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First Season Effects of Managed Flooding on the Invasive Species *Phalaris arundinacea* L. And Shoreline Vegetation Communities in an Urban Wetland

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THESIS APPROVAL

The abstract and thesis of Noah John Jenkins for the Master of Science in Environmental Sciences and Resources were presented April 29, 2005, and accepted by the thesis committee and the department.

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ABSTRACT

An abstract of the thesis of Noah John Jenkins for the Master of Science in Environmental Science and Resources presented April 29, 2005.

Title: First Season Effects of Managed Flooding on the Invasive Species *Phalaris arundinacea* L. and Shoreline Vegetation Communities in an Urban Wetland.

Recent management efforts in the Smith and Bybee Lakes Wildlife Area (SBL), a 700-ha preserve in north Portland, Oregon, have included using a water control structure to suppress invasive reed canarygrass (*Phalaris arundinacea* L.) by flooding during spring and early summer growth periods. For the first year of managed flooding, I sought to determine: (a) the extent and distribution of reed canarygrass at SBL; (b) the effectiveness of the change in water level at suppressing reed canarygrass; and (c) the effects of the change in water level on other plant species.

I established 30 vegetation transects throughout SBL before completion of the water control structure. These transects were randomly distributed, placed perpendicularly to the shorelines of the wetlands, and had a cumulative length of 3.1 km. I measured vegetation on the transects in autumn 2003 and autumn 2004 using the line intercept method at 10-cm intervals. I surveyed the transects to generate elevation profiles, accurate to 0.15 cm, to determine depth and duration of flooding, which I correlated with vegetative changes. I also monitored inundation depth, growth, and phenological response of individual stands of reed canarygrass during the 2004 growing season.

Baseline measurements showed that reed canarygrass had 43.7% cover in the shoreline areas of SBL in 2003. This invasive plant had a strong impact on plant diversity; correlation of reed canarygrass cover with Shannon diversity yielded a Spearman's rho value of 0.69 ($P<0.05$). Measurements of reed canarygrass stands during the 2004 growing season indicated a reduction in mid-season growth and alteration of growth habit for inundated patches as compared to upland stands. Monitoring of the transects during 2004 showed a 2.5% decrease in reed canarygrass cover and an increase in cover for several important native taxa. Using multinomial logistic regression, I determined that deeper inundation and the presence of regenerating willow forest were more likely to produce a decrease in reed canarygrass cover; peak flooding of more that 0.85 m was most likely to yield a net loss of reed canarygrass. My results indicate that deep inundation, particularly when combined with shading, can be effective in suppressing reed canarygrass.

FIRST SEASON EFFECTS OF MANAGED FLOODING ON THE INVASIVE SPECIES *PHALARIS ARUNDINACEA* L. AND SHORELINE VEGETATION COMMUNITIES IN AN URBAN WETLAND

by

NOAH JOHN JENKINS

A thesis submitted in partial fulfillment of the
requirements for the degree of

MASTER OF SCIENCE
in
ENVIRONMENTAL SCIENCES AND RESOURCES

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2005

DEDICATION

For Mom.

Thank you for all the ways you let me know you're in my corner, come what may.

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I had phenomenal help from Marsha Holt-Kingsley, Susan Garland, Emily Smith, and Kelli Hoffman in the field; they spent hours and hours and HOURS

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

Introduction and Background

Introduction: Managed Flooding

Managed flooding is often an important component of habitat restoration. Alteration of historic hydrological patterns often degrades ecological function (Warren et al. 2002); hence, re-establishing those patterns—or approximating them through management efforts—is often necessary to restore the ecological balance of a natural area. This is particularly true in wetlands, where incomplete restoration of hydrology can result in the failure of the overall restoration of the wetland (Mulhouse and Galatowitsch 2003).

Numerous examples have illustrated the success of this approach to habitat restoration. A twenty-year study of salt-marsh recovery in Connecticut after dikes were breached to restore tidal flooding found that more flooding and greater hydroperiod were important to the recovery of vegetation; this led to restored sites being used more than twice as often by birds. Indeed, angiosperms, macroinvertebrates, birds and fish all began to recover on the restored sites, leading the researchers to assert that the return of tidal flooding set the marshes on a course to the full return of ecological function (Warren et al. 2002). In another study in Washington, dike breaching led to the establishment of a tidal freshwater plant assemblage; invertebrate and fish populations likewise began to mirror those of surrounding reference estuaries (Tanner et al. 2002).

Similarly, hydrologic restoration played a critical role in a study of prairie pothole wetlands, where vegetative recovery was impaired in areas where mid-summer flooding was absent or minimal; flooding frequency was also the best predictor of species richness in these wetlands (Mulhouse and Galatowitsch 2003).

In each of these studies, the suppression of invasive plant species was an important component of the restoration. Hydroperiod was an important factor in the replacement of *Phragmites australis* by salt marsh angiosperms (Warren et al. 2002); flooding likewise played a role in controlling reed canarygrass (*Phalaris arundinacea* L.), which died back significantly four years after dike breaching (Tanner et al. 2002).

In cases where a return to historic patterns is impossible or impractical, managed flooding may be substituted to approximate the ecological benefits that natural flooding provided (Michener and Haeuber 1998). Ellis et al. (1999) found that managed flooding began the process of restoring ecosystem function in a riparian context. Managed flooding can also be critical in the control of invasive species; Paveglio and Kilbride (2000) urged the maintenance of consistent water levels through winter and early spring in conjunction with other control measures to prevent the spread of reed canarygrass.

In December 2003, Metro—the regional metropolitan governing authority headquartered in Portland, Oregon—finished the construction of a new water control structure at Smith and Bybee Lakes Wildlife Area (SBL), a protected area of more than 700 ha containing two lakes and associated palustrine wetlands, in North Portland. Being responsible for the environmental conditions in SBL, Metro sought to use this new control structure to enable water level management to achieve a number of objectives:

- Control of reed canarygrass (*Phalaris arundinacea* L.) by flooding during spring and early summer growth periods;
- Support of native emergent and bottomland hardwood plant communities via annual drawdown;
- Provision of fish passage into and out of the wetlands, allowing them to be used by juvenile salmonids (Chinook and Coho) for off-channel rearing and refugia habitats;
- Exposure of mudflats for migrating shorebirds in mid- to late summer;
- Retention of water in the wetlands during winter for use by waterfowl, including flooded emergent wetlands with food resources for dabbling ducks.

Background

Smith and Bybee Lakes Wildlife Area

The Smith and Bybee Lakes Wildlife Area is located in North Portland, near the confluence of the Columbia and Willamette Rivers. It encompasses Smith Lake, Bybee Lake, and the St. Johns landfill, which covers roughly 100 ha. SBL is home to a variety of wildlife; more than 100 species of birds have been sighted there, including nesting ospreys (*Pandion haliaetus*) and bald eagles (*Haliaeetus leucocephalus*), and numerous mammals such as black-tailed deer (*Odocoileus hemionus*), beaver (*Castor canadensis*) and nutria (*Myocastor coypus*) live there. One of two remaining large populations of western painted turtles (*Chrysemys picta bellii*) in Oregon resides at SBL, along with Pacific chorus frogs (*Pseudacris regilla*) and salamanders (Friends of Smith and Bybee Lakes 2004). Since completion of the water control structure, coho (*Oncorhynchus kisutch*) and chinook (*O. tschawytscha*) salmon have been using the fish ladder to access the lakes.

In a 1994 study, Lev and others identified 14 major plant assemblages at SBL, based on field observations and on color infrared photographs taken in July 1992. The largest of these was dominated by water smartweed (*Polygonum amphibium*), which was also a significant component of the "dead Piper's willow" assemblage. These two assemblages combined to cover 298 ha, or 41% of the total study area of 722 ha. An "upland" assemblage—primarily the St. Johns landfill—covered 124 ha (17%). Forest wetland assemblages, dominated by willows (*Salix* spp.), Oregon ash (*Fraxinus latifolia*), or black cottonwood (*Populus balsamifera*) accounted for a combined 113 ha (16%). The driest herbaceous assemblage, which occurred on areas subjected to the least inundation, was dominated by reed canarygrass, and covered 86 ha (12%). Reed canarygrass was also an important component of several other assemblages (Lev et al. 1994). That study also established and monitored ten transects, each meant to represent one of the assemblages identified from the aerial photograph.

At the time of the Lev et al. report, an earth dam was in place at the point where the North Slough arm of the Columbia Slough enters the Smith and Bybee Lakes system. This dam, installed in 1982, prevented much of the normal, seasonal variation in lake levels. Additionally, by holding lake levels high throughout the year, the dam led to the death of much of the willow forest that occupied a significant percentage of the area surrounding the lakes. This gave rise to the "dead Piper's willow" assemblage referred to above. Subsequently, several dry years in the early 2000s—during which the lakes were dry or nearly so for part of the growing season—have allowed for the establishment of a regenerating forest of Pacific willow (*Salix lucida*) along the shores of the lakes (Elaine Stewart, *pers. comm.*).

A follow-up study of several of the transects previously established and monitored by Lev and others, conducted in 2001, indicated that reed canarygrass has been increasingly invasive there, increasing in cover from 25% to over 45% (Figure 1).

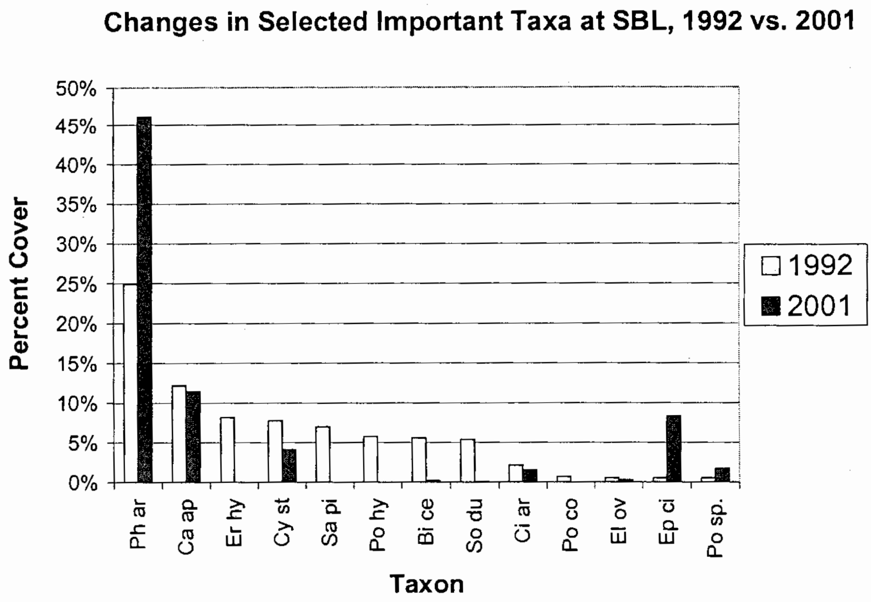


Figure 1. Comparison of transects at SBL in 1992 vs. 2001. "Ph ar" = *Phalaris arundinacea*; "Ca ap" = *Carex aperta*; "Er hy" = *Eragrostis hypnoides*; "Cy st" = *Cyperus strigosus*; "Sa pi" = *Salix piperi*; "Po hy" = *Polygonum hydropiper*; "Bi ce" = *Bidens cernua*; "So du" = *Solanum dulcamara*; "Ci ar" = *Cirsium arvense;* "Po co" = *Polygonum coccineum* (= *P. amphibium*); "El ov" = *Eleocharis ovata*; "Ep ci" = *Epilobium ciliatum*; "Po sp." = *Polygonum* species.

Two of ten transects established in 1992 were monotypic in reed canarygrass (Lev et al. 1994); this species was dominant or co-dominant along five of the transects. By 2001, all six of the transects that were relocated and measured were dominated or co-dominated by reed canarygrass (Elaine Stewart, *unpub. data*). Moreover, reed canarygrass had become the dominant species on two of the transects where it had not been significant in 1992.

Reed Canarygrass

Reed canarygrass is a long-lived, cool-season, perennial grass, with vigorous growth supported by an aggressive underground rhizome system. It is a facultative wetland species that occurs in palustrine wetlands, wet prairies, riparian areas and some uplands (Paveglio and Kilbride 2000). Individual plants can reach 2 m in height (Barnes 1999); total productivity in terms of biomass can be as high as 9 tons per acre (20 tonnes per hectare) (Stannard and Crowder 2001). Reed canarygrass establishes either by seed or rhizomes/root wads (Antieau 2003); it has a germination rate comparable to that of other cool-season pasture grasses (Stannard and Crowder 2001).

Growth typically begins in early spring (Stannard and Crowder 2001); in some cases, though, growth may start as early as mid-December, with plants reaching heights of up to 0.6 m by mid-March (Antieau 2003). Much of the growth of reed canarygrass occurs in mid-spring, with five to seven weeks of vertical growth following germination (Antieau 2003). Tillering—the production of new shoots from rhizomes—then begins, and continues until hard frost (Antieau 2003). Although summer drought conditions typically cause mature shoots to senesce, growth can continue in the fall if sufficient water becomes available (Stannard and Crowder 2001). Culms of the plant can also root, particularly after anthesis, and nodes become meristematically active once the panicles are removed (Stannard and Crowder 2001). Seeds are produced in early summer, and may germinate soon after ripening or form a seed bank in the soil, where they remain viable for an extended period of time, from several months to perhaps more than ten years (Barnes 1999; Stannard and Crowder 2001).

Reed canarygrass has a shallow root system, with most rhizomes occurring in the upper 20 cm of soil (Stannard and Crowder 2001). The rhizomes, which grow outwardly from the mother plant until the terminal bud develops a shoot (Stannard and Crowder 2001), account for much of the localized spread of reed canarygrass; 74% of new shoots come from rhizomes (Antieau 2003).

Due to its aggressive growth, reed canarygrass can develop into dense, monotypic stands (Naglich 1994). In a study of 41 restored prairie pothole wetlands, reed canarygrass went from being infrequent (occurring on fewer than 14 sites) in 1989 to occurring on all sites in 2000, and increased its percent cover by more than 60% on 19 sites (Mulhouse and Galatowitsch 2003). Reed canarygrass monocultures may remain dominant in wetlands for decades, resulting in the decrease of diversity of wetland flora (Emers 1990). For example, Volker and Smith (1965; cited in Barnes 1999) reported that 12 species disappeared from an Iowa wetland following the establishment of reed canarygrass. Native, herbaceous species that begin growth in late spring are particularly vulnerable to competition with reed canarygrass (Stannard and Crowder 2001). This competitive exclusion of native species changes wetland function and habitat (Antieau 2003). Wildlife dependent on the food, shelter, and habitat values provided by diversity in the wetland plant community may be affected. Shallow groundwater hydrology may also change, owing to increased evapotranspiration by reed canarygrass (Antieau 2003). It is thus desirable to find means of controlling reed canarygrass.

Response to Control Measures

Efforts to control the spread of reed canarygrass have included tillage, flooding, herbicides, defoliation (mowing, grazing), shading, biocontrol (release of pathogens), scalping (removal of topsoil), and combinations of the above measures (Stannard and Crowder 2001). Methods that have been applied at SBL over the past 5 years have included defoliation and shading, with limited success. For example, two transects that were monotypic in reed canarygrass in 1992 have been mowed annually in September for the past several years (Elaine Stewart, *pers. comm.*). While percent cover of reed canarygrass was lower in 2001, it was still the dominant species on both transects. On another mowed transect where reed canarygrass was only a trace species in 1992, it had become co-dominant by 2001, despite management efforts.

With the new water control structure, flooding at SBL became a viable option for managing this invasive species. Reports vary as to how much flooding reed canarygrass can tolerate, but there is general agreement that prolonged, deep inundation will kill it. Although reed canarygrass can survive periodic flooding by producing rhizomes tolerant of anoxic conditions, studies have shown that it may only tolerate deep inundation (at least 1 ft, or 30 cm, of water) for two years before it succumbs (Antieau 2003). Stannard and Crowder (2001) report that continual flooding or ponding, especially in warm weather, will eventually cause root death in reed canarygrass, leaving the plant without access to its carbohydrate reserves. Reed canarygrass tolerance for inundation is also dependent on the prevailing current and the silt content of the water (Antieau 2003).

Ordinarily, reed canarygrass undergoes a significant depletion in carbohydrate reserves in late May and June, as its growing point is elevated and a seed head develops (Antieau 2003). Inundation during this critical period may strongly inhibit tillering in this species, resulting in reduced vigor and/or abundance. Klimesova (1994) concluded that flooding was most effective in controlling reed canarygrass when timed to coincide with rhizome growth and tillering. This may lead to greater cover for other plant species at SBL, which could further inhibit reed canarygrass; seedlings of reed canarygrass are sensitive to competition (Stannard and Crowder 2001). However, past studies have indicated that as many as three years of permanent flooding under more than 5 ft (1.5 m) of water were required to eliminate reed canarygrass (Antieau 2003). In a report prepared for the Washington State Department of Transportation, Antieau (1999) recommended perennial inundation of more than 30 cm for most of the year to control reed canarygrass. It will likely require several years of response information to determine the effectiveness of the proposed regime at SBL.

Response of other taxa to flooding

Effective control of reed canarygrass consists of suppressing the growth of this invasive species, then "filling in the void" with species that are more desirable for the function of the wetland (Stannard and Crowder 2001). Hence, of interest to the management of SBL is both whether flooding will be effective at controlling reed canarygrass and, if so, what will be the response of the rest of the plant community. Research concerning the response of the other important taxa at SBL shows that their response to flooding is generally favorable. Table 1 summarizes the responses of the most common plant species at SBL.

Table 1. Summary of responses of major taxa at SBL to flooding experiments. Positive response (+); Negative response (-); Data conflicting or unavailable (?)

Species	Flooding Response
Phalaris arundinacea L.	-
Ludwigia palustris	?
Polygonum amphibium	+
P. persicaria	+
P. hydropiperoides	+
Bidens cernua	+
Salix lucida	+
Cyperus strigosus	?
Eragrostis hypnoides	+
Eleocharis ovata	+
Rumex maritimus	-
Veronica spp.	?

A number of flooding experiments have indicated that *Polygonum* species, such as *P. amphibium, P. persicaria,* and *P. hydropiperoides*, thrive under flooded conditions. *P. persicaria*, for instance, maintained growth and reproduction at close to maximum levels when flooded; by growing superficial and adventitious roots, this species was able to avoid any oxygen deficit resulting from being underwater (Sultan and Bazzaz 1993). Three varieties of *P. amphibium* had similar or higher mean leaf area in a submerged treatment, as compared to controls (Mitchell 1976). Carter and Grace (1990) found that a flooded treatment increased total size, shoot mass and stolon production in *P. hydropiperoides*, and concluded, based on their experiments, that both *P. amphibium* and *P. hydropiperoides* grow best in flooded conditions.

Bidens species also appear to respond well to inundation, although germination experiments on *Bidens* species provide conflicting evidence of their flooding tolerance. *B. cernua* germination went from 70% to 35% as water depth was increased from 0 cm to 10 cm in an experiment by Keddy and Ellis (1985). On the other hand, inundated seeds of a closely related species, *B. laevis*, had a higher germination percentage than seeds in drained or saturated soil, though seeds in closed containers—which simulated hypoxic conditions—did not germinate (Leck et al. 1994). In a study of an experimental wetland, *B. cernua* was the dominant species in all microcosms after one season; it maintained this position only in seasonally flooded treatments after five years (Weiher et al. 1996). *B. cernua* also produced the most biomass of the 17 wetland taxa studied by Kercher and Zedler (2004) in three out of four hydrologic regimes, including the most flooded treatment, in an outdoor pot experiment.

Ludwigia species, as obligate wetland plants (Cooke 1997), are typically tolerant of flooded conditions. Bonyongo et al. (2000) found that two such species, *L. stolonifera* and *L. leptocarpa*, were strongly present on the wettest part of the Okavango River floodplain, where flooding depths can exceed 1 m. On the other hand, Sahid and Hossain (1995) found that flooding significantly reduced the shoot length, dry weight, and root length of *L. hyssopifolia*. This result was particularly true when plants were flooded at the time of germination. Flooding significantly reduced the percentage of plants emerging from seed, although it did not affect the survival rate (Sahid and Hossain 1995).

Baskin et al. (1993) studied the flooding response of four summer annual species of Cyperaceae, and concluded that the primary effect of flooding was to prevent germination while seeds are underwater; once the water receded, seed dormancy ended for the summer. In another experiment (Baskin et al. 1991), the same authors also showed that seeds remained non-dormant throughout the growing season, regardless of flooding, for a variety of annual mudflat species; some, such as *Eragrostis hypnoides*, came out of dormancy at higher percentages in flooded conditions.

Salix species are listed as "very tolerant" of flooding. Good adventitious root growth allows them to withstand more than two growing seasons of inundation (Walters et al. 1980). Cooke and Azous (1997) found that two willow species common in Pacific Northwest wetlands—*S. scouleriana* and *S. sitchensis*—could grow in flooded conditions at all times of year, tolerating inundation of up to 1 m even during the growing season.

Eleocharis species are likewise tolerant of inundation. Two common species, *E. ovata* and *E. palustris*, are obligate wetland plants (Cooke 1997), indicating that they grow in commonly inundated areas. In a study of riverine plants in Australia, Blanch et al. (1999) found that *E. acuta*, while uncommon on the floodplain, was locally abundant in areas where inundation was less frequent but water remained after the flooding had receded.

Nabben et al. (1999) conducted a comparative study of the response of three *Rumex* species to submergence; results varied with life history. *R. maritimus* was intermediate in its susceptibility to flooding. Half of the young plants died after 56 days underwater in light conditions; mortality was total after only 35 days in the dark. Mature plants fared better, but were still negatively affected by flooding.

In sum, most of the major species at SBL, other than reed canarygrass, should at least tolerate the new water regime, and thus have the opportunity to fill the void left by any diminution of reed canarygrass cover. It remains unclear whether other non-native and/or invasive plant species will take advantage of the opening created by reed canarygrass suppression.

Little information is available on the response of wetland plant communities to water-level management (Paveglio and Kilbride 2000). Given this, as well as the lack of consensus on effective approaches to the use of flooding for reed canarygrass control, there is a need among wetland managers for greater study of the effectiveness and impacts of managed flooding.

This Study

Annual Water Management

With the new water control structure in place, the annual water management cycle at Smith and Bybee was planned to proceed as follows: In late fall, the manager for SBL—which, in 2003-2004, was Elaine Stewart of Metro—will close the structure (with the exception of 3 reverse tidegates and the fish ladder) to trap rising waters with heavy rains and seasonally high tides. This closure will hold as much water in the wetlands as possible, up to a maximum elevation of approximately 11.0 ft (3.35 m) above sea level (NGVD 29 datum), allowing for the greatest possible inundation of reed canarygrass. At this point, after the cool-season reed canarygrass had been suppressed, drawdown will begin. This drawdown will provide a progression of mudflats on which the native emergent plant community will grow; these natives are warm-season plants that do not grow earlier in the spring. However, the drawdown will have to progress rapidly enough to stimulate the salmonids to leave the wetlands. The wetlands will be dry or nearly dry by the end of September annually. These prescriptions are all subject to adjustment, depending on how the fish and plants respond.

During this study, water was retained until early June, when drawdown began. The drawdown proceeded slowly, at approximately 15 cm/week (Elaine Stewart, *pers. comm.*). By early August, the structure was open to daily tidal flow; it remained open until November 10, 2004, when the stoplogs were added to begin catching winter rains.

During the fall of 2003, I established 30 vegetation transects at SBL. The transects are randomly distributed throughout the Wildlife Area; I used them to analyze the response of reed canarygrass, native emergent and bottomland forest species, and potentially problematic invasive species to the new water regime. Initial monitoring of the transects took place from October to December of 2003, prior to completion of the water control structure, providing baseline information. During the 2004 growing season, I measured several characteristics of stands of reed canarygrass on the transects, along with depth and duration of inundation. I used this information to determine the effects of flooding on the growth of reed canarygrass. In addition, I monitored the transects late in the 2004 growing season, and then compared results to the baseline data to determine any changes in community composition. Finally, I surveyed the transects to get their topographic profiles, and used these—in combination with water level data—to determine the magnitude and duration of flooding on all parts of the transects. I found an inverse correlation between several measures of stand health and degree of inundation, as well as a reduction in percent cover for reed canarygrass, particularly in areas with significant willow cover and/or flooding of 0.85 m or more. I also found an increase in cover for most native plant species, and a decrease in cover for most invasive species other than *Phalaris*.

Research Questions and Hypotheses

In this thesis, I addressed the following questions:

1. What is the present extent and distribution of reed canarygrass at Smith and Bybee, particularly on lands that will be affected by the change in water regime?
2. How does the change in water regime affect the distribution of reed canarygrass at SBL? Does it suppress the abundance of reed canarygrass?
3. Does the change in water regime affect the distribution/abundance of other plant species at SBL?
    a. What is the response of late-germinating native herbaceous species, especially emergents?
    b. What is the response of other "weed" species (e.g., *Cirsium* spp.)?
    c. What is the response of bottomland forest species?

My hypotheses were:

1. Retaining water in the Smith and Bybee Lakes Wildlife Area until late May/June will reduce the abundance of reed canarygrass, as measured by percent cover.

2. The suppression of reed canarygrass will lead to a positive response in the percent cover of late-germinating emergent wetland species.

3. Longer and deeper inundation will be more effective in suppressing reed canarygrass in terms of both percent cover and phenological characteristics.

Methods

My overall approach to capturing vegetation responses to the new hydrologic regime was the establishment and annual monitoring of 30 new transects positioned perpendicularly to the shoreline of the open water surfaces throughout the Smith and Bybee Lakes area. This consisted of marking transects and measuring vegetation along transects during autumn 2003 to obtain baseline information, then repeating those measurements late in the 2004 growing season and comparing the resulting percent cover data. After the second round of monitoring, I surveyed the transects, referencing them to known elevation benchmarks, and combined this information with water-level data taken at the new water control structure to find depth and duration of flooding.

I also measured the depth of inundation of selected reed canarygrass stands, along with several indicators of the health of those stands, during the growing season, to determine whether deeper flooding had greater effect on reed canarygrass phenology.

Response of Abundance of Taxa

Transect point selection

Using an aerial photograph of SBL, taken in July 1992, Minot Kerr of Metro created shapefiles of the open water surfaces and a 50-m band of all land surfaces surrounding open water in ArcView 3.2. He then gridded the land surface shapefile into 10m x 10m cells using an avenue script, which selected 60 points at random from the gridded land surface shapefile. These points were entered into Microsoft Excel in numerical order. I then randomized them, using the Random Number Generation function from the Data Analysis toolpak. After consultation with SBL Wildlife Manager Elaine Stewart, we rejected several points as likely outliers; the rejected points were situated such that a transect drawn through that point, starting from the lakeshore, would strike another water body before reaching an upland area. We rejected a few other points due to extreme difficulty in accessibility. I replaced rejected points with points from further down the randomized list.

The aerial photograph of SBL, with the selected transects marked in white, appears as Figure 2.

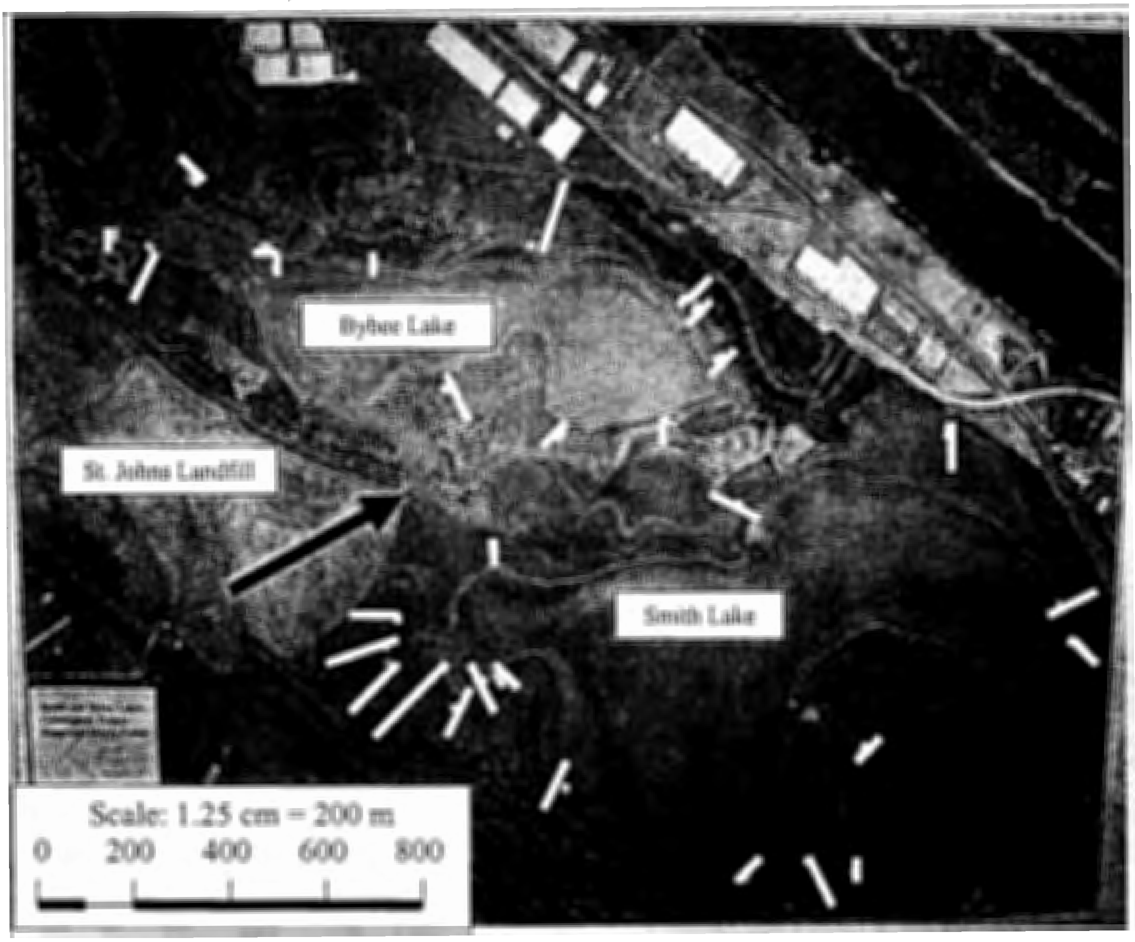


Figure 2. Aerial photograph of the Smith and Bybee Lakes Wildlife Area, taken in July 1992. Transects for the present study are marked in white. The arrow shows the location of the water control structure.

Transect point location

ArcView supplied decimal latitude and longitude to five decimal places for each survey point. I entered this information into a GPS unit (Trimble TDC 1 Asset Surveyor), which I used to find the points in the field. I entered altitude as 1 m. I located all points included in the study to within 1 m. For a subset of 12 transect points, the average number of satellite views was 7.83, with an average of 6.75 satellites communicating; the average position dilution of precision (PDOP) was 2.77. Three-dimensional co-ordinates require at least four satellite views; PDOP of less than 4 is considered “excellent” (Los Alamos National Laboratory 2003).

Transect marking procedure

I marked survey points with 3/8” x 4’ pieces of rebar and with plastic field markers anchored with fabric staples. In a few cases, when the point was either less than 10 m or more than 50 m from the shoreline, I later moved the rebar to a point 50 m from the shore to conserve materials, since leaving those pieces at the survey points would have meant the use of more rebar than necessary for the transect. I left the field marker in place to mark the survey point.

Using this mark as a reference, I placed rebar and a field marker at the edge of the water, such that a line drawn between the survey point marker and the shoreline marker was perpendicular to the shoreline. I read the direction from the survey point to the shoreline marker using a Starter compass, and recorded the reading on the first data sheet for the transect. I measured slope from the shoreline with a Suunto PM-5/360 PC clinometer, using an upland reference point. Next, I stretched a 50-m metric tape measure between the two pieces of rebar, with vegetation and woody debris moved as necessary to make the line as straight as possible. I then took a final compass reading to verify the initial reading, which I changed if necessary.

I marked points on the transect every 50 m, or less where such segment lengths were not possible or not needed. I marked these points in a similar manner to the others, using the compass bearing and the previous marker to site each new point. Where possible, I tied flagging tape to vegetation on or very close to the transect line to assist in the marking and future location of the transect. I stretched the tape measure from the last transect point in the appropriate compass direction, making periodic stops to check that the line was straight, until the end of the segment. I then drove the rebar and marker into the ground, and stretched the tape measure between this and the previous piece of rebar, making any necessary corrections to the line as above.

Each transect followed an elevation gradient from the present water level to a point that appeared high enough to remain dry at the maximum possible water level (11.0' or 3.35 m, NGVD 29 datum). I used a bench that had formed along the previous lake shoreline as a rough reference for this during initial installation, in places where it was present. The length of the transects thus varied with topography. Transects ranged from 21.5 m to 279.8 m in length; the average length was 102.8 m. In all, the transects totaled 3082.9 m.

After installing and measuring vegetation along 17 full transects, I began abbreviating transects. A band of regenerating willow forest surrounds most areas of the lakes; I included this in the full transects. Because I installed the remaining 13 transects late in the season, the water level in the lakes had risen to inundate the mudflats between the water and this willow forest, and the leaves had fallen off of the willows, making it impossible to get reliable percent cover data for these lower transect segments. Consequently, it was necessary to start transects that would have run through willow forest from the upland edge of the willows.

Transecting procedure

All transect monitoring took place during the dry part of the year. The initial round of monitoring began on October 20, 2003, and continued through December 3 of that year; I began abbreviating new transects to exclude the willow forest on November 20. I used the line intercept method (O'Neill 1999, Sharp 2002, Youngman 2002) to measure vegetation along each transect. Once the tape measure was in place, I recorded vegetation at all heights every 1 dm. I placed a piece of rebar or a straight stick next to the tape measure at each dm mark, and noted any live plant touching this on the data sheet (such that, if more than one plant of more than one species occupied a given point vertically, this method captured all vertical layers). If I could not identify a species that occurred on a transect in the field, I took a sample from off the transect when available. Very young—i.e., not past the cotyledon phase—or unrecognizable plants were noted as "unknown" or "unknown seedling."

Using the same procedure as above, I monitored the transects a second time, starting on September 9, 2004; I finished this round on November 8. To keep the data consistent, I did not extend the abbreviated transects.

Surveying Methods

I did all surveying with a Topcon AT-G2 autolevel (Precision Instruments, Portland, OR), tripod, and survey rod, following the methods outlined in Herubin (1982) for differential leveling and profile leveling. Elaine Stewart of Metro found information on seven benchmarks of known elevation around SBL, and converted all elevations to the NGVD 29 datum to match the datum used for water depth measurement at the water control structure. Four of the benchmarks were installed and maintained by Multnomah County; one was a piezometer installed by the staff of the St. Johns landfill; one was installed by the general contractor in charge of construction work being done on the north shore of Smith lake; and one was the staff gauge mounted on the water control structure, which had been calibrated to read the elevation of the lake level above mean sea level using the NGVD 29 datum.

Beginning on November 5, 2004, I ran level circuits to between one and three transects. I always closed the circuit to the original starting point, which was either one of the seven benchmarks or a known elevation on a transect, usually the upland end, established through my survey work. I took all surveying data to the nearest 0.005 ft (0.15 cm), and recorded any error of closure after double-checking all calculations. All errors of closure were positive; this was likely systematic error. It is possible that the swampy ground at SBL caused the survey level to sink slightly between backsight and foresight readings; since the latter are subtracted from instrument height in calculating turning point elevations, this would cause elevation readings to be consistently higher than actual.

Once I had established an elevation on a transect, I did profile leveling ("rod shots") to develop a topographical profile of the transect. After positioning the tape measure to define the line of the transect, I typically took rod shots at 3-m intervals. I occasionally deviated from this interval to establish elevations at each rebar marker on the transect, or because the terrain of the transect warranted longer or shorter intervals—e.g., abrupt changes in slope required a smaller increment, while long segments with little change in elevation allowed a larger increment to speed the work.

After noticing some discrepancies in the elevation data, I also ran level circuits from the staff gauge on the water control structure to four of the benchmarks to reconcile these differences. I used the results of these circuits to make necessary corrections to the elevation data.

Water level data

Elaine Stewart took water level readings from a staff gauge at the point where the North Slough arm of the Columbia Slough enters the lake system approximately once per week during 2003. The last reading before construction on the new water control structure began was on September 22, 2003; no readings were available from that date until December 1, 2003, when construction was completed. Weekly gauge readings resumed on that date, and continued through July 27, 2004. A few staff gauge readings were also taken in November and December of 2004. Additionally, Ducks Unlimited installed a pressure transducer next to the staff gauge on May 6, 2004. This recorded water level data at daily intervals. On August 6, 2004, the last of the stoplogs in the water control structure were removed, leaving the lakes subject to the tidal fluctuations of the North Slough. Daily mean levels for the North Slough were available from the USGS website (USGS 2005).

Response of Reed Canarygrass Phenology

Stand Selection

I defined a "stand" of reed canarygrass as a patch between 1 m and 5 m long where reed canarygrass appeared in each dm, based on the baseline transects done in fall 2003. Distinct stands had to be separated by at least 1 m. This definition yielded 87 stands from across the 30 transects. I normalized the transects so that each was spread over a distance equivalent to that of the longest transect to achieve a relatively even distribution of inundation depths among all transects, and selected 45 stands—representing 13 transects—for monitoring. I selected stands to represent different depths of inundation. Because access to the stands was by canoe for this portion of the study, it was necessary to select stands from transects that were fairly close together to be able to monitor all stands regularly. Thus, I selected stands meeting the above criteria from two clusters of transects—one on each lake. I was unable to find three of the selected stands in the field, and two other stands appeared to die off during the study; hence, the final analysis of phenology data encompassed 40 stands, representing 11 transects.

Stand location and installation

I located stands in the field using a 100-m tape to measure distance from the upland end of a transect and a Starter compass to determine direction. Access to the stands was via canoe. Distances and compass directions from the markers to the stands were already known, based on the transect data. Once I located the stands, I marked them with ½"-diameter PVC, which I cut in the field to a length that would leave at least one foot of pipe above the level of the water at the site I was marking.

Monitoring

I monitored the stands every two weeks, from May 19 to July 21, 2004; in all, each stand received four rounds of measurement. For the first two rounds, I took three replicate measurements for water depth and straightened plant height for each stand—i.e., I straightened plants to their full height to measure them. I averaged these measurements to determine stand height. Because variability of plant height within stands was greater than expected, later rounds of monitoring used six to ten replicates; I determined the exact number in the field, with replicate measurements made until the last measurement changed the average by less than 2 cm. I took all measurements to the nearest mm with a metric metal tape mounted on a 1"x 2" x 7' piece of wood.

During the last round of monitoring, I took data on plant height *in situ*—i.e., without straightening—before recording the straightened plant height. I later used the ratio of these two measurements as an indicator of the health of the stands (Northup and Nichols 1998). Additionally, I recorded the number of shoots per plant and the height of the tallest "live" (actively growing) shoot, and water depth, as before. I replicated all of these measurements six times per stand, and averaged these to give characteristics for the stand as a whole.

Statistical Methods: Abundance Response of Taxa

After I entered the transect data into Excel spreadsheets—representing the presence of a given taxon on a decimeter by a "1" in the cell—I calculated the percent cover for each taxon present on a given transect as the number of points (dm) where the taxon appeared, divided by the total number of points on the transect. I repeated this for all 30 transects for each of the two years.

I then performed a summary calculation of percent cover for each taxon by dividing the total number of counts for each taxon from all of the transects by the total length of the transects. I sorted the results from highest to lowest average percent cover and graphed them.

I computed diversity indices both on an individual transect level and overall, using the Shannon diversity index. I calculated p_i for each taxon on each transect as the number of counts for that taxon divided by the number of occupied decimeters—i.e., bare ground was eliminated from consideration—and then used these to find diversity (H') for each transect:

$$H' = - \Sigma\, p_i \ln (p_i)$$

I calculated overall diversity for both years using the same formula, but with p_i computed as the total number of counts for each taxon across all transects, divided by the total number of occupied decimeters for that year.

Analysis of changes in taxa

I used McNemar's test for significant changes (Zar 1974) to analyze differences in the percent cover of individual taxa between 2003 and 2004. I was particularly interested in the significance of the response of *Phalaris arundinacea,* other taxa covering significant portions of the study area, desirable native taxa, and potentially problematic invasive taxa other than *Phalaris.* I considered changes to be significant if the test yielded a p-value of 0.05 or less. I analyzed data from 27 of the 30 transects as a statistical representation of the plant community at Smith and Bybee Lakes. I eliminated the other three transects from the analysis because a legitimate comparison of the data on these transects was not possible between the two years. One of these transects ended in a revegetation project that was mowed during 2003, but not in 2004.

I monitored the other two at apparently different stages of the growing season; the first round of monitoring was in late November of 2003, while the second occurred in early September of 2004. The results were radically different, as an annual species (*Bidens cernua*) had become dominant on both transects in 2004; however, informal observation of the transects later showed that *P. arundinacea* had returned to dominant status after *B. cernua* died back. Thus, the data I took for 2004 for these transects did not reflect the same state of the plant community as those taken in 2003. All of the statistical analysis I report here is based on the 27 remaining transects.

I also examined the effects of flooding on reed canarygrass in the willow forest portions of the transects to determine whether the response of *P. arundinacea* differed in this habitat from its overall response. Using the chi-squared test, I compared the net loss of reed canarygrass in the willow forest to the total net change on the 17 full-length transects. I considered the influence of the willow forest as extending to: 1) points on the transect where willow was present; and 2) a two-meter band to the north or south of a willow patch—depending on the aspect of the transect—corresponding to the maximum extent of shading. I based maximum shading on the assumption of regenerating willow patches being two meters tall and an estimated growing-season solar angle of 45°. I applied this only to the regenerating willow forest, not to upland forested areas, and considered only those areas affected by willow forest in both 2003 and 2004. I assumed that a p-value of 0.05 or less indicated a significant difference. I then repeated this analysis for the portions of the 17 transects located in the elevation zone that supports regenerating willow forest, which I defined as all decimeters between the lowest and highest elevations where young willow was present in both years of the study.

Lastly, I examined the interaction between reed canarygrass and other taxa. By subtracting the 2004 transect data from the 2003 data using arrays, I determined the change (if any) of each taxon on each decimeter on all 27 transects included in the analysis. I then analyzed the response of all taxa on decimeters where reed canarygrass was eliminated, counting the number of times each taxon took the place of *Phalaris*; I also determined the number of times bare ground replaced *Phalaris*. I performed the same analysis on those decimeters where reed canarygrass, having been absent in 2003, appeared in 2004. I used McNemar's test for significant differences to determine whether a given taxon displaced, or was displaced by, reed canarygrass, considering p-values of 0.05 or less to be significant.

Determination of depth and duration of flooding

Using a Visual Basic script in Excel, I interpolated elevation values for each decimeter, assuming a linear trend between data recorded in the field. I calculated maximum depth of inundation at each decimeter by subtracting its elevation from the maximum water depth recorded at the water control structure for 2004.

I used data from the staff gauge for September 22, 2003, and for the period from December 1, 2003 through May 5, 2004, assuming a linear change between readings, to interpolate daily water levels at SBL. From May 6 to June 21, 2004, I used the daily transducer data; data past this date exhibited discrepancies when compared to the staff gauge readings, so I reverted to using gauge readings and interpolating daily values through August 6, 2004. On this date, I began using USGS data on mean water levels in the North Slough at Portland, which I converted to NGVD 29, to represent lake levels for the remainder of the water year. Using these data and the interpolated elevations, I estimated the duration of flooding at each decimeter on the transects.

Impact of flooding and willows on reed canarygrass response

I then used multinomial logistic regression to determine the relationship between several flooding variables and the response of reed canarygrass. Specifically, I considered maximum inundation, duration of inundation, and the presence or absence of willows or willow shading in the regression. I used Systat 11 (Systat Software Inc., Richmond, CA) to regress these against the change in reed canarygrass for each decimeter from 2003 to 2004, assigning the label "4" to those decimeters where reed canarygrass disappeared (the reference case in the regression), "3" to decimeters where there was no reed canarygrass in either year, "2" to decimeters where reed canarygrass was present in both years, and "1" to decimeters where reed canarygrass was absent in 2003 but present in 2004. I performed this analysis on 26 of the original

30 transects, eliminating the three transects referred to above as well as one transect for which the elevation data were not reliable, owing to very high error of closure (0.9 ft) on the level circuit I used to determine its elevation relative to benchmark.

To assure that errors of closure from surveying did not change the results of the analysis, I also performed a duplicate analysis that incorporated these errors. Since all errors were positive—i.e., observed elevations were likely higher than actual—I added all errors applicable to a transect to the maximum inundation of each decimeter on that transect, and ran the same regression to compare the results. Because these results were not materially different, I report only the original analysis.

Statistical Methods: Response of Reed Canarygrass Phenology

After ranking the growth and inundation data from the stand measurements, I used a nonparametric correlation coefficient—Spearman's Rho—to look for relationships between each of the variables described above and water depth. I used a t-test to determine whether or not these relationships were significant.

Results

Response of Abundance of Taxa

Diversity

The baseline data from the initial round of transecting indicate that *P. arundinacea* had a strong impact on diversity at SBL. Percent cover of reed canarygrass showed a strong, negative correlation with Shannon diversity (Spearman's rho = 0.69; p < 0.001), as seen in Figure 3.

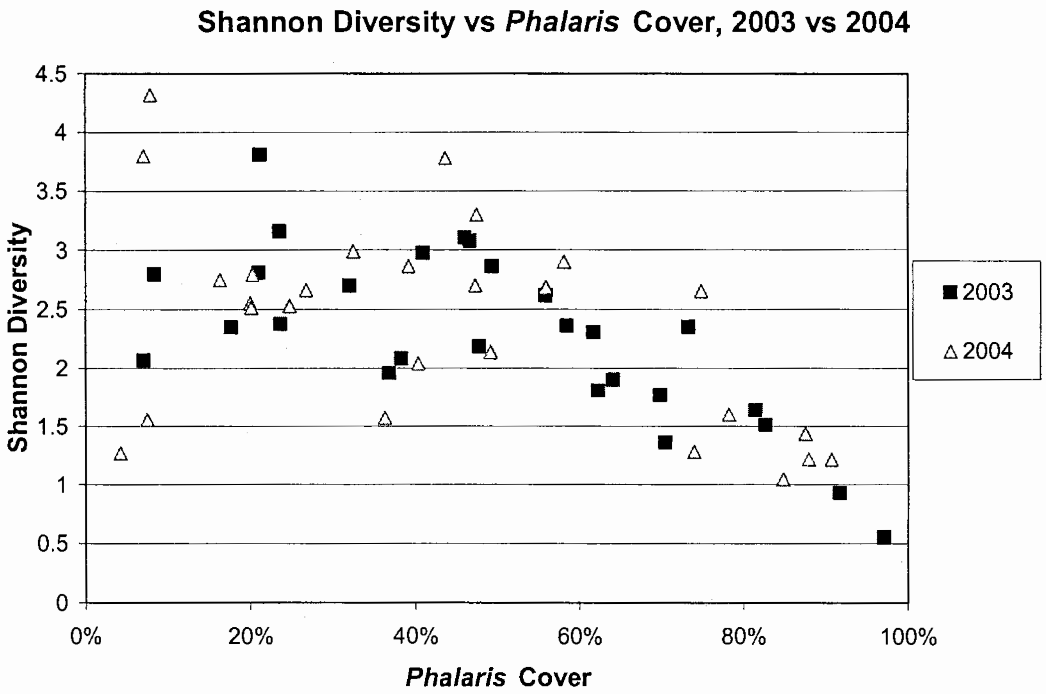


Figure 3. Relationship of Shannon diversity index to percent cover of reed canarygrass at Smith and Bybee Lakes in fall 2003 and fall 2004. Each data point represents a measured transect in the study.

This relationship was less strong, though still significant, in 2004; Spearman's rho = 0.41, $p < 0.05$. The overall Shannon Diversity index (H') for the transects in 2003 was 3.06. Transect data from 2004 show a small but significant ($p < 0.001$) increase in H' to 3.18.

Table 2. Transect lengths, *Phalaris* cover, cover for the next most dominant taxon (or most dominant, if *Phalaris* was not dominant), and Shannon Diversity for 2003 and 2004.

Transect	Length (dm)	% Ph ar 2003	% Cover 2003 (next dom sp.)	2003 H'	% Ph ar 2004	% Cover 2004 (next dom sp.)	2004 H'
43	778	41.9	20.6 (Sa lu)	1.81	40.4	32.3 (Sa lu)	2.04
178	1264	39.2	12.4 (Sa lu)	2.86	39.2	31.2 (Lu pa)	2.86
398	477	69.2	23.5 (Lu pa)	1.77	78.2	35.0 (Lu pa)	1.6
937	1202	32.5	22.6 (Po sp)	3.10	26.8	39.8 (Po sp)	2.66
1121	801	6.7	43.3 (Lu pa)	2.80	7.1	51.9 (Po sp)	3.80
1441	1113	36.1	61.5 (Bi sp)	1.96	36.3	60.6 (Bi sp)	1.57
1692	1681	20.1	45.2 (Lu pa)	2.81	20.0	46.0 (Bi sp)	2.55
1864	875	15.3	48.7 (Lu pa)	2.35	20.1	41.5 (Po sp)	2.51
2027	2798	66.3	20.3 (Sa lu)	2.35	55.9	32.0 (Bi sp)	2.69
2075	776	28.6	33.4 (Po sp)	2.70	32.5	58.1 (Po sp)	2.99
2339	1000	13.6	22.3 (Bi sp)	3.81	7.9	32.2 (Ve sp)	4.32
2812	743	82.6	14.8 (Po sp)	1.51	84.8	33.5 (Po sp)	1.04
3248	1000	6.9	63.7 (Lu pa)	2.07	7.5	81.7 (Po sp)	1.55
3395	756	22.3	65.2 (Po sp)	2.38	4.2	82.4 (Po sp)	1.27
3830	739	69.1	25.6 (Po sp)	1.36	73.9	28.8 (Po sp)	1.28
3851	387	41.9	26.1 (Lo co)	3.08	47.5	33.1 (Ve sp)	3.30
4373	1000	64.0	42.5 (Er hy)	1.90	87.5	21.3 (Po sp)	1.43
4514	621	48.1	18.5 (Lu pa)	2.36	47.3	28.0 (Lu pa)	2.70
4880	658	32.2	45.7 (Lu pa)	2.08	24.8	63.1 (Po sp)	2.53
4886	429	45.7	47.8 (Lu pa)	2.19	49.2	62.9 (Po sp)	2.14
5277	215	91.6	11.6 (Lu pa)	0.93	87.9	15.3 (Lu pa)	1.21
5310	379	56.5	16.9(My aq)	2.31	58.0	35.1 (Ve sp)	2.90
5769	334	48.5	26.3 (Bi sp)	2.62	43.7	45.8 (Ve sp)	3.78
5790	2097	33.4	42.0 (Po sp)	2.97	20.3	52.6 (Po sp)	2.79
5861	1500	97.0	3.5 (Po sp.)	0.55	90.7	7.5 (Bi sp)	1.21
6038	500	80.6	13.2 (Er hy)	1.64	74.8	29.6 (Bi sp)	2.66
6478	619	21.6	35.9 (Cy st)	3.16	16.3	71.6 (Ve sp)	2.75

Table 2 summarizes the changes on the individual transects represented in Figure 3. In most cases, a decrease in *Phalaris* cover coincided with an increase in diversity; exceptions to this were cases where another taxon—e.g., *Polygonum* species—had become dominant on the transect, or had substantially increased in percent cover. Conversely, an increase in reed canarygrass cover typically produced a decrease in diversity, except on transects that changed dominant or sub-dominant species from 2003 to 2004.

Percent Cover

The data show that P. *arundinacea* was the dominant species at SBL, with 43.7% cover in 2003. Other important species included water purslane (*Ludwigia palustris*), smartweeds (*Polygonum* spp.), beggar's tick (*Bidens* spp,), Pacific willow (*Salix lucida*), false nutsedge (*Cyperus strigosus*), creeping lovegrass (*Eragrostis hypnoides*), *Veronica* species, ovoid spikerush (*Eleocharis ovata*), and golden dock (*Rumex maritimus*). I grouped the *Polygonum* species at the genus level, since the first round of monitoring was too late in the 2003 growing season to permit identification of individual species. The same was true for *Bidens, Veronica, Carex* and *Solanum* species. I ignored another common species, *Azolla mexicana*, for purposes of this study, as it is an aquatic plant that was left on the soil surface as the water in the lakes receded during the growing season. No other species averaged more than 5% cover across the 30 transects.

The first year of flooding at SBL caused some change to the plant community there. Overall, *P. arundinacea* cover decreased by 2.5 percentage points; other plant taxa underwent more substantial changes. Figure 4 shows the changes in percent cover for the most common taxa at SBL from 2003 to 2004.

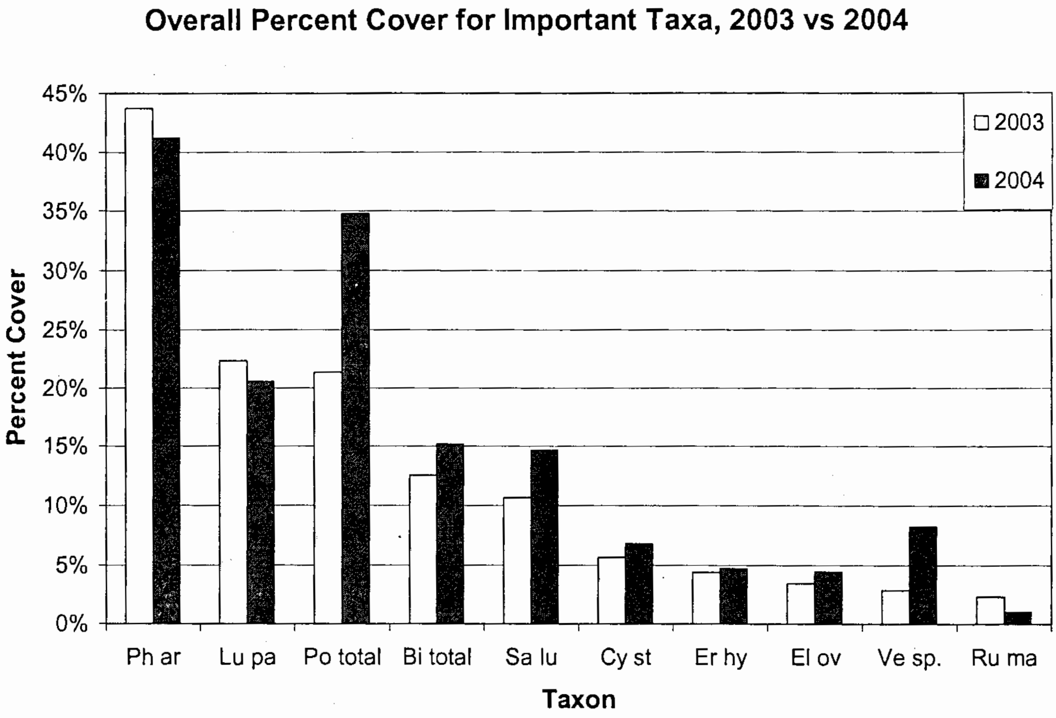


Figure 4. Percent cover for the 10 most common taxa at Smith and Bybee Lakes in 2003 and 2004. "Ph ar" = *Phalaris arundinacea*; "Lu pa" = *Ludwigia palustris*; "Po total" = all *Polygonum* species; "Bi total" = all *Bidens* species; "Sa lu" = *Salix lucida*; "Cy st" = *Cyperus strigosus*; "Er hy" = *Eragrostis hypnoides*; "Ve sp." = *Veronica* species; "El ov" = *Eleocharis ovata*; "Ru ma" = *Rumex maritimus.*

Also of interest was the change in percent cover for potentially problematic invasive species other than reed canarygrass, shown in Figure 5. With the exception of *Solanum* species, *C. vulgare*, and *L. salicaria*, percent cover for these taxa decreased or was not appreciably different.

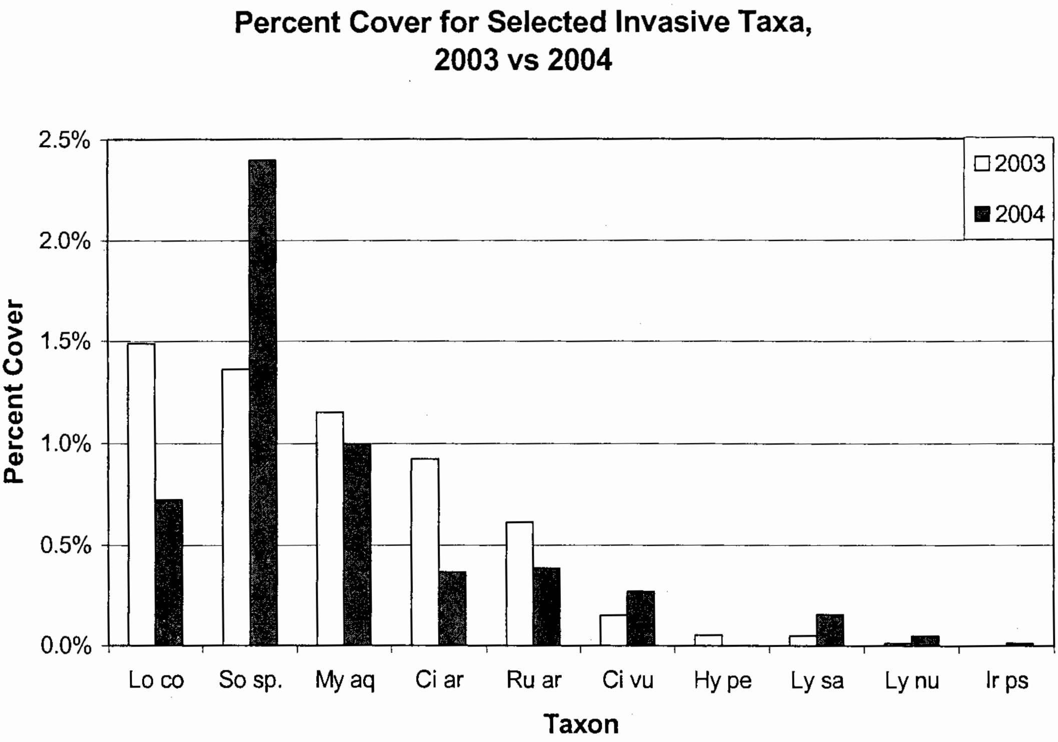


Figure 5. Comparison of percent cover of selected invasive species in 2003 and 2004 at SBL. "Lo co" = *Lotus corniculatus*; "So total" = all *Solanum* species; "My aq" = *Myriophyllum aquaticum*; "Ci ar" = *Cirsium arvense;* "Ru ar" = *Rubus armeniacus*; "Ci vu" = *Cirsium vulgare*; "Hy pe" = *Hypericum perforatum*; "Ly nu" = *Lysimachia nummularia*; "Ly sa" = *Lythrum salicaria*; "Ir ps" = *Iris pseudacorus.*

The response of comparatively rare, desirable species was generally positive; this was especially true for *Carex* species, *Leersia oryzoides*, and *Myosotis laxa*. Figure 6 shows these changes.

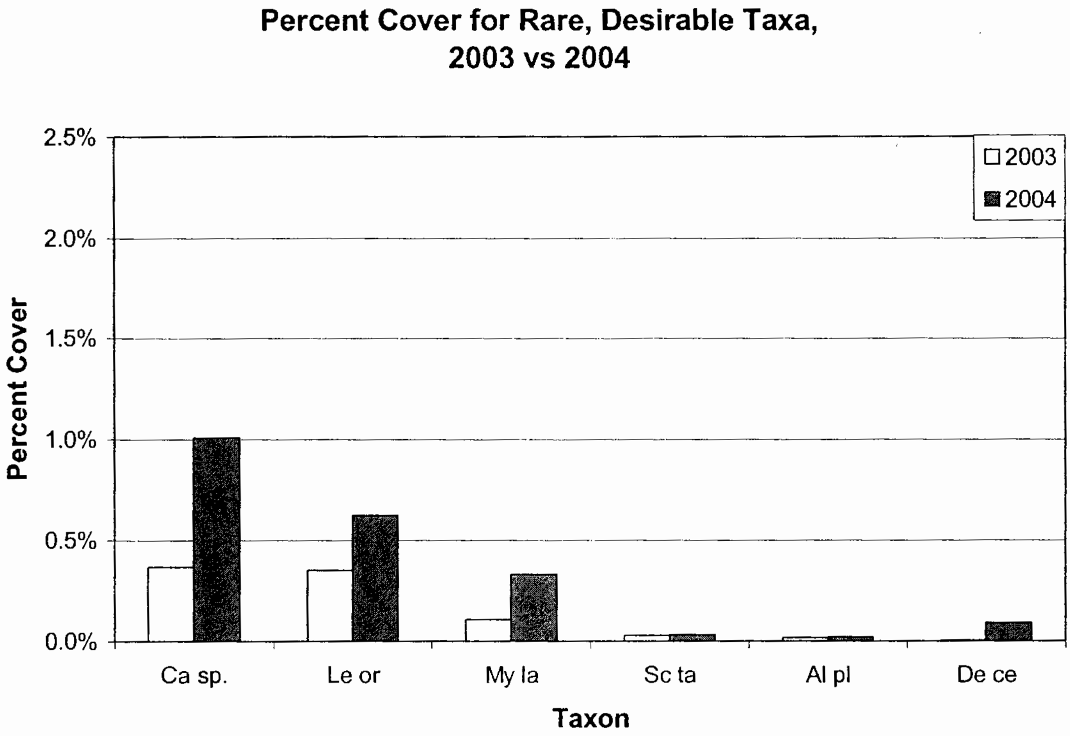


Figure 6. Comparison of percent cover for low-frequency, desirable native species at SBL, 2003 vs. 2004. "Ca sp." = *Carex* species; "Le or" = *Leersia oryzoides*; "My la" = *Myosotis laxa*; "Sc ta" = *Scirpus tabernaemontanii*; "Al pl" = *Alisma plantago-aquatica*; "De ce" = *Deschampsia cespitosa*.

Significance of Changes

Table 3 summarizes the statistical significance of the changes observed in the graphs above. Changes for all nearly all taxa were significant; the small increases in *E. hypnoides*, *I. pseudacorus*, *S. tabernaemontanii* and *A. plantago-aquatica*, and the decrease in cover of *M. aquaticum*, were the only exceptions.

Table 3. Summary of significance of changes in taxa of interest from 2003 to 2004 at Smith and Bybee Lakes.

Category	Species	Δ % cover	p-value	Significant?
Important Taxa	*P. arundinacea*	-2.5%	< 0.001	Yes
	L. palustris	-1.8%	< 0.001	Yes
	Polygonum spp.	+13.4%	< 0.001	Yes
	Bidens spp.	+2.6%	< 0.001	Yes
	S. lucida	+4.0%	< 0.001	Yes
	C. strigosus	+1.2 %	< 0.001	Yes
	E. hypnoides	+0.3 %	>0.05	No
	Veronica spp.	+5.3 %	< 0.001	Yes
	E. ovata	+1.0 %	< 0.001	Yes
	R. maritimus	-1.3 %	< 0.001	Yes
Other invasive taxa	*L. corniculatus*	-0.8 %	< 0.001	Yes
	Solanum spp.	+1.0 %	< 0.001	Yes
	M. aquaticum	-0.2 %	>0.05	No
	C. arvense	-0.6 %	< 0.001	Yes
	R. armeniacus	-0.2 %	< 0.001	Yes
	C. vulgare	+0.1 %	<0.025	Yes
	H. perforatum	-0.1 %	< 0.001	Yes
	L. salicaria	+0.1 %	< 0.001	Yes
	L. nummularia	+0.04%	<0.05	Yes
	I. pseudacorus	+0.01 %	>0.05	No
Uncommon native taxa	*Carex* spp.	+0.6 %	< 0.001	Yes
	L. oryzoides	+0.3 %	< 0.05	Yes
	M. laxa	+0.2 %	< 0.001	Yes
	S. tabernaemontanii	+0.004%	>0.05	No
	D. cespitosa	+0.09 %	< 0.001	Yes
	A. plantago-aquatica	+0.004%	>0.05	No

Willow forest

Flooding in areas of regenerating Pacific willow forest suppressed *P. arundinacea* to a markedly greater degree. Reed canarygrass was significantly ($p < 0.001$) less likely to occur in willow forest than it was on the transects as a whole; overall percent cover on the 17 full-length transects in 2003 was 42.4%, while cover in the willow forest was 32.7%.

The decrease in reed canarygrass cover in willows was, by contrast, significantly higher than the overall change ($p < 0.001$). Reed canarygrass cover decreased by 4.9% (to 37.8%) on the 17 full-length transects overall. In willow forest portions of these transects, this decrease was more than doubled, as reed canarygrass cover dropped to just under 22%, a loss of 10.8% (Figure 7).

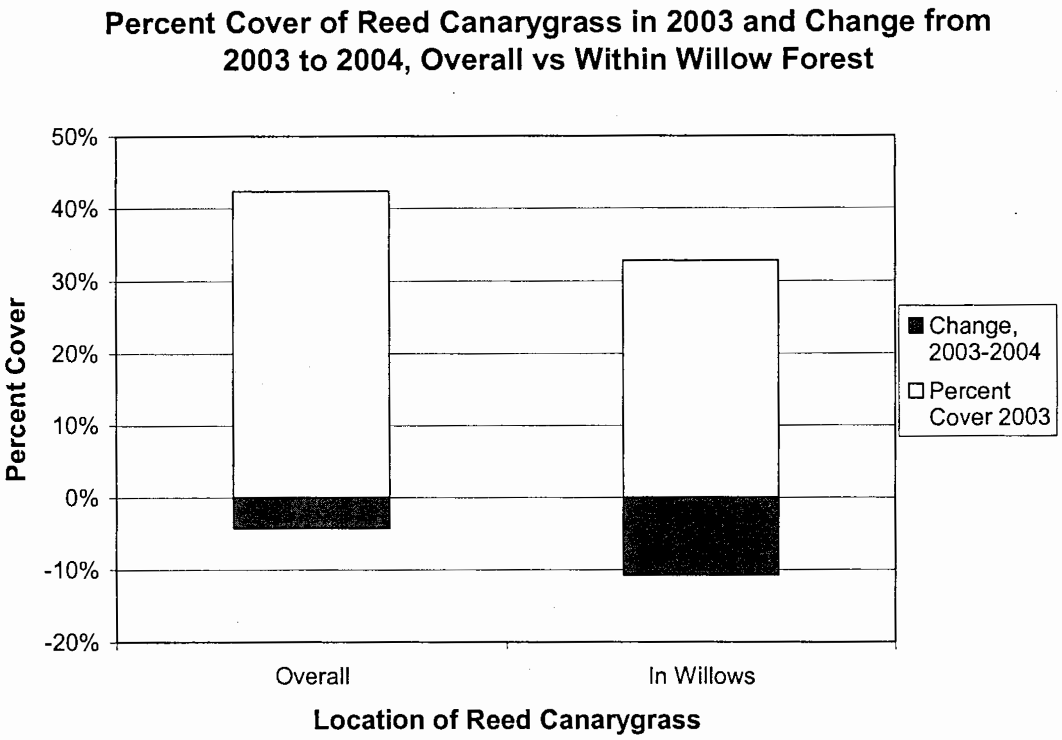


Figure 7. Reed canarygrass percent cover in 2003 and change from 2003 to 2004, in regenerating willow forest vs. overall for 17 full-length transects.

The distribution of reed canarygrass within the elevation zone that supported regenerating willow forest showed no significant relationship ($p > 0.05$) to the presence or absence of willow; percent cover with or without willow canopy was 32.3%. However, the influence of willow cover did have a significant impact on the change in reed canarygrass cover, which dropped by 3.4% in open areas but by 10.7% under canopy ($p < 0.001$) (Figure 8).

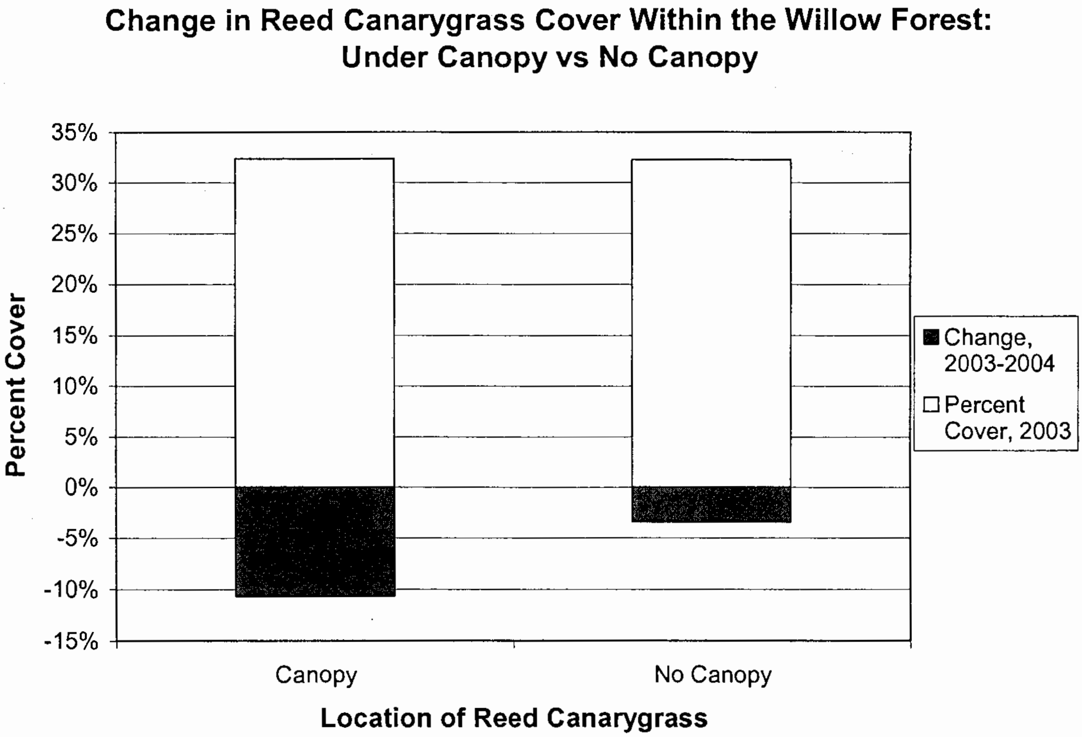


Figure 8. Reed canarygrass percent cover in 2003 and change from 2003 to 2004 in the elevation zone that supported regenerating willow forest; canopy influence vs. no canopy influence.

Interaction of reed canarygrass with other taxa

The suppression of reed canarygrass had the greatest benefit for native species that already had significant percent cover at SBL. *Polygonum* and *Bidens* species, as well as *S. lucida*, *L. palustris*, and *Veronica* species, filled many of the vacancies left by reed canarygrass. The two *Solanum* species also benefited.

Conversely, reed canarygrass out-competed some taxa, both native and invasive, under the new water regime. *E. hypnoides* and the two *Eleocharis* species were displaced by reed canarygrass, as were the invasive species *L. corniculatus* and *C. arvense.* Figure 9 summarizes these results for a selection of native and invasive taxa.

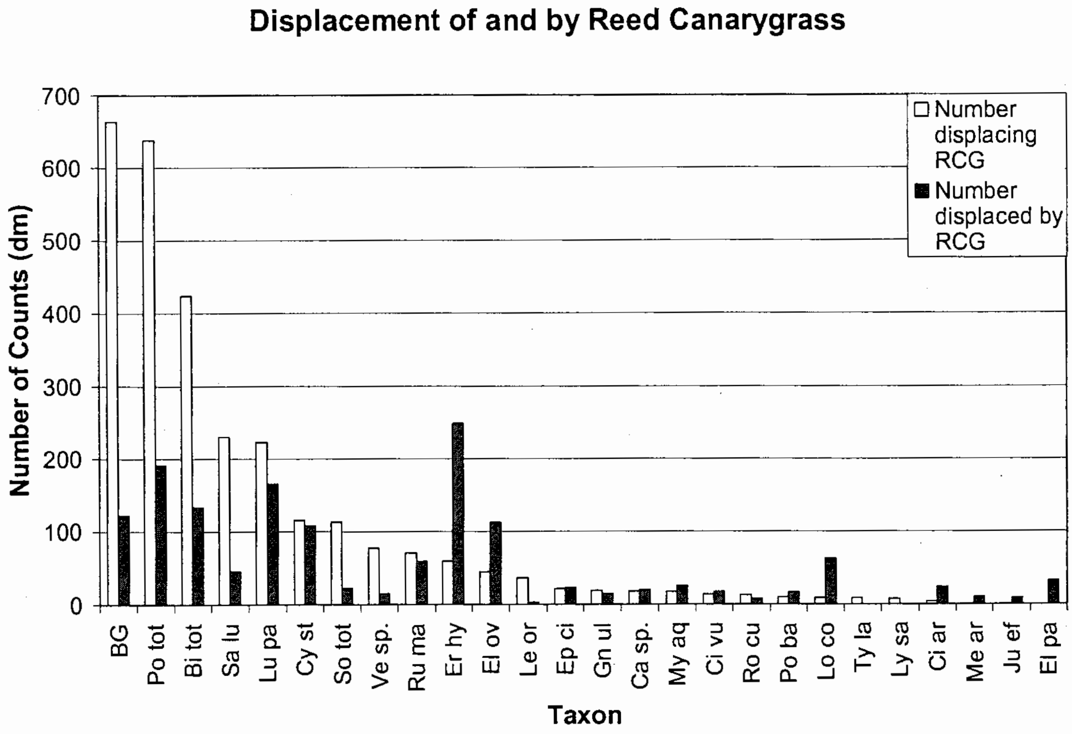


Figure 9. Interactions between reed canarygrass and a mix of native and invasive taxa at SBL. "BG" = Bare ground; "Ep ci" = *Epilobium ciliatum*; "Gn ul" = *Gnaphalium uliginosum*; "Ro cu" = *Rorippa curvisiliqua*; "Po ba" = *Populus balsamifera*; "Ty la" = *Typha latifolia*; "Me ar" = *Mentha arvensis*; "Ju ef" = *Juncus effusus*; "El pa" = *Eleocharis palustris*; other species abbreviations are as in Figures 4, 5 and 6.

Depth vs. duration

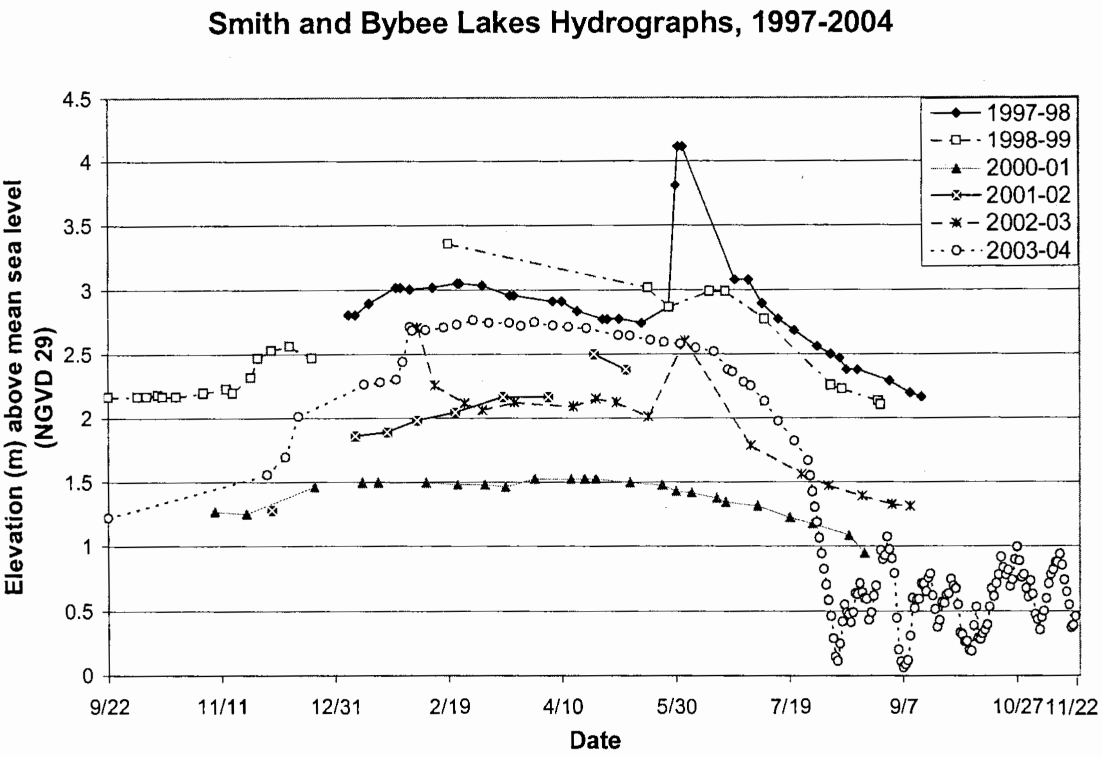


Figure 10. Hydrographs of Smith and Bybee Lakes for 1997-2004. Data for all years except 2003-2004 are based solely on staff gauge readings.

Figure 10 shows hydrographs for Smith and Bybee Lakes for the past several years. These indicate that, prior to the year of this study, lake levels had been dropping since 1999. Moreover, there was little fluctuation in lake levels during the water year. With the completion of the new water control structure, the level of the lakes during winter and spring was noticeably higher, while summer drawdown produced much greater variation in surface elevation.

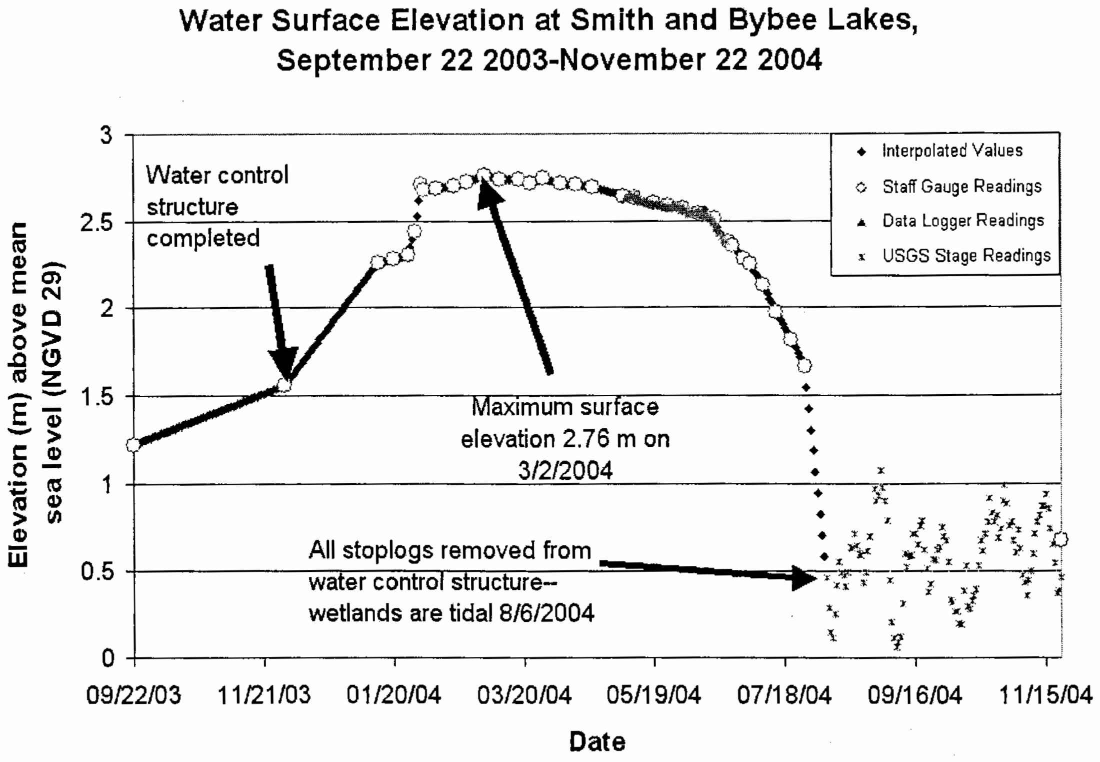


Figure 11. Hydrograph of Smith and Bybee Lakes from September 22, 2003 to November 22, 2004. Daily interpolated values are based on a linear interpolation between staff gauge readings.

Figure 11 shows the hydrograph of the 2003-2004 water year in greater detail. The highest recorded surface elevation was 2.76 m, recorded on March 2, 2004. I used this elevation to determine the maximum inundation for each decimeter of all transects. Using interpolated values for water level, I estimated the duration of inundation for each decimeter; this ranged from 0 to 305 days. It is likely that the actual lake levels during fall 2003 stayed fairly constant, except in response to storms, since the surface connection between the lakes and the Columbia Slough was interrupted by the construction of the water control structure. Once all the stoplogs had been removed from the water control structure on August 6, 2004, the lakes were subject to tidal fluctuation. Because the highest tide was below the lowest elevation on the transects, this did not affect duration of flooding.

There was a very strong relationship between maximum inundation and duration of flooding; the plot of ln (maximum inundation) vs. ln (duration) is close to linear (Spearman's rho = 0.9997; $p < 0.001$) (Figure 12).

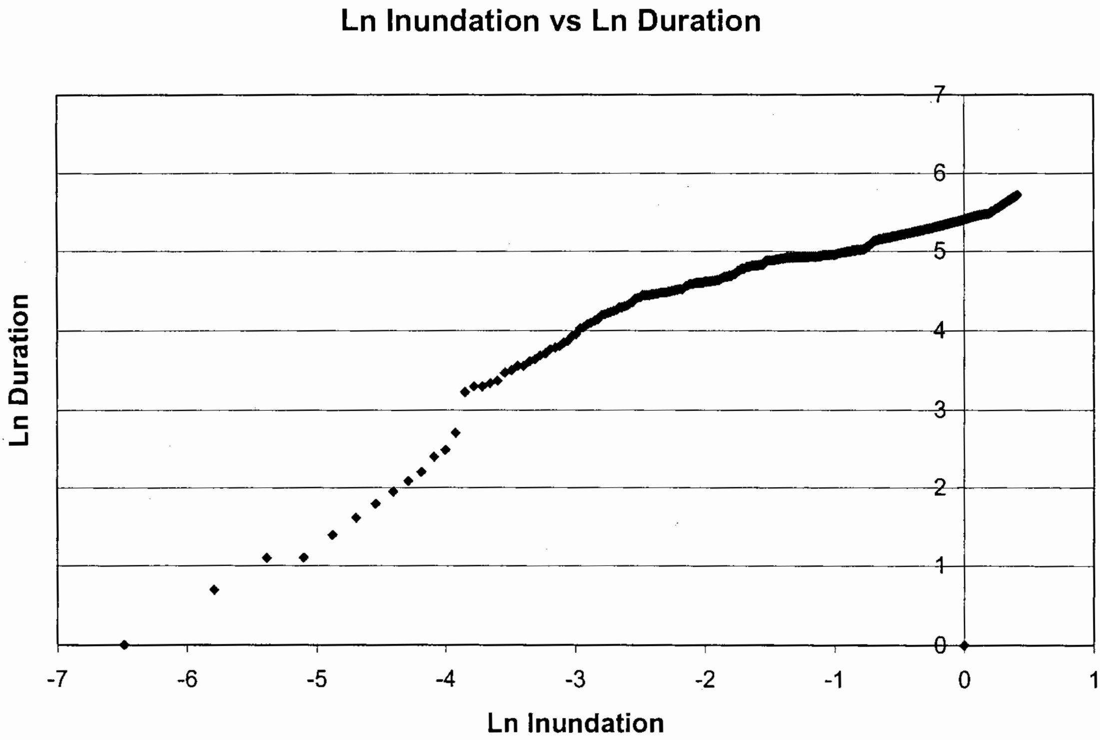


Figure 12. Plot of ln inundation vs. ln duration for flooding at SBL.

Because depth and duration were very closely related, and because the duration data are based largely on interpolated water level values, I have only included depth of inundation as a flooding variable in the remainder of the analysis.

Impact of flooding and willow cover on reed canarygrass

The presence-absence response of reed canarygrass showed a significant relationship with degree of inundation. Increased inundation was more likely to produce a decrease in reed canarygrass than it was to yield an increase or no change in existing reed canarygrass. Greater flooding was more likely to be associated with locations that had no reed canarygrass in either year, indicating that this species was less common in areas closer to the dry-season lake shore; historic hydrologic patterns would have kept these areas under significant inundation, thus making reed canarygrass unlikely to grow there. In 2003, deeply flooded (> 0.85 m) areas had 33.5% reed canarygrass cover, significantly ($p < 0.001$) less than the overall cover for the study area.

The presence or absence of regenerating willow forest was also a significant factor in the response of reed canarygrass. Decimeters that were influenced by willows were less than half as likely to see an increase in reed canarygrass, and less than one-third as likely to fall into either of the no-change categories, as they were to undergo a decrease in reed canarygrass. Odds ratios, with 95% confidence limits, are given in

Table 4. The odds ratios indicate whether an outcome is more (greater than 1) or less (less than 1) likely, given an increase in the independent variable; ratios different from "1" indicate that the independent variable significantly affects the dependent variable.

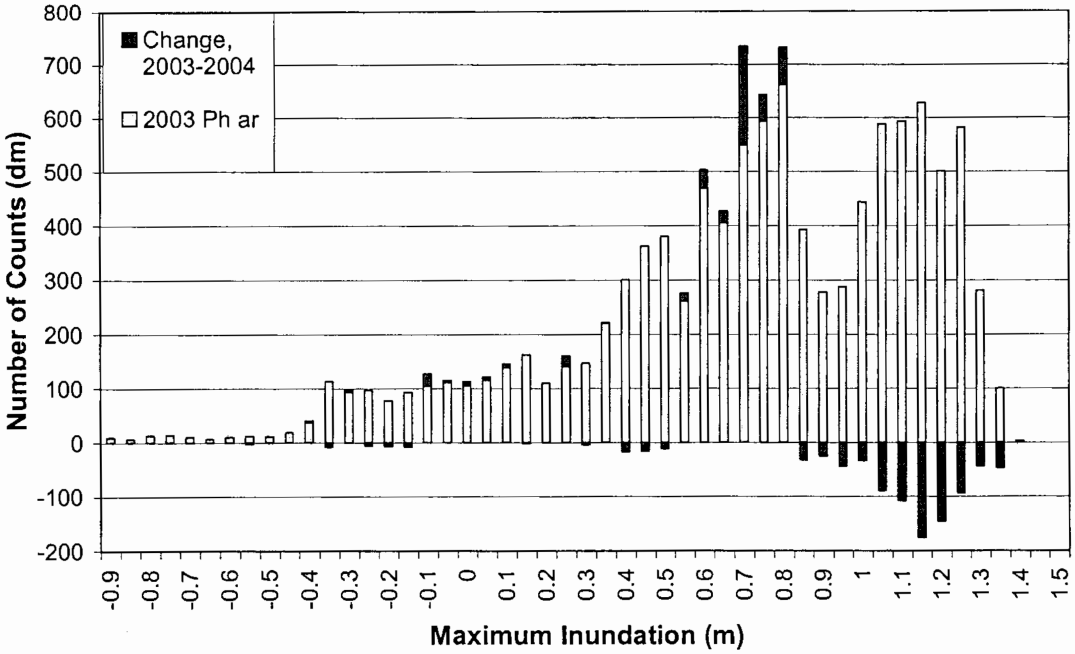
Table 4. Odds ratios and 95% confidence limits for multinomial logistic regression. Variables included were maximum inundation and presence/absence of regenerating willow forest. Case 1 = increase in reed canarygrass; Case 2 = reed canarygrass present both years; Case 3 = reed canarygrass absent both years. Reference case is disappearance of reed canarygrass. McFadden's Rho-squared = 0.102. All odds ratios, and the model as a whole, are significant; $p < 0.001$.

Variable	Case	Odds Ratio	Upper Bound	Lower Bound
Max. Inundation	1	0.800	0.846	0.756
	2	0.715	0.747	0.684
	3	1.719	1.802	1.639
Willow present	1	0.460	0.562	0.377
	2	0.319	0.366	0.277
	3	0.292	0.330	0.258

Taxon responses by degree of inundation

A higher degree of flooding generally led to a positive growth response in the native plants of the emergent zone at SBL, while causing a decrease in reed canarygrass. The following series of graphs (Figure 13 a-k) illustrates this. Of particular note here is the response of reed canarygrass, which generally expanded its cover in areas flooded less than 0.85 m, but lost cover in areas with more flooding. This result suggests a "threshold" value for the effectiveness of flooding for this species. Reed canarygrass cover dropped by 6.07% in these areas; this loss was significantly ($p < 0.001$) more than the overall loss of this species during the study.

Distribution and Change of *Phalaris arundinacea* by Inundation Category (a)



Distribution and Change of *Polygonum* Species by Inundation Category (b)

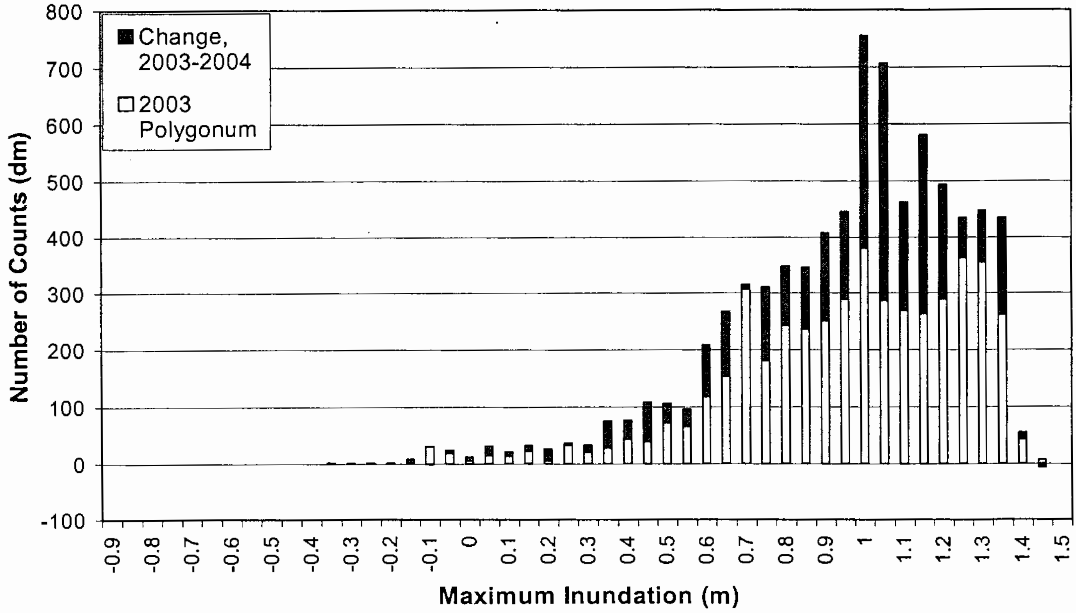
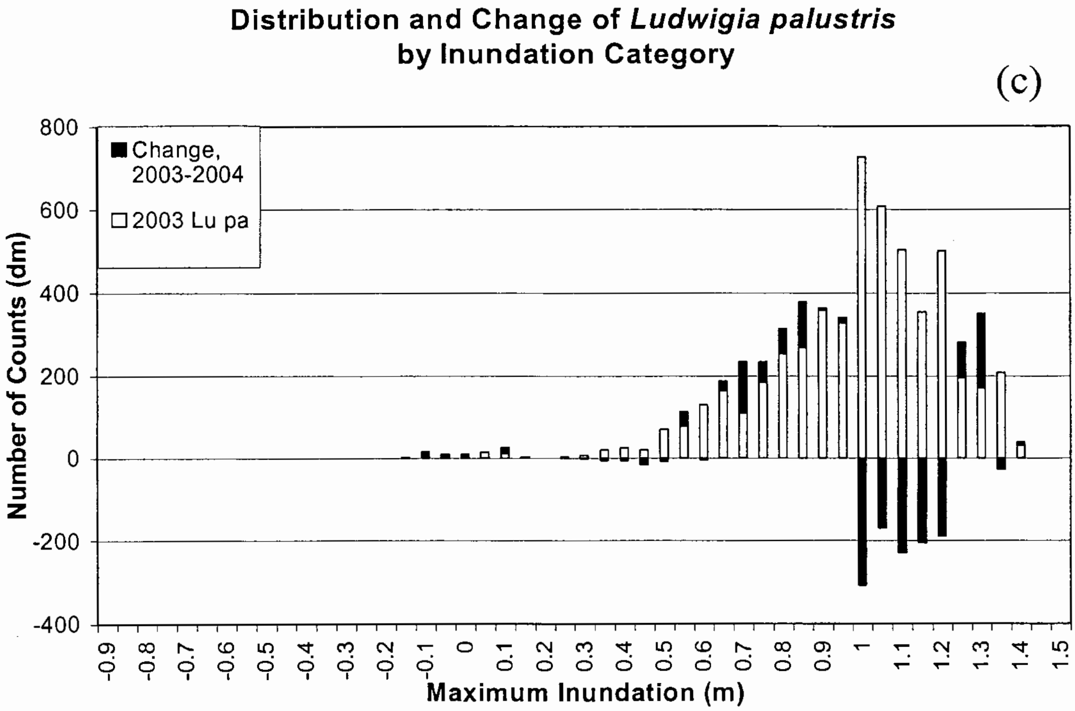


Figure 13 a-b. Response to flooding of *Phalaris arundinacea* and *Polygonum* species by inundation category. Negative numbers on the x-axis are vertical distance above maximum water level.



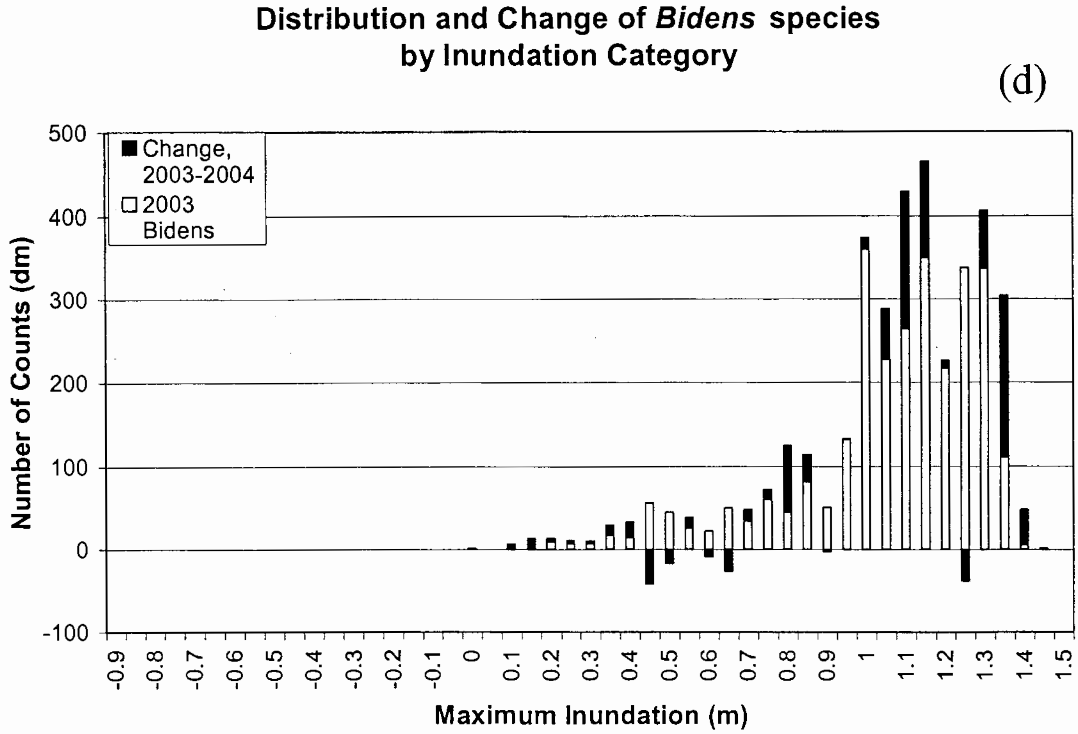


Figure 13 c-d. Response to flooding of *Ludwigia palustris* and *Bidens* species by inundation category. Negative numbers on the x-axis are vertical distance above maximum water level.

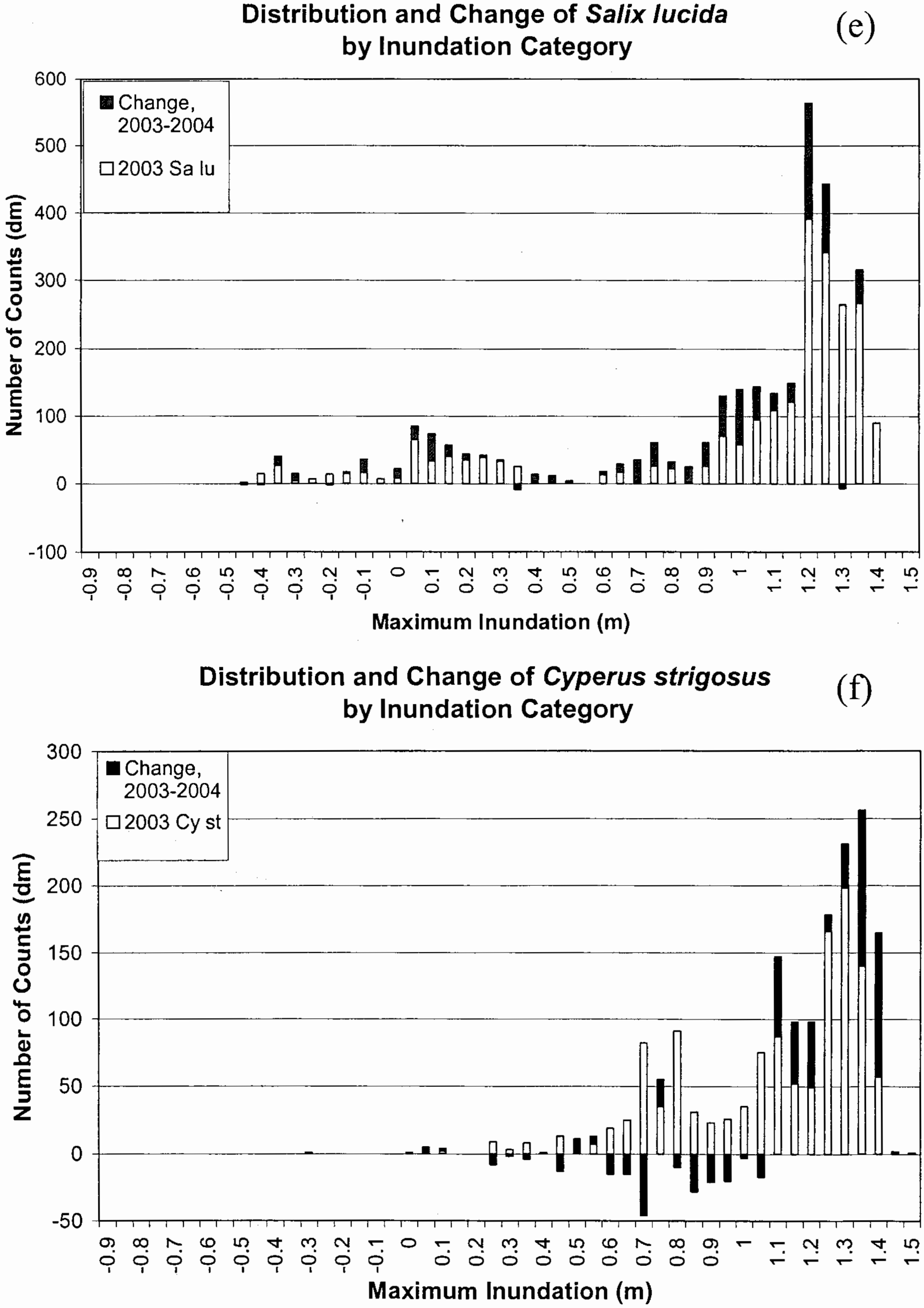
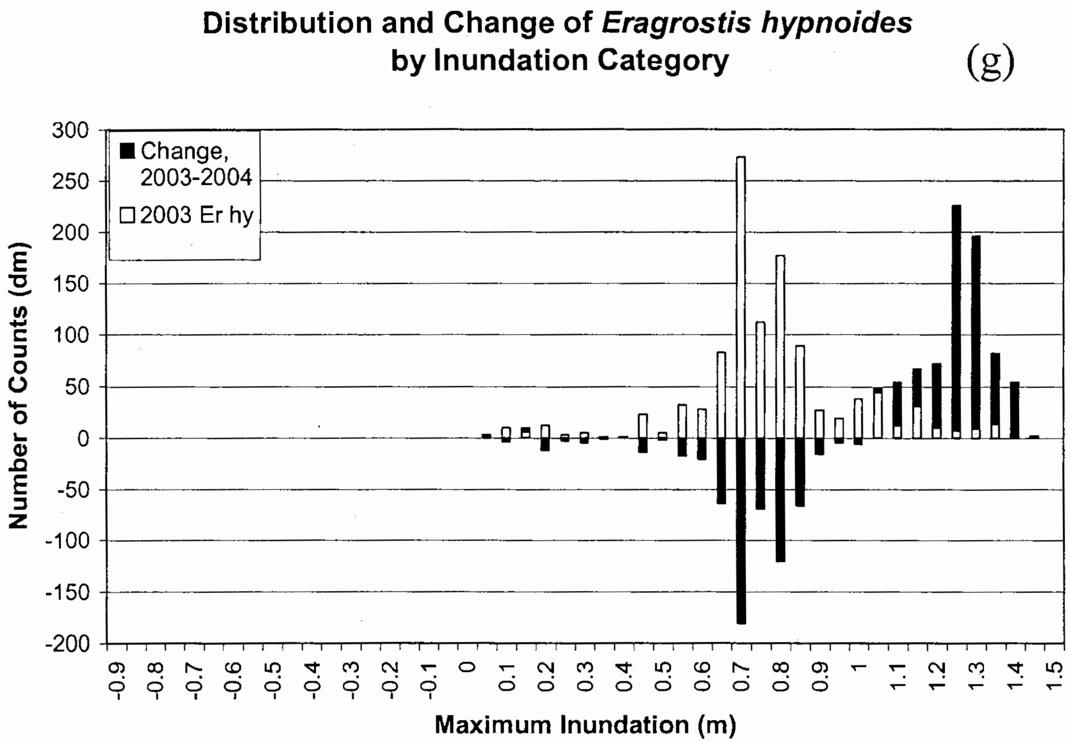


Figure 13 e-f. Response to flooding of *Salix lucida* and *Cyperus strigosus* by inundation category. Negative numbers on the x-axis are vertical distance above maximum water level.



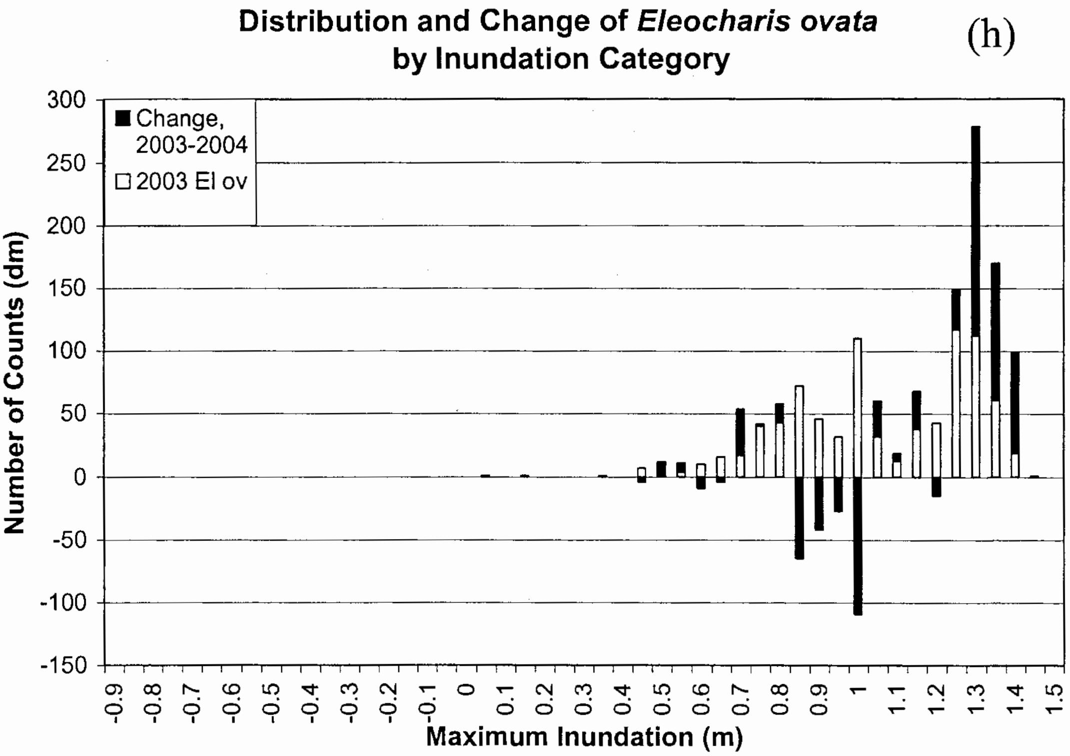
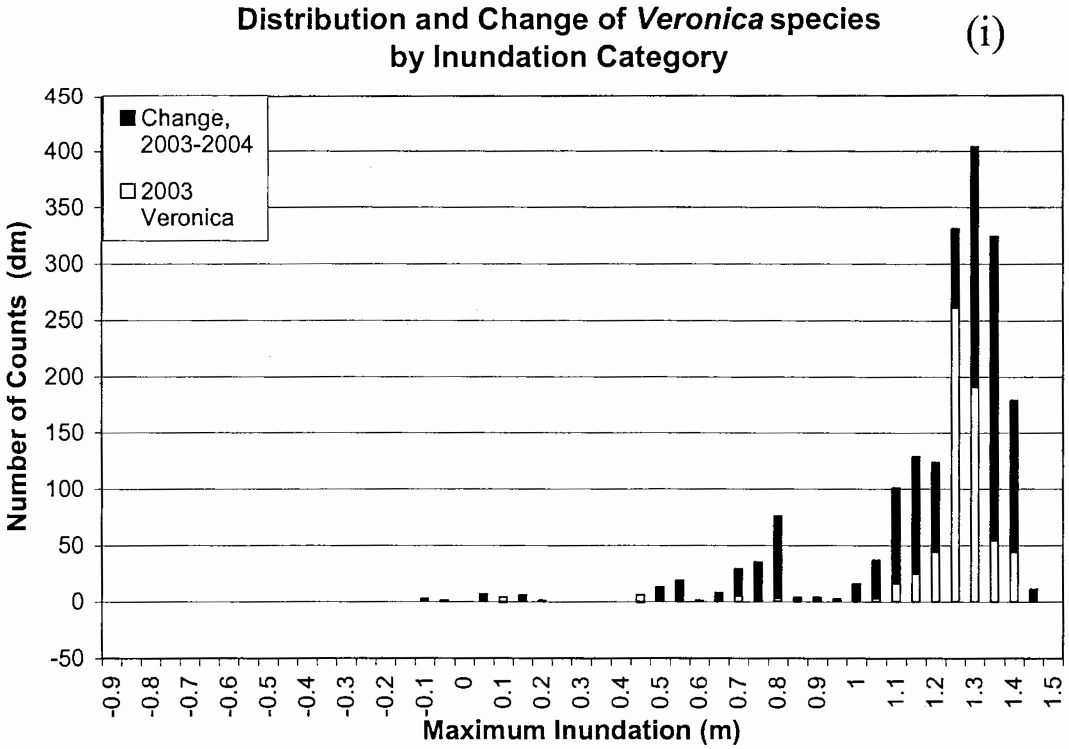
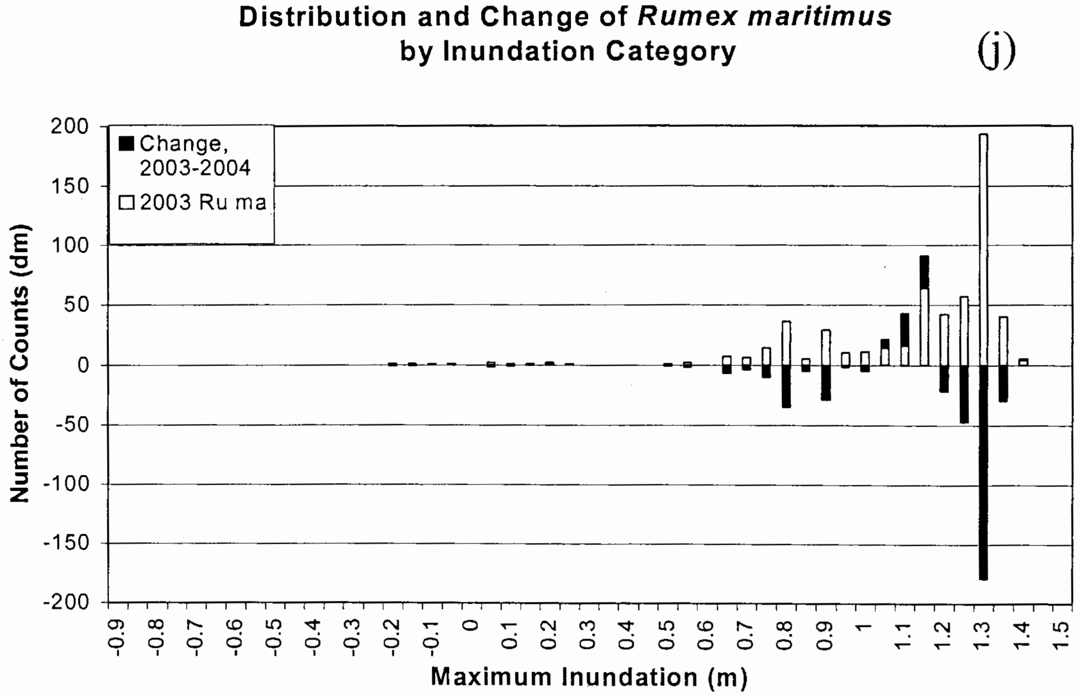


Figure 13 g-h. Response to flooding of *Eragrostis hypnoides* and *Eleocharis ovata* by inundation category. Negative numbers on the x-axis are vertical distance above maximum water level.


Distribution and Change of *Veronica* species by Inundation Category

(i)


Distribution and Change of *Rumex maritimus* by Inundation Category

(j)

Figure 13 i-j. Response to flooding of *Veronica* species and *Rumex maritimus* by inundation category. Negative numbers on the x-axis are vertical distance above maximum water level.

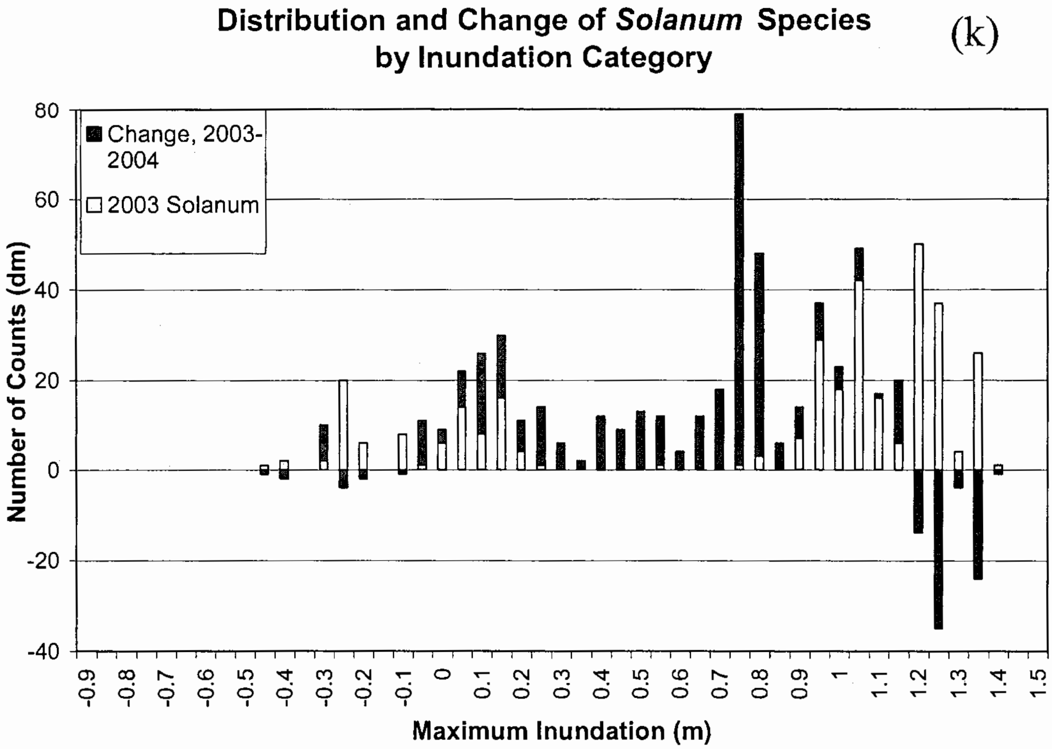


Figure 13 k. Response to flooding *Solanum* species by inundation category. Negative numbers on the x-axis are vertical distance above maximum water level.

Duration and timing of flooding

Points that were subjected to deep inundation were under water for long periods of time; these points also spent more time under "intermediate" flooding of 0.5-0.6 m (Table 5). The differences in time spent under these conditions were due primarily to different end dates (Table 6). Points that received less than 0.85 m maximum inundation were under less than 0.6 m of water by May or June, while points with more flooding remained under 0.6 m until late June or July.

Table 5. Total duration of flooding and duration of 0.5 m and 0.6 m inundation for points with maximum flooding of 0.7, 0.75, 0.8, 0.85 and 1.15 m.

Maximum flooding	Time under water	At least 0.5 m	At least 0.6 m
1.15 m	241 d	189 d	177 d
0.85 m	207 d	142 d	137 d
0.8 m	202 d	139 d	125 d
0.75 m	197 d	137 d	104 d
0.7 m	191 d	125 d	89 d

Table 6. Start and end dates of 0.5 m and 0.6 m inundation for points with maximum flooding of 0.7, 0.75, 0.8, 0.85 and 1.15 m.

Maximum flooding	0.5 m		0.6 m	
	Start	End	Start	End
1.15 m	January 4	July 9	January 11	July 4
0.85 m	January 30	June 19	January 31	June 15
0.80 m	January 31	June 17	February 1	June 4
0.75 m	January 31	June 15	February 1	May 14
0.70 m	February 1	June 4	February 2	April 30

Response of Reed Canarygrass Phenology

Correlation of phenological characteristics with flooding

A significant positive correlation existed between water depth and stand height, according to data taken during the first round of monitoring (Figure 14).

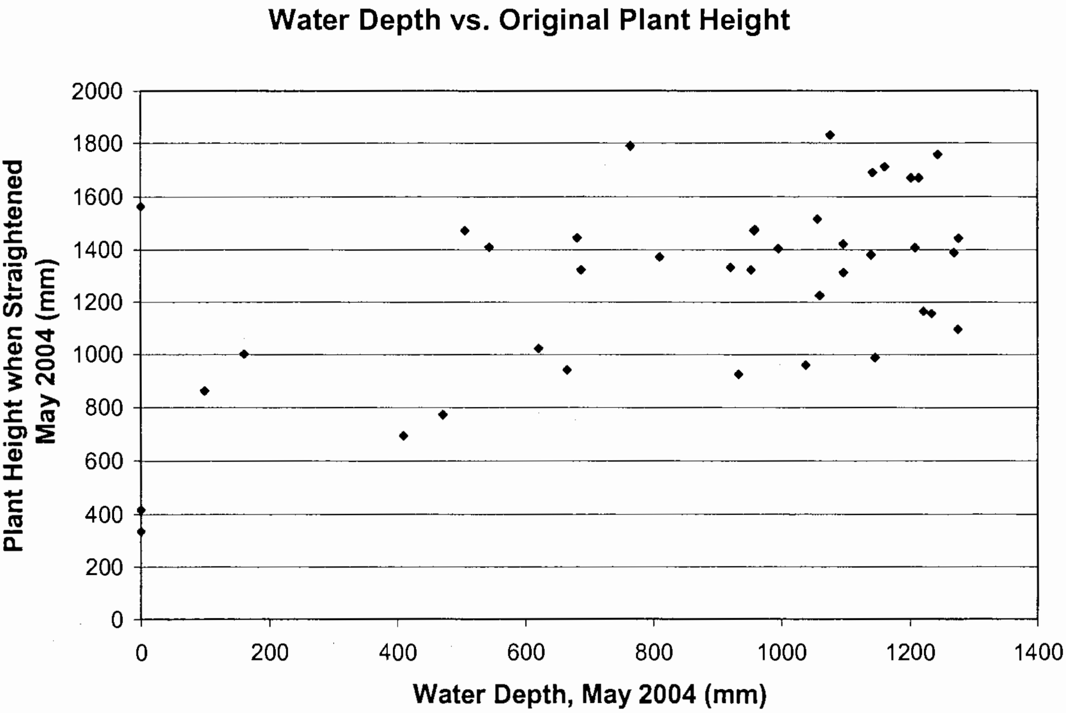


Figure 14. Relationship between stand height and water depth as of May 19-21, 2004. Spearman's rho = 0.38, $p < 0.05$.

Other measures of stand health indicated that greater inundation had a negative effect on reed canarygrass over the course of the study. Both the erectness—the ratio of plant height *in situ* to plant height when straightened—and the height of the tallest live shoot were significantly inversely correlated with the original water depth—i.e., the depth measured on May 19-21—as shown in Figures 15 and 16.

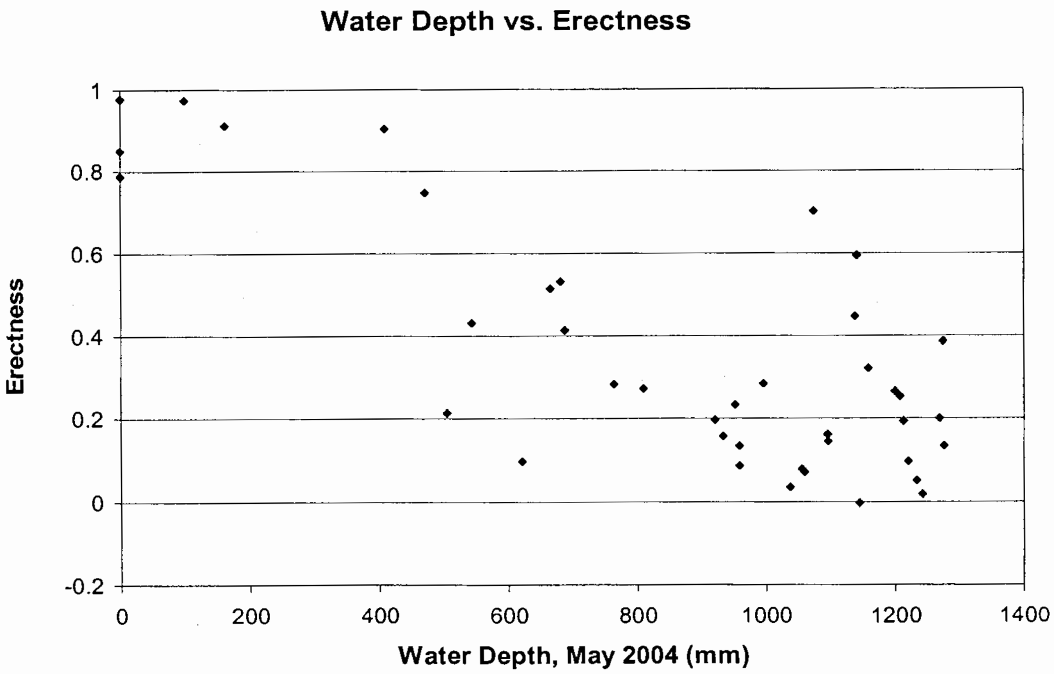


Figure 15. Relationship between erectness as of July 21, 2004 and original water depth. Spearman's rho = 0.59 ($p < 0.05$).

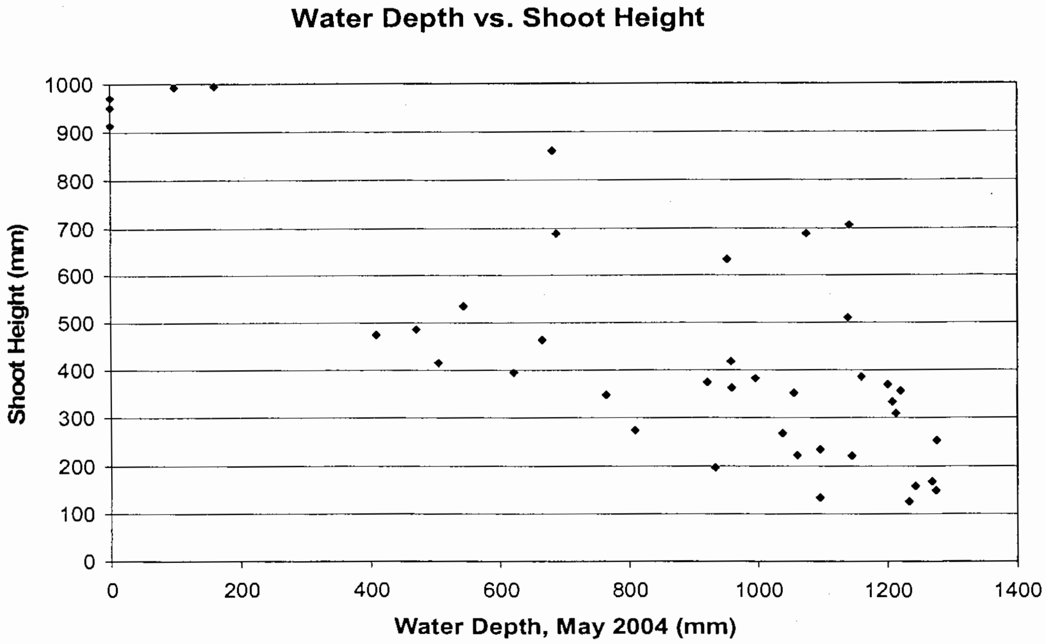


Figure 16. Relationship between height of tallest live shoot as of July 21, 2004 and original water depth. Spearman's rho = 0.72 ($p < 0.05$).

Growth of the stands also appeared to be impaired by inundation, though this relationship was not significant (Figure 17).

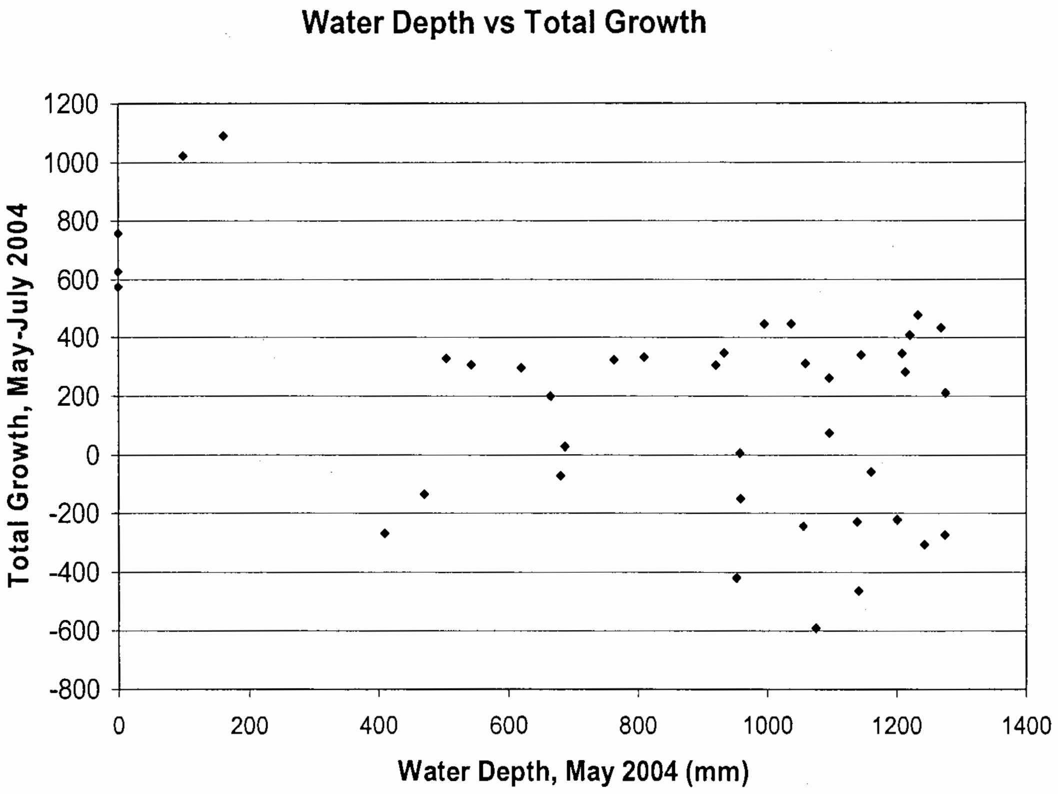


Figure 17. Stand growth from May 19 to July 21, 2004 vs. original water depth. Spearman's rho = 0.31 (p > 0.05).

"Negative growth" on this graph likely means that the plants that made up the stand had fallen over by early July and rooted at the nodes; it is also possible that the original plants died back, and that stand measurements in July were of second-growth plants.

Finally, there was a significant positive correlation between original water depth and number of shoots per plant (Figure 18).

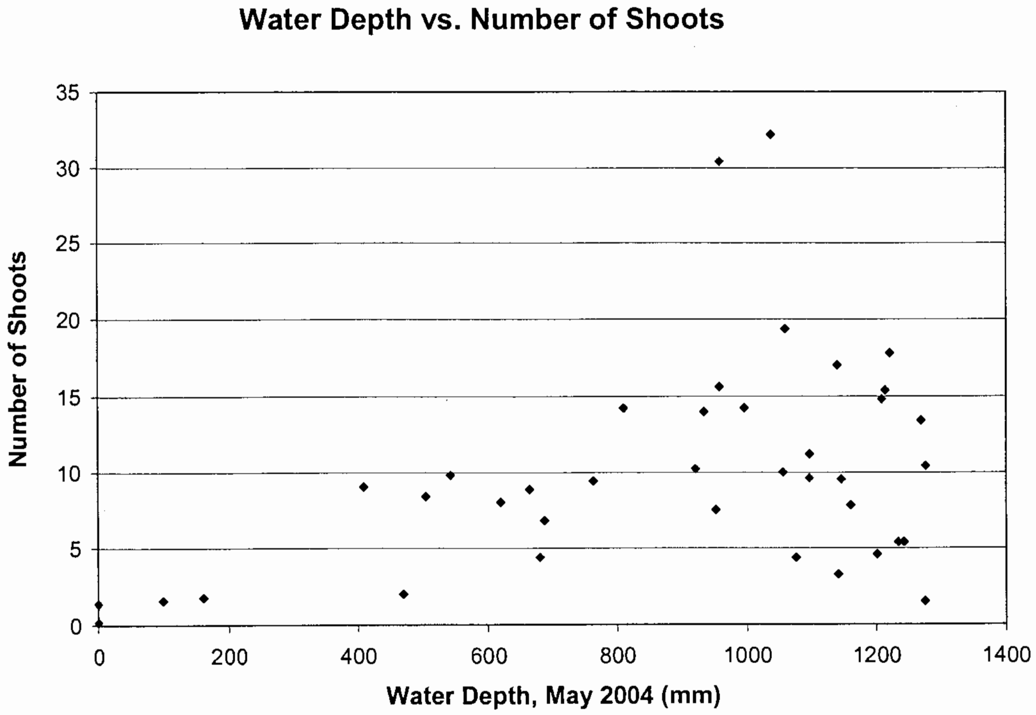


Figure 18. Number of shoots per plant on July 21, 2004 vs. original water depth. Spearman's rho = 0.38 ($p < 0.05$).

Discussion and Conclusions

Reduction in Reed Canarygrass Abundance

The results of this study support the first hypothesis. After the first year of managed flooding at Smith and Bybee Lakes, the percent cover of reed canarygrass decreased from 43.7% to 41.2%, a small but significant drop. The decrease takes on added significance in light of previous studies of the plant community at SBL, which showed a dramatic increase in reed canarygrass from 1992 to 2001. Reversal of a decade-long trend of increasing reed canarygrass cover argues that the first year of flooding was effective in the suppression of this invasive species.

Reed canarygrass effect on diversity

Reed canarygrass clearly plays a role in suppressing native plant species at SBL, as evidenced by its negative impact on diversity. This relationship, while still significant, was less strong after the first year of managed flooding. This result may indicate that flooding delayed the growth of reed canarygrass, allowing other taxa to grow in what had previously been monotypic stands of reed canarygrass. It could also mean that other taxa, such as *Polygonum* species, came to dominate some transects in response to the new hydrologic regime; this was the case on two transects, where both diversity and *Phalaris* cover were low in 2004. In either case, flooding lessened the negative pressure that reed canarygrass.exerts on the native plant community at SBL.

Reed canarygrass in the willow forest

The decrease in reed canarygrass was particularly high in areas dominated by Pacific willow. This change was likely due to the combination of flooding and shading. The regenerating willow forest at SBL occurred primarily in areas subject to significant inundation (> 1 m), which was more effective in eliminating reed canarygrass; hence, the latter was more likely to suffer the effects of flooding in these areas.

Nevertheless, the additional effect of shading almost certainly played a role. Willow shading of reed canarygrass more than doubled the chance of a decrease in reed canarygrass as compared to an increase; a decrease was more than three times as likely as continued presence when regenerating willow played a role. Furthermore, willow canopy significantly increased the loss of reed canarygrass cover as compared to non-canopy areas, even at similar elevations. Light transmittance under dense canopies is primarily in the red or far-red regions of the spectrum (Grant 1997); reed canarygrass does not germinate well under far-red light (Lindig-Cisneros and Zedler 2002). Shading reduces both total biomass and root: shoot ratio of reed canarygrass (Perry and Galatowitsch 2004), and lower light transmission under canopy has also been found to inhibit the survival of existing reed canarygrass plants (Maurer and Zedler

2002). The canopy in the regenerating willow forest had a significant effect on the survival of reed canarygrass by this mechanism. The combination of inundation and shading was very effective in controlling reed canarygrass.

Response of Other Taxa

Native taxa

Late-germinating, emergent wetland species showed a positive response to the new flooding regime. Nearly all of the important native taxa at SBL increased in percent cover from 2003 to 2004, as did several less-common, desirable taxa. Prolonged maintenance of constant water levels, such as was the case at SBL during the 1980s and 1990s, is detrimental to emergent aquatic plants. By contrast, annual moist soil wetland species can establish quickly after drawdown, once dominant vegetative species are flooded out; these stands were favored by early June drawdown in one study (Harris and Marshall 1963). This was certainly true of the emergent community at SBL; *Polygonum* species, *Bidens* species, *C. strigosus*, *E. ovata*, and *Veronica* species all significantly increased in percent cover after the first year of managed flooding. *E. hypnoides* increased slightly; *L. palustris* and *R. maritimus* decreased significantly, especially where maximum inundation exceeded 1 m. There was no indication that wetland indicator status was a predictor of the change. All of the "rare" native taxa increased in percent cover, though the increase was only significant for *Carex* species, *L. oryzoides*, and *M. laxa*.

Invasive taxa

Invasive taxa other than reed canarygrass had a mixed response to the new hydrologic regime. *Solanum* species benefited significantly, as did *C. vulgare* and *L. salicaria*, though the latter was still uncommon at SBL. Other species, such as *L. corniculatus*, *E. ciliatum*, *R. armeniacus*, and *C. arvense* had significantly less cover after the first year of managed flooding; *H. perforatum* disappeared from the transects in 2004.

Solanum species have a variety of morphologies that allow them to thrive in differing habitats; prolonged inundation promoted larger leaf area in *S. dulcamara* in one study (Braun and Toth 1994). This helps to explain some of the increase observed in this taxon's percent cover. Much of this increase may also be attributed to a difference in the stage of the growing season at which I observed the plants in 2003 vs. 2004: Many *Solanum* leaves—particularly those of the annual species *S. nigrum*— had already senesced when I monitored the transects in 2003, whereas in 2004 the plants were still leafed out. It is also worth noting that *Solanum* cover increased less—or, in some cases, decreased—in areas subject to deep inundation (Figure 13 k). Nonetheless, continued expansion of *Solanum* species could be a concern under the new water regime, and should be monitored further.

Also of possible concern is *L. salicaria*, which, while not a significant factor yet at SBL, is well-known as a problematic invasive species in wetlands (Zedler and Kercher 2004). At least one microcosm experiment determined that establishment and dominance of this species was limited by seasonal flooding, particularly on low-fertility soils (Weiher et al. 1996), indicating that water management efforts at SBL could help prevent its expansion. Conversely, *L. salicaria* appeared in palustrine wetlands following the restoration of tidal flooding at Spencer Island, WA (Tanner et al. 2002). *L. salicaria* did not appear frequently on the transects in this study; it is difficult to assess or predict its response to the new hydrologic conditions based on this small number of observations. Given this species' ability to invade, this question merits further study.

Competition with reed canarygrass

The interactions between reed canarygrass and other taxa were less clear-cut, as some taxa fared far better at the expense of reed canarygrass, while others succumbed to competition with it. Among native taxa, *Polygonum*, *Veronica*, and *Bidens* species, *S. lucida*, *L. palustris*, and *L. oryzoides* all successfully out-competed reed canarygrass under the hydrologic conditions created by managed flooding; *Solanum* species were the only invasive taxa to overtake large areas previously inhabited by reed canarygrass.

Despite making gains in overall cover, *Eragrostis hypnoides* and *Eleocharis ovata* both failed to effectively compete with reed canarygrass; these two and a second *Eleocharis* species, *E. palustris*, were the only native species to lose significant ground to *Phalaris*. Reed canarygrass also significantly out-competed *L. corniculatus* and *C. arvense,* two invasive species. It is worth noting that the losses of *E. ovata* and, especially, *E. hypnoides* occurred primarily in areas subject to less than 1 m inundation, where reed canarygrass was both more abundant and more likely to increase in percent cover.

Overall, it appears that the change in hydrologic regime had a greater effect on the response of native taxa than did the suppression of reed canarygrass during the first year of management. It is encouraging, however, that invasive taxa other than reed canarygrass are not the primary taxa replacing the latter where it is eliminated by flooding. The great majority of places where reed canarygrass disappeared in this study were either left bare or occupied by native taxa, in accordance with my second hypothesis. If the trends evident in the first year continue, reed canarygrass suppression will likely lead to even greater increases in cover for desirable species at SBL. This has important implications for the variety of wildlife that resides there for all or part of the year, many of whom depend on the seeds, shoots or roots of native plant species for food.

Phenological Impacts

Stand height showed a positive correlation with inundation. This is consistent with the way many species respond to flooding stress; stem elongation and greater allocation of resources to shoot growth are common adaptations to flooding (Kercher and Zedler 2004). The greater length of stems, as well as the increase in branching produced by inundation, indicate that reed canarygrass uses more resources for aboveground growth when flooded. This comes at the expense of root development, as evidenced by the increase of shoot: root ratios in studies of flooded reed canarygrass (Miller and Zedler 2003). Further, Conchou and Fustec (1988) found that reed canarygrass shifted stocks of carbon, nitrogen, phosphorous and potassium from belowground to aboveground tissues in response to flooding. Inundation caused a shift in the use of resources away from rhizome development; this could have implications for the long-term survival of reed canarygrass.

Stems of heavily flooded reed canarygrass lacked turgor when I monitored them during the growing season; this was evident in the relationship between erectness and inundation. Having to devote resources to stem elongation, branching, and the production of adventitious roots, reed canarygrass has less energy to store in its rhizomes. This has a negative effect on clonal reproduction later in the growing season; tillering is reduced under more prolonged inundation (Rice and Pinkerton 1993).

Effectiveness of Inundation

Deeper inundation was negatively correlated with several measures of the health of reed canarygrass during the growing season. It also increased the likelihood that reed canarygrass would disappear on a given decimeter, as compared to the likelihood of appearing or remaining there. Flooding was thus a principal cause of the reduction in reed canarygrass in this study.

The analysis of the effects of willow forest indicate that reed canarygrass was less likely to be found in the elevation zone where this forest was likely to be present; this zone was heavily inundated. Moreover, the regression showed that reed canarygrass was more likely to have been absent in both years than it was to decrease as inundation became more severe. These results indicate that reed canarygrass has not historically inhabited areas around the lakes that are subject to deep flooding. Since deep inundation has discouraged past establishment of reed canarygrass and has reduced its percent cover in this study, it seems reasonable to infer a causal relationship between depth of inundation and suppression of reed canarygrass. This supports my third hypothesis.

Flooding of more than 0.85 m appears to have been particularly effective in reducing reed canarygrass cover. Areas of SBL that were subject to less than this amount of inundation primarily saw a net increase of reed canarygrass, while those that were flooded to at least this depth underwent significant losses of this species.

The duration and timing of flooding may also play a key role in determining the effect on reed canarygrass cover. Points that were flooded to at least 0.85 m were under water for a longer period of time. Moreover, these points spent at least two weeks more under 0.6 m inundation than did less-flooded points. This additional time coincided with the typical tillering period of reed canarygrass in late May and June, such that points subjected to less than 0.85 m maximum inundation were under less than 0.6 m of water at the time when reed canarygrass usually produces new shoots from its rhizomes. This may account for the difference in the response of reed canarygrass at different degrees of maximum inundation.

Further study will be necessary to determine whether the reduction in reed canarygrass resulted from the 0.85-m maximum degree of inundation, the duration of that inundation, or the timing of the flooding that allowed for inundation of at least 0.6 m during the tillering period. It is also possible, given the bimodal distribution of reed canarygrass with respect to elevation (Figure 12 a), that genetic differences in plants at different locations produced differing responses to flooding. I did not perform a genetic study, and thus cannot rule this out as a contributing factor; however, the phenological responses to flooding (Figures 14-18) were consistent, which reinforces the case for flooding as a causal agent in the reduction in reed canarygrass.

Management Implications

Managed flooding was effective in its first year at Smith and Bybee Lakes, and thus shows promise as a means of controlling reed canarygrass. This is particularly true when it is combined with shading, another common control measure for this species. Flooding was most effective at depths of 0.85 m or greater; this degree of inundation reduced reed canarygrass cover, and also produced a positive response in many desirable native taxa. *S. lucida*, which affected reed canarygrass by shading it in this study, responded most positively to flooding at these depths; hence, flooding can contribute both directly and indirectly to reed canarygrass suppression.

Conversely, flooding of between 0.5 m and 0.85 m increased reed canarygrass cover. While it is difficult to draw long-term conclusions from this, it appears that inundation must be implemented to the appropriate threshold or it may prove counterproductive.

The overall significance of this study is that it shows the effectiveness of an active management strategy on one of the invasive plants on the "most wanted" list in the Pacific Northwest and in wetlands across North America. These results provide information for wetland and lakes managers anywhere on the effects of inundation on an invasive species such as reed canarygrass and on the response of the remainder of the vegetative community. Moreover, future studies may be able to consider these effects in conjunction with other control measures; the latter can be more effective when combined with water management capabilities that allow for consistent water levels through spring (Paveglio and Kilbride 2000). The effectiveness of the combination of flooding and shading seen in this study bears this out; multiple means of reed canarygrass control have a greater chance of success than any single approach.

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