Chapter 4
Management of Plant Invaders Within a Marsh: An Organizing Principle for Ecological Restoration?

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Abstract Controlling plant invaders is often one aspect of ecological restoration. However, the planning and application of control measures can lead to difficult questions regarding project goals and measures of success. We present a case study of a coastal wetland system in South Carolina, USA, where two plant invaders, *Phragmites australis* and *Typha domingensis*, were targeted for control. As project participants gradually accepted the concept that success must be measured in terms of long-term system parameters rather than short-term invader control, the methods and approaches changed. As an alternative to applying herbicides, a method of reconnecting the system to the ocean was pursued. Instead of simply measuring plant control, a before-after-control-impact monitoring design was implemented that allowed comparison among restored and multiple reference systems in the immediate area. Attempts to reestablish tidal flow and modify environmental conditions to alter system attributes were variable with both unplanned positive and negative effects. Most of these impacts were associated with the fact that the wetland existed in a state park used by large numbers of people for passive recreation. The case study demonstrates that plant invasion and the willingness of people to control plant invaders can provide a useful starting point for eventual development and implementation of scientifically meaningful attempts at ecological restoration.

Keywords BACI • Ecological restoration • *Phragmites australis* • Reference site • *Tidal reconnection* • *Typha domingensis*.

4.1 Introduction

Salt marshes along the eastern USA coast are susceptible to dramatic changes in community composition and structure when hydrological connections to the ocean are restricted (Warren et al. 2002). Restrictions can result from natural,

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nearshore geological processes or from human activities such as marsh filling and diking. Generally, salt marshes cut-off from the ocean are converted to fresh or brackish marshes that can be invaded by perennial monocots including the common reed grass (*Phragmites austalis*) and cattails (*Typha* spp.). Ecological effects of *Phragmites* invasions and attempts at removal have been studied extensively in the Northeast USA from Massachusetts to Maryland (Boumans et al. 2003; Gratton and Denno 2006; Kimball and Able 2007; Teal and Weishar 2005). *Phragmites* invasions generally are less common and less well studied in the southeastern US.

The economic value of various ecological functions associated with tidal salt marshes (Costanza et al. 1997) has increased the interest of coastal managers to restore modified and/or invaded salt marsh sites. Restoration goals for impounded marshes typically, when stated, include reestablishment of tidal exchange, elimination of salt-intolerant and/or invasive grasses, and eventual development of ecological attributes similar to natural or reference tidal marshes (Roman et al. 2002; Warren et al. 2002). However, many problems and questions associated with marsh restoration remain because of difficulties in defining which ecological characteristics and/or which reference marshes should be considered. Evidence to date indicates certain marsh characteristics remain dissimilar to natural marshes decades after restoration (Craft et al. 2003; Zedler and West 2008). Coastal systems also can be unstable and characterized by a long history of switching from one ecological state to another (Booth et al. 1999; Zedler and West 2008), making determination of an appropriate restoration target difficult. Furthermore, coastal marshes increasingly are fringed by residential and commercial development making marsh conditions inextricably connected to and dependent on the social, political, and economic environment of coastal communities. The connection between marshes and local human communities means that any restoration activities will be influenced by a diverse array of stakeholders.

Plants frequently form the foundation for categorizing systems (e.g., *Spartina*-marsh), and plant management emerges as a primary activity in most restoration projects (Young 2000; Young et al. 2005). When plant invasions are present, plant eradication or control generally are required. Although the successful management and restoration of plant invasions must be guided by ecological theory and accepted research approaches (Neckles et al. 2002), managers universally recognize that a strong theoretical background is only one part of the restoration process. Along with ecological theory, a range of economic, political, and social factors can influence efforts to manage plant invasions as was shown in the case study involving knapweed, an introduced species invading pastures and rangeland in Colorado (Luken and Seastedt 2004)

This chapter focuses on what ostensibly is a salt marsh restoration project located in coastal South Carolina, USA. We use the restoration project to illustrate processes common to many efforts involving habitat restoration and invasive species, namely how an initially simple instance of controlling plant invaders eventually developed into a complex case of ecosystem management. We attempt to clarify the frequently conflicted nexus among researchers focused on testing for scientific generalizations, resource managers mandated to change or manipulate a system for the benefit of others, and the general public or stakeholders who see the environment as a resource that supports various recreational activities.

4.2 Methods and Approaches

4.2.1 Study Site

Sandpiper Pond is a 15 ha brackish to freshwater marsh located within the barrier beach system at Huntington Beach State Park, South Carolina, USA (Fig. 4.1). Prior to the 1980s, Sandpiper Pond was connected to the ocean by a narrow channel. The pond proper was composed of open water, tidal mud flats, and salt marsh vegetation (e.g., *Spartina alterniflora*, *Juncus roemerianus*). After the 1980s the channel closed. Sandpiper Pond became a mostly freshwater system and was invaded by two perennial monocot species, the common reed (*Phragmites australis*) and southern cattail (*Typha domingensis*) (Fig. 4.2).

4.2.2 Project Inception

Located within a state park, Sandpiper Pond (33:30:53 N, 79:03:06 W) is managed for multiple uses that include recreation and tourism. The name Sandpiper Pond was derived from the diversity of wading birds that historically used the site, and the state park is considered one of the premier birding sites on the southeast

Fig. 4.1 A map of South Carolina, USA (SC) and the location of Huntington Beach State Park (HB) within South Carolina

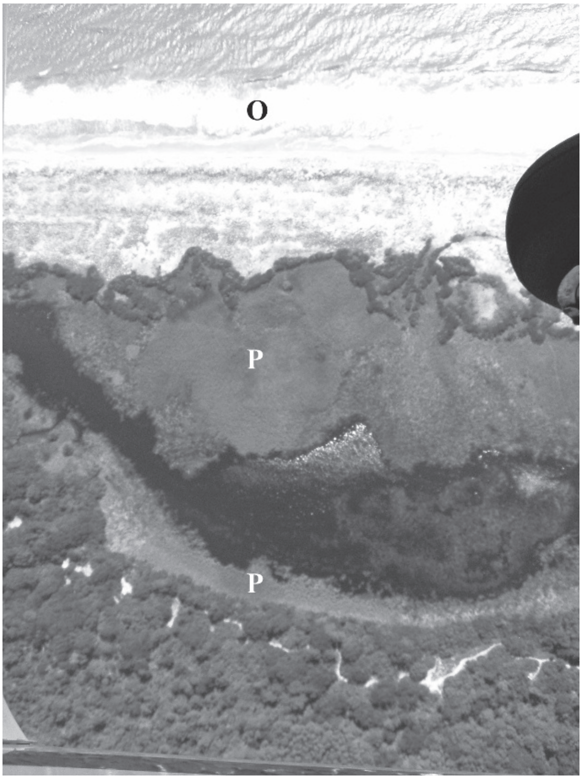

Fig. 4.2 The central portion of Sandpiper Pond long after closure of the channel to the ocean. Fringing areas (P) surrounding the central water are stands of *P. australis* and *Typha* sp. The open ocean (O) is at the *top*

coast (Luken and Moore 2005). Initial concerns about the ecological condition of Sandpiper Pond were expressed to park personnel by a group of volunteers, the Friends of Huntington Beach State Park. The Friends, composed mostly of bird watchers, concluded that avian use of Sandpiper Pond had declined in recent years. Evidence for the decline in bird sightings mainly was anecdotal, although observations occasionally were recorded in a communal log located near the site. The Friends obtained a small grant from the US Environmental Protection Agency to restore the site focusing primarily on control of the two plant invaders.

Once funding was secured, park personnel arranged a meeting among park managers, Friends of Huntington Beach, and faculty from Coastal Carolina University to discuss the project and to determine how to proceed. There was a wide range of opinions regarding how to proceed and how to define success. Two proposed approaches for plant control, spraying of herbicide and excavation of a channel reconnecting Sandpiper to the ocean, were debated. Participants eventually came to the conclusion that simply controlling plants would not adequately define restoration success and that long-term monitoring to characterize trends in various ecological parameters was essential.

4.2.3 Project Design

Previous research suggested that salt stress may be sufficient to control certain plant invaders (Burdick et al. 2001; Bart and Hartman 2002; Farnsworth and Meyerson 2003), and thus it was decided that a new channel would be excavated to bring salt water into the system. A modified before-after-control-impact (BACI) monitoring design was chosen to assess the effects of reconnection on various ecosystem characteristics. BACI designs typically are applied in situations where the before condition represents a predisturbance state (Green 1979; Stewart-Oaten et al. 1992; Wiens and Parker 1995). The question in most BACI studies is whether changes in an unimpacted environment can be attributed to an identified disturbance yet to occur. At Sandpiper Pond we were interested in whether the current reconnection would lead to system changes.

Selection of a BACI experimental design required both a delay in channel excavation to allow collection of "before" data and the selection of reference or control sites. A relatively short 3–4 months delay in channel excavation was agreed to because of funding and scheduling concerns. For example, channel construction had to be completed before the beginning of spring loggerhead sea turtle nesting. The limited delay only allowed for collection of seasonally restricted, winter to early spring, before data. The selection of control sites and ability to optimize the detection of significant impact effects in typical BACI studies is a subject of much concern (Underwood 1994; Benedetti-Cecchi 2001). For the Sandpiper study, we selected two control sites: a saltwater pond (Jetty Pond) created when the Murrells Inlet jetties were constructed and at approximately the same time as the impoundment of Sandpiper Pond, and a salt marsh (Huntington Marsh) located on the backside of Huntington Beach State Park just west of Sandpiper Pond. Jetty Pond (33:31:30 N, 79:02:12 W) was approximately the same size as Sandpiper but remained connected to the ocean and was surrounded by developing salt marshes predominated by *Spartina alterniflora*, *Spartina patens*, *Salicornia virginica*, and *Limonium carolinianum*. Huntington Marsh (33:30:48 N, 79:03:22 W) specifically was chosen to represent a high-marsh environment and was predominated by *Juncus roemerianus*. Rationale for selecting a high-marsh site was driven by the observation that remnant populations of *J. roemerianus* still existed around Sandpiper Pond, and *J. roemerianus* represented a likely early "colonist" if opening the channel had the desired effect. Multiple control or reference locations also were chosen to reduce limitations of the BACI design (Underwood 1994) and because a priori information did not suggest selection of a "correct" restoration target in view of the potentially strong system modification by the plant invaders.

4.2.4 Inlet Construction and Maintenance

Excavation of the new inlet to the ocean began in April 2005 (Fig. 4.3). Equipment operators began excavating near the pond with the goal of developing a channel ca. 30 m wide and 400 m long. Large quantities of sand were pushed laterally up or down the beach during excavation because the channel cut through a 3+ m tall barrier dune system. Provisions for removing sand from the site were not economic. The inlet was opened successfully on 17 April 2005 during a low tide. Water immediately began flowing from Sandpiper Pond and into the ocean. However, subsequent wave action deposited sand at the mouth of the inlet, and tidal exchange was stopped except for times of very high tides. The inlet was reopened again on 14 September 2005 after turtle nesting season was over, but the opening was again closed shortly thereafter.

During the brief April period when the inlet was open, a coastal storm in combination with a high tide flooded Sandpiper Pond with sea water. On the subsequent low tide, large quantities of detritus were mobilized and washed out of Sandpiper Pond into the ocean. The wrack spread along the coast and eventually was deposited on the beach in long windrows (Fig. 4.4). The unanticipated wrack deposition was a problem as large

Fig. 4.3 The initial reexcavation of a channel from Sandpiper Pond through the barrier beach and extending into the ocean in April 2005. Stands of *P. australis* surround either side of the inlet and the excavator is located just in front of the ocean

Fig. 4.4 Wrack deposited along Huntington Beach as a result of inlet excavation and mobilization of Sandpiper Pond detritus

numbers of people utilize the beach for passive recreational activities. Fortunately, subsequent high tides eventually broke up and transported the wrack off site.

In 2006, sea-beach amaranth (*Amaranthus pumilus*), a federally threatened and endangered plant species, was found growing in the excavated inlet. Presumably excavation activities stimulated germination of dormant seeds. The emergence of *A. pumilus* quickly ended any further attempts to excavate the inlet and plants were flagged while human traffic was restricted in the areas around the plants.

During 2006/2007 park personnel redirected their activities to the opposite side of Sandpiper Pond to increase tidal exchange between the pond and the salt marsh. Efforts focused on a small culvert and tidal creek connecting Sandpiper Pond to the salt marsh on the backside of the barrier beach. The small culvert eventually was replaced with a larger culvert and at present allows tidal exchange during high tides.

4.2.5 Monitoring Efforts

Additional funding through South Carolina's Sea Grant Program was obtained to monitor the effects of the restoration and control efforts. Permanent vegetation and soil monitoring stations were established at Sandpiper Pond (n = 12), Jetty Pond (n = 6), and Huntington Marsh (n = 4). At each monitoring station, samples were collected from 3 to 4 elevations above, at, and below the terrestrial-marsh boundary. Stations were monitored twice yearly, once during the winter or dormant growing season (November to January) and once during the summer or active growing season (May to July). Plant species richness and cover were determined from within 1 m^2 quadrats. Plant biomass was collected from either 0.25 (in *J. roemerianus* stands only) or 0.5 m^2 quadrats in which all above-ground stems were clipped at the sediment surface and later dried at 60°C for 2 + d before determining dry mass. Sediment cores (2.1 cm dia., 5 cm depth) were collected from each station and elevation and processed to determine soil pore water salinity and organic content. Pore water salinity was measured by adding a know volume of deionized water (DW) to the sediment sample, agitating multiple times, determining salt content of the supernatant using a refractometer, and calculating total salinity by standardizing to the total water (DW + soil pore water) in the sample. Organic content was determined by placing a known amount of sediment in a furnace at 500°C for 4 + h and calculating the ash-free dry mass (AFDM) by subtraction.

Along with plant and soil monitoring, CCU faculty and students and Huntington Beach volunteers monitored a variety of other system characteristics such as fish communities, bird use, and basic water chemistry. Some sampling efforts are ongoing, but others have ceased as grants expired and students graduated.

4.2.6 Analyses

Data were analyzed using a variety of parametric, nonparametric, and multivariate approaches. The lack of extensive before and, to a lesser extent, after sampling times by necessity limits the actual application of BACI analyses (Underwood 1994). Instead, we applied one- and two-way ANOVA models and nonparametric Kruskal-Wallis tests to the data where appropriate (e.g., normality and homogeneity assumptions satisfied). To compare among treatment levels (e.g., Sandpiper, Jetty, Huntington locations) appropriate pairwise comparison tests for parametric, Ryan-Einot-Gabriel-Welsch F or REGW F (Day and Quinn 1989), and nonparametric, Dunn's multiple comparison test (Hollander and Wolfe 1973), were applied. All tests, where possible, were run using SPSS v14. A nonmetric multidimensional scaling (NMDS) ordination approach was used to analyze species composition data (Cox and Cox 2001). Sorenson's distance measure weighed by relative coverage was calculated for species compositions from the before or winter season samples, and PC-ORD v5 used for NMDS.

4.2.7 Educational Efforts

In concert with the restoration and monitoring, there also were educational and outreach activities. A new observation deck and interpretive display were constructed during 2005. The goal of these structures was to inform park visitors about the ongoing restoration efforts (Fig. 4.5). In addition, Sandpiper Pond regularly is used as an educational resource both in park programs and courses taught at CCU.

Fig. 4.5 Placard constructed near an observation deck explaining the goals and procedures of the Sandpiper Pond restoration project

4.3 Preliminary Results

Species richness and predominant species composition for the plant communities within each marsh prior to reopening the inlet are presented in Table 4.1. Although predominated by *P. australis* and *T. domingensis*, Sandpiper Pond supported greater numbers of plant species than either reference location (Table 4.1). Jetty Pond and Huntington Marsh were relatively species poor and supported species characteristic of high marsh communities (Table 4.1). Ordination of the initial species compositions for each location using NMDS also suggested that the plant community at Sandpiper Pond was different from the reference locations (Fig. 4.6). Jetty Pond and Huntington Marsh were more similar to each other (clusters C and D), except for a few sites where *Spartina patens* predominated, than to Sandpiper Pond (clusters A and B, Fig. 4.6).

Initial sediment pore water salinity and organic content for samples collected from within each marsh also are presented in Table 4.1. Pore water salinity was significantly different among marshes (Kruskal-Wallis $\chi^2 = 18.96$, df = 2) with the freshwater condition at Sandpiper Pond significantly different from either reference marsh (Dunn's multiple comparisons test, $p < 0.05$). Sediment AFDM also was significantly different among marshes ($F_{2, 19} = 4.89$, $p < 0.02$). Organic content was significantly greater within Sandpiper compared with Jetty Pond sediments (REGW F, $p < 0.05$).

Table 4.1 Plant species richness and predominant species composition and mean (±SE) soil salinity and organic content (ash-free dry mass) from below shoreline samples at the three marsh locations within Huntington Beach State Park, SC, USA

	Location		
Characteristics	Huntington Marsh	Jetty Pond	Sandpiper Pond
Plant species richness	8	5	17
Predominant species	*Borrichia frutescens*	*Salicornia virginica*	*Phragmites australis*
	Juncus roemerianus	*Spartina patens*	*Typha domingensis*
Soil salinity (ppt)	22.4 ± 3.6^{a}	27.5 ± 3.2^{a}	0.0 ± 0.0^{b}
Soil AFDM (mg/g)	10.5 ± 5.0^{a}	18.0 ± 4.5^{ab}	43.2 ± 8.1^{b}

Superscripts (e.g., a, b) indicate significant subsets determined either by REGW-F (soil AFDM) or Dunn's multiple comparison procedure (soil salinity)

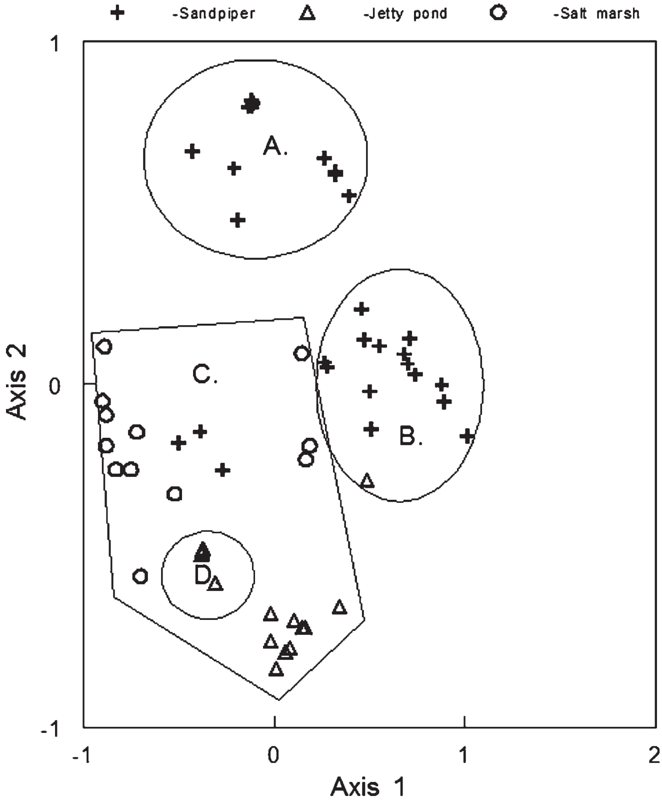

Fig. 4.6 Nonmetric multidimensional scaling ordination of community samples from three locations (indicated by *symbols*) at Huntington Beach State Park, South Carolina, USA. Groups identified by Sorenson's distance and NMDS at a final stress of 10.3 are clustered and identified by *letters*

Changes in pore water salinity and above-ground biomass in the winter after opening the channel were modest or not consistent with significant "after impact" effects. Pore water salinities rose slightly at Sandpiper Pond (5.9 ± 0.9 ppt) remaining

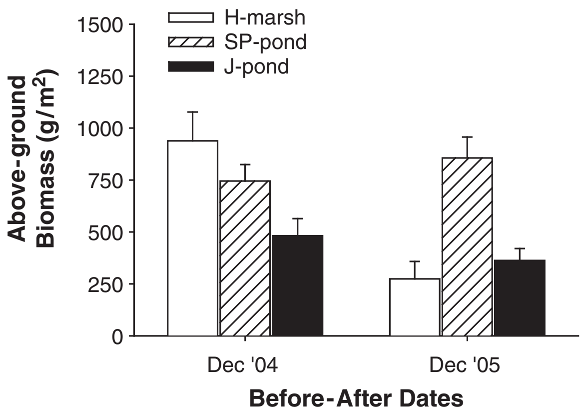

Fig. 4.7 Mean (±SE) live and dead above-ground biomass from Huntington marsh and Jetty pond control locations and Sandpiper pond impacted location before, December 2004, and after, December 2005, restoration of channel flow

fresh to brackish and not approaching salinities in the reference marshes, which remained similar to the before levels (Table 4.1). Above-ground biomass exhibited a significant interaction ($F_{2,\,38} = 6.16$, $p < 0.006$) between date (before, after) and location (Sandpiper, Jetty, Huntington), but results did not suggest an effect of increasing salinity on the vegetation in Sandpiper Pond (Fig. 4.7). Instead, Fig. 4.7 results suggest that the modified system, Sandpiper Pond, was more stable than either reference systems. Total above-ground biomass declined from 24.3% to 74.9% in Jetty Pond and Huntington Marsh compared with the 14.7% increase at Sandpiper Pond between winter 2004 and 2005.

Although enough quantitative data do not yet exist to conduct a full BACI analysis, qualitatively Sandpiper Pond has changed appreciably between spring 2004 and the most recent visit in 2007 (Fig. 4.8). A comparison of the images in Fig. 4.8 suggest that open water area has increased and previous sections of live *T. domingensis* and/or *P. australis* contain noticeable increases in standing-dead stems. However, the invasive species and extensive detrital mat are still present 3 year after initial efforts at control.

4.4 Sandpiper Pond in Retrospect

Original motivations for the Sandpiper Pond project were based on perceived, negative differences between the current invaded marsh and the historical uninvaded marsh. Notably, the project was initiated on the basis of anecdotal (e.g., greater wading bird use prior to invasion) and observational information (e.g., tidal flats now covered with vegetation). However, critical attributes of the invaded system were never characterized fully before plans to manage the system were conceived.

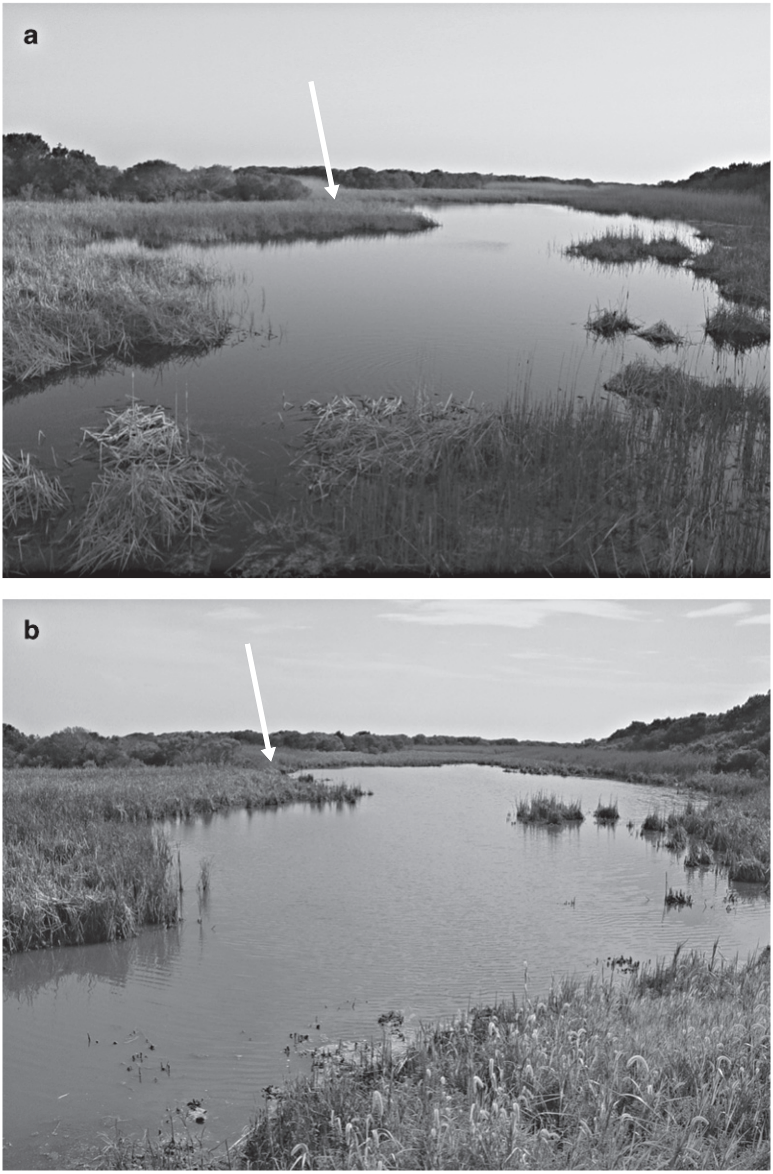

Fig. 4.8 The north end of Sandpiper Pond in spring 2005 after channel opening (**a**) and spring 2007 after additional restoration efforts involving installation of a culvert connecting Murrells Inlet Marsh to Sandpiper pond (**b**). *Arrows* are pointing to the same relative point in each picture

Obviously, wading birds were precluded from shallow open water and mud flats that were no longer part of the system in 2004, but a complete census of bird communities was never conducted. The perceived decline in overall bird observations also could be the result of invasion, a global decline in bird populations or even a decline in the ability of birders to make observations because of visual impairment from the invasive plants. The initial lower pore water salinity, higher soil organic content, higher plant species richness, and biomass measured at Sandpiper compared with control marshes also could have been confounded. Reliance on a single sampling time or even short-term (e.g., months) monitoring periods to characterize system attributes can be misleading (Underwood 1994; Chapman 1999; O'Connor and Crowe 2005). Regardless of the lack of convincing data on system degradation, funds were provided for plant control under a grant program that stressed broad involvement of the public in on-the-ground restoration activities.

Sandpiper Pond represents an unambiguous demonstration of the inherent difficulties associated with integrating research/monitoring objectives into a management/restoration project. For example, park and volunteer personnel could only delay construction of the channel until April/May 2004, long enough to collect one set of "before" data but not long enough to provide even one complete set of seasonal data for a rigorous BACI study design (Underwood 1994). Difficulties in melding research and management efforts also become more acute when funds are limited (typically a universal factor), treatments or management activities are not controlled fully, and/or many people are involved in a volunteer capacity. Clearly, attempts to manage plant invasions without commitments (i.e., funding) for long-term monitoring may not lead to anticipated restoration goals (Luken 1997). The initial inlet construction at Sandpiper did not lead to a measurable control of invasives during the 1 year of research funding available. Only after park personnel followed an adaptive management approach and restored tidal flow through the culvert did changes to the system become visible. Unfortunately, no funding currently is available to document if the changes to Sandpiper four years after initial efforts are measurable or not. However, Sandpiper Pond also demonstrates the potential positive outcomes associated with involving diverse groups of people in large-scale ecosystem manipulations. Numerous CCU faculty and students, park personnel, community stakeholders, and park visitors have participated either actively or passively in the Sandpiper restoration activities. The educational experiences associated with community-based restoration projects alone can represent a measurable outcome and result in a more scientifically and environmentally aware public (Brumbaugh et al. 2000a, b).

A fundamental problem associated with marsh restoration is that pre-restoration conditions may set limits (i.e., restoration thresholds) on postrestoration development (Hobbs and Harris 2001). For example, Lindig-Cisneros et al. (2003) found that sediment sterility interfered with attempts to influence the height of a restored California *Spartina* marsh through repeated nitrogen fertilization. Impounded marsh dominance by *P. australis* and *Typha* spp. also affected the pace at which salt marsh species colonized after reconnection with the ocean (Warren et al. 2002). The widespread existence of restoration thresholds reinforces the need for rigorous assessment protocols (Hobbs and Harris 2001) that include appropriate experimental designs and accurate indicators of success. In the absence of such protocols, resource managers may assume that management goals are being achieved when in fact the restored systems are simply making the transition from one degraded state to another degraded state. Even the existence of appropriate indicators may not be sufficient to assess the progress of a project as it often is necessary to recalibrate protocols when applied outside the "home" region (Pennings et al. 2003).

Rigorous assessment protocols for complex ecological systems require true replication of treatments and controls. When such replication is not possible, as was the case with Sandpiper Pond, alternative experimental designs are required (e.g., BACI). Difficulties and assumptions associated with applying alternative designs and statistical approaches continue to be identified (Underwood 1994; Walters and Coen 2006). The use of a BACI approach to design Sandpiper sampling efforts overcame the lack of restored treatment replication but suffered from a single before sampling and a limited number of control sites. Even if sampling was more extensive, a BACI design assumes that the impact is identifiable in terms of time, intensity, and spatial distribution. Unfortunately, the "impact" at Sandpiper Pond turned out to be variable in all aspects. Impact variation often was unpredictable and uncontrollable and included the recent prevalence of extremely high tides and coastal storms that have accelerated beach erosion leading to occasional overwash events at sites other than the excavated inlet. The very method of applying the impact (i.e., inlet vs. culvert) changed during the course of the study and is likely to change again if park personnel determine original restoration goals have not been met.

The initial monitoring of Sandpiper and control marshes at best may provide background for developing a set of new hypotheses that can be tested with more precise approaches. Salt tolerance is an important factor limiting the invasion of *Phragmites australis* (Burdick et al. 2001; Bart and Hartman 2002). Greenhouse experiments examining plant growth responses to various salt concentrations and species combinations would increase understanding about the responses of marsh vegetation to the reestablishment of tidal influence. Marshes in transition from brackish conditions to salt water conditions also may provide excellent opportunities for field experiments aimed at understanding the structure and composition of salt marsh communities (Pennings et al. 2005). Previous research suggests that salt marsh recovery from *Phragmites* invasion can be slow (Warren et al. 2002), and one possible explanation could be the significantly greater amounts of invader biomass and detritus that persist in invaded marshes (e.g., Sandpiper Pond). Experiments designed to tease out the role of detritus accumulation in inhibiting rapid community change and the connections to fundamental differences in patterns of senescence between invaders (e.g., *P. australis*) and native marsh plants (e.g., *S. alterniflora*) easily could be conducted. The unexpected establishment of sea-beach amaranth, a threatened and endangered species, at the site of inlet excavation also solicits further studies on the interaction between marsh and dune seed banks and soil disturbance. Finally, the Sandpiper project provides an excellent arena for addressing the issue of appropriate restoration targets and the value of adaptive management to achieving stated targets.

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