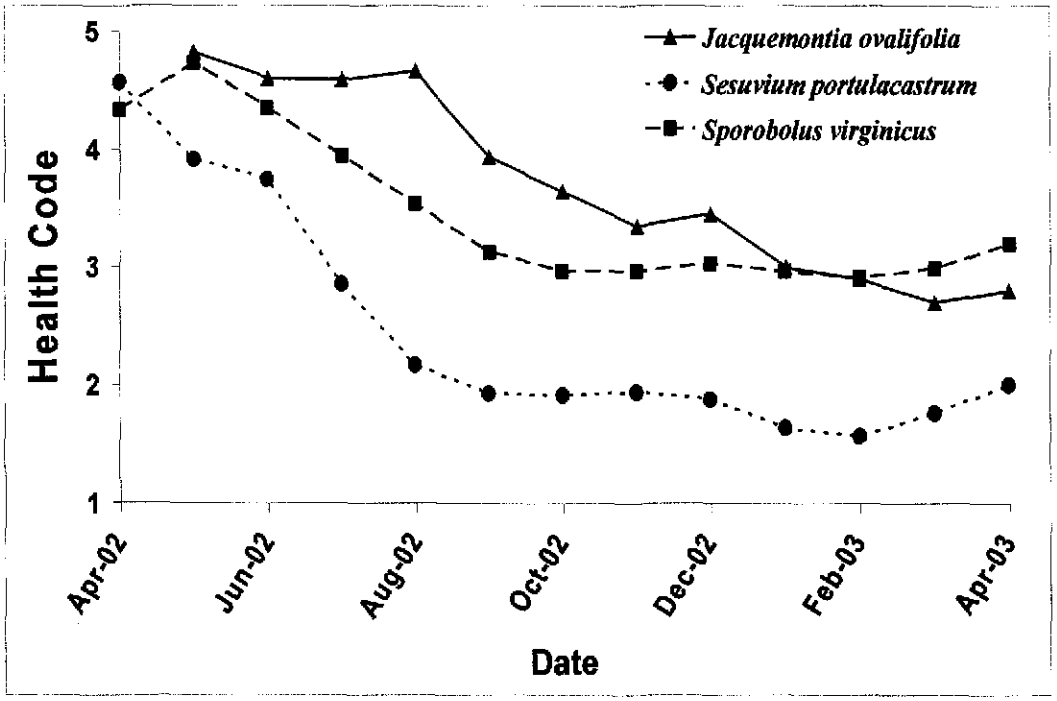


Figure 2. Monthly physical appearance of (a) outplanted sedges and (b) outplanted groundcovers. Health-code: 5 = 0-20% dead or dying; 4 = 21-40% dead or dying; 3 = 41-60% dead or dying; 2 = 61-80% dead or dying; 1 = 81-99% dead or dying.

Average physical appearance of *Cyperus javanicus* individuals showed the least fluctuation throughout the study period, with a maximum average health-code of 4.07 ± 1.25 (SD) on May 1, 2002, and a minimum average health-code of 2.97 ± 1.16 (SD) in mid-February 2003. Interestingly, beginning in April 2003, the health of almost all species had started to rise. This was partially due to the increase in production of new, healthy shoots/tillers by most of the study species, which began in late March.

Growth

Average maximum height for all species showed an expected initial increase in height from April through June 2002 as outplanted individuals developed from seedlings to mature plants. Despite declining physical appearance, average maximum height of *Cyperus javanicus*, *C. laevigatus,* and *Sporobolus virginicus* individuals remained relatively constant from July 2002 to April 2003. *Bolboschoenus maritimus* and *Cyperus polystachyos* individuals, however, exhibited a decline in average maximum height from June 2002 through April 2003 (Figure 3). This decline in average maximum height corresponds with a decline in overall physical appearance and survival of *C. polystachyos* and *B. maritimus* individuals.

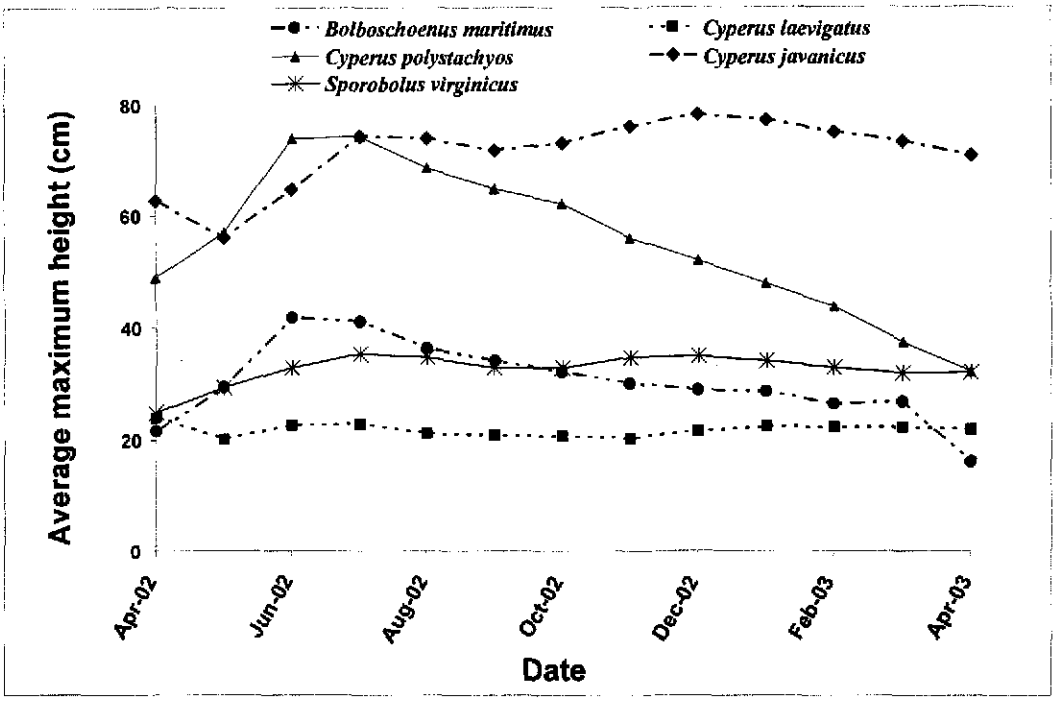


Figure 3. Average maximum height of outplanted individuals.

The number of shoots per individual was counted for *Bolboschoenus maritimus* and *Cyperus javanicus* and estimated for *Sporobolus virginicus*, but was not recorded for the other four species. The number of shoots for both *C. javanicus* and *S. virginicus* gradually increased throughout the study period. On April 1, 2002, the average number of shoots per individual was 5.3 ± 4.1 (SD) for *C. javanicus* and less than six for *S. virginicus*. By the end of the study period, April 1, 2003, the average number of shoots per individual had increased to 12.6 ± 9.7 (SD) for *C. javanicus* and greater than 15 tillers per individual for *S. virginicus* (Figure 4).

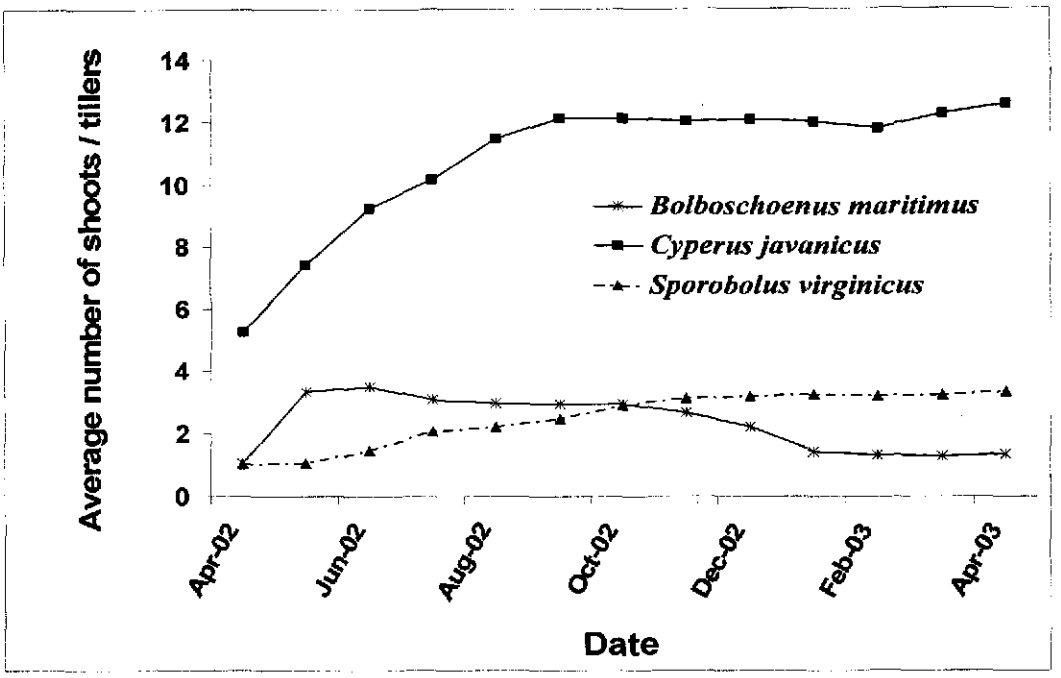


Figure 4. Monthly average number of shoots or tillers per live individual. Numbers for *B. maritimus* and *C. javanicus* are actual counts. Numbers for *S. virginicus* are based on estimated number of tillers per individual. Estimates for *S. virginicus*: 1= 1-5, 2=6-10, 3=11-15, 4=16-20, and 5=21-25 tillers per individual.

Bolboschoenus maritimus individuals showed an initial increase in the average number of shoots per individual from April 1, 2002, to June 1, 2002 (Figure 4). After June the number of shoots declined to a low of 1.18 ± 0.40 (SD) shoots per live individual in mid-March 2003. However, production of new shoots began in April 2003, and by May 2003 the number of shoots had again increased to an average of 1.38 ± 0.54 (SD) shoots per live *B. maritimus* individual. As for total shoots, on April 1, 2002, the total number of shoots for *B. maritimus* transplants was 276. Between April and June 2002, vast production of new shoots by transplanted individuals increased the total number of shoots to 839. New shoot production decreased rapidly at this point and many existing shoots began to senesce. By March 1, 2003, the total number of live *B. maritimus* shoots had dropped to 14.

Reproduction

Reproduction in all species occurred rapidly after outplanting. By the beginning of June 2002, two months after outplanting, at least a few individuals of each species had begun to produce flowers or reproductive shoots. From May 2002 through February 2003, *Cyperus laevigatus* and *C. polystachyos* displayed the highest percentage of reproductive individuals, with close to 100% of outplanted individuals bearing reproductive stems (Figure 5). The proportion of reproductive individuals never exceeded 79% for *Bolboschoenus maritimus, Cyperus javanicus,* or *Sporobolus virginicus.*

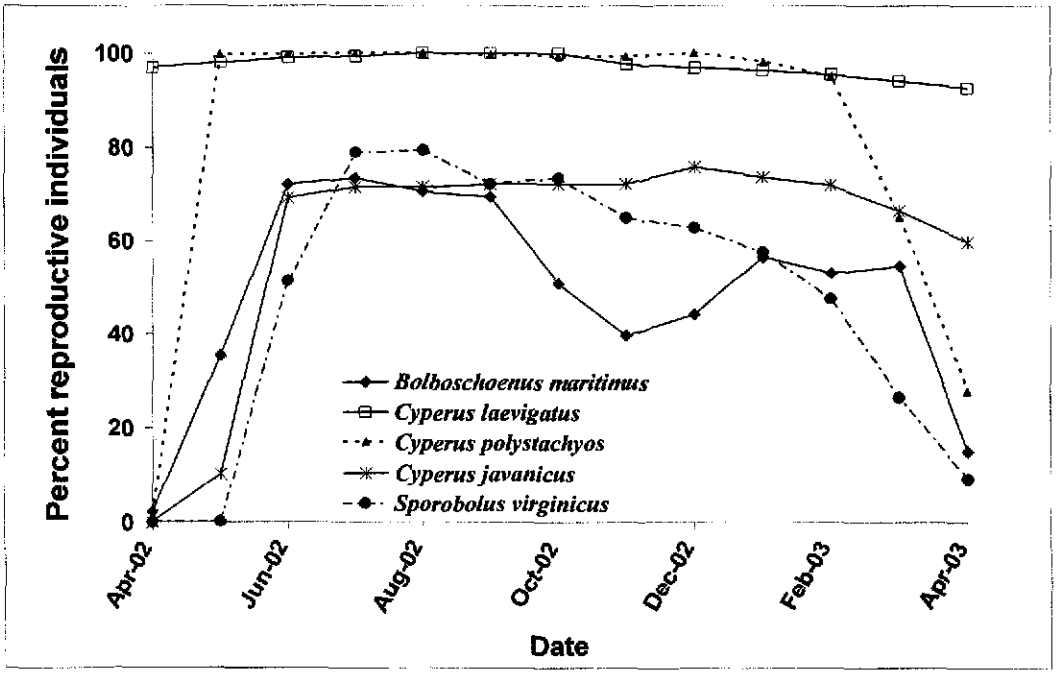


Figure 5. Monthly percentage of outplanted individuals with at least one reproductive stem or tiller.

For all sedge species, initiation of reproduction (i.e., elongation of a reproductive stem) had begun by May 1, 2002. Three of the four sedge species (*Bolboschoenus maritimus, Cyperus laevigatus,* and *C. polystachyos)* showed similarities in their reproductive patterns (Figure 6). Peak reproduction, as measured by the average number of reproductive stems per individual, for these three species occurred by July 14, 2002. After this peak, initiation of new reproductive stems declined, and thus, the average number of reproductive stems per individual declined steadily from the middle of July 2002 through April 2003. Reproduction in *Cyperus javanicus* exhibited a gradual climb from June through December 2002; and from December through April 2003, the average number of reproductive stems/individual declined only slightly (Figure 6). Reproduction in *Sporobolus virginicus* increased sharply from May through December 2002 and then steadily declined from mid-December 2002 through April 2003.

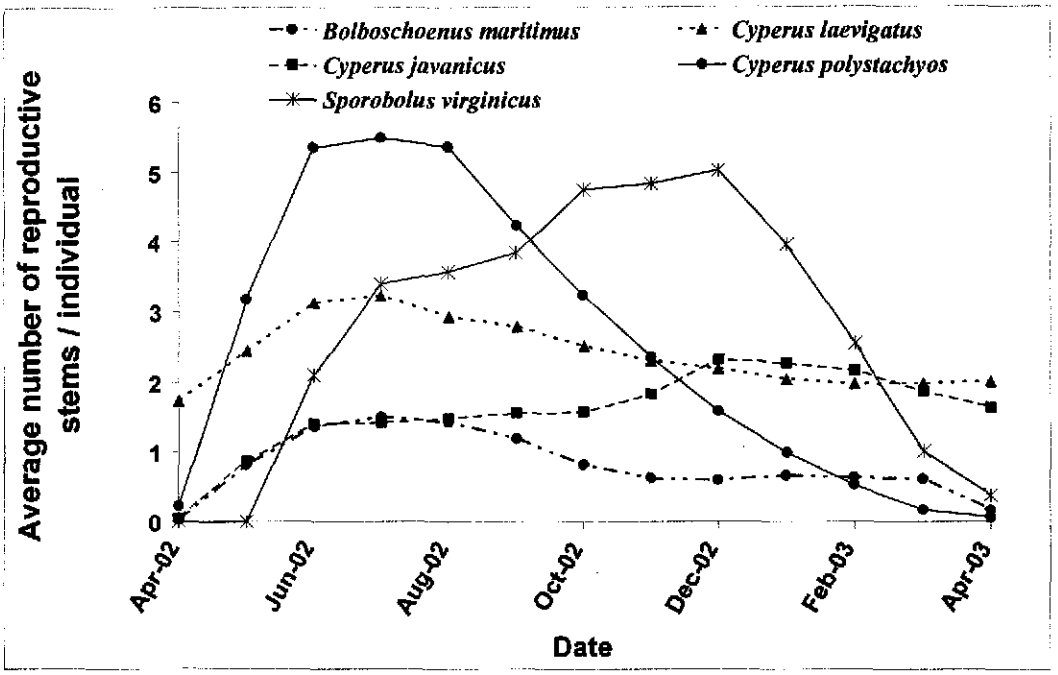


Figure 6. Monthly average number of reproductive stems/individual for *B. maritimus, C. javanicus, C. laevigatus, C. polystachyos,* and *S. virginicus*. Numbers for *B. maritimus, C. javanicus, C. polystachyos,* and *S. virginicus* are based on actual counts. Numbers for *C. laevigatus* are based on estimated number of reproductive stems per individual. Estimates for *C. laevigatus*: 0 = no reproductive stems, 1 = 1-10, 2 = 11-20, 3 = 21-30, 4=31-40 reproductive stems.

The final average number of reproductive stems was correlated with final average health-code in *Cyperus laevigatus* ($r = 0.828$; $p<0.0001$; $n=40$), *C. javanicus* ($r = 0.692$; $p<0.0001$; $n=40$), *C. polystachyos* ($r = 0.591$; $p = 0.0003$; $n=32$), and *Sporobolus virginicus* ($r = 0.450$; $p = 0.004$; $n=40$). Due to high mortality of *Bolboschoenus maritimus*, analysis of correlation between final number of reproductive stems and final average health-code was not conducted.

The remaining two species, *Jacquemontia ovalifolia* and *Sesuvium portulacastrum,* did not exhibit a straightforward reproductive pattern. The average number of flowers per plot for these two species fluctuated throughout the study period (Figure 7). *Sesuvium portulacastrum* showed an initial burst of flower production in

June of 2002 followed by an abrupt drop in number of flowers per plot. There was a second slight increase in flower production in October 2002, followed by a decline in the numbers of flowers per plot for much of the study period. *Jacquemontia ovalifolia* showed two distinct peaks of flower production, one in July 2002 and the second in February 2003. Final number of flowers per plot was correlated with final average health-code per plot for both *J. ovalifolia* and *S. portulacastrum* ($r = 0.440$; $p = 0.0045$; $n = 40$ and $r = 0.347$; $p = 0.031$; $n=38$ respectively).

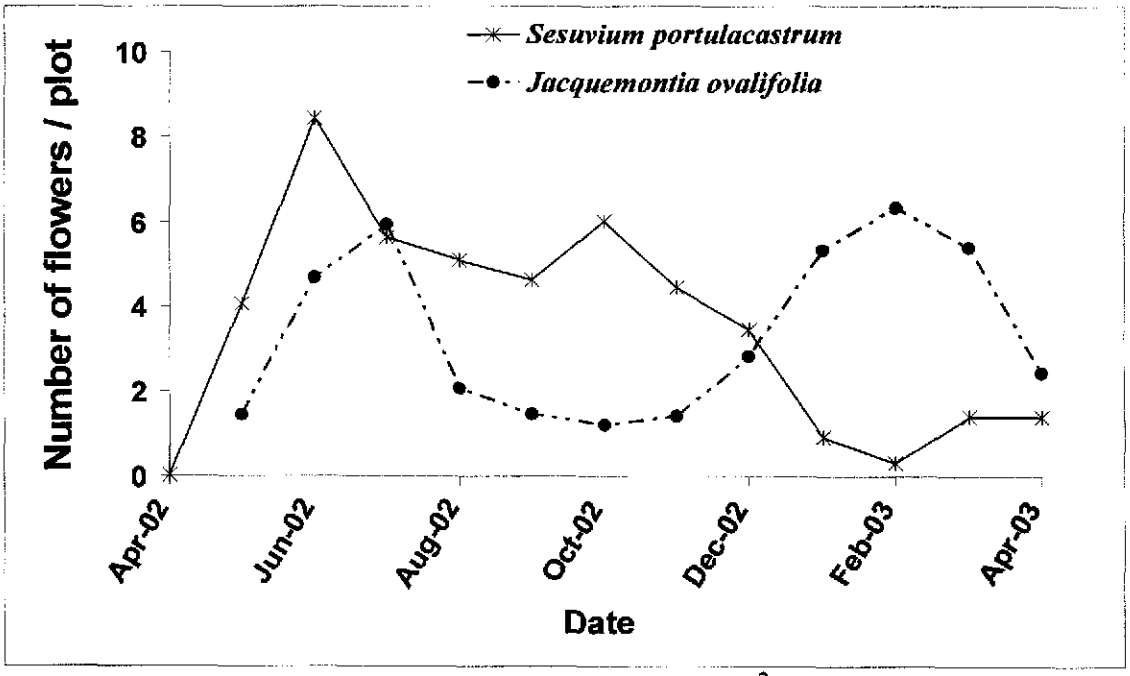


Figure 7. Monthly average number of flowers per 0.25 m^2 outplanting plot for *S. portulacastrum* and *J. ovalifolia.*

Percent Cover

Percent Cover of Outplanted Native Species

Shortly after outplanting, overall average percent cover of native species in planting plots ranged from 21.8 percent for *Bolboschoenus maritimus* to 60.1 percent for *Cyperus*

polystachyos (Figure 8). The range in percent cover varied primarily due to morphological characteristics of each native species. For example, *B. maritimus* has erect stems and produces relatively few linear leaves, whereas *C. polystachyos* forms many basal leaves which give the plant a somewhat bushy appearance.

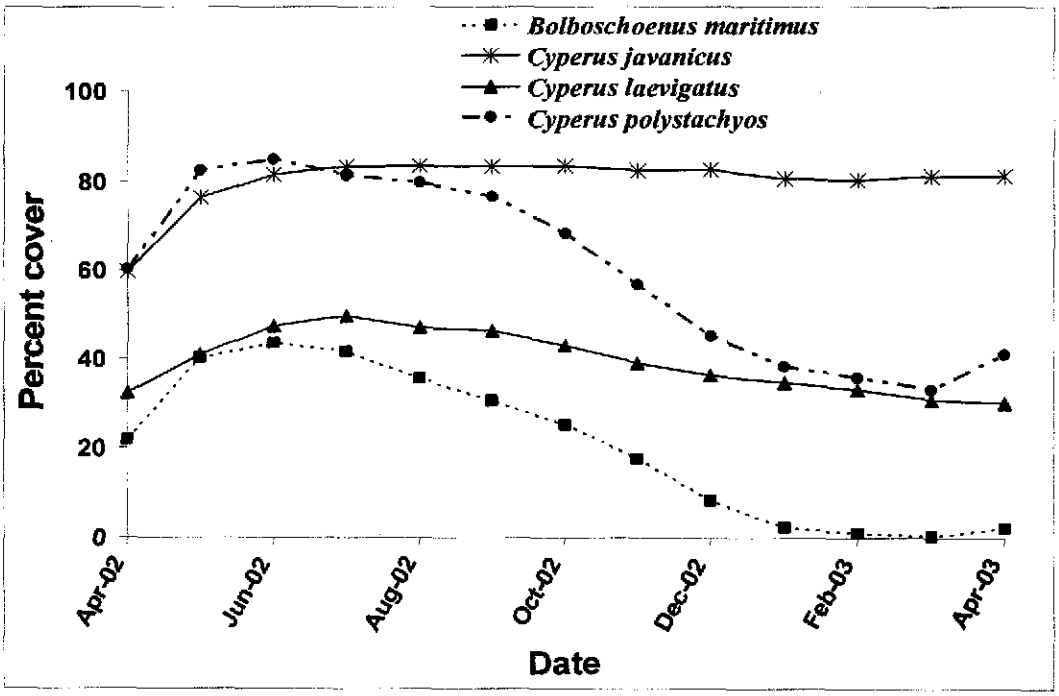


Figure 8. Monthly average percent cover (mean of absolute cover ± 1 SD) of outplanted native sedge species. Averages are based on planting plots for each species individually; control plots and planting plots with 100% mortality are not included.

The greatest overall average percent cover recorded during the study period was for *Cyperus polystachyos* (85.0 ± 14.9 (SD)) in June 2002 and *Cyperus javanicus* (83.7 ± 17.9 (SD)) in September 2002. The lowest average percent cover during the study period was recorded for *Bolboschoenus maritimus* with 0.4 ± 0.9 (SD) in March 2003 and 24.5 ± 23.9 (SD) for *Sesuvium portulacastrum* in April 2003 (Figure 8; Figure 9).

Average percent cover in planting plots steadily increased for all species from April 2002 to July 2002. From July 2002 through April 2003, average percent cover decreased for *Bolboschoenus maritimus, Cyperus laevigatus, C. polystachyos,* and *Sesuvium portulacastrum.* This decline in average percent cover corresponds to declining health and survival of these four species.

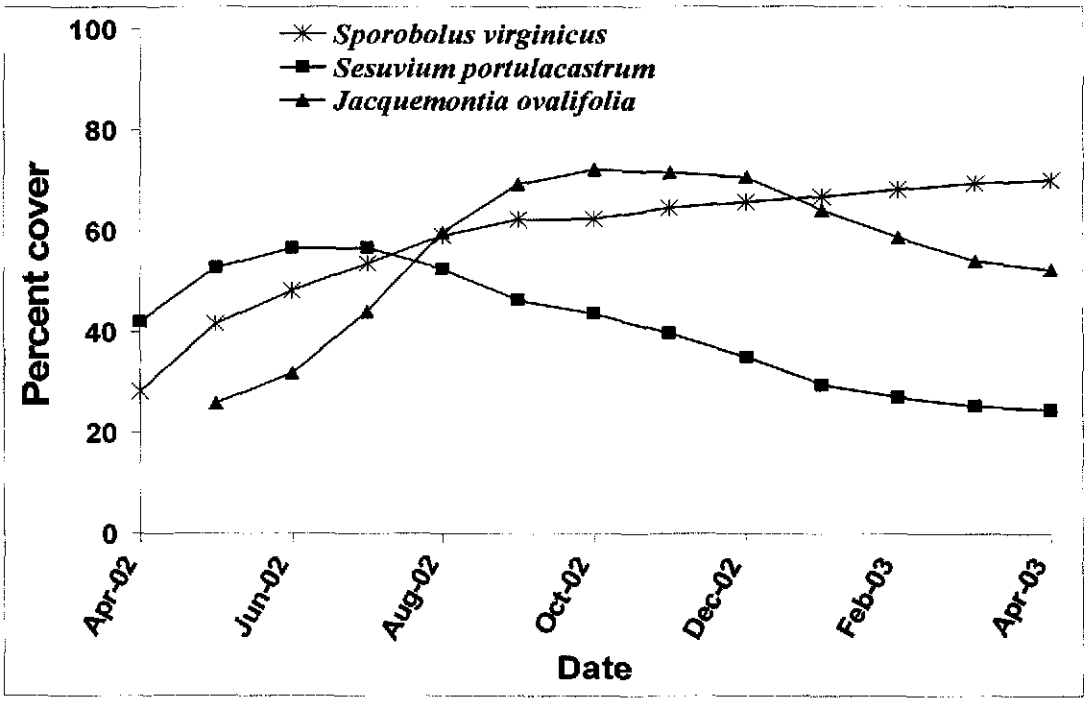


Figure 9. Monthly average percent cover (mean of absolute cover ± 1 SD) of outplanted native groundcover species. Averages are based on planting plots for each species individually; control plots and planting plots with 100% mortality are not included.

Despite slightly declining health, average percent cover of *Sporobolus virginicus* continued to rise throughout the study period. Average percent cover of *Cyperus javanicus* increased through August 2002, then plateaued and remained around 83 percent for the duration of the study. Average percent cover of *Jacquemontia ovalifolia* steadily increased

until mid-October 2002, when it reached its maximum average percent cover of 72.4 ± 16.0 (SD). At this point, average percent cover began declining in *J. ovalifolia* outplanting plots.

Non-planted Species Composition

Prior to outplanting of native species, existing vegetation in sedge outplanting and control plots was dominated by *Pluchea indica, Bacopa monnieri, Batis maritima,* and *Typha latifolia* (Table 3). Almost 97 percent of existing vegetation in these plots was composed of only five species. Existing vegetation in native groundcover outplanting and control plots was dominated by similar species, with *P. indica* and *B. maritima* providing almost 80 percent of total existing vegetative cover (Table 3).

Table 3. Species composition in study plots prior to outplanting (March 2002). Relative cover of dominant species in outplanting and control plots prior to outplanting of native species.

	Relative Cover (%)	
Species	**Sedge plots**	**Groundcover plots**
Pluchea indica	32.7	58.9
Bacopa monnieri*	30.8	5.0
Batis maritima	17.3	19.3
Atriplex suberecta	--	9.0
Typha latifolia	10.2	2.1
Bolboschoenus maritimus*	5.7	1.7
Total	**96.8**	**96.1**

* = native (indigenous) species

Shortly after outplanting of native species, non-planted vegetation began reappearing in the study plots. After one year, four of the five species (*Bacopa monnieri,*

Batis maritima, Pluchea indica, and *Typha latifolia)* that were most abundant in sedge plots prior to outplanting had re-colonized and were again among the most common species recorded in sedge planting plots (high density / competition and low density / competition) and control (control / competition) plots (Figure 10; Appendix B).

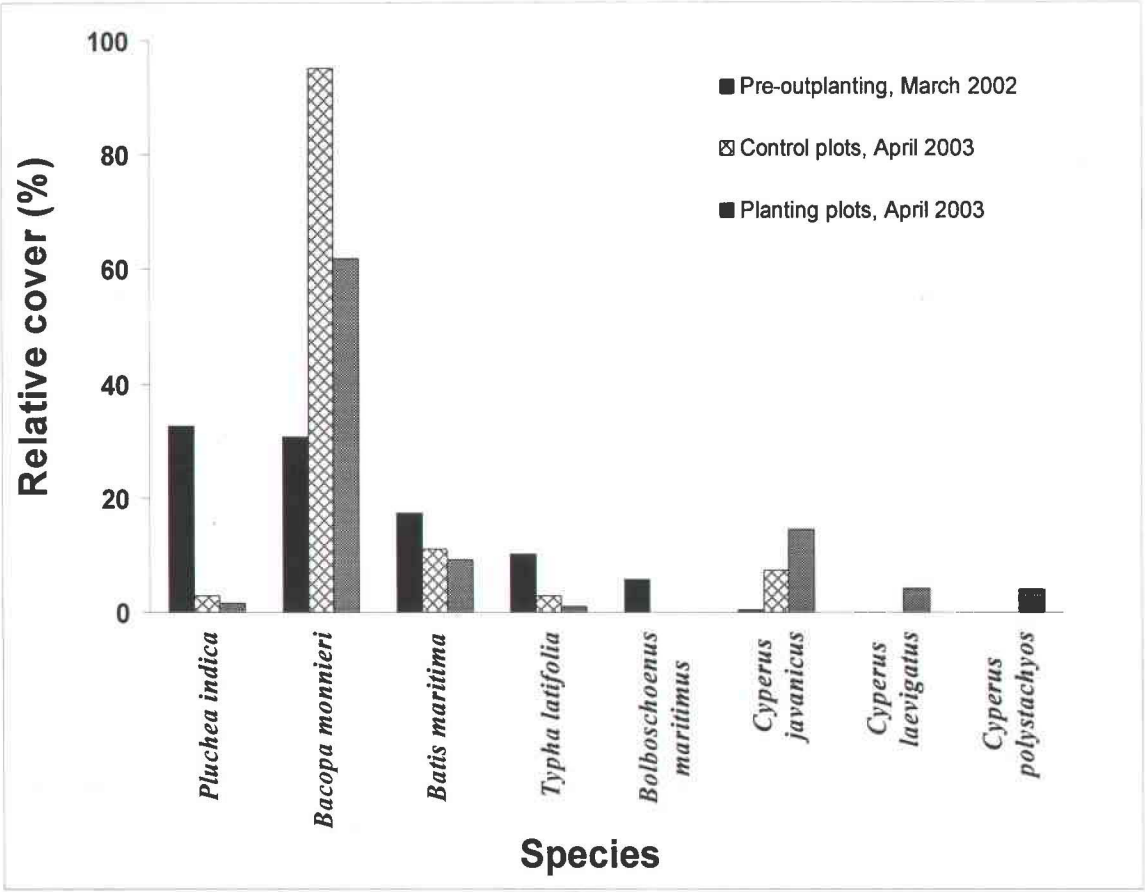


Figure 10. Overall relative cover of dominant species in sedge planting and control plots ("competition" plots only). Columns indicate relative cover prior to outplanting of native species and one year later (April 1, 2003) in planting plots and control plots. "Planting plots" = high density / competition + low density / competition plots combined. Control plots = control / competition treatment plots.

Although seedlings of non-planted species were observed in planting and control plots, the majority of the cover of non-planted species was the result of vegetative growth of established plants growing into planting and control plots. The most widespread non-

planted species in competition treatment plots at the end of the study was *Bacopa monnieri.* Ninety-five percent of all vegetative cover in sedge control plots consisted of *B. monnieri* (Figure 10). Relative cover of *B. monnieri* in sedge planting plots was 62 percent.

Despite being among the most common species recorded in sedge planting and control plots at the end of the study, vegetative cover of *Pluchea indica, Batis maritima,* and *Typha latifolia* had not attained pre-outplanting levels. Relative cover prior to outplanting was 32.7, 17.3 and 10.2 percent for *P. indica, B. maritima,* and *T. latifolia* respectively. One year post-outplanting, these numbers had dropped to less than 12 percent for *B. maritima* and less than 3 percent for *P. indica* and *T. latifolia* (Figure 10).

A similar pattern occurred in groundcover plots (planting plots and control plots). After one year, three of the five most common species in groundcover plots prior to outplanting had re-colonized, and were again the three most dominant species in groundcover planting (high density / competition and low density / competition) and control (control / competition) plots (Figure 11). However, *Pluchea indica,* which was responsible for almost 60 percent of total vegetative cover in groundcover plots prior to outplanting (Table 3), did not regain this dominance after a year's time. In April 2003, *P. indica* provided just 14.6 percent of total cover in planting plots and 24.1 percent of total cover in control plots. As in the sedge plots, the most widespread species in groundcover plots at the end of the study was *Bacopa monnieri.* Approximately 32 percent of vegetative cover in planting plots and 34 percent in control plots consisted of *Bacopa monnieri.*

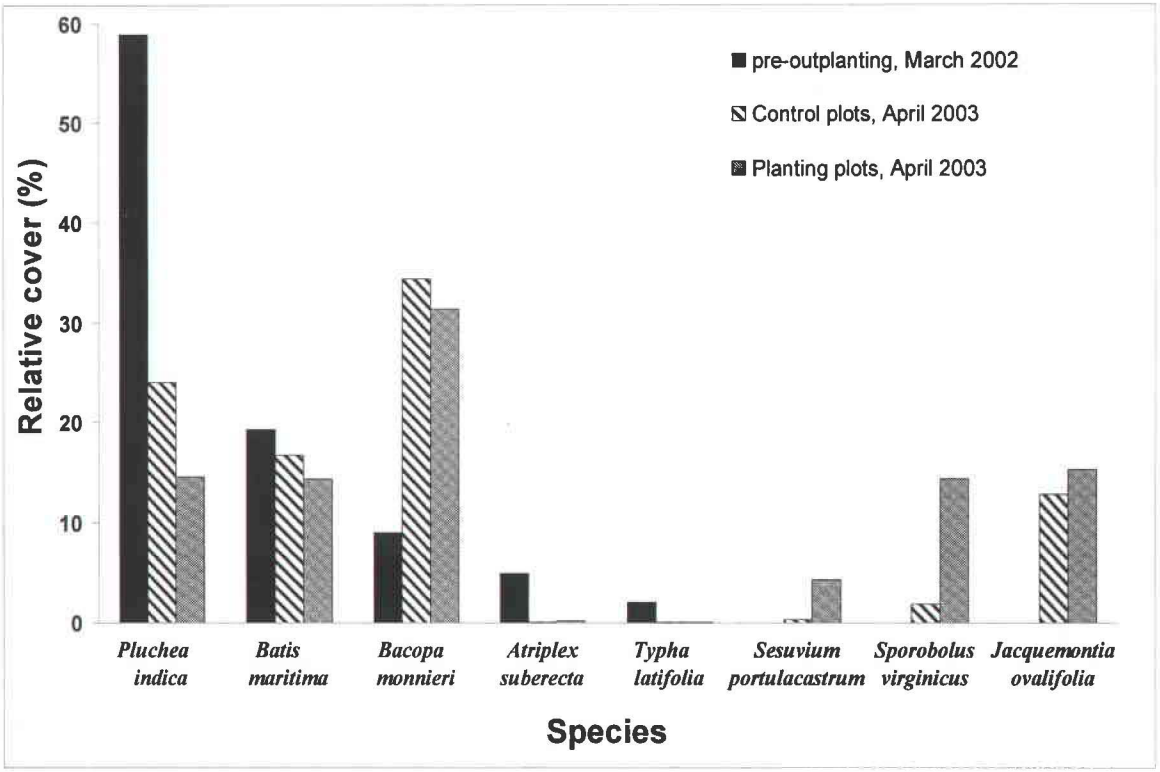


Figure 11. Overall relative cover of dominant species in groundcover planting and control plots ("competition" plots only). Columns indicate relative cover prior to outplanting of native species and one year later (April 1, 2003) in planting plots and control plots. "Planting plots" = high density / competition + low density / competition plots combined. Control plots = control / competition treatment plots.

Batis maritima, with 16.7 percent of total cover, was the third most abundant cover in control plots. However, cover in planting plots (i.e., control plots not included) was dominated first by *Bacopa monnieri* and second by *Jacquemontia ovalifolia.* Plantings of *J. ovalifolia* produced dense cover in plots where seedlings had been planted. As well, *J. ovalifolia* had managed to spread into adjacent planting plots and control plots of the other groundcover species. *Atriplex suberecta* and *Typha latifolia*, which had been the fourth and fifth most abundant species prior to outplanting, were virtually absent from planting plots and control plots in April 2003 (Figure 11).

Outplanting Treatments

Density

Initial planting density did not significantly affect final percent survival, or average health-code, maximum height, number of shoots, or number of reproductive stems in any of the outplanted species (Appendix C). Final average percent cover (absolute cover), in general, depended more on whether there was competition present than on initial planting density. Final average percent cover in both high density treatments of *Cyperus javanicus* (i.e., with competition and without competition) was greater than percent cover in either of the low density plots (Figure 12).

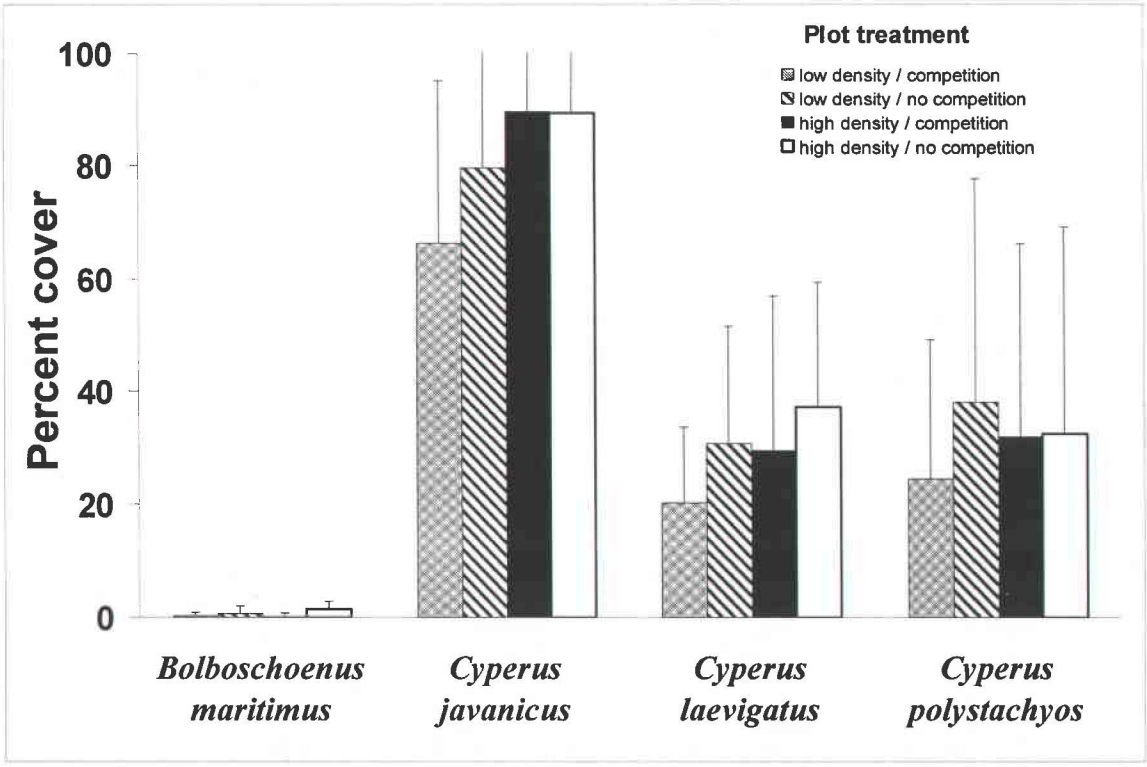


Figure 12. Final average percent cover (mean ± 1 SD) of outplanted native sedge species in "density" and "competition" treatment plots.

For the other six species, however, final average percent cover in both high density and low density plots with no competition was greater than final percent cover in high density and low density plots with competition (Figure 12; Figure 13). However, the differences in final average percent cover were not statistically significant for any species except *Sesuvium portulacastrum* and *Sporobolus virginicus* (Figure 13). Final average percent cover of *S. portulacastrum* was significantly greater in high density plots with no competition than in low density plots with competition ($p = 0.003$). Final average percent cover of *S. virginicus* was significantly greater in high density plots without competition (80.2 ± 20.9 S.D.) and low density plots without competition (83.8 ± 13.7 S.D.) than in low density plots with competition (50.0 ± 27.9 S.D.; $p=0.02$ and $p=0.008$ respectively).

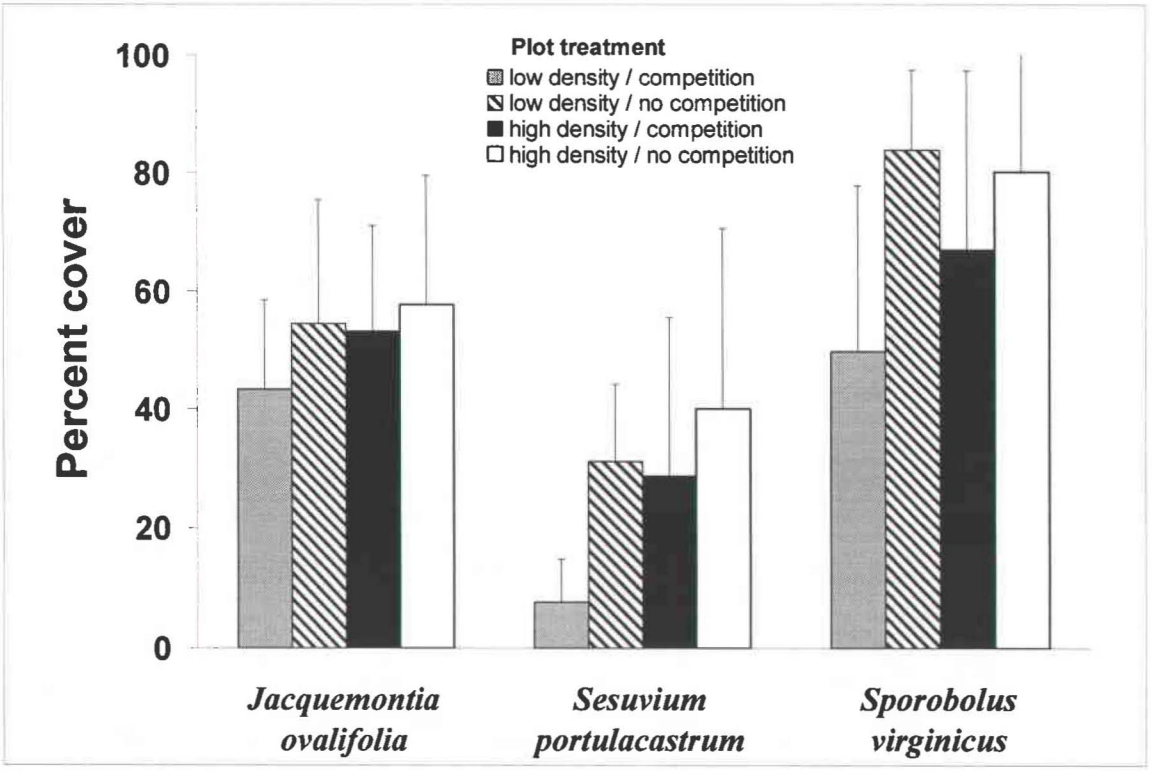


Figure 13. Final average percent cover (mean ± 1 SD) of outplanted native groundcover species in "density" and "competition" treatment plots.

Final average percent cover of all outplanted species was significantly greater in all high density and low density plots when compared to control plots except for *Jacquemontia ovalifolia* and *Bolboschoenus maritimus* (all tests of significance were conducted at the $p<0.05$ level). Final average percent cover of *J. ovalifolia* in low and high density plots was not significantly greater than final average percent cover in control plots. High mortality of *B. maritimus* resulted in low final percent cover in all planting plots.

Final average percent cover of non-planted species was similar in high density and low density planting plots of all outplanted species. Although final average percent cover of non-planted species was greater in control / competition plots than in low density / competition or high density / competition plots of all species (except *Bolboschoenus maritimus)*, these differences were statistically significant in only *Sporobolus virginicus* (Figure 14; Figure 15). A two-way ANOVA with block and density as fixed factors showed a significant difference among density treatments in final average percent cover of non-planted species ($F_{2,18}=6.16$, $p=0.009$). Tukey's HSD showed that final average percent cover of non-planted species in control plots (154.2 ± 34.2 (SD)) was significantly greater than in low density (105 ± 45.7 (SD); $p=0.04$) or high density (90.4 ± 44.6 (SD); $p=0.01$) plots.

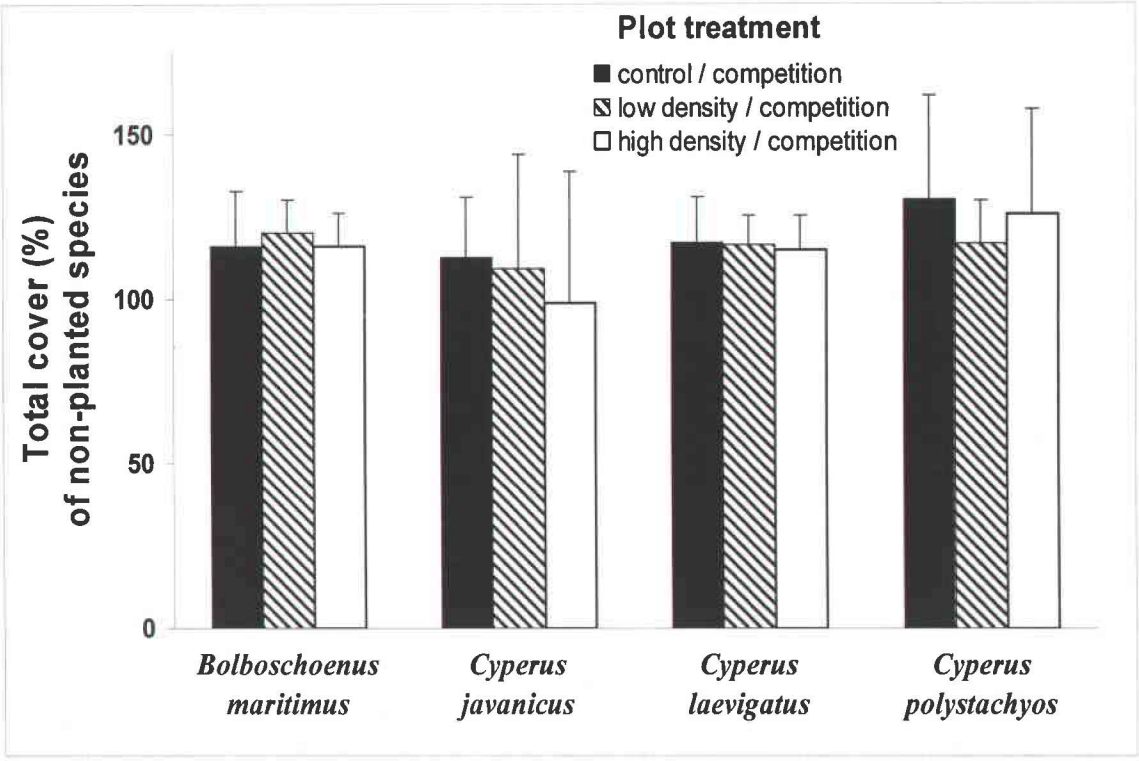


Figure 14 Absolute cover (mean ± 1 SD) of non-planted species in sedge planting plots on April 1, 2003. Values are the average of the total percent cover of all non–planted species documented in each planting plot. Values may exceed 100% as multiple species occurred in many plots.

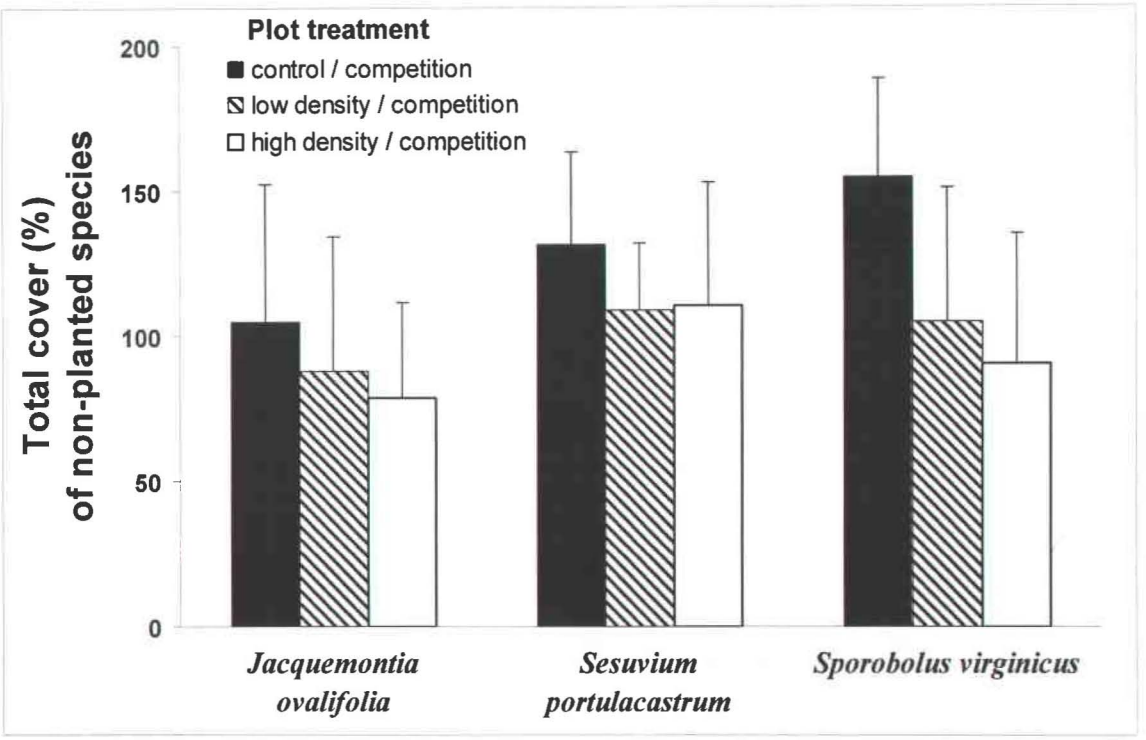


Figure 15. Absolute cover (mean ± 1 SD) of non-planted species in groundcover planting plots on April 1, 2003. Values are the average of the total percent cover of all non–planted species documented in each planting plot. Values may exceed 100% as multiple species occurred in many plots.

Initial density of outplanted native species did, however, influence final average biomass of non-planted species. Final average dry weight of non-planted species, removed from no competition plots, was significantly greater in control plots than in high density plots of all species (except *Bolboschoenus maritimus, Jacquemontia ovalifolia,* and *Sesuvium portulacastrum*) (Table 4). Final average dry biomass in control /no competition plots was significantly greater than in low density / no competition plots for all species except *B. maritimus, J. ovalifolia, S. portulacastrum,* and *Sporobolus virginicus.* Final average dry biomass was not significantly different in low density / no competition than in high density / no competition plots in any species.

Table 4. Dry biomass (g) (mean ± 1 SD) of non-planted species removed from no competition planting plots on April 1, 2003. Averages are listed by each of three different planting treatments.

	Mean Biomass (g)		
	Treatment (initial planting density)		
Planting plots of:	control /no competition	low density / no competition	high density / no competition
Bolboschoenus maritimus	16.56 ± 6.69[a]	17.92 ± 7.67[a]	17.69 ± 5.40[a]
Cyperus javanicus	15.83 ± 10.87[a]	7.18 ± 5.61[b]	5.94 ± 4.47[b]
Cyperus laevigatus*	17.52 ± 9.57[a]	12.35 ± 4.19[b]	9.21 ± 5.49[b]
Cyperus polystachyos	18.13 ± 6.33[a]	10.79 ± 5.66[b]	9.78 ± 4.70[b]
Jacquemontia ovalifolia	2.95 ± 2.37[a]	3.38 ± 4.92[a]	4.23 ± 3.71[a]
Sesuvium portulacastrum	11.05 ± 9.31[a]	6.40 ± 5.19[a]	5.44 ± 4.40[a]
Sporobolus virginicus	7.98 ± 4.90[a]	4.29 ± 4.16[ab]	2.52 ±2.35[b]

[a] All values are means ± 1 SD. For each species, values that have the same letter are not significantly different. All tests of significance were conducted at the $p<0.05$ level.
*data were square-root transformed prior to analysis to meet normality assumption.

Competition

Removal of non-planted species had little effect on final survival, health, maximum height, number of shoots or number of reproductive stems produced by outplanted species, with the exception of *Bolboschoenus maritimus* and *Sporobolus virginicus* (Appendix C). Final average survival of *B. maritimus* transplants was significantly greater in no competition plots (12.9% ± 18.5 SD)) than in competition plots (1.8% ± 6.0 SD)) (Mann-Whitney Rank Sum Test: T=328.5; p=0.03; n=20). Final average number of reproductive stems per *B. maritimus* individual was also greater in no competition plots (0.70 ± 0.86 SD)) than in competition plots (0.05 ± 0.22 SD)). The other exception was noted in the final physical appearance of *S. virginicus*. Final average health-code of *S. virginicus* individuals

in no competition plots (3.67 ± 0.90 SD) was significantly greater than in competition plots (2.78 ± 0.63 SD) (t = -3.564; df =37; p=0.001) (Appendix C).

Although final average percent cover was greater in no competition plots than in competition plots of all species, this difference was only significant in *Bolboschoenus maritimus* (median = 4.0 and 2.0 for no competition and competition plots; Mann-Whitney Rank Sum Test: T= 277.5; p<0.0001; n=20), *Sesuvium portulacastrum* (median = 24.0 and 8.0 for no competition and competition plots respectively; Mann-Whitney Rank Sum Test: T=334.5; p=0.042; n=20), and *Sporobolus virginicus* (mean = 82.0 ± 17.3 SD) and 58.6 ± 29.6 SD) in no competition and competition plots respectively; *t*=3.049; df=38; p=0.004) (Appendix C).

Environmental Parameters

Soil Salinity

Final average percent survival, physical appearance, and maximum height were not correlated with average soil salinity (per plot) in *Bolboschoenus maritimus, Jacquemontia ovalifolia, Sesuvium portulacastrum,* and *Sporobolus virginicus.* Final average percent survival was negatively correlated with average soil salinity in *Cyperus javanicus* (r = -0.743, p=0.014; n=40), *C. laevigatus* (r = - 0.379, p=0.016; n=40), and *C. polystachyos* (r = -0.723, p<0.0001; n=40). Final average health-code was not correlated with average soil salinity in plots of *B. maritimus, C. laevigatus, J. ovalifolia, S. portulacastrum,* and *S. virginicus.* Final average health-codes were negatively correlated with average soil salinity in outplanting plots of *C. javanicus* (r = -0.343; p=0.03; n=40) and *C. polystachyos* (r = -0.675; p<0.0001; n=40). Final average number of shoots and

number of reproductive stems were not correlated with average soil salinity in any species.

Final average percent cover was only correlated with overall average soil salinity in *Cyperus polystachyos* and *Sesuvium portulacastrum.* In *C. polystachyos* outplanting plots, final average percent cover was negatively correlated with average plot soil salinity ($r = -0.603$; $p < 0.0001$; $n=40$), whereas in *S. portulacastrum* average percent cover was positively correlated with average soil salinity ($r = 0.413$; $p = 0.008$; $n=40$).

Soil Temperature

Effects of soil temperature were only correlated with survival and growth parameters in *Cyperus javanicus* and *C. polystachyos.* Final average percent survival ($r = -0.381$; $p = 0.02$; $n=40$), health-code ($r = -0.565$; $p=0.001$; $n=40$), maximum height ($r = -0.541$; $p = 0.002$; $n=31$), and percent cover ($r = -0.436$; $p=0.005$; $n=40$) were all negatively correlated with average soil temperature in plots of *C. polystachyos*. Final average number of reproductive stems per plot was not correlated with soil temperature in *Cyperus polystachyos* plots. Final average percent cover was also negatively correlated with average soil temperature ($r = -0.410$; $p = 0.03$; $n=40$) in outplanting plots of *C. javanicus.*

Average plot soil temperature was not correlated with final average percent survival, health-code, maximum height, number of shoots, and number of reproductive stems in plots of *Bolboschoenus maritimus, Cyperus javanicus, C. laevigatus, Jacquemontia ovalifolia, Sesuvium portulacastrum,* and *Sporobolus virginicus.*

Individual Native Species Results

Bolboschoenus maritimus

Transplants of *Bolboschoenus maritimus* initially flourished after outplanting. Beginning in November 2002, however, physical appearance of transplants began to decline and shoot senescence was becoming widespread in all planting plots. By December 2002 only 51.5% of transplanted *B. maritimus* plants still had live stems and average health-code had dropped from 4.75 ± 0.56 (SD), as recorded in April 2002, to 2.30 ± 1.25 (SD). When monitored in early March 2003, only 11 (4.2%) of the transplanted *B. maritimus* individuals had live shoots. However, beginning in late March 2003, new shoots started appearing in planting plots next to tags marking the original transplanted individuals. Presumably the underground tubers/culms of many *B. maritimus* transplants had come out of dormancy and begun to produce new shoots. By May 1, 2003, 72 *B. maritimus* plants (27.7% of the original transplants) had produced new shoots and production of new, healthy shoots caused the health-code to climb to an average of 4.88 ± 0.34 (SD).

Bolboschoenus maritimus propagules had an average maximum height of 21.42 cm ± 6.40 (SD) and an average of 1.1 ± 0.3 (SD) shoots when transplanted. At the conclusion of the study, the average maximum height had dropped to 16.25 cm ± 7.53 (SD) and the average number of shoots per individual was only 1.4 ± 0.6 (SD). This drop in height was primarily because the majority of live shoots at the conclusion of the study were newly emerged and had not attained the stature of mature *B. maritimus* shoots. The peak of vegetative growth for *B. maritimus* individuals occurred in June 2002 when the

average number of shoots per individual was 3.50 ± 1.70 (SD) and the average maximum height was 41.77 cm ± 12.56 (SD).

Initially transplants of *Bolboschoenus maritimus* showed high reproductive rates. By July 2002, 73 percent of transplanted individuals had at least one shoot in some stage of reproduction (i.e., flowering, seed ripening, seed setting or seedhead senescing). By April 2003 only 15 percent of live individuals were reproductive. Despite prolific seed set, no *B. maritimus* seedlings were observed in any of the planting or control plots during the study period.

Initial average percent cover of *Bolboschoenus maritimus* was 24.6 ± 3.9 (SD) in high density (high density / competition plus high density / no competition) plots and 13.7 ± 3.3 (SD) in low density (low density / competition plots plus low density / no competition plots). For the first three months of the study, average percent cover increased in all planting plots. After this point, percent cover steadily declined through early April 2003. One year post-planting, average percent cover had dropped to 0.9 ± 1.2 (SD) in high density and 0.5 ± 1.1 (SD) in low density plots. By May 2003, average percent cover had risen to 3.6 ± 3.2 (SD) in high density plots, 2.1 ± 2.1 (SD) in low density plots and 0.4 ± 0.8 (SD) in control plots. Final cover averaged 1.1 ± 1.0 percent in competition plots (high density / competition + low density / competition) and 4.5 ± 2.9 percent in no competition (high density / no competition + low density / no competition).

Cyperus javanicus

In May 2003, over a year after outplanting, 97 percent of outplanted *Cyperus javanicus* plants possessed live shoots and the majority of *C. javanicus* individuals appeared quite healthy and robust. Eighty-five percent of live plants in April 2003 had a health-code of 3 or above. Eight of the nine plants that did not survive were located in one outplanting block. This block had the highest overall average soil salinity (11.9 mmhos / 9.5 ppt) of all *C. javanicus* planting blocks and average plot soil salinity was negatively correlated with both final average survival and final average health-code ($r = -0.743$; $p=0.014$; $n=40$ and $r = -0.343$; $p = 0.03$; $n=40$, respectively). Average health-code for entire planting plots ranged from 1.25 to 5.

Initial number of shoots for *Cyperus javanicus* seedlings averaged 5.3 ± 4.1 (SD) per individual. Within three months, this number had almost doubled to 10.2 ± 5.5 (SD). Production of new shoots continued throughout the study, although production slowed between September 2002 and March 2003. One year after outplanting, the average number of shoots per individual had risen to 12.6 ± 9.7 (SD). The range in final number of shoots per individual was considerable. Individuals exhibiting poor health had as few as one shoot, whereas the healthiest individual possessed 88 shoots. Average maximum height remained relatively stable throughout the study period. Initial average maximum height was 62.56 cm ± 12.95 (SD) and final average of maximum height was 70.76 cm ± 36.31 (SD).

Reproduction occurred rapidly after outplanting in *Cyperus javanicus*. By June 1, 2002, two months after outplanting, almost 70 percent of *C. javanicus* plants had produced at least one reproductive stem. Although reproductive individuals averaged between 70 and 76 percent from mid-June 2002 through February 2003, production of

new reproductive stems slowed sharply between July and November 2002. The majority of stems recorded as reproductive during this period were in the seed ripening, seed setting and senescent stages of reproduction (Figure 16). There was a slight surge in production of new reproductive stems in November and December 2002, but, once again, from January through April 2003, there were essentially no new reproductive stems being produced. During the last monitoring period in May 2003, another large increase in the number of new reproductive stems was documented (Figure 14).

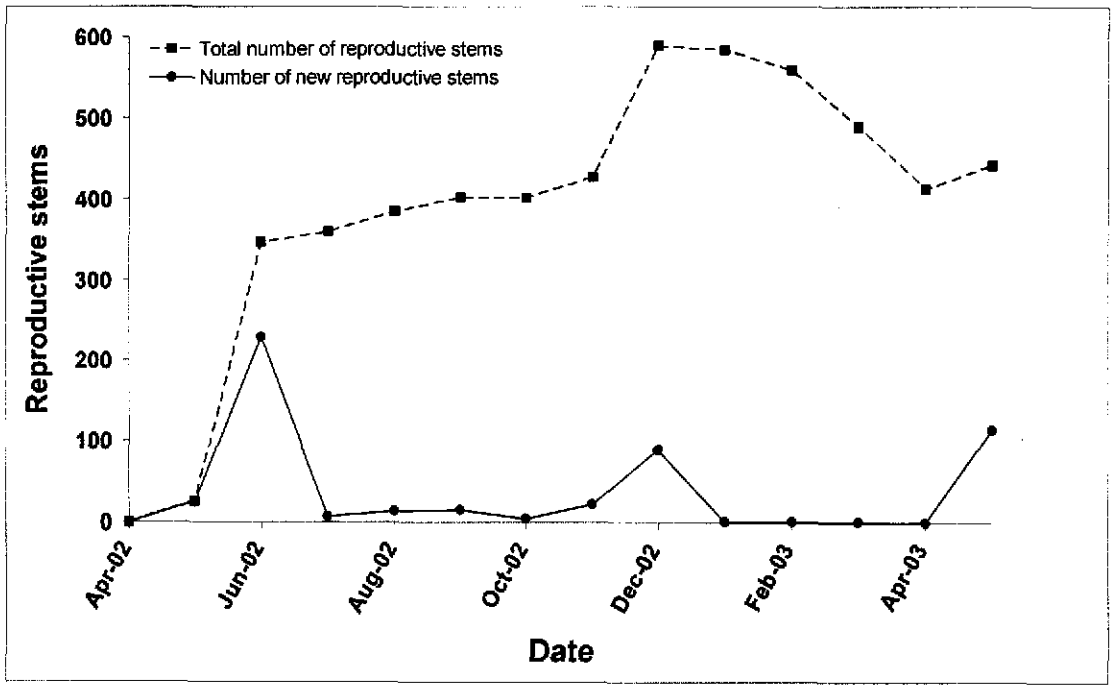


Figure 16. Monthly totals of all reproductive stems and newly emerging reproductive stems in *C. javanicus* outplanting plots. Total reproductive stems includes stems in all stages of reproduction (i.e., elongation of reproductive stem, flowering, seed ripening seed setting and senescence). Number of new reproductive stems includes only stems in the "elongation of reproductive stem" stage of reproduction.

Cover of *Cyperus javanicus* increased considerably in both low density and high density plots over the course of the study period. Initial percent cover was, on average, 38.7 ± 7.5 (SD) in low density plots and 62.5 ± 10.4 (SD) in high density planting plots. One year later, average percent cover was 90.2 ± 21.5 (SD) in high density plots, 75.6 ±

25.0 (SD) in low density plots, and 14.7 ± 31.2 (SD) in control plots. Final average percent cover was 78.1 ± 26.3 (SD) in competition plots and 84.3 ± 24.6 (SD) in no competition plots.

Cyperus laevigatus

Although 76 percent of outplanted *Cyperus laevigatus* individuals were still alive one year after outplanting, the physical appearance of the majority of the plants was quite poor. In fact, 61 percent of live plants had a health-code of 2 (i.e., 61-80% of plant dead or dying) or worse on April 1, 2003. The poorest overall health for *C. laevigatus* individuals occurred in early March of 2003 when the average health-code dropped to 2.19 ± 1.02 (SD). Physical appearance did begin to improve in mid-April, perhaps due to the onset of heavy rains in March. By May 2003 the average health-code had climbed slightly to 2.76 ± 1.21 (SD). Final average planting plot health-codes in May 2003 ranged from 1.8 to 4.3.

The range in average physical appearance and percent survival in plots of *Cyperus laevigatus* tended to correspond with planting block. Mortality in three of the ten planting blocks accounted for 58 percent of total mortality in *C. laevigatus*. High soil salinity may have contributed to some of the mortality and poor health observed in *C. laevigatus*. Planting plots in Block 4, the planting block with the highest death rates and lowest final health-code, also had the highest average soil salinity (12.3 mmhos (9.8 ppt) ± 0.6 SD). Planting plots in a nearby block, which had the second highest soil salinity (overall average 12.1 mmhos (9.7 ppt) ± 1.2 SD), however, had a 93 percent survival rate Individuals in this block also appeared quite healthy and robust (average health -code = 3.85 ± 1.42 SD). Overall average soil salinity was negatively correlated with final

percent survival in each planting plot ($r = -0.379$; $p=0.016$; $n=40$), but was not correlated with final average health in each planting plot.

Average height of *Cyperus laevigatus* individuals showed little fluctuation during the study period. The lowest average maximum height of 19.56 cm ± 5.24 (SD) was recorded on April 14, 2002. The greatest average maximum height, 23.93 cm ± 6.46 (SD), actually occurred at the time of outplanting. For the duration of the study, average maximum height fluctuated around 21 cm. The final average maximum height of live *C. laevigatus* individuals was 21.92 ± 6.73 (SD) cm.

Outplanting stock of *Cyperus laevigatus* was propagated from divisions of mature nursery plants. Thus, many plants already had reproductive stems prior to outplanting. In fact at the beginning of the study, 97 percent of *C. laevigatus* individuals had at least one reproductive stem. This prolific reproduction continued throughout the study period. Percent reproductive individuals never dropped below 92 percent, and from August through November 2002, 100 percent of *C. laevigatus* plants had at least one reproductive stem. Average number of reproductive stems per plant ranged from greater than 10 in April 2002 to greater than 30 in July of 2002. At the end of the study, the average number of reproductive stems was slightly greater than 20 per plant.

Initial average percent cover of *Cyperus laevigatus* was 35.6 ± 8.6 (SD) in high density plots and 19.6 ± 6.9 (SD) in low density plots. Maximum average percent cover for high density plots (60.3 ± 12.2 (SD)) and low density plots (39.4 ± 14.6 (SD)) occurred in July 2002. One year after outplanting, average percent cover had decreased to 33.3 ± 24.7 (SD) in high density plots and 25.4 ± 17.9 (SD) in low density plots. *Cyperus laevigatus* was never observed in any control plots during the course of the

study. Final average percent cover was 33.9 ± 21.3 (SD) in no competition and 24.8 ± 21.6 (SD) in competition plots.

Cyperus polystachyos

Initial survival rates of *Cyperus polystachyos* were quite high. In September 2002, six months after outplanting, more than 96 percent of *C. polystachyos* plants were still alive. Although physical appearance of *C. polystachyos* started rapidly declining by August 2002, it was not until mid-November that survival rates declined sharply. This decline continued through the end of the study. One year after outplanting, only 173 of the original 300 plants (58%) were alive. Almost 50 percent of mortality occurred in two planting blocks and more than 60 percent of mortality occurred in three planting blocks. All individuals in planting plots in Blocks 3 and 4 died before the end of the study. Similar to *Cyperus javanicus* and *C. laevigatus,* average soil salinity recorded in Block 4 (12.2 mmhos (9.7 ppt)) was the highest of the *C. polystachyos* outplanting blocks. Average soil salinities in Blocks 3 and 5, the other blocks with high mortality rates, were 8.1 mmhos (6.5 ppt) and 11.3 mmhos (9.0 ppt) respectively. Highest final average survival rates for *C. polystachyos* occurred in planting plots with lower average salinity measurements. Final percent survival in *C. polystachyos* planting plots was strongly negatively correlated with average soil salinity ($r = -0.723$; $p<0.0001$; $n=40$).

Physical appearance of *Cyperus polystachyos* individuals declined rapidly beginning in August 2002. This decline continued through March 2003. Production of new vegetative and reproductive shoots resumed in mid-April, after a period of virtually no new production, causing an increase in the average health-code. Average health-code

of live *C. polystachyos* individuals increased from a low of 1.46 ± 0.79 (SD), in mid-February 2003, to 2.37 ± 1.35 (SD) by May 2003.

Height of *Cyperus polystachyos* plants increased dramatically in the first two months after outplanting. Average maximum height of C. *polystachyos* seedlings in April 2002 was 48.79 ± 5.15 (SD) cm. By June 2002 the average maximum height had increased to 73.99 ± 17.49 (SD) cm. Average maximum height declined beginning in mid-July, corresponding with declining health of *C. polystachyos* plants. Final average maximum height of live plants was 32.14 ± 12.12 (SD) cm. Presumably, this average would have, once again, started to rise with the increase in production of new shoots and reproductive stems that began in April 2003.

Reproductive stems developed in the majority of *Cyperus polystachyos* plants shortly after outplanting. By May 1, 2002, 99 percent of outplanted individuals had produced at least one reproductive stem. More than 99 percent of individuals possessed reproductive stems through mid-December. These stems contained primarily ripening and senescing seed heads. Production of new reproductive stems essentially halted in early August. The average number of reproductive stems per live individual declined from a high of 56.36 ± 23.56 (SD) in mid-July 2002, to a low of 0.70 ± 1.19 (SD) in late March 2003. Production of new reproductive stems began to recur in mid-April 2003. This rise in reproduction at the end of the study increased the average number of reproductive stems per individual to 1.71 ± 2.98 (SD) by May 2003.

Healthy *Cyperus polystachyos* plants provided dense cover in both high density and low density plots. Initial average percent cover was 70.4 ± 11.5 (SD) in high density plots and 42.3 ± 8.0 (SD) in low density plots. By mid-May 2002, average percent cover

had risen to 92.1 ± 14.0 (SD) in high density and 77.2 ± 14.6 (SD) in low density plots. Average percent cover for all plots rose until mid-June, at which point percent cover began to decline. Final average percent cover was 32.1 ± 34.8 (SD) in high density and 31.2 ± 33.0 (SD) in low density plots. Final average percent cover was 35.2 ± 37.4 (SD) in no competition plots and 28.1 ± 29.5 (SD) in competition plots. *Cyperus polystachyos* was never observed in any control plots.

Jacquemontia ovalifolia

After one year, 71.3 percent of the original 300 *Jacquemontia ovalifolia* plants were still alive. Unlike all other outplanted species (except *Bolboschoenus maritimus*, in which mortality was widespread), mortality of *J. ovalifolia* individuals was randomly distributed among planting blocks and planting plots. Final overall percent survival was higher in no competition plots (75.5%) than in competition plots (69%) and higher in low density plots (75%) than in high density plots (69.5%). However, the differences in final percent survival were not significantly different between competition and no competition treatment plots or between high density and low density treatment plots.

Physical appearance (health-code) of *Jacquemontia ovalifolia* remained relatively steady from May through August 2002. Physical appearance dropped sharply from August to November 2002 and then declined more gradually from November 2002 through April 2003. Despite overall declining health, the majority of *J. ovalifolia* plants were relatively healthy at the end of the study period. More than 63 percent of planting plots had an average health-code of 3 or more on April 1, 2003. Planting plot health-code at this point averaged 2.80 ± 0.99 (SD).

Two distinct flowering peaks were exhibited by *Jacquemontia ovalifolia* during this study period. The first peak in reproductive output occurred two months after outplanting. Average number of flowers per plot rose from an average of 1.45 ± 1.34 (SD) at the time of outplanting to an average of 7.73 ± 5.69 (SD) on July 1, 2002. Flowering production slowed after July and reproductive output dropped to an average of 1.15 ± 2.89 (SD) flowers per plot in October 2002. Very few flowers were produced in October or November 2002, but flower production surged again from December 2002 through February 2003. Average number of flowers per plot in mid-February 2003 was 6.35 ± 5.68 (SD). At this point, reproduction slowed again and number of flowers declined steadily from the end of February 2003 to April 2003. Final average number of flowers per plot was 2.43 ± 4.72 (SD).

Jacquemontia ovalifolia plants grew rapidly and spread throughout entire planting blocks. Initial cover of *J. ovalifolia* in outplanting plots averaged 32.4 ± 5.3 percent in high density plots and 19.0 ± 3.2 percent in low density plots. By mid-August, growth of *J. ovalifolia* from planting plots had extended into 95 percent of the control plots and by November 2002 average percent cover had risen to 77.8 ± 20.4 (SD) in high density, 65.6 ± 16.7 (SD) in low density, and 44.3 ± 30.3 (SD) in control plots. Final average percent cover of *J. ovalifolia* was similar in all density treatments, 51.8 ± 27.0 (SD) in control, 49.0 ± 18.7 (SD) in low density, and 55.6 ± 19.5 (SD) in high density plots. Final average percent cover in no competition plots was 57.2 ± 22.2 (SD) and 47.1 ± 20.6 (SD) in competition plots.

Sesuvium portulacastrum

Despite its rapidly declining physical appearance, survival of *Sesuvium portulacastrum* was greater than 90 percent for the first six months after outplanting. By the end of June, although survival was 98 percent, the majority of *S. portulacastrum* plants were in poor health. Beginning in October 2002, survival began to steadily decrease. By the end of January 2003, percent survival had dropped to 73 percent and average plot health-code reached its lowest level of 1.62 ± 0.80 (SD). One year after outplanting, overall survival had dropped to 65 percent and appeared as though it would continue to drop. Forty-one percent of *S. portulacastrum* mortality was located in two planting blocks and 70 percent of mortality was located in four of the ten planting blocks. However, unlike many of the other species, the planting areas with high *S. portulacastrum* mortality were not the areas with the highest soil salinity or highest soil temperatures. In fact the planting areas with the highest final percent survival were the planting blocks with the highest overall average salinities and final percent survival was positively correlated with average soil salinity.

Sesuvium portulacastrum outplantings did not produce flowers until one month after outplanting. Maximum reproduction occurred in June 2002 with an average of 9.8 ± 8.8 (SD) flowers per plot. Reproduction declined from the end of June through September 2002. A slight surge of reproduction occurred in October, after which reproduction sharply declined. The lowest average number of flowers per plot, 0.2 ± 0.8 (SD), was recorded in February 2003. Reproduction increased again slightly from February through April 2003 and at the conclusion of the study, the average number of flowers per plot was 1.4 ± 3.1 (SD).

Dense cover of *Sesuvium portulacastrum* occurred in relatively few outplanting plots. Initial percent cover of *S. portulacastrum* in outplanting plots averaged 52.3 ± 10.9 (SD) in high density and 31.8 ± 8.4 (SD) in low density plots. Maximum average percent cover of 68.7 ± 17.0 (SD) in high density and 45.1 ± 16.7 (SD) in low density plots occurred in late July 2002. From August 2002 through the end of the study, average percent cover of *S. portulacastrum* gradually declined. Final average percent cover was 34.5 ± 28.6 (SD) in high density, 14.5 ± 12.4 (SD) in low density, and 2.8 ± 6.7 (SD) in control plots. Final average cover in no-competition plots was 30.7 percent ± 24.9 (SD), and 18.3 percent ± 22.0 (SD) in competition plots.

Sporobolus virginicus

Outplanted populations of *Sporobolus virginicus* maintained high survival rates during the entire study period. After one year, 87 percent of the original outplanted individuals were still alive. Overall survival did not drop below 90 percent until October 2002, six months after outplanting. Fifty-two percent of final mortality was recorded from one planting block. Mortality in this area occurred rapidly after outplanting. Within one month after outplanting, only 14 of 38 *S. virginicus* plants located in this block were still alive. The block exhibiting the highest mortality also had the highest overall average soil salinity (12.4 mmhos (9.9 ppt)). However, plants in the area with the second highest average soil salinity measurements exhibited 100 percent survival after one year and final percent survival was not correlated with overall average salinity.

Prior to outplanting, many *Sporobolus virginicus* seedlings were quite tall and spindly with very few leaves. Physical appearance improved dramatically within one

month. Physical appearance declined from June through November 2002, but then remained relatively stable from November 2002 through March 2003. After one year, 75 percent of *S. virginicus* plants had a health-code of 3 or higher and average health-code at the end of the study was 3.24 ± 0.90 (SD). However, plants in no-competition plots appeared more robust than plants in competition plots. In fact final average health-code in no-competition plots (3.67 ± 0.90 (SD)) was significantly greater than in competition plots (2.78 ± 0.63 (SD)) ($t = -3.564$; $df=37$; $p=0.001$).

Outplanted *Sporobolus virginicus* individuals exhibited extensive vegetative growth and spread in the year following outplanting. Average number of tillers per individual at the time of outplanting was less than 5. The number of tillers per individual increased consistently through the study period and at the end of the study period the average number of tillers per individual was greater than 15. Many individuals had more than 25 live tillers at the end of the study. The average maximum height of seedlings at the time of outplanting was 24.87 ± 9.16 (SD) cm. Average maximum height increased through July until it reached a maximum average of 35.37 ± 9.63 (SD) cm. After July 2002, average maximum height remained relatively stable, fluctuating between 32 and 35 cm. After one year, average maximum height was 32.05 ± 7.89 (SD) cm.

Production of reproductive stems by *Sporobolus virginicus* did not occur until June of 2002. By mid-July approximately 80 percent of *S. virginicus* individuals had reproductive tillers and the average number of reproductive stems was 3.6 ± 3.4 (SD) per individual. Number of reproductive stems per individual continued to climb to its highest level, 5.1 ± 6.4 (SD), in mid-December 2002. Reproduction steadily declined from mid-December through the end of the study. By April 2003, less than 10 percent of *S.*

virginicus plants were reproductive and average number of reproductive stems per individual had dropped to 0.4 ± 1.7 (SD). The quantity of viable seed produced by *S. virginicus*, however, was questionable. Flowering stalks often contained no seeds, or if seeds existed, they often appeared white, indicating that they were not viable.

Initial percent cover of *Sporobolus virginicus* in outplanting plots averaged 31.4 ± 3.8 (SD) in high density and 19 ± 5.0 (SD) in low density plots. By August 2002 average percent cover in low density plots (54.0 ± 19.5 (SD)) was similar to average cover in high density plots (59.6 ± 21.6 (SD)). One year post-outplanting, average percent cover of *S. virginicus* had risen to 73.7 ± 26.2 (SD) in high density, 66.9 ± 27.5 (SD) in low density and 14 ± 21.4 (SD) in control plots. Final average percent cover was not significantly different in low density and high density plots. Final average percent cover in no-competition plots (82.0 ± 17.3 (SD)) was significantly greater than in competition (58.6 ± 29.6 (SD)) plots (t = 3.049; df=38; p=0.004).

Discussion

Survival and Growth

Prior to this study, there has been little documentation or quantification of survival rates of outplanted native Hawaiian wetland plant species. The results of this study indicate that native wetland plant species may vary in their ability to survive outplanting in brackish coastal wetlands. Based on their ability to survive, grow, and reproduce for the duration of this study, several native wetland plant species that would be useful to coastal wetland restoration efforts were identified.

Outplanted seedlings of all species initially thrived in their new environment at Honouliuli. However, at the conclusion of the study, only 67% of outplanted individuals were still alive. The majority of outplanted *Cyperus javanicus, Jacquemontia ovalifolia,* and *Sporobolus virginicus* plants were healthy and robust at the end of the study. Outplantings of *Bolboschoenus maritimus, Cyperus laevigatus, Cyperus polystachyos,* and *Sesuvium portulacastrum* were not as successful. Mortality in these species was quite high and many of the surviving individuals appeared chlorotic, stunted, and generally unhealthy. High mortality in *B. maritimus, C. laevigatus,* and *S. portulacastrum* was quite unexpected as these three species are common components of other brackish wetlands around the state (Stemmermann 1981; Koob 1999; Wagner 1999; pers. obs.).

Hydrological conditions during the study period, particularly low soil moisture, may have hindered survival and growth of outplanted species, particularly *Cyperus laevigatus* and *Bolboschoenus maritimus.* Hydrology and water level play an important role in growth and establishment of wetland plant species (Callaway *et al.* 1997; Keddy 1999; Middleton 1999). Many wetland plant species demonstrate stunted growth under lowered water levels (Hunter *et al.* 2000). High rates of survival and shoot production have been shown to occur in stands of *B. maritimus* that were grown in 5 to 25 cm of standing water (Kantrud 1996; Clevering and Van Gulik 1997). *Cyperus laevigatus* may also require standing water for increased growth and survival (Koob 1999), although some studies suggest that waterlogging is not required for normal growth of *C. laevigatus* (Onkware 2000). Although the plantings were done near the edge of Pond 2 at Honouliuli, reduction in the pond water level throughout the study period resulted in

planting plots being quite distant from open water at the conclusion of the study. Presumably, this lowered water level also resulted in a substantial reduction in soil moisture in planting plots from the beginning to the end of the study. While *C. laevigatus* and *B. maritimus* are obligate wetland plants that may grow best when submerged in shallow water, the other species in this study are facultative wetland species, and thus may not have been as affected by this decrease in water level and soil moisture (Reed 1988).

It is unclear why high mortality and poor health were exhibited in outplanting plots of *Sesuvium portulacastrum.* This species is a halophyte and should be able to tolerate the conditions of high salinity and drought encountered during this study period (Joshi 1981). Further, in other restoration projects in the state, this species has become established after outplanting (Sustainable Resources Group International 2000; Harrison 2002). Mortality in *S. portulacastrum*, as well as other outplanted species, may have been related to the genetic stock received from the nursery. Survival of restored vegetation may depend not only on the traits of a particular species, but also on the specific genotype of the seeds or propagules (Lesica and Allendorf 1999; Middleton 1999; Montalvo and Ellstrand 2000). Outplanting stock grown from seed or propagules from a distant source may not be adapted to conditions found at a restoration site (Keddy 1999; Lesica and Allendorf 1999; Montalvo and Ellstrand 2000; Fattorini 2001). The only species grown from locally collected seeds or propagules were *Bolboschoenus maritimus* and *Cyperus javanicus*. Seeds of *Cyperus polystachyos* were collected from James Campbell NWR. The rest of the species were grown from nursery stock, and the source populations were unknown.

Declining health, shoot death and cessation of reproduction, observed in many species beginning in mid-July, may also be related to the life history strategies of these species. Outplanted populations, most notably of *Bolboschoenus maritimus* and *Cyperus polystachyos,* appeared to go through a die-back and/or senescent period which began in late summer of 2002 and continued through winter. Beginning in late March 2003, production of new vegetative and reproductive shoots resumed in several outplanting plots. Growth and reproduction of clonal wetland plant species in temperate regions usually occurs in a similar cyclical pattern (Bernard 1990). *Carex* species in temperate regions demonstrate peak growth in spring after winter-induced dormancy is broken (Yetka and Galatowitsch 1999). Populations of *B. maritimus* in northern California exhibit winter dormancy, which begins in fall. This dormancy is followed by new shoot growth from underground corms in early spring, flowering in late spring, and peak growth in mid-summer (Kantrud 1996). The climate of coastal wetlands in Hawai`i would not necessitate winter dormancy of sedge populations. However, cyclic die-back (with winter senescence and shoot production in spring) has been observed in natural populations of *B. maritimus* (Eric Guinther, 'Ahahui Malama I Ka Lokahi, pers. comm., 2003), and experimental populations of *Cyperus laevigatus* (Van Dyke 2001). Long–term monitoring of the temporal occurrence of phenological stages in native sedge populations, natural and outplanted, would help elucidate whether the observed pattern of winter-dormancy and spring re-growth is truly cyclical, or whether it was related to environmental conditions during this study period.

Soil Salinity and Soil Temperature

Environmental conditions of a restoration site greatly influence the establishment of outplanted species (Bradshaw 1987; Fattorini 2001; Zedler 2001). Salinity is one abiotic factor that can profoundly restrict survival in outplanted populations of wetland species (Webb and Mendelssohn 1996; Callaway *et al.* 1997; Hootsmans and Wiegman 1998; Keddy 1999). Final survival rates of *Cyperus javanicus, C. laevigatus,* and *C. polystachyos* indicated that survival in these species was negatively correlated with high levels of soil salinity.

High mortality in all sedge species was recorded from "Block Four". This planting area consistently recorded the highest levels of soil salinity of all outplanting blocks. However, individuals of *Cyperus javanicus* and *Cyperus laevigatus* survived and thrived in other planting blocks with relatively high salinity levels. Further, these species are known to tolerate high levels of soil salinity (Stemmermann 1981; Koob 1999; Onkware 2000). There may be a threshold of soil salinity above which these species may not be able to survive. Concurrently, there may have been other abiotic factors specific to Block Four which contributed to the high mortality observed in this area. Survival of *Cyperus polystachyos,* on the other hand, appeared to be directly associated with soil salinity. High mortality in *C. polystachyos* occurred in all planting plots with average salinities exceeding 8 mmhos (6.4 ppt).

Widespread mortality in *Bolboschoenus maritimus* made it difficult to discern the effects of soil salinity on survival and health in this species. Survival and health of the three outplanted groundcover species, *Jacquemontia ovalifolia, Sesuvium portulacastrum,* and *Sporobolus virginicus* did not appear to be related to salinity levels.

This is not surprising, as those three species are known to tolerate saline and drought conditions (Joshi 1981; Blits and Gallagher 1981; Marcum and Murdoch 1991; White 1999; Onkware 2000).

Average soil temperature had little influence on the growth and survival parameters measured throughout this study. The range in average soil temperatures across planting plots was minimal. Further, daily minimum and maximum temperatures in planting plots were generally within 2-3°C of each other. The influence of fluctuating ambient and soil temperatures on a seasonal basis probably had more of an impact on survival and health of outplanted species than did variations in soil temperature across planting plots measured over one day at Honouliuli.

Competition and Density

Numerous studies have indicated that control of invasive and non-planted species may be necessary for successful establishment of restored plant populations (Kantrud 1996; D'Antonio and Meyerson 2002; Perry and Galatowitsch 2003; Seabloom and van der Valk 2003). Results from this study, however, demonstrated almost no significant impacts of invasive and non-planted species on survival of outplanted species. *Bolboschoenus maritimus* was the only study species that recorded higher final survival and reproduction in "no competition" plots than in competition plots.

Widespread mortality and poor health in *Cyperus laevigatus, C. polystachyos,* and *S. portulacastrum* may have obscured any effects of removal of non-planted species on final survival and reproduction in these species. Lack of significant results could also be due to small sample sizes (10 plots per treatment).

Bolboschoenus maritimus, Sesuvium portulacastrum, and *Sporobolus virginicus* were the only species whose growth appeared to be affected by colonization of non-planted species. Final percent cover of these species was greater in plots where non-planted species were removed than in plots where non-planted species were not removed. *Bacopa monnieri,* an herbaceous species that forms dense prostrate mats, was the primary non-planted "competitor" observed in planting plots during this study. Although the overall primary "competitor" across all outplanting plots was *Bacopa monnieri,* plots of groundcover species (i.e., *S. portulacastrum, S. virginicus,* and *Jacquemontia ovalifolia*) were subject to higher levels of competition from woody species such as *Pluchea indica* and *Batis maritima.* This was due to the proximity of groundcover plots to existing populations of *P. indica* and *B. maritima.* Higher levels of competition from woody species may have resulted in the decrease in cover of groundcover species that was not observed in sedge planting plots. Differences in life forms of colonizing species may influence the ability of that species to competitively displace another species (Grace 1987).

Although results of this study did not demonstrate impacts of non-planted species on survival of outplanted species, observations in the field indicated that there was a differential effect of non-planted species on survival of outplanted species. *Cyperus javanicus* (in plots with low mortality) formed tall and dense stands that were less readily overgrown by non-planted species than plots of species such as *Sesuvium portulacastrum* and *Cyperus laevigatus*, which did not produce abundant cover. Several plots of *C. laevigatus* and *S. portulacastrum* were rapidly overgrown and essentially smothered by *Bacopa monnieri,* as well as other non-planted species. Mortality in these plots appeared

to be hastened by colonization of non-planted species. Further, without any vegetation control, all outplanted species would most likely succumb to the continued growth of aggressive invasive species such as *Batis maritima, Pluchea indica, Typha latifolia*, and eventually *Leucaena leucocephala* and *Prosopis pallida.* Periodic control of non-desired vegetation would likely be necessary to maintain populations of outplanted native species.

Wetland plant restoration studies have examined the effects of such variables as water regime (Clevering and van Gulik 1997; Seabloom *et al.* 1998; Budelsky and Galatowitsch 2000) and soil amendments (Merrilees *et al.* 1995; van der Valk *et al.* 1999; Handa and Jefferies 2000) on establishment of outplanted species. However, few studies have looked at the effect of initial planting density on establishment and vegetative cover of outplanted species. Results of this study did not reveal any significant differences in survival, growth or reproduction of outplanted individuals based on initial planting density. Rapid growth of *Cyperus javanicus, Jacquemontia ovalifolia, Sporobolus virginicus,* and *C. polystachyos* (in planting plots with low mortality) quickly obscured any differences in percent cover due to initial planting density. This indicates that these species may be able to rapidly spread and provide abundant cover even when planted sparsely.

Extensive research has documented that plant species that first become established at a site have an advantage over later arriving species (Friedman 1971; Ross and Harper 1972; Fowler 1981; Grace 1987; Bertness 1991). This preemption can inhibit invasion of other species by making conditions unsuitable for establishment or by interfering with growth (Ross and Harper 1972; Grace 1987; Keddy 1998). This study

strove to determine not only the effect of preemption of outplanted native plant species on establishment of invasive species, but also whether initial planting density influenced the effects of preemption.

Results of this study only partially supported the preemption hypothesis. Biomass of non-planted species removed from "competition" planting plots was greater in control and low density plots than in high density plots of several species. This would indicate that higher densities of outplanted species precluded invasion to a greater extent than in low densities or control plots. However, final percent cover indicated that non-planted species were able to colonize planting plots to the same extent regardless of initial density of outplanted species. Lack of conclusive evidence of the influence of initial planting on exclusion of non-planted species could partially be explained by the extensive growth of *Bacopa monnieri* observed in almost all competition plots. Colonization by non-planted species other than *B. monnieri* was slow to occur in many planting plots. Although certain plots were covered by *Batis maritima, Pluchea indica,* and *Typha latifolia,* in many competition plots the only non-planted species observed at the end of the study was *Bacopa monnieri.* Additionally, the bulk of colonization of non-planted species occurred through expansion of existing populations growing into study plots. Few seedlings or vegetative shoots were observed in study plots. Previous studies on preemption and competitive exclusion examined the influence of existing plant cover on germination and seedling growth, and not vegetative expansion, of a colonizing species (Ross and Harper 1972; Friedman 1971; Grace 1987).

This study should help establish criteria for introduction and restoration of native wetland plant species. Selection of plant species for restoration should be based on

environmental tolerances, growth requirements, competitive abilities, and, if known, whether the species previously existed at the restoration site (Wilcox and Whillans 1999). Several of the study species, based on these requirements, would be recommended for restoration of other coastal wetlands in Hawai`i. However, the limited success of the other study species should not exclude these species from consideration. Instead further research, including field and greenhouse experiments, should be conducted to test environmental factors, which may hinder survival and growth of these species. Factors examined should include tolerances to salinity, water and nutrient levels and soil composition. Further research should also directly test the ability of native wetland plant species to compete with invasive wetland plant species.

Discussion and Management Recommendations for Individual Native Species

Bolboschoenus maritimus

Bolboschoenus maritimus is a common component in many coastal wetlands around the state and thus is a desired species for revegetation projects. *Bolboschoenus maritimus* transplants at Honouliuli readily matured and reproduced from transplanted stems as long as the underground rhizomes and tubers were kept intact. Successful restoration of *B. maritimus* using transplanted rhizomes has occurred in temperate wetlands (Clevering and van Gulik 1997). However, despite being culled from healthy populations, transplanted populations of *B. maritimus* never appeared as robust or reached the density of natural populations at Honouliuli.

As mentioned previously, hydrology may have played a key role in the extremely high mortality of *Bolboschoenus maritimus.* A noticeable trend in transplanted and

natural stands of *B. maritimus* was that plants closer to open water or growing in shallow water appeared taller, thicker stemmed, darker green, and generally more robust than plants farther from open water. Water level in Pond 2 declined during the study period from 50 cm in April 2002 to 27 cm in April 2003. *Bolboschoenus maritimus* planting plots that were originally 0.25 m from the water's edge in April 2002 were, on average, 1.57 m from the edge at the end of the study.

Although *Bolboschoenus maritimus* readily produces new vegetative shoots, it may take a while for sparse stands to fill in. Initially planting *B. maritimus* in dense stands directly in shallow water may increase survival and shoot production and allow dense stands to develop more rapidly. As the majority of newly emerged *B. maritimus* shoots recorded in May 2003 appeared in no-competition plots, control of invasive or non-native species may be required for the spring re-emergence of *B. maritimus* shoots from underground tubers. Control of invasive vegetation may also allow for germination of seeds produced the previous summer and fall.

Cyperus javanicus

Outplantings of *Cyperus javanicus* were among the most successful of all outplanting efforts. This species could be a valuable component of future restoration efforts in both brackish and freshwater wetlands in the state. *Cyperus javanicus* plants managed to survive and thrive in a variety of conditions at Honouliuli. Further, many outplanted individuals rapidly reproduced and spread vegetatively, resulting in dense cover in outplanting plots. Even lower-density plantings quickly provided abundant cover in outplanting plots. Colonization of non-planted species did not appear to affect

the final percent cover of *C. javanicus* to any measurable extent. Even when planted in lower densities and subjected to competition from invasive species, *C. javanicus* was able to survive, grow and reproduce. Despite its apparent ability to thrive, periodic vegetation control, especially of woody species such as *Pluchea indica* and *Prosopis pallida*, would be necessary to ensure the continued success of *C. javanicus* populations.

Cyperus laevigatus

After one year, *Cyperus laevigatus* outplantings were only moderately successful. Poor health and low survival in this species was surprising as *C. laevigatus* forms healthy dense stands in other saline and brackish wetlands around the state (Duvall, pers. comm., 2002; pers. obs.) In a few planting blocks, *C. laevigatus* plants were quite healthy and robust, but in many other sites around Pond 2, plants were small and chlorotic. High soil salinity may have contributed to mortality and poor health observed in a few plots. Block Four, the planting block with the highest mortality, had the highest overall average soil salinity. However, the blocks with the second through fourth highest average salinities had high survival rates. Some of the healthiest plants were found growing in plots with an average salinity of 12.1 mmhos (9.7 ppt). Further, robust stands of *C. laevigatus* are found growing in other brackish and saline wetlands around the state (Stemmermann 1981; Wagner 1999; pers. obs.). Clearly there were other environmental factors besides salinity that inhibited *C. laevigatus'* ability to survive and thrive at Honouliuli. Soil moisture, nutrient levels, and/or soil type may have played a role in limiting the success of *C. laevigatus* outplantings. In a study investigating the potential use of *C. laevigatus* for wastewater treatment in Hawai`i, Van Dyke (2001) observed that *C. laevigatus* plants

subjected to higher nutrient levels were substantially taller and darker green than those subjected to lower nutrient levels.

During this study, very few *Cyperus laevigatus* planting plots achieved and maintained abundant cover. Planting *C. laevigatus* in high densities may increase the likelihood of establishing abundant cover of *C. laevigatus*. Until dense stands are attained, *C. laevigatus* outplanting areas may require removal of invasive plants for the plants to thrive. Planting *C. laevigatus* at the water's edge and/or in shallow water could also potentially increase survival and improve health of this species (Koob 1999).

Cyperus polystachyos

Initially, the outplanting of *Cyperus polystachyos* appeared as though it would be quite successful. For the first six months of the study, plants in most plots were very healthy, reproduced prolifically and grew in both stature and diameter. However, less than 60% of outplanted individuals were still alive by the end of the study.

Survival rate of *Cyperus polystachyos* outplanting plots was associated with location at Honouliuli. Dense, healthy plots were located on the southwest corner of Pond 2. This area is closest to the freshwater input to the pond. The three planting blocks with the highest final survival rates had the lowest average soil salinities (4-5.5 mmhos / 2.6 – 3.5 ppt). Plots with average salinities exceeding 8 mmhos (6.4 ppt) exhibited close to 100 percent mortality.

Planting plots with high survival and healthy individuals provided very dense cover and exhibited high reproductive output. In areas favorable to *Cyperus polystachyos,* even lower-density plantings rapidly filled in and provided abundant cover.

This species, due to its height and bushy nature, can tolerate some competition from other species and, given the right environmental conditions, *C. polystachyos* may require less maintenance, in the form of non-native vegetation control, than other species in order to survive and thrive. *Cyperus polystachyos* appears to be a more valuable component in freshwater than in brackish restoration efforts.

Jacquemontia ovalifolia

Jacquemontia ovalifolia not only became established in many planting plots, but also spread into many adjacent control plots and planting blocks. Soil salinity and soil temperature were not correlated with final survival or physical appearance of *J. ovalifolia. Jacquemontia ovalifolia* plants tolerated and even thrived in a range of environmental conditions at Honouliuli. *Jacquemontia ovalifolia* provided relatively dense cover whether initially planted in low or high densities. Due to its ease of propagation and ability to survive with fairly minimal maintenance, this species may be useful in both brackish and freshwater wetland restoration projects elsewhere in the state.

Dense growth and encroachment of *Bacopa monnieri* did not seem to affect survival and growth of *Jacquemontia ovalifolia* compared to its effect on other species. Due to its rapid vine-like growth, *J. ovalifolia* plants managed to grow on top of the thick mats of *B. monnieri*, as opposed to species such as *Sesuvium portulacastrum,* which ended up being smothered by *B. monnieri. Jacquemontia ovalifolia* may be able to effectively compete with invasive low-growing groundcover and grass species. However, without periodic vegetation control, especially of woody species such as

Pluchea indica, Prosopis pallida, and *Leucaena leucocephala, J. ovalifolia* would probably eventually be overgrown and would possibly die out.

Sesuvium portulacastrum

Although *Sesuvium portulacastrum* is often found growing in areas exposed to high salinity and high temperatures, outplantings of this species did not fare well at Honouliuli. Percent survival was 65 percent after one year and physical appearance of many of the surviving plants at this time was quite poor. Survival of *S. portulacastrum* was highest in planting plots with the highest overall average salinities. It is unclear why such high mortality and poor health was observed in outplanting plots of *S. portulacastrum.*

Poor health and low survival in *Sesuvium portulacastrum* resulted in sparse cover in many outplanting plots. During this study, lower density plantings never achieved dense cover and were rapidly overcome by non-planted species, such as *Bacopa monnieri* and *Batis maritima.* Initially planting *S. portulacastrum* in dense stands may increase the likelihood of establishing abundant cover of this species. Because *S. portulacastrum* is such a low-growing groundcover, its ability to compete with woody or bushy invasive species such as *B. maritima* and *Pluchea indica* is limited. Without frequent vegetation control, this species would inevitably be displaced by other rapidly growing shrubs and groundcovers.

Sporobolus virginicus

Outplanted *Sporobolus virginicus* populations had the second highest survival rate of all outplanted species. Plants survived and flourished in almost all planting plots at Honouliuli. Vegetative spread exhibited by *S. virginicus* was quite extensive. In fact *S. virginicus* plants had spread to 15 of 20 control plots by the end of the study period. As with other species, no seedlings were observed during the study period and the viability of seeds produced by flowering stems was questionable. Many seeds produced by *S. virginicus* were white and appeared to lack an embryo. Spread of *S. virginicus* at Honouliuli will most likely continue through vegetative means rather than through seed dispersal.

Dense cover was quickly provided by *Sporobolus virginicus* plants in both high and low density planting plots. During this study lower-density planting plots provided cover as abundant as high density planting plots within four months. Higher density outplantings of *S. virginicus* may not be necessary to quickly restore or revegetate an area with this species. However, as with all the other study species, it is unlikely that *S. virginicus* could effectively compete with woody invasive species, and thus, to maintain abundant cover of *S. virginicus*, periodic vegetation control of non-desired species should be undertaken.

CHAPTER 3: GERMINATION OF NATIVE WETLAND SEDGE SPECIES UNDER DIFFERENT MOISTURE, TEMPERATURE AND SALINITY LEVELS

Abstract

Germination response of four native wetland sedges under different water, salinity and temperature regimes was investigated to assess the potential for use of seeds in restoring populations of native wetland plants. Seeds of *Cyperus polystachyos* and *C. javanicus* exhibited higher average germination percentages under moist and saturated conditions than under one cm of standing water. The effect of water-level on germination, however, was not strong and was associated with temperature regime. In general, germination percentages for seeds of *C. polystachyos* and *C. javanicus* were greatest under alternating temperatures of 20/15 and 24.3/18.2 °C than under alternating temperatures of 29/23 °C. Final germination percentage decreased with an increase in salinity. Salinity levels greater than 0.85% resulted in 0 percent germination. Under alternating temperatures of 24.3/18.2 and 20/15 °C, seeds of *C. javanicus* exhibited high germination percentages in salinity levels of 0.85%. *Cyperus polystachyos* exhibited high germination in 0.85% salinity in the 20/15 °C temperature regime. Germination in *C. polystachyos* was restricted in 0.85% salinity in higher temperatures. Final germination did not exceed 25 percent in any treatment for seeds of *Bolboschoenus maritimus* and *Cyperus laevigatus.* Germination of *B. maritimus* seeds only occurred after bleach scarification. Results of this study indicate that restoring or augmenting populations of *C. javanicus* and *C. polystachyos* with seeds may have a high likelihood of success. Further study of germination requirements and limitations of *C. laevigatus* and

B. maritimus should be conducted prior to extensive use of seeds of these species in restoration efforts.

Introduction

Determining the influence of abiotic conditions, such as water depth, temperature, and salinity, on germination of native wetland species is vital not only for restoration efforts but also for those attempting to manage native wetland plant communities (Budelsky and Galatowitsch 1999; Masuda *et al.* 1999). Sedges and other wetland plant species often spread primarily by vegetative or clonal growth (Hartman 1988; van der Valk *et al.* 1999). However, initial establishment of new populations usually occurs through seed dispersal and germination. Many wetland restoration projects rely on natural colonization for establishment of native species (Galatowitsch and van der Valk 1996; Yetka and Galatowitsch 1999). Unfortunately, wetland restoration sites are often either degraded, distant from a seed or propagule source of target species, or lack a native seed bank (Bradshaw 1983; Galatowitsch and van der Valk 1996; Middleton 1999; Streever and Zedler 2000).

Transplanting seedlings is one alternative for returning native vegetation to a restoration site. Unfortunately, growing and transplanting seedlings is expensive and time consuming. Direct sowing of native seed may provide a simpler and cheaper option (van der Valk *et al.* 1999). However, information on requirements for germination of wetland plants is limited (Budelsky and Galatowitsch 1999; Baskin and Baskin 2003). This is especially true for native Hawaiian wetland sedge species.

After dispersal, seeds often require specific environmental conditions in order to germinate (Middleton 1999; Baskin and Baskin 2001; Baskin *et al.* 2003). Seeds of

many wetland species germinate in response to changes in hydrology, such as water drawdown or flooding (Galinato and van der Valk 1986; LaDeau and Ellison 1999). In fact, many emergent wetland species require bare mudflats or shallow water for germination and seedling establishment (Galinato and van der Valk 1986; Shipley and Parent 1991; ter Heerdt and Drost 1993). Seeds of many emergent wetland species will not germinate underwater (Seabloom *et al.* 1998; Middleton 1999; Baldwin *et al.* 2001).

Temperature and salinity are other key factors that influence germination of wetland species (Martinez *et al.* 1992; Baldwin and Mendelssohn 1998; Middleton 1999). Seeds of many wetland species are unable to germinate under conditions of high salinity (Galinato and van der Valk 1986; Badger and Ungar 1989; Baskin and Baskin 2001). Even germination of halophytes often occurs at seasonally low salinities (Badger and Ungar 1989). Fluctuating temperatures have been shown to produce the highest germination in many wetland sedge species (Thompson and Grime 1983; Budelsky and Galatowitsch 1999; Baskin and Baskin 2001). Further, seeds of many wetland plant species, including several *Cyperus* species, show higher germination under high alternating temperatures (Shipley and Parent 1991; Baskin and Baskin 2001).

The objective of this study was to assess the germination response of four native wetland sedge species under different conditions of water-level, salinity and temperature. This study tested the hypotheses that 1) germination percentages would be highest under conditions of moist soil and would decrease with increasing standing water depth, 2) germination percentages would be highest in freshwater and would decrease with increasing salinity, and 3) germination percentages would be higher under alternating temperatures of 29/23 °C and 24.3/18.2 °C than at 20/15 °C. The results of this study are

intended to provide recommendations on the use of seed for establishment and restoration of native sedge species.

Methods

Three germination trials were conducted between August 2002 and April 2003. Each trial lasted for eight weeks. Seeds of four native wetland sedges, *Bolboschoenus maritimus, Cyperus javanicus, C. laevigatus,* and *C. polystachyos*, were used in all germination trials. Seeds were collected on three different dates (August 2002, November 2002 and January 2003) from outplanted populations in the Honouliuli Unit of Pearl Harbor National Wildlife Refuge. Seeds were cleaned and examined under a dissecting scope to determine that the seed coat was intact and that the seed contained an embryo. Seeds with damaged seed coats or that appeared to lack an embryo were removed. Within two weeks of collection, seeds were placed under experimental conditions in Percival I-30VL temperature-controlled growth chambers (Percival Scientific, Inc., Perry, Iowa) at the University of Hawai'i at Manoa. Seeds were germinated in Petri dishes on Whatman No. 1 filter paper. Distilled water was used for all treatments. Petri dishes with tight fitting lids were used to reduce water loss. Water levels were checked and germinated seeds were removed every other day.

Percent germination data were arcsine-square-root transformed prior to analysis to meet normality assumptions. Data were analyzed for statistical significance using one-way and two-way ANOVA. Multiple comparisons of means were made using Tukey's test of 'honestly significant differences' (Clewer and Scarisbrick 2001). All tests of significance were conducted at the $p<0.05$ level. Percent germination in the dry water-

level treatment was 0 percent for all species in all temperature regimes; this treatment was excluded from data analysis. All other treatments that resulted in 0 percent germination were also excluded from further data analysis.

Water Level and Temperature

The initial test of germination response was conducted from August through early October 2002. To investigate germination response under a range of water levels, five replicates of 20 seeds of each species were placed under four water levels: one cm of standing water, saturated (water just covering the top of the seeds), moist (moistened filter paper), and dry. Seeds were placed in temperature controlled growth chambers with an alternating temperature regime of 29/23 °C and 12-hour photoperiods with light corresponding with the higher temperature. The temperatures were chosen based upon the average monthly temperature for southwest O`ahu for July through October.

In mid-October through mid-December 2002, a second experiment testing germination response under different water levels was conducted. The moisture treatments are as described above, but based on lack of results from the first trial, seeds were placed in a growth chamber with an alternating 24.3/18.2 °C temperature regime, with 12-hour alternating light/dark photoperiods. Five replicates of twenty seeds were used for each treatment.

A final experiment to test the effect of moisture and temperature was conducted beginning in February of 2003. The results of the first two trials demonstrated that temperature played an important role in the germination of each species, except for *Bolboschoenus maritimus* which showed no germination in any treatment or temperature.

For the third trial, a 4 x 3 factorial design was utilized with four moisture levels (one cm standing water, saturated, moist, dry) and three temperature regimes (29/23, 24.3/18.2, and 20/15 °C). As in previous trials, five replicates of 20 seeds for each species and treatment were placed in growth chambers with alternating 12-hour light/dark photoperiods. Due to lack of germination in previous trials, seeds of *B. maritimus* were scarified by soaking them in a bleach solution (3% sodium hypochlorite) for three days (following Clevering 1995) prior to being placed under treatment conditions.

Salinity and Temperature

Germination trials comparing four different levels of salinity (0.0, 0.85, 1.7, and 3.4%) were initiated in October of 2002. Saline solutions were added until just the top of the seeds were covered (i.e., saturated conditions in the water level experiment). Five replicates of 20 seeds for each combination of species and salinity were placed in growth chambers with fluctuating temperature (24.3/18.2 °C) and photoperiods (12-hour light/dark photoperiods). Petri dishes with tight-fitting lids were used to reduce evaporation, which might alter salinity concentrations. Saline solutions were prepared using filtered sea water (salinity measured as 34 ppt) diluted to the correct concentrations. Seawater was collected from the Waikiki Aquarium in Honolulu, Hawaii, no more than one day before germination trials were initiated. Germinated seeds were removed and saline solutions and water levels were refreshed every other day. Fresh saline solutions were prepared once a week.

A second trial was initiated in February of 2003. A 4 x 3 factorial design was utilized with four salinity levels (0.0, 0.85, 1.7, and 3.4%) and three temperature regimes (29/23, 24.3/18.2, and 20/15 °C). As in previous trials, five replicates of 20 seeds for

each combination of species, salinity and temperature treatment were placed in growth chambers with alternating 12-hour light/dark photoperiods. As in the water-level treatments, seeds of *Bolboschoenus maritimus* were scarified by soaking them in a bleach solution (3% sodium hypochlorite) for three days prior to being placed under treatment conditions.

Results

Trial 1: August 2002 – October 2002

After eight weeks, very little germination was observed in any species except for *Cyperus polystachyos* (Figure 17). The final germination percentage for *C. polystachyos* was highest under moist conditions (94 ± 7 SD), followed by saturated conditions (64 ± 8), and one cm of water (40 ± 8 SD). No seeds germinated under dry conditions (Figure 17). A one-way ANOVA of final germination percentages showed a significant effect of water level on overall germination percentage ($F_{3,16}$=129.611; $p<0.001$). Final germination was significantly greater under moist conditions than under saturated conditions ($p<0.001$) or under one cm of standing water ($p<0.001$). Final germination was also significantly greater under saturated conditions than under one cm of standing water ($p=0.017$).

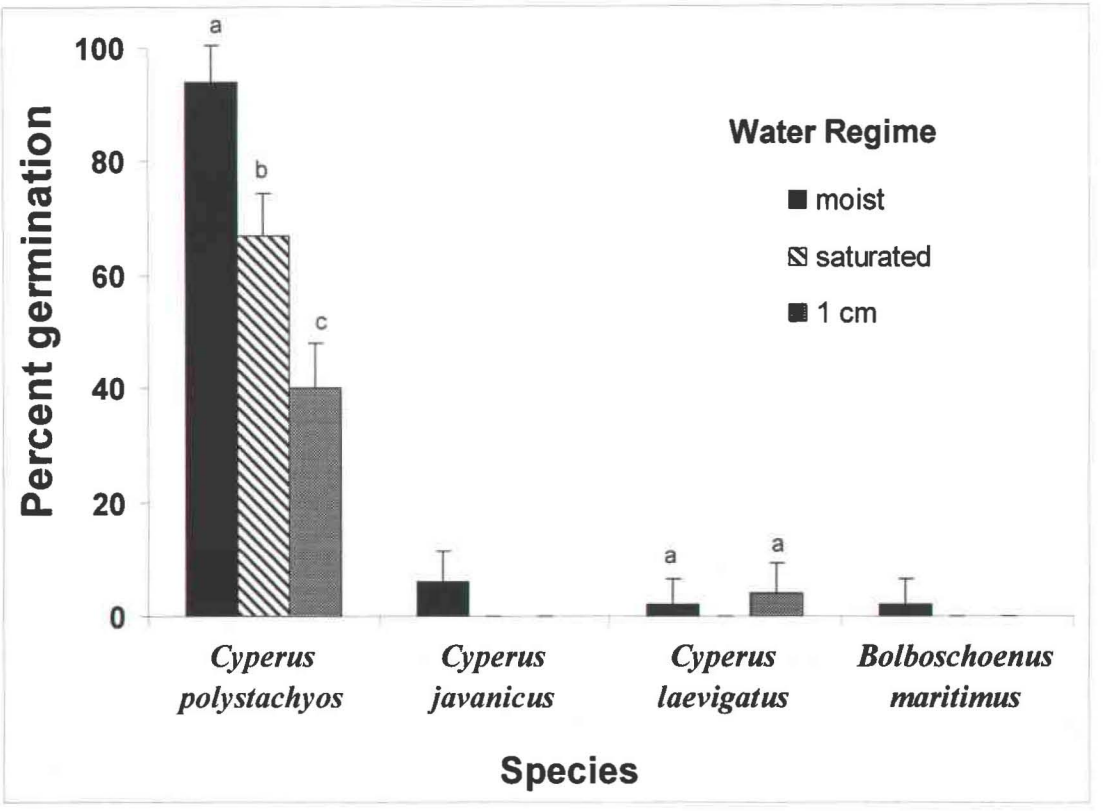


Figure 17. Final percent germination (mean ± 1 SD) of native wetland sedge seeds after eight weeks in four water-level treatments under alternating light (12-hour light/dark) and temperature (29/23 °C) . Different letters denote significant differences ($p<0.05$; Tukey test) between water regimes within each species.

Seeds of all other species showed very low germination, with percentages between 0 and 6 percent under all moisture levels. Final germination percentage in these species was too low to analyze for differences due to water-level treatment.

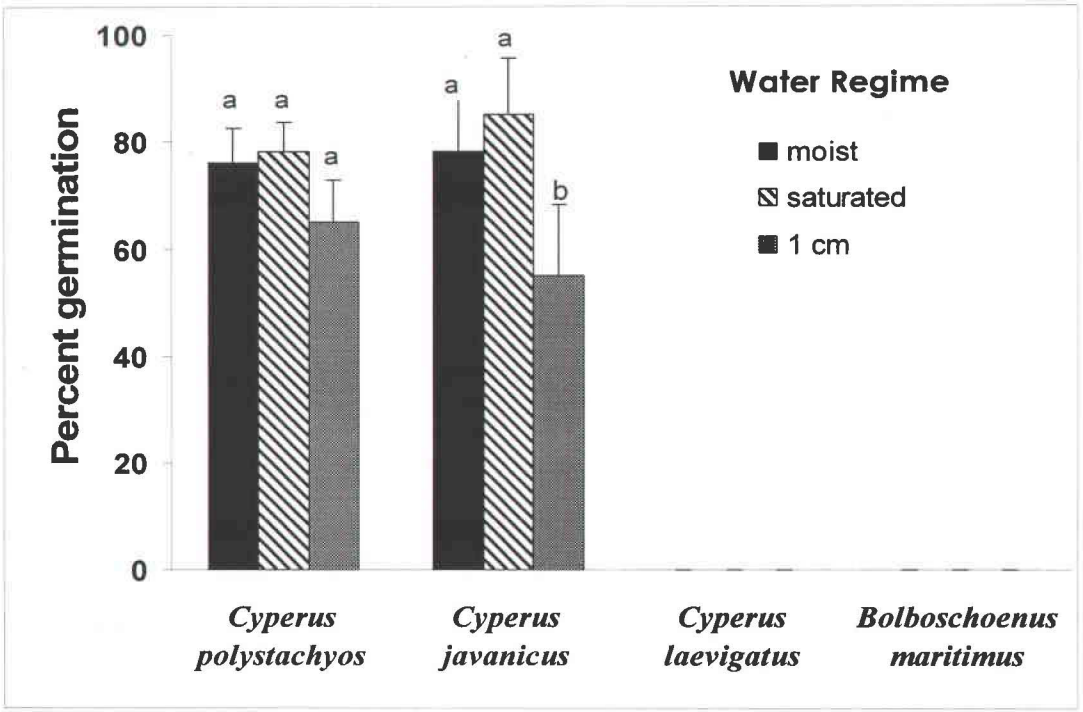
Trial 2: October 2002 - December 2002

Water Level

Final germination percentages for water-level treatments ranged from 0 percent, for seeds of *Bolboschoenus maritimus* and *Cyperus laevigatus* under all treatments, to 85 percent for seeds of *Cyperus javanicus* in the saturated treatment (Figure 18). Final

germination percentages for *C. javanicus* were highest under saturated conditions (85 ± 11%) (mean ± SD), followed by moist conditions (78 ± 10%), and one-cm standing water (55 ± 13%). A one-way ANOVA of final germination percentages showed a significant effect of water level on overall germination percentage of *C. javanicus* ($F_{2,12}$= 6.61; p = 0.01). Final percent germination was significantly greater under saturated conditions than under one cm of standing water (p=0.01). Germination under moist conditions was not significantly different than germination under saturated or one-cm standing water treatments.

a)



b)

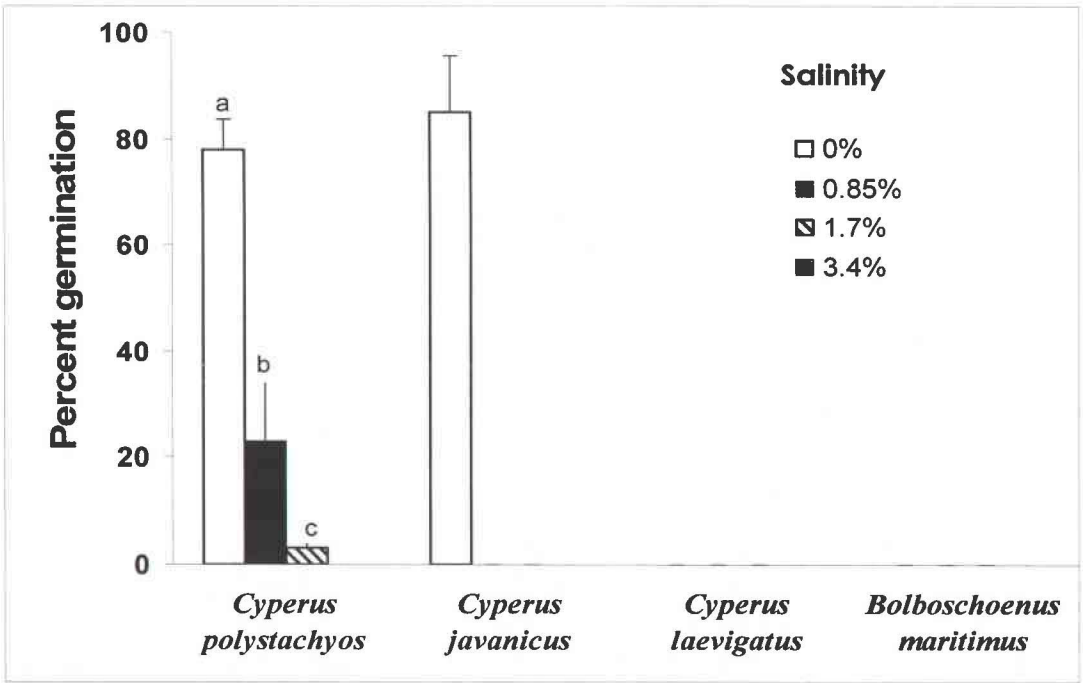


Figure 18. Final percent germination (mean ± 1 SD) of native wetland sedge seeds after eight weeks in (a) four water-level treatments and (b) four salinity treatments under a alternating light (12-hour dark/light) and temperature (24.3/18.2°C) regimes. Different letters denote significant differences ($p<0.05$; Tukey test) among treatment levels within each species.

Final germination percentages for *Cyperus polystachyos* ranged from 65 ± 8 percent (mean ± SD) under one cm of standing water to 78 percent ± 6 (SD) under saturated conditions. Final germination in moist soil conditions (76 percent ± 7 (SD) was only slightly lower than under saturated conditions. There were no significant differences in final germination percentage of *C. polystachyos* in the water-level treatments.

Seeds of all species, except *Cyperus polystachyos,* failed to germinate under salinity conditions of 0.85, 1.7, and 3.4 percent salinity. Final germination for *C. polystachyos* was 23 percent ± 14 (SD) under 0.85 percent salinity; 3 ± 4 percent under 1.7 percent salinity, 0 percent under 3.4 percent salinity, and 78 percent ± 6 (SD) in freshwater (0 percent salinity). There was a significant effect of salinity on overall germination percentage of *C. polystachyos* ($F_{2,12} = 48.754$; $p<0.001$). Final germination percentage was significantly greater in the 0 percent salinity treatment than in either the 0.85 or 1.7 percent salinity treatments ($p<0.001$ for both combinations). Germination was also significantly greater in the 0.85 percent treatment than in the 1.7 percent salinity treatment ($p=0.005$).

Trial 3: February 2003 - April 2003

Germination results under the various water-level, salinity, and temperature treatments are summarized in Table 5. Similar to the results of trials one and two, germination percentages for seeds of *Bolboschoenus maritimus* and *Cyperus laevigatus* were very low under all treatments and temperature regimes. However, unlike the first two trials, in which *B. maritimus* germination was 0 percent in almost all treatments, seeds of *B. maritimus* did germinate under several treatments in this trial. This is

presumably due to the bleach scarification treatment. Overall, mean germination percentages of all species, except *Bolboschoenus maritimus,* were higher under alternating temperatures of 20/15 °C (44 percent) and 24.3/18.2 °C (45 percent) than under alternating temperatures of 29/23 °C (26 percent). Final mean germination percentages, when considering all temperatures, were similar regardless of water-level. Higher salinity levels resulted in decreased germination in all species and germination was not recorded in any species at salinity levels above 0.85 percent.

Table 5. Final percent germination (mean ± 1 SD) for seeds of native wetland sedges after eight weeks under different water-levels, salinity levels, and temperature regimes.

Species/ Temperature	**Water Regime**				**Salinity**			
Cyperus polystachyos	**dry**	**moist**	**saturated**	**1 cm**	**0%**	**0.85%**	**1.7%**	**3.4%**
20/15 °C	0	88 ± 9	95 ± 6	82 ± 7	95 ± 6	60 ± 9	0	0
24.3/18.2 °C	0	98 ± 3	82 ± 12	84 ± 7	82 ± 12	30 ± 6	0	0
29/23 °C	0	74 ± 19	78 ± 4	68 ± 12	78 ± 4	26 ± 13	0	0
Cyperus javanicus								
20/15 °C	0	80 ± 6	90 ± 12	87 ± 7	90 ± 12	70 ± 5	0	0
24.3/18.2 °C	0	90 ± 6	80 ± 7	88 ± 6	80 ± 7	74 ± 7	0	0
29/23 °C	0	46 ± 7	24 ± 7	28 ± 6	24 ± 7	1 ± 2	0	0
Cyperus laevigatus								
20/15 °C	0	2 ± 3	3 ± 4	7 ± 3	3 ±4	0	0	0
24.3/18.2 °C	0	0	1 ± 2	0	1 ± 2	0	0	0
29/23 °C	0	0	3 ± 4	8 ± 5	3 ± 4	0	0	0
Bolboschoenus maritimus								
20/15 °C	0	0	0	0	0	0	0	0
24.3/18.2 °C	0	17 ± 13	16 ± 12	24 ± 10	16 ± 12	0	0	0
29/23 °C	0	12 ± 6	8 ± 6	11 ± 7	8 ± 6	0	0	0

Water Level

There was no unilateral effect of water level on final germination percentages; effect depended on the temperature regime. Two-way ANOVA indicated that there was a statistically significant interaction between water level and temperature regime on final germination in all species, except *B. maritimus* (Table 6).

Table 6. Summary of two-way analysis of variance (ANOVA) results of the effects of temperature and water level on final germination of native wetland sedge species.

	B. maritimus		***C. javanicus***		***C. laevigatus***		***C. polystachyos***	
	F	**p**	**F**	**p**	**F**	**p**	**F**	**p**
Temperature	6.09	0.021	123.27	<0.001	9.16	<0.001	11.86	<0.001
Water level	0.75	0.482	1.098	0.344	9.60	<0.001	5.483	0.008
Temperature x water level	0.64	0.53	5.303	0.002	3.35	0.02	3.634	0.014
Temperature	***	***	151.881	<0.001	***	***	13.31	<0.001
Salinity	***	***	40.666	<0.001	***	***	105.02	<0.001
Temperature x salinity	***	***	6.335	0.006	***	***	5.49	0.011

*** Analysis not performed due to 0 percent germination in all salinity treatments for *B. maritimus* and *C. laevigatus*. For *C. polystachyos* and *C. javanicus*, only the 0% and 0.85 percent salinity treatments were used when analyzing data for salinity and temperature x salinity effects.

Final percent germination for *Cyperus javanicus* was highest under moist conditions at 29/23 and 24.3/18.2 °C, but was lowest under moist conditions at 20/15 °C (Table 5). For *Cyperus polystachyos*, mean germination percent was highest under saturated conditions at 20/15 and 29/23 °C, but was lowest under saturated conditions at 24.3/18.2 °C.

Final germination percentage for *Cyperus javanicus* in the 29/23 °C temperature regime was significantly greater ($F=16.05$; $p<0.001$) under moist conditions than in either

saturated conditions ($p<0.001$) or under one cm of standing water ($p=0.008$). Percent germination was not significantly different for *C. javanicus* seeds under different water-level conditions in the 24.3/18.2 and 20/15 °C temperature regimes or when comparing water-level treatments across all temperature regimes.

Overall, mean germination of *Cyperus polystachyos* was similar in moist (87 percent ± 12 SD) and saturated (85 percent ± 9 SD) conditions and one cm of standing water (78 percent ± 9 SD). Final germination percentages of *C. polystachyos* in the various water-level treatments were not significantly different in the 29/23 and 20/15 °C temperature regimes. Seeds of *C. polystachyos* in the 24.3/18.2 °C temperature regime germinated at a significantly higher level ($F=10.925$; $p=0.002$) in moist conditions than they did in saturated conditions ($p=0.004$) or under one cm of standing water ($p=0.005$).

Maximum germination (24 ± 10%) (mean ± SD) for *Bolboschoenus maritimus* occurred under conditions of one cm of standing water at 24.3/18.2 °C (Table 5). Seeds of *B. maritimus* germinated at higher percentages under one cm of water and moist conditions than under saturated conditions in the 24.3/18.2 and 29/23 °C temperature regimes. However, mean germination percentages, across all temperatures, were not significantly different regardless of water level. Further, there were no significant differences in *B. maritimus* germination due to water-level treatments in either the 24.3/18.2 °C or the 29/23 °C temperature regimes. Seeds of *B. maritimus* did not germinate in any treatment under alternating temperatures of 20/15 °C.

Overall, germination of *Cyperus laevigatus* seeds was greater in one cm of standing water (5 percent ± 5 SD), than in saturated (2 percent ± 4 SD) or moist (0.7 percent ± 2 SD) conditions. These differences were significantly different ($F=10.248$; df

= 12; p=0.003). Germination was significantly greater under one cm of standing water than under moist (p=0.002) or saturated (p=0.04) conditions.

Salinity

Salinity had a strong effect on final germination in all species. Regardless of temperature regime, seeds of *Bolboschoenus maritimus* and *Cyperus laevigatus* did not germinate in any of the salinity treatments. Seeds of *Cyperus polystachyos* and *C. javanicus* did not germinate at salinity levels greater than 0.85 percent.

Across all temperatures tested, *Cyperus javanicus* germinated at slightly higher levels under 0.85 percent salinity (48 percent ± 35 SD) than did *C. polystachyos* (39 percent ± 18 SD). Germination of *C. javanicus* in the 0.85 percent salinity treatment was similar under alternating temperatures of 20/15 and 24.3/18.2 °C, but was much lower under the 29/23 °C temperature regime (Table 5). In fact, final average germination percent was significantly greater in 0.85 percent salinity in alternating temperatures of 20/15 °C and 24.3/18.2 °C than in the 29/23 °C temperature regime ($p<0.001$ for both combinations). Overall, mean germination of *C. javanicus* was not significantly greater in 0 percent salinity than in 0.85 percent salinity. However, when comparing germination in 0 percent and 0.85 percent salinity treatments in each temperature regime, final percent germination was significantly greater in 0 percent salinity than in 0.85 percent salinity in both the 29/23 °C ($t=7.462$; $df=8$; $p<0.0001$) and 20/15 °C ($t=3.508$; $df = 8$; $p = 0.008$), but not in the 24.3/18.2 °C temperature regimes.

Cyperus polystachyos, overall, showed greater mean germination in 0% salinity than in 0.85 percent salinity ($t = 8.511$; $df = 28$; $p<0.001$). Germination was also significantly greater in 0 percent salinity than in the 0.85 percent salinity treatment for

seeds of *C. polystachyos* in each temperature regime (Table 5; Table 7). *Cyperus polystachyos* seeds in the 0.85 percent salinity treatment also exhibited significantly higher germination ($F_{2,12}$; p=0.001) when in alternating temperatures of 20/15 °C than in alternating temperatures of 24.3/18.2 °C (p=0.004) or 29/23 °C (p=0.001).

Table 7. Summary of *t*-test results comparing final germination percentage of *Cyperus polystachyos* in 0 percent and 0.85 percent salinity treatments under alternating temperature regimes.

Temperature (°C)	t-value	df	p-value
29/23	7.051	8	p<0.001
24.3/18.2	7.446	8	p<0.001
20/15	5.838	8	p<0.001

Temperature

Average percent germination for both *Cyperus javanicus* ($F_{2,42}$=158.44; p<0.001) and *C. polystachyos* ($F_{2,42}$=8.10; p=0.001) was significantly higher under alternating temperatures of 20/15 and 24.2/18.3°C than under alternating temperatures of 29/23 °C. Overall, seeds of *Bolboschoenus maritimus* germinated to a significantly greater extent in the 24.3/18.2 °C (19 percent ± 11 SD) temperature regime than in the 29/23 °C (10 percent ± 6 SD) temperature regime (t = 2.628; df=28; p=0.014). Seeds of *B. maritimus* did not germinate in any treatment under alternating temperatures of 20/15 °C.

Very low germination rates were observed in all treatments for *Cyperus laevigatus*. However, final germination was significantly greater under alternating temperatures of 29/23 °C (4 percent ± 5 SD) and 20/15 °C (4 percent ± 4 SD) than under

the 24.3/18.2 °C (0.3 percent ± 1.0 SD) temperature regime ($p=0.04$; and $p=0.009$ respectively).

Discussion

Although seeds of many emergent wetland species require exposed mudflats in order to germinate, saturated conditions and shallow flooding did not appear to substantially hinder germination of sedge species examined in this study (Galinato and van der Valk 1986; Shipley and Parent 1991; Baskin and Baskin 2001). Further, although seeds of many wetland plant species, including several *Cyperus* species, show higher germination under high alternating temperatures (Shipley and Parent 1991; Baskin and Baskin 2001), germination in *Cyperus* species in this study was greater under lower alternating temperatures than under higher alternating temperatures.

Of the four species studied, only *Cyperus polystachyos* and *C. javanicus* exhibited substantial germination under experimental treatment conditions. Seeds of *Cyperus polystachyos* collected at three different time periods germinated readily in a wide variety of temperatures and water-levels. The results indicate that seeds of *C. polystachyos* are non-dormant at maturity and will germinate under a wide variety of environmental conditions that may be found in the field. Sowing seed in addition to or even instead of transplanting seedlings may be a viable restoration option for this species.

Seeds of *Cyperus javanicus* germinated readily under lower fluctuating temperatures; however, germination was restricted under higher alternating temperatures. This indicates that seeds of *C. javanicus* may be conditionally dormant at time of maturity. Seeds exhibiting conditional dormancy will germinate only under a narrow

range of environmental conditions such as temperature (Baskin and Baskin 2001). Seeds of *C. javanicus,* in the lower temperature regimes, exhibited high germination under all water-levels. Sowing seeds in areas that experience shallow flooding or waterlogged soils may not hinder germination of *C. javanicus.* However, spreading seeds during early spring, when temperatures are lower, may increase germination in this species.

Not surprisingly, seeds of both *Cyperus polystachyos* and *C. javanicus* germinated only under conditions of low salinity. Increased salinity levels are known to reduce germination in wetland plants, including many halophyte species (Badger and Ungar 1989; Masuda *et al.* 1999; Baskin and Baskin 2001). Further, very few halophytes will germinate at salinity levels exceeding 3.0% (Masuda *et al.* 1999). Germination of *C. javanicus* and *C. polystachyos* under low salinity levels increased with a decrease in alternating temperatures. Other studies on emergent wetland plants have demonstrated that germination under saline conditions decreases with an increase in temperature (Badger and Ungar 1989; Masuda *et al.* 1999; Baskin and Baskin 2001).

Seeds of *Cyperus polystachyos* and *C. javanicus,* however, may remain viable even after exposure to saline conditions. Although seeds of *C. polystachyos* and *C. javanicus* did not germinate at higher salinities (1.7 and 3.4%), when seeds from these salinity treatments were placed in fresh water germination rapidly occurred. This indicates that seeds of these two species will remain viable and may come out of dormancy when levels of salinity decrease.

Seeds of both *Bolboschoenus maritimus* and *Cyperus laevigatus* exhibited low germination under all temperatures, water-levels and salinities. Studies have demonstrated that germination of *Bolboschoenus maritimus* can be increased with

exposure to temperature stratification and chemical scarification (Clevering 1995). Bleach-scarification increased germination percentage of *B. maritimus* in this study; however, final germination percent of bleach-scarified seeds never exceeded 25% in any of the experimental conditions. Unlike the other species in this study, bleach-scarified seeds of *B. maritimus* did not germinate under low temperatures. Germination requirements and techniques for breaking dormancy in this species should be further tested prior to using seeds to restore or augment populations of *B. maritimus.*

In all three trials, final germination of *Cyperus laevigatus* never exceeded 10 percent regardless of temperature, water-level, or salinity conditions. Other researchers in Hawai'i have been able to achieve high rates of *C. laevigatus* germination in nursery settings, but the conditions to which germinating seeds were exposed have not been documented (Van Dyke, pers. comm., 2002.). Seeds of many wetland plant species in temperate regions require temperature stratification, specifically exposure to cold temperatures, in order for dormancy to be broken (Shipley and Parent 1991; Baskin and Baskin 2001). Preliminary results from this study indicate that seeds of *C. laevigatus* may require exposure to cold temperatures in order to germinate.

Initially, due to absence of germination, seeds of *C. laevigatus* were presumed unviable. However, by placing seeds of *C. laevigatus* in colder temperatures (20/15 °C) for several weeks and then placing them under a higher temperature regime (29/23 °C), relatively high rates of germination were achieved. This indicates that seeds of *C. laevigatus* may be dormant at maturity and require temperature stratification in order for dormancy to be broken and germination to occur. *Cyperus laevigatus* is readily

propagated through division of culms and this method would be recommended for restoration efforts until germination requirements of this species are understood.

The results of this study provide preliminary information on germination requirements of several native sedge species. However, in these experiments, only fresh seeds were used. In many restoration efforts, fresh seeds may not be readily available. Seed viability and germination rates of wetland plants have been shown to be significantly affected by a combination of temperature, storage conditions and storage duration (Badger and Ungar 1989; Budelsky and Galatowitsch 1999). Studying germination response of seeds of native wetland species after different lengths of storage or under different storage regimes would provide valuable information for those interested in using seeds to augment or restore native sedge populations.

Germination phenology studies in the field, for example recording when seeds are set and when seeds germinate in natural populations, would also greatly increase our knowledge of germination requirements of native wetland sedges. Date of seed collection may have an influence on the dormancy state of seeds, which would influence germination rates. These differences may have to do with seasonal shifts dormancy and / or changes in temperature requirements for germination (Baskin and Baskin 2001). Further study on dormancy patterns and germination requirements of these and other native wetland sedges would be recommended prior to widespread use of native seeds in wetland plant restoration efforts.

LITERATURE CITED

Athens, J.S., J.V. Ward, and S. Wickler. 1992. Late Holocene lowland vegetation, O`ahu, Hawai`i. *New Zealand Journal of Archaeology* 14:9-34.

Athens, J. S. 1997. Hawaiian native lowland vegetation in prehistory. In P.V. Kirch and T.L. Hunt, eds. *Historical Ecology in the Pacific Islands,* pp. 248-270. Yale University Press. New Haven.

Badger, K.S., and I.A. Ungar. 1989. The effects of salinity and temperature on the germination of the inland halophyte *Hordeum jubatum. Canadian Journal of Botany* 67:1420-1425.

Baldwin, A.H., and I.A. Mendelssohn. 1998. Effects of salinity and water level on coastal marshes: an experimental test of disturbance as a catalyst for vegetation change. *Aquatic Botany* 61:255-268.

Baldwin, A.H., M.S. Egnotovich, and E. Clarke. 2001. Hydrologic change and vegetation of tidal freshwater marshes: field, greenhouse and seed-bank experiments. *Wetlands* 21: 519-531.

Baskin, C.C., and J.M. Baskin. 2001. *Seeds: Ecology, Biogeography, and Evolution of Dormancy and Germination.* Academic Press. San Francisco.

Baskin, C.C., and J.M. Baskin. 2003. Seed germination and propagation of *Xyris tennesseensis,* a federal endangered wetland species. *Wetlands* 23: 116–124.

Baskin, C.C., J.M. Baskin, E.W. Chester, and M. Smith. 2003. Ethylene as a possible cue for seed germination of *Schoenoplectus hallii* (Cyperaceae), a rare summer annual of occasionally flooded sites. *American Journal of Botany* 90: 620-627.

Bernard, J.M. 1990. Life history and vegetative reproduction in *Carex. Canadian Journal of Botany* 68:1441-1448.

Bertness, M.D. 1991. Interspecific interactions among high marsh perennials in a New England salt marsh. *Ecology* 72:125-137.

Blits, K.C., and J.L. Gallagher. 1991. Morphological and physiological responses to increased salinity in marsh and dune ecotypes of *Sporobolus virginicus* (L.) Kunth. *Oecologia* 87:330-335.

Bradshaw, A.D. 1983. The reconstruction of ecosystems. *Journal of Applied Ecology* 20:1-17.

Bradshaw, A.D. 1987. The reclamation of derelict land and the ecology of ecosystems. In W.R. Jordan, M.E. Gilpin and J.D. Aber, eds. *Restorations Ecology: a Synthetic Approach to Ecological Research*, pp. 52-74. Cambridge University Press, New York

Budelsky, R.A., and S.M. Galatowitsch. 1999. Effects of moisture, temperature, and time on seed germination of five wetland *Carices*: implications for restoration. *Restoration Ecology* 7:86-97.

Budelsky, R.A., and S.M. Galatowitsch. 2000. Effects of water regime and competition on the establishment of a native sedge in restored wetlands. *Journal of Applied Ecology* 37:971-985.

Callaway, J.C., J.B. Zedler and D.L. Ross. 1997. Using tidal salt marsh mesocosms to aid wetland restoration. *Restoration Ecology* 5:135-146.

Chang, P.R. 1990. Strategies for managing endangered waterbirds on Hawaiian National Wildlife Refuges. Master's Thesis. University of Massachusetts at Amherst.

Clark, D.L., and M.V. Wilson. 2001. Fire, mowing and hand-removal of woody species in restoring a native wetland prairie in the Willamette Valley of Oregon. *Wetlands* 21:135-144.

Clewer, A.G., and D.H. Scarisbrick. 2001. *Practical Statistics and Experimental Design for Plant and Crop Science.* John Wiley and Sons, Ltd. New York.

Clevering, O.A. 1995. Germination and seedling emergence of *Scirpus lacustris* L. and *Scirpus maritimus* L. with special reference to the restoration of wetlands. *Aquatic Botany* 50:63-78.

Clevering, O.A., and W.M.G. van Gulik. 1997. Restoration of *Scirpus lacustris* and *Scirpus maritimus* stands in a former tidal area. *Aquatic Botany* 54:229-246.

Clevering, O.A., and M.P.J. Hundscheid. 1998. Plastic and non-plastic variation in growth of newly established clones of *Scirpus (Bolboschoenus) maritimus* L. grown at different water depths. *Aquatic Botany* 62:1-17.

Cuddihy, L.W., and C.P. Stone. 1990. *Alteration of Native Hawaiian Vegetation: Effects of Humans, Their Activities and Introductions.* University of Hawai'i Press. Honolulu, Hawaii.

D'Antonio, C., and L.A. Meyerson. 2002. Exotic plant species as problems and solutions in ecological restoration: a synthesis. *Restoration Ecology* 10:703-714.

Dahl, T.E. 1990. *Wetlands Losses in the United States 1780's to 1980's.* U.S. Department of Interior, USFWS. Washington, D.C.

Ducks Unlimited, Inc. 2000. *Hawaiian Islands Wetlands Conservation Plan.* (Pamphlet). Honolulu, Hawaii.

Duvall, F. Wildlife Biologist, Department of Land and Natural Resources. Personal Communication. February 2003.

Elliott, M.E. 1981. Wetlands and wetland vegetation of the Hawaiian Islands. Master's thesis (Geography), University of Hawai'i, Honolulu, Hawai'i.

Elzinga, C.L., D.W. Salzer, and J.W. Willoughby. 1998. *Measuring and Monitoring Plant Populations.* U.S. Department of Interior, Bureau of Land Management Technical Reference 1730-1. Denver, Colorado.

Falk, D.A., C.I. Miller, and M. Olwell (eds.). 1996. *Restoring Diversity: Strategies for Reintroduction of Endangered Plants.* Island Press. Washington, D.C.

Fattorini, M. 2001. Establishment of transplants on machine-graded ski runs above timberline in the Swiss Alps. *Restoration Ecology* 9:119-126.

Fowler, N. 1981. Competition and coexistence in a North Carolina grassland: II. The effects of the experimental removal of species. *Journal of Ecology* 69:843-854

Friedman, J. 1971. The effect of competition by adult *Zygophyllum dumosum* Boiss. on seedlings of *Artemisia herba-alba* Asso in the Negev desert of Israel. *Journal of Ecology* 775-782.

Galatowitsch, S.M., and A.G. van der Valk. 1996. The vegetation of restored and natural prairie wetlands. *Ecological Applications* 6:102-112.

Galinato, M.I., and A.G. van der Valk. 1986. Seed germination traits of annuals and emergents recruited during drawdowns in the Delta Marsh, Manitoba, Canada. *Aquatic Botany* 26:89-102.

Grace, J.B. 1987. The impact of preemption on the zonation of two *Typha* species along lakeshores. *Ecological Monographs* 57:283-303.

Guinther, E. Biologist, Ahahui Malama I ka Lokahi. Personal Communication. May 2003.

Handa, I.T., and R.L. Jefferies. 2000. Assisted revegetation trials in degraded salt-marshes. *Journal of Applied Ecology* 37:944-958.

Harrison, L. 2002. Kanaha Pond Report. *Native Hawaiian Plant Society Annual Newsletter* 9:1.

Hartman, J.M. 1988. Recolonization of small disturbance patches in a New England salt marsh. *American Journal of Botany* 75:1625-1631.

Hootsmans, J.M., and Wiegman, F. 1998. Four helophyte species growing under salt stress: their salt or life? *Aquatic Botany* 62:81-94.

Hunter, R.G., D.L. Combs, and D.B. George. 2000. Growth of sofstem bulrush (*Scirpus validus*) in microcosms with different hydrologic regimes and media depths. *Wetlands* 20:15-22.

Jordan, W.R., M.E. Gilpin and J.D. Aber. 1987. *Restorations Ecology: a Synthetic Approach to Ecological Research.* Cambridge University Press. New York.

Joshi, A.J. 1981. Amino acids and mineral constituents in *Sesuvium portulacastrum* L., A salt marsh halophyte. *Aquatic Botany* 10:69-74.

Juvik, S.P., and J.O. Juvik. 1998. *Atlas of Hawaii.* 3rd edition. University of Hawai`i Press. Honolulu. Hawaii.

Kantrud, H. A. 1996. The alkali (*Scirpus maritimus* L.) and saltmarsh (*S. robustus* Pursh) bulrushes: A literature review. National Biological Service, Information and Technology Report 6. Jamestown, ND: Northern Prairie Wildlife Research Center Home Page. http://www.npwrc.usgs.gov/resource/literatr/bulrush/bulrush.html

Keddy, P.A., L.H. Fraser, and I.C. Wisheu. 1998. A comparative approach to examine competitive response of 48 wetland plant species. *Journal of Vegetation Science* 9:777-786.

Keddy, P. 1999. Wetland restoration: the potential for assembly rules in the service of conservation. *Wetlands* 19:716-732.

Kent, D. M. 2001. *Applied Wetlands Science and Technology.* Lewis Publishers. New York.

Kirch, P.V. 1982. Transported landscapes. *Natural History* 91:32-35.

Koob, G.A. 1999. *Propagation of Native Hawaiian Wetland Plant Species: Ten Species with Potential for Use in Wetland Restoration and Revegetation Projects.* Study prepared for USFWS. Honolulu, Hawaii.

LaDeau, S.L., and A.M. Ellison. 1999. Seed bank composition of a Northeastern U.S. tussock swamp. *Wetlands* 19:255-261.

Loope, L.L. 1998. Hawai`i and the Pacific Islands. In M.J. Mac, P.A. Opler, C.E. Puckett Haecker and P.D. Doran, eds. *Status and Trends of the Nation's Biological Resources*, Volume 2, pp. 747-774. U.S. Department of the Interior, U.S. Geological Survey. Reston, Virginia.

Marcum, K.B., and Murdoch, C.L. 1991. Salt tolerance of the coastal salt marsh grass, *Sporobolus virginicus* (L.) kunth. *New Phytologist* 120:281-288.

Martinez, M.L., T. Valvere, and P. Moreno-Casasola. 1992. Germination response to temperature, salinity, light and depth of sowing of ten tropical dune species. *Oecologia* 92:343-353.

Masuda, M., M. Masayuki, and T. Yahara. 1999. Effects of salinity and temperature on seed germination in a Japanese endangered halophyte *Triglochin maritimum* (Juncaginaceae). *Journal of Plant Research* 112:457-461.

Meier, K.Z., Laurel, J.P., and J.E. Maragos. 1993. State of Hawai'i (USA). In D.A. Scott, ed. *A Directory of Wetlands in Oceania,* pp. 179-197. International Waterfowl and Wetlands Research Bureau, Slimbridge, U.K. and Asian Wetland Bureau, Kuala Lumpur, Malaysia.

Merrilees, D.W., G.E.D. Tiley and D.C. Gwynne. 1995. Restoration of wet heathland after opencast mining. In Wheeler, B.D., S.C. Shaw, W.J. Fojt, and R.A. Robertson. *Restoration of Temperate Wetlands,* pp 523-532. John Wiley & Sons. New York.

Middleton, B. 1999. *Wetland Restoration, Flood Pulsing, and Disturbance Dynamics.* John Wiley & Sons, Inc. New York.

Mitsch, W.J., and J.G. Gosselink. 2000. *Wetlands 3rd edition.* John Wiley & Sons, Inc. New York.

Montalvo, A.M., and N.C. Ellstrand. 2000. Transplantation of the subshrub *Lotus scoparius*: I. Test of the home site advantage hypothesis. *Conservation Biology* 14: 1034–1045.

National Research Council. 1995. *Wetlands: Characteristics and Boundaries.* National Academy Press. Washington, D.C.

Onkware, A.O. 2000. Effect of soil salinity on plant distribution and production at Loburu delta, Lake Bogoria National Reserve, Kenya. *Austral Ecology* 25:140-149.

Perry, L.G., and S.M. Galatowitsch. 2003. A test of two annual cover crops for controlling *Phalaris arundinaceae* invasion in restored sedge meadow wetlands. *Restoration Ecology* 11:297-308.

Reed, P.B. 1988. National list of plant species that occur in wetlands: Hawai'i (Region H). National Wetlands Inventory, U.S. Fish and Wildlife Service, Washington, D.C.

Reinartz, J.A., and E.L. Warne. 1993. Development of vegetation in small created wetlands in southeastern Wisconsin. *Wetlands* 13:154-164.

Ross, M.A., and J.L. Harper. 1972. Occupation of biological space during seedling establishment. *Journal of Ecology* 60:77-88.

Seabloom, E.W., A.G. van der Valk, and K.A. Moloney. 1998. The role of water depth and soil temperature in determining initial composition of prairie wetland coenoclines. *Plant Ecology* 138:203-216.

Seabloom, E.W., and A.G. van der Valk. 2003. Plant diversity, composition and invasion of restored and natural prairie pothole wetlands: implications for restoration. *Wetlands* 23:1-12.

Shallenberger, R.J. (ed.). 1981. *Hawai`i's Birds.* Hawai`i Audubon Society. Honolulu, Hawaii.

Shipley, B., and M. Parent. 1991. Germination responses of 64 wetland species in relation to seed size, minimum time to reproduction and seedling relative growth rate. *Functional Ecology* 5:111-118.

Silbernagle, M. U.S. Fish and Wildlife Refuge biologist. Honolulu, Hawaii. Personal Communication. May 2001.

Society of Wetland Scientists. 2000. Position Paper on the Definition of Wetland Restoration. http://www.sws.org/wetlandconcerns/restoration.pdf (17 March 2003).

Stemmermann, L. 1981. *A Guide to Pacific Wetland Plants.* U.S. Army Corps of Engineers, Honolulu District.

Streever, B., and J. Zedler. 2000. To plant or not to plant. *BioScience* 50:188-189.

Sustainable Resources Group Int'l, Inc. 2001. *Planting Site Report- Final Report: Watershed restoration using Marine Corps Base Hawaii environmental geographic information system (EGIS).* N47408-98-D-2408, DO 0003. Prepared for: Environmental Department, Marine Corps Base Hawaii, Kaneohe Bay HI 96863. Prepared by: Sustainable Resources Group Int'l, Inc. 970 N. Kalaheo Avenue, C300 Kailua, HI. Under Contract to Geo InSight International, Inc. 33 E. Matilija St., Suite 215, Ojai, CA 93023.

ter Heerdt, G.N.J., and H.J. Drost. 1994. Potential for the development of marsh vegetation from the seed bank after a drawdown. *Biological Conservation* 67:1-11.

Thompson, K., and J.P. Grime. 1983. A comparative study of germination responses to diurnally-fluctuating temperatures. *Journal of Applied Ecology* 20:141-156.

Turner, C.E., T.D.Center, D.W. Burrows, and G.R. Buckingham. 1998. Ecology and management of *Melaleuca quinquenervia,* an invader of wetlands in Florida, U.S.A. *Wetlands Ecology and Management* 5:165-178.

Ungar, I.A. 1978. Halophyte seed germination. *Botanical Review* 44:233-264.

U.S. Department of Agriculture, Soil Conservation Service. 1972. *Soil Survey of the Islands of Kauai, O`ahu, Maui, Molokai, and Lanai, State of Hawaii.* U.S. Dept. of Agriculture in cooperation with the University of Hawaii Agricultural Experiment Station. Washington, D.C.

U.S. Department of Agriculture, Natural Resources Conservation Service. 2001. 2001 Wetlands Reserve Program, Hawai`i Summary. http://www.nrcs.usda.gov/programs/wrp/factsheets/2001/HI_WRP.PDF (December 2001).

U.S. Fish and Wildlife Service. 2001. *O`ahu National Wildlife Refuge Complex* (pamphlet). O`ahu National Wildlife Refuge Complex, Hale`iwa, Hawaii.

U.S. Fish and Wildlife Service. Pacific Islands – National Wildlife Refuges. http://pacificislands.fws.gov/wnwr/nwrindex.html#hawaii (10 March 2003).

U.S. Geological Survey. 1997. National Water Summary on Wetland Resources. Water Supply Paper 2425. http://water.usgs.gov/nwsum/WSP2425/state_highlights_summary.html (7 March 1997).

van der Valk, A.G., T.L. Bremholm, and E. Gordon. 1999. The restoration of sedge meadows: seed viability, seed germination requirements and seedling growth of *Carex* species. *Wetlands* 19:756-764.

Van Dyke, P. 2001. Growing makaloa (*Cyperus laevigatus* L.) in constructed wetlands for weaving and treating wastewater. Report for U.S. Geological Survey Grant (no. 99 CRGR003) Bishop Museum. Honolulu, Hawaii.

Van Dyke, P., Manager, Amy B.H. Greenwell Ethnobotanical Garden, Captain Cook, Hawai`i. Personal Communication. November 2002.

Wagner, W.L., D.R. Herbst, and S.H. Sohmer. 1999. *Manual of the Flowering Plants of Hawaii.* University of Hawaii Press: Bishop Museum Press. Honolulu, HI.

Walker, L.R., and S.D. Smith. 1997. Impacts of invasive plants on community and ecosystem properties. In J.O. Luken and J.W. Thieret, eds. *Assessment and Management of Plant Invasions,* pp. 69-86. Springer, New York.

Webb, E.C., and I.A. Mendelssohn. 1996. Factors affecting vegetation dieback of an oligohaline marsh in coastal Louisiana: Field manipulation of salinity and submergence. *American Journal of Botany* 83:1429-1435.

Wheeler, B.D., S.C. Shaw, W.J. Fojt, and R.A. Robertson. 1995. *Restoration of Temperate Wetlands.* John Wiley & Sons, New York.

Wilcox, D.A., and T.H. Whillans. 1999. Techniques for restoration of disturbed coastal wetlands of the Great Lakes. *Wetlands* 19:835-857.

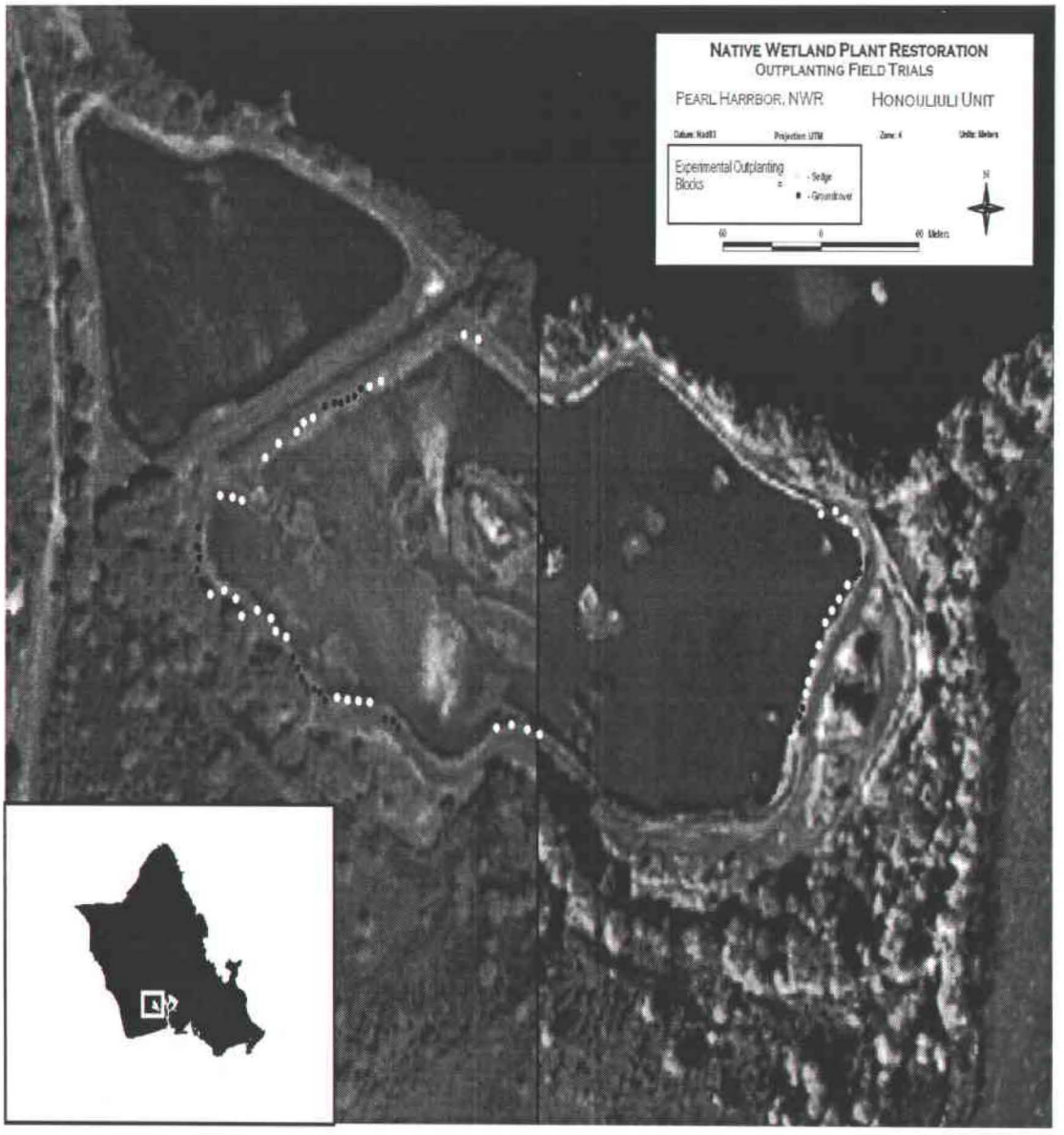
Yates, C.J., and Hobbs, R.J. 1997. Temperate eucalypt woodlands: a review of their status, processes threatening their persistence and techniques for restoration. *Australian Journal of Botany* 45:949-973.

Yetka, L.A., and Galatowitsch, S.M. 1999. Factors affecting revegetation of *Carex lacustris* and *Carex stricta* from rhizomes. *Restoration Ecology* 7:162-171.

Zedler, J.B. 2000. Progress in wetland restoration ecology. *Trends in Ecology and Evolution* 15:402-407.

Zedler, J.B., J.C. Calloway, and G. Sullivan. 2001. Declining biodiversity: why species matter and how their functions might be restored in Californian tidal marshes. *BioScience* 51:1005–1017.

APPENDIX A: Map of Outplanting Field Trials Study Site.



APPENDIX B: Composition of Non-planted Species in Outplanting Treatment Plots, April 1, 2003.

Table B.1. Final average cover of non-planted species recorded in sedge planting and control plots, broken down by treatment, on April 1, 2003.

Plots planted with	*Bacopa monnieri*	*Pluchea indica*	*Batis maritima*	*Typha latifolia*	all other species	Total
Bolboschoenus maritimus						
control / competition	100 ± 0	3.2 ± 7.5	8.4 ± 8.9	1.8 ± 3.8	2.4 ± 7.6	115.8 ± 16.8
low density / competition	100 ± 0	0.2 ± 0.6	17.4 ± 11.7	1.0 ± 2.5	1.6 ± 5.1	120.2 ± 10.0
high density / competition	100 ± 0	2.2 ± 7.0	9.6 ± 10.7	3.0 ± 4.3	1.2 ± 2.7	116.0 ± 9.8
Cyperus javanicus						
control / competition	96.6 ± 10.8	1.2 ± 3.8	8.4 ± 11.2	4.4 ± 5.5	1.6 ± 5.0	112.4 ± 18.5
low density / competition	86.0 ± 30.7	5.2 ± 9.6	9.6 ± 13.6	2.2 ± 4.0	6.2 ± 12.9	109.2 ± 34.9
high density / competition	82.4 ± 37.3	3.4 ± 6.4	7.6 ± 10.0	1.8 ± 3.3	3.6 ± 7.6	98.8 ± 40.0
Cyperus laevigatus						
control / competition	100 ± 0	3.2 ± 7.5	8.4 ± 8.9	1.8 ± 3.9	3.8 ± 11.3	117.2 ± 13.6
low density / competition	100 ± 0	0.2 ± 0.6	17.4 ± 11.7	1.0 ± 2.5	0	118.6 ± 11.9
high density / competition	100 ± 0	2.2 ± 7.0	9.6 ± 10.7	3.0 ± 4.3	0	114.8 ± 10.4
Cyperus polystachyos						
control / competition	99.0 ± 3.2	5.0 ± 8.0	10.0 ± 13.6	3.4 ± 4.8	12.6 ± 28.7	130.0 ± 31.4
low density / competition	100 ± 0	0	6.8 ± 7.7	2.4 ± 5.4	7.6 ± 15.5	116.8 ± 12.8
high density / competition	95.2 ± 10.5	6.6 ± 17.1	14.6 ± 17.5	2.0 ± 3.4	7.2 ± 22.1	125.6 ± 31.7

APPENDIX B (Continued): Composition of Non-planted Species.

Table B.2. Final average cover of non-planted species recorded in groundcover planting and control plots, broken down by treatment, on April 1, 2003.

Plots planted with	*Bacopa monnieri*	*Pluchea indica*	*Batis maritima*	all other species	Total
Jacquemontia ovalifolia					
control / competition	47.0 ± 49.8	30.6 ± 37.8	17.0 ± 21.1	10.2 ± 16.2	104.8 ± 47.2
low density / competition	47.0 ± 48.8	13.6 ± 20.3	16.2 ± 26.1	11.0 ± 14.1	87.8 ± 46.2
high density / competition	49.8 ± 47.1	11.2 ± 15.9	11.8 ± 18.4	6.0 ± 13.5	78.8 ± 32.9
Sesuvium portulacastrum					
control / competition	62.0 ± 40.5	34.8 ± 34.9	21.4 ± 33.2	13.0 ± 18.5	131.2 ± 32.0
low density / competition	45.2 ± 35.8	29.4 ± 33.8	25.4 ± 33.8	18.2 ± 28.0	108.8 ± 22.8
high density / competition	13.6 ± 20.1	13.6 ± 20.1	21.2 ± 28.4	8.2 ± 10.8	110.4 ± 42.4
Sporobolus virginicus					
control / competition	38 ± 48.4	46.4 ± 39.1	35.8 ± 34.2	34.0 ± 48.7	154.2 ± 34.2
low density / competition	45.2 ± 48.7	26.4 ± 26.0	22.4 ± 31.6	11.0 ± 12.6	105.0 ± 45.7
high density / competition	22 ± 41.6	25.6 ± 27.0	20.0 ± 21.4	22.8 ± 32.6	90.4 ± 44.6

APPENDIX C: Results of Density and Competition Treatments.

Table C.1. Final mean (± 1 SD) percent survival, health-code, maximum height (cm), number of shoots, and number of reproductive stems for outplanted species, in the different density and competition treatments (measured on April 1, 2003). "High density" means are the average of high density / competition plus high density / no competition plots. "Low density" means are the average of low density / competition plus low density / no competition plots. "Competition" means are the average of high density / competition plus low density / competition plots. "No competition" means are the average of high density / no competition plus low density / no competition plots.

	Survival (%)	Health-Code	Maximum height (cm)	Number of shoots	Number of reproductive stems
Bolboschoenus maritimus					
high density / competition	1.1 ± 3.5	5 ± 0	15 ± 0	1 ± 0	0 ± 0
high density / no competition	13.2 ± 11.4	4.6 ± 0.7	15.1 ± 6.4	1.5 ± 6	0.1 ± 0.4
low density / competition	2.5 ± 7.9	1 ± 0	15.5 ± 9.2	1 ± 0	1 ± 0
low density / no competition	12.5 ± 24.3	4.7 ± 0.6	15.7 ± 3.8	1.4 ± 0.5	0 ± 0
high density	7.2 ± 10.5	4.7 ± 0.7	15.1 ± 5.9	1.4 ± 0.6	0.1 ± 0.3
low density	7.5 ± 18.3	3.8 ± 1.9	15.6 ± 5.3	1.3 ± 0.4	0.3 ± 0.5
competition	1.8 ± 6	3 ± 2.8	15.3 ± 6.5	1 ± 0	0.5 ± 0.7
no competition	12.9 ± 18.9	4.7 ± 0.7	15.3 ± 6.2	1.5 ± 0.6	0.1 ± 0.4
Cyperus javanicus					
high density / competition	98.9 ± 3.5	2.9 ± 1.0	71.9 ± 38.6	10.0 ± 5.0	1.2 ± 1.2
high density / no competition	93.3 ± 17.1	3.1 ± 1.1	66.8 ± 32.2	11.7 ± 6.6	1.6 ± 1.4
low density / competition	97.5 ± 7.9	2.9 ± 1.1	69.3 ± 35.1	15.2 ± 11.3	2.1 ± 3.2
low density / no competition	97.5 ± 7.9	3.3 ± 1.1	69.8 ± 38.2	16.2 ± 8.9	2.4 ± 2.7
high density	96.1 ± 12.7	3.0 ± 1.0	69.3 ± 34.7	10.9 ± 5.7	1.3 ± 1.4
low density	97.5 ± 7.7	3.1 ± 1.1	69.6 ± 35.7	15.7 ± 9.9	2.3 ± 2.9
competition	98.2 ± 6.0	2.9 ± 1.0	70.6 ± 36	12.6 ± 8.9	1.7 ± 2.4
no competition	95.4 ± 13.5	3.2 ± 1.1	68.3 ± 34.4	13.9 ± 8.0	1.9 ± 2.2
Cyperus laevigatus					
high density / competition	64 ± 35.0	2.2 ± 0.8	23.2 ± 4.6	******	1.9 ± 1.1
high density / no competition	84 ± 18.4	2.1 ± 0.6	19.6 ± 3.5	******	1.7 ± 1.0
low density / competition	80 ± 21.1	2.0 ± 0.7	22.2 ± 4.8	******	1.7 ± 1.1
low density / no competition	82 ± 29.0	2.5 ± 0.7	23.4 ± 3.3	******	2.0 ± 1.4
high density	74 ± 29.1	2.2 ± 0.7	21.3 ± 4.4	******	1.8 ± 1.0
low density	81 ± 24.7	2.3 ± 0.7	22.8 ± 4.1	******	1.9 ± 1.2
competition	72 ± 29.3	2.1 ± 0.7	22.7 ± 4.6	******	1.8 ± 1.1
no competition	83 ± 23.6	2.3 ± 0.7	21.5 ± 3.8	******	1.9 ± 1.2

Table C.1. (Continued) Density and Competition treatment results

	Survival (%)	Health-Code	Maximum height (cm)	Number of shoots	Number of reproductive stems
Cyperus polystachyos					
high density / competition	53 ± 39.7	1.5 ± 1.1	27.5 ± 13.2	******	0.3 ± 0.3
high density / no competition	64 ± 39.5	1.9 ± 0.9	27.9 ± 11.2	******	0.5 ± 0.4
low density / competition	60 ± 40	1.7 ± 0.8	31.4 ± 8.3	******	0.6 ± 0.9
low density / no competition	62 ± 41.6	2.4 ± 1.4	32.8 ± 14.7	******	1.3 ± 1.3
high density	58.5 ± 39.0	1.7 ± 1.0	27.7 ± 11.8	******	0.4 ± 0.3
low density	61 ± 39.7	2.0 ± 1.2	32.1 ± 11.5	******	1.0 ± 1.1
competition	56.5 ± 39.0	1.6 ± 0.9	29.5 ± 10.8	******	0.5 ± 0.7
no competition	63 ± 39.5	2.1 ± 1.2	30.3 ± 12.9	******	0.9 ± 1.0
Jacquemontia ovalifolia					
high density / competition	68 ± 9.2	2.5 ± 1.1	*****	*****	1.4 ± 2.4
high density / no competition	71 ± 12.9	3.0 ± 1.2	*****	*****	4 ± 8.7
low density / competition	70 ± 14.1	2.8 ± 0.8	*****	*****	2.4 ± 2.7
low density / no competition	80 ± 24.9	2.9 ± 0.9	*****	*****	1.9 ± 1.9
high density	69.5 ± 11.0	2.8 ± 1.2	*****	*****	2.7 ± 6.3
low density	75 ± 20.4	2.9 ± 0.8	*****	*****	2.2 ± 2.3
competition	69 ± 11.7	2.7 ± 0.9	*****	*****	1.9 ± 2.5
no competition	75.5 ± 19.9	3.0 ± 1.1	*****	*****	3.0 ± 6.2
Sesuvium portulacastrum					
high density / competition	64 ± 18.4	1.8 ± 1.0	*****	*****	0.7 ± 0.9
high density / no competition	67 ± 35.6	2.4 ± 1.3	*****	*****	2.5 ± 4.9
low density / competition	56 ± 28.0	1.7 ± 1.0	*****	*****	0.8 ± 2.3
low density / no competition	70 ± 34.3	2.1 ± 1.3	*****	*****	1.6 ± 3.0
high density	65.5 ± 27.6	2.1 ± 1.2	*****	*****	1.6 ± 3.6
low density	63 ± 31.3	1.9 ± 1.1	*****	*****	1.2 ± 2.6
competition	60 ± 23.4	1.7 ± 1.0	*****	*****	0.7 ± 1.7
no competition	68.5 ± 34.1	2.3 ± 1.3	*****	*****	2.1 ± 4.1
Sporobolus virginicus					
high density / competition	84.6 ± 30.3	2.4 ± 1.0	29.6 ± 11.3	2.3 ± 1.0*	0
high density / no competition	83.1 ± 26.3	3.5 ± 1.0	31.4 ± 6.4	3.7 ± 2.4*	0.6 ± 1.1
low density / competition	90.0 ± 16.1	2.9 ± 0.7	32.0 ± 7.6	3.9 ± 1.7*	0.3 ± 0.5
low density / no competition	96.7 ± 7.0	3.9 ± 0.8	33.3 ± 4.8	5.5 ± 2.1*	0.9 ± 2.0
high density	83.8 ± 27.7	2.9 ± 1.1	30.5 ± 9.0	3.0 ± 2.0*	0.3 ± 0.8
low density	93.3 ± 12.6	3.4 ± 0.9	32.6 ± 6.2	4.7 ± 2.0*	0.6 ± 1.5
competition	87.3 ± 23.8	2.6 ± 0.9	30.1 ± 9.4	3.1 ± 1.6*	0.1 ± 0.4
no competition	89.9 ± 20.0	3.7 ± 0.9	32.3 ± 5.6	4.6 ± 2.4*	0.7 ± 1.6

*** parameter not measured for this species

APPENDIX C (Continued): Results of Density and Competition Treatments.

Table C.2. Final mean (± 1 SD) percent cover for outplanted sedge species, in the different density and competition treatments (measured on April 1, 2003). "High density" means are the average of high density / competition plus high density / no competition plots. "Low density" means are the average of low density / competition plus low density / no competition plots. "Competition" means are the average of high density / competition plus low density / competition plots. "No competition" means are the average of high density / no competition plus low density / no competition plots.

	Absolute Cover (%)			
Treatment	***Bolboschoenus maritimus***	***Cyperus javanicus***	***Cyperus laevigatus***	***Cyperus polystachyos***
control / competition	0	15.4 ± 34.2	0	0
low density / competition	0.3 ± 0.7	66.4 ± 28.7	20.2 ± 13.4	24.4 ± 24.8
high density / competition	0.2 ± 0.6	89.8 ± 18.1	29.4 ± 27.6	31.8 ± 34.5
control / no competition	0.8 ± 1.0	14 ± 29.5	0	0
low density / no competition	0.7 ± 1.3	79.6 ± 24.6	30.6 ± 21.0	38.0 ± 39.7
high density / no competition	1.5 ± 1.3	89.0 ± 25.0	37.2 ± 22.3	32.4 ± 36.9
high density	0.9 ± 1.2	90.2 ± 21.5	33.3 ± 24.7	32.1 ± 34.8
low density	0.4 ± 0.8	75.6 ± 25.0	25.4 ± 17.9	31.2 ± 33.0
control	0.5 ± 1.2	14.7 ± 31.2	0	0
competition	1.1 ± 1.0	78.1 ± 26.3	33.9 ± 21.3	28.1 ± 29.5
no competition	4.5 ± 2.9	84.3 ± 24.6	24.8 ± 21.6	35.2 ± 37.4

Table C.3. Final mean (± 1 SD) percent cover for outplanted groundcover species, in the different density and competition treatments (measured on April 1, 2003). "High density" means are the average of high density / competition plus high density / no competition plots. "Low density" means are the average of low density / competition plus low density / no competition plots. "Competition" means are the average of high density / competition plus low density / competition plots. "No competition" means are the average of high density / no competition plus low density / no competition plots.

	Absolute Cover (%)		
	Jacquemontia ovalifolia	***Sesuvium portulacastrum***	***Sporobolus virginicus***
Treatment			
control / competition	44.4 ± 27.5	1.4 ± 2.7	8.2 ± 14.0
low density / competition	43.4 ± 15.2	7.8 ± 7.1	50.0 ± 27.9
high density / competition	53.4 ± 17.7	28.8 ± 26.9	67.2 ± 30.2
control / no competition	59.2 ± 25.7	4.2 ± 9.2	19.8 ± 26.4
low density / no competition	54.6 ± 20.9	21.2 ± 13.2	83.8 ± 13.7
high density / no competition	57.8 ± 21.8	40.2 ± 30.5	80.2 ± 20.9
high density	55.6 ± 19.5	34.5 ± 28.6	73.7 ± 26.2
low density	49.0 ± 18.7	14.5 ± 12.4	66.9 ± 27.5
control	51.8 ± 27.0	2.8 ± 6.7	14.0 ± 21.4
competition	47.1 ± 20.6	18.3 ± 22.0	58.6 ± 29.6
no competition	57.2 ± 22.2	30.7 ± 24.9	82.0 ± 17.3