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"Assessment of Four Years of Marsh Restoration at the Jones Farm Experimental Restoration Facility in Northeast Ohio: Water Quality, Plant Community Development, and Adaptive Management"

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Spring 2008

Abstract. In order to characterize water quality, plant community diversity, and invasive species management at a restored wetland, I have analyzed data collected from June 2004 to August 2007 at the George Jones Memorial Farm in New Russia Township, Ohio. The Jones wetlands site is comprised of six emergent, herbaceous marshes that were restored on an old-field site in 2003. The six cells were constructed using a uniform physical restoration treatment, managed uniformly for invasive species, and replanted using three planting treatments. Each planting treatment was applied to two wetlands; treatments included two "designer" plantings of native taxa and one "self-design" control. Water quality data was collected weekly during the growing seasons of 2004-7 and plant diversity data was collected each summer. Restoration at the Jones wetlands has engendered the development of six stable, diverse marshes. Wetlands planted with native species have higher macrophyte diversity than unplanted wetlands and may show signs of different ecosystem functioning. *Phalaris arundinacea* displaced cattail (*Typha* sp.) as the most troublesome invasive taxon, although management of invasive taxa was progressively less time-consuming each year of the study. Continued post-restoration monitoring at the Jones wetlands is of great importance. Additional management recommendations are also offered.

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Chapter 1: Introduction

This thesis describes research using a set of six aligned artificial marshes located at the George Jones Memorial Farm in northeast Ohio. The Jones wetlands project is a long-term experiment in restoration ecology.

Terminology. The NRC (1992) describes restoration as "the return of an ecosystem to a close approximation of its condition prior to disturbance" and treats "created," "rehabilitated" and "mitigation" wetlands as special cases of restored wetlands. Creation involves constructing new wetlands that did not once exist in a specific place. Rehabilitation involves fairly superficial improvements to a degraded ecosystem. Mitigation involves wetland restoration designed to "avoid, reduce, or compensate for the effects of environmental damage." Thus creation and rehabilitation are technical terms while mitigation is generally defined in terms of specific policies. I will use restoration in the broad sense suggested by the NRC. In discussing the relationship between biodiversity and ecosystem functioning, I will follow Jax's (2005) recommendations by distinguishing between ecosystem processes (e.g. nitrogen fixation), functional groups (e.g. nitrogen fixers), ecosystem functioning ("the sum of those processes that maintain the [eco]system"), and ecosystem services (e.g. nitrogen amendment to the soil). Because these closely related concepts are often described under the blanket term "function," discussions of BEF benefit from greater semantic precision (Jax 2005).

Preliminary Acknowledgements. The research that I present in this paper is the result of collaborative work. Specifically, John Petersen (Associate Professor, Oberlin College), David Benzing (Emeritus Professor, Oberlin College), Rob Stenger (Oberlin College '05), Kate Weinberger (Oberlin College '06), and Joshua Smith (The Ohio State University, M.S. '06) laid the foundation for a significant portion of this thesis (see Acknowledgements for further details). I will indicate research that I conducted myself by using the singular first person, research conducted jointly with others by using the plural first person or the passive voice, and work conducted by others by using the passive voice. Previous publications describing research at the Jones wetlands include an honors thesis (Weinberger 2006), a Master's thesis (Smith 2006), and a poster presentation (Grossman and Petersen 2007). Petersen (2002) describes the project's original research proposal.

Dedication. I dedicate this thesis to my grandparents, who have never stopped believing in and loving me.

A. Restoration Ecology

Restoration Ecology is a young field that has emerged in a period of unprecedented ecosystem degradation. The Society for Ecological Restoration International (SERI) defines ecological restoration as, "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SERI 2004). Restoration ecology, then, is a field of applied ecology that develops techniques for recovering "a natural range of ecosystem composition, structure, and dynamics" through the enlistment of "ecological theory and application of the scientific method" (Palmer et al. 2006). Restoration ecology both informs and is informed by general ecological research. I will address issues of ecological importance such as biogeochemical cycling, ecosystem succession and stability, the relationship between biodiversity and ecosystem function, and the problem of invasive species. Restoration projects deal with issues such as these at biological scales ranging from the molecular to the landscape. Because it provides ways of assessing ecological theory in human-constructed ecosystems, some have suggested that restoration ecology constitutes an "acid test" of ecological theory (Bradshaw 1987).

Restoration of a broad range of ecological systems is more relevant than ever as a strategy for increasing the supply of services that originate in ecosystems and sustain human populations (Zedler and Kercher 2005, Costanza et al. 2006). Naeem (2006) presents a partial list of these "ecosystem services," including provision of lumber, biofuels, and potable water; carbon sequestration and storage; soil formation; and recreation. This juxtaposition between dwindling supply and rising demand is only exacerbated by the threat of unpredictable, non-linear climate change (Stern 2006).

There is a deficit in research addressing the relationship between the management of restored ecosystems and the quantity and quality of services they supply (Andersson et al. 2007). The origins of some ecosystem services, such as pollination, can be easily traced to discrete ecosystem structures or processes (in this case, habitat integrity and the maintenance of specific biotic communities of pollinator organisms). Other services, such as retention of dissolved phosphorus from agricultural run-off by wetland soils and vegetation, are the product of much more complex ecosystem functioning. To effectively argue that ecological restoration will increase the availability of these services, ecologists must describe more clearly the relationship between ecological functioning and the supply of ecosystem services. We must test restoration methodologies, assess which are the most effective at shaping ecosystem function in a way that increases ecosystem service supply, and use this knowledge to supplement the broader conceptual framework describing restoration ecology (Halle 2007). Filling these current gaps in knowledge with a definitive understanding of the specific mechanisms by which restored ecosystems provide valued ecosystems services is necessary to building a political and economic justification for ecological restoration.

B. Why Wetland Restoration?

Although once considered "sinister," places of little or no economic value (Mitsch and Gosselink 2000), wetlands are understood today to be threatened ecosystems that are

important as components of human and "natural" landscapes. Defined as ecosystems with either periodically or permanently flooded substrates, wetlands can be characterized by hydrology, location in the watershed, vegetation, and water chemistry. Wetlands facilitate important ecosystem functions; they are often highly productive and diverse and sequester nutrients over both the short and the long term (Mitsch and Gosselink 2000, McKenna 2003, Mulhouse and Galatowitsch 2003). Wetlands are important to humans as well. They provide a range of ecosystem services including flood prevention, runoff retention, waste treatment, water purification, and maintenance of other ecosystem services through their role in sustaining biodiversity. By acting as habitat for migratory birds, wetlands sustain threatened populations and provide aesthetic value to humans. Wetlands are also a source of natural capital that benefits the human economy even though this value is not included in most economic measures (Costanza and Daly 1992, Hawken et al. 2000, Balcombe et al. 2005). Chapter 3 includes further discussion of the humanistic and economic value of wetlands.

Current legislation has failed to deter substantial wetland degradation. McKenna (2003) estimates that roughly 86% of American wetlands have been altered. Complete wetland loss may be as high as 50% of all wetlands extant prior to European settlement. Though rates of degradation have declined from a pre-1970's high of nearly 460,000 acres per year, present-day loss is no less sustainable (NRC 1992). Wetland loss is of special concern in northeastern Ohio, where once-plentiful wetlands have now been almost entirely degraded, resulting in a loss of ecosystem services (Petersen 2003). The legislation currently protecting domestic wetlands comprises a mixed bag of policies that followed the 1977 Clean Water Act (CWA). Section 404 of the CWA instituted a permit requirement for the potentially damaging discharge of dredged or fill material into wetlands (EPA 2008). Yet this legislation and the toothless offspring it has fostered over the last thirty years are not sufficient to combat the cumulative wetland degradation threatening these ecosystems. The Army Corps of Engineers must approve permits for development that may result in wetland degradation when other options are financially infeasible. At the same time, many federal programs designed to encourage restoration, like the Conservation Reserve Enhancement Program (CREP), are voluntary. Private landowners, who are not subject to Section 404 permitting, are, under CREP encouraged with financial incentives to restore degraded habitat on their property (FSA 2008).

George H.W. Bush's "no-net-loss" policy, established in the late 1980's, has further slowed the rate of domestic wetland degradation. The policy mandates that all wetland destruction be matched by the creation of "mitigation" wetlands (Balcombe et al. 2005). Yet the degree to which mitigation, like restoration, can produce wetlands compositionally and functionally like natural wetlands is debatable (Kentula 2000, Seabloom and Van der Valk 2003, Spieles 2005, Zedler and Kercher 2005). Though mitigation wetlands are better than nothing, they do not match natural wetlands in terms of ecological function. And mitigation wetlands, if not located thoughtfully in the broader landscape or restored unsuccessfully, may not provide desired ecosystem services. This may, however, be a moot point. Though mitigated wetlands may not fully replace natural wetlands, ecologists, economists, and policymakers are increasingly interested in the possibility of widespread domestic wetland restoration (NRC 1992, Dobson et al. 1997, Zedler 2003, Costanza et al. 2006).

Most restorationists want to achieve several objectives, both legal and ecological (Comin et al. 2001). Ecological objectives – those that are measured in the field through post-restoration monitoring – describe thresholds for successful restoration of either ecological processes or ecological patterns (NRC 1992, Ryder and Miller 2005). The goal of process- or function-oriented projects is the restoration of complex ecosystem dynamics such as removal of nutrients between inflow and outflow from a wetland (Moreno et al. 2007). The goal of pattern- or structure-oriented restoration is to restore ecosystem qualities such as biodiversity or primary productivity (Comin et al. 2001). Another restoration objective extrinsic to the properties of the restored ecosystem is compliance with either public or private mandates. So, a restored marsh that is dominated by cattail but that removes most of the nitrogen and phosphorus that flow into it might be said to have met standards of "functional success" and (depending on the legislation effecting its restoration) "compliance success" while perhaps not meeting pattern-oriented objectives (Kentula 2000, Matthews and Endress 2008). Furthermore, wetlands must be located at appropriate places in the landscape for restorationists to achieve "landscape success" (Kentula 2000). Wetlands placed upstream of sources of water pollution or isolated hydrologically do not function as filters for water pollution. Likewise, isolated wetlands in fragmented landscapes may not provide for landscape-level heterogeneity, which is often a restoration objective (Comin et al. 2001). Thus, siting of wetlands within the landscape affects the success of ecological restoration as both an ecosystem- and landscape-level process.

C. Research Questions in Restoration Ecology

Research in restoration ecology has yet to adequately address a number of the field's foundational questions. Kusler and Kentula (1990), Palmer et al. (2005), Halle (2007b), and Choi (2007) summarize these knowledge gaps, including questions of which ecosystem attributes should be restored, what restoration methods should be used, and how ecosystem structure is related to ecological functioning. The relationship between these general gaps of knowledge in ecological research and the Jones wetlands project will be addressed in the subsequent section of this chapter.

Restored Attributes. Restoration researchers have attempted to monitor and induce change in wide range of ecosystem attributes (Erwin 1990, Kentula et al. 1992, NRC 1992, Tchobanoglous 1993). Hobbs and Norton (1996) suggest that this array of ecosystem attributes for ecosystem restoration can be categorized into six classes: composition, structure, pattern, heterogeneity, function, and dynamics and resilience. Projects that focus on several attributes and monitor the relationship between restoration methodologies and outcomes over the long-term can help to clarify the appropriate scope of research in restoration ecology and contribute to the development of a consistent toolbox of restoration methods.

Restoration Methods. If restorationists use consistent restoration methods and monitor the results of their work, their findings can inform general restoration practice instead of simply standing alone as individual case studies (Hobbs and Norton 1996). There are many ways to restore ecosystems and many ways of monitoring the results of ecological restoration. Most projects entail both modifying the substrate or hydrology of the restoration site and, subsequently, sowing of seeds or vegetative propagules of desired plants. Different methods are used in wetlands that differ in type, climate, or location in

the watershed (Mitsch and Gosselink 2000). Some methods may be more appropriate than others given specific objectives; research focused on the relationship between restoration methods and outcomes helps practitioners choose the right restoration methods.

Post-restoration monitoring constitutes a much-neglected restoration methodology. Monitoring of structural and functional ecological parameters should optimally occur periodically over 20 years following restoration, but researchers and practitioners alike rarely persevere with costly and time-intensive monitoring protocols (NRC 1992). Many restoration studies, including this one, report data for only four to five years after restoration occurs (Callaway et al. 2003, McKenna 2003, De Steven et al. 2006). Longer-term monitoring is usually associated with research wetlands, such as those at the Ohio State University (Fink and Mitsch 2007). Mulhouse and Galatowitsch (2003) reported that 41 prairie potholes failed to support typical plant communities nine to eleven years after restoration. Long-term monitoring of this nature is necessary to provide feedback about restoration success (Matthews and Endress 2008). Only the type of data that long-term monitoring provides can provide information about the effectiveness of restoration strategies.

Structure and Function. The relationship between ecosystem structure and function is one topic of general ecological research that can be addressed through ecological restoration (Ehrenfeld and Toth 1997, Palmer et al. 2006). Ecologists can carry out restoration experiments to better understand how composition and arrangement of components of a system generate emergent properties ranging from resilience to productivity to trophic complexity.

D. Study Objectives

Because the Jones wetlands is an experimental system restored using three planting methods, research there can address basic issues in the field of restoration ecology. Goals of the project include assessing the relationship between these different restoration treatments and the patterns of diversity and ecosystem functioning that develop among treatments and the relationships between ecosystem structure and function that develop in maturing ecosystems regardless of treatment (Petersen 2002). In addressing these questions, research at the Jones wetlands speaks to the gaps in knowledge confronting general restoration research.

Restored Attributes. Ecosystem attributes targeted for restoration at the Jones wetlands are similar to those that Mitsch and his colleagues at the Ohio State University have chosen in designing their constructed wetlands (Mitsch et al. 2005, Fink and Mitsch 2007). These attributes include both structural (macrophyte diversity and arrangement, water quality) and functional (invasibility, stability, nutrient cycling, hydrology, community metabolism) parameters.

Restoration Methods. The Jones wetlands is also part of a larger trend in research addressing the effectiveness of various restoration methods. These methods include planting with local species (Harter and Mitsch 2003, Seabloom and Van der Valk 2003), restoration of pre-drainage hydrology (McKenna 2003, Mulhouse and Galatowitsch 2003), and control of invasive species (Zedler and Kercher 2005). Research at the Jones wetlands will help to determine the effectiveness of these methods in inducing restoration success and help restoration ecologists develop replicable methodologies. Furthermore,

research at the Jones wetlands addresses the impacts of three different planting strategies (described below) on wetland ecosystem functioning. Few studies adopt this approach; most focus on the functional effects of restoration compared to either "natural" reference sites, unrestored but degraded ecosystems, or one other restoration strategy. The Jones wetlands project is designed to assess multiple restoration strategies.

The Jones wetlands are also maintained to facilitate long-term monitoring. Assessment of water quality, plant community diversity, and invasive species impacts is currently in its fifth season. Future research will be able to address the long-term outcome of the restoration project as it relates to the different restoration treatments employed at the farm.

Structure and Function. The Jones project is also designed to create a stable marsh ecosystem suitable for long-term ecological study, including research into the relationship between biodiversity and ecosystem function. The biodiversity-ecosystem functioning ("BEF" *sensu* Naeem [2006]) perspective addresses the degree to which biodiverse ecosystems display augmented rates of productivity, complexity, or supply of ecosystem services. The consideration of the structure-function relationship in restoration research is one example of the relevance of restoration experiments to the advancement of ecological theory. I will discuss the application of BEF to research at the Jones wetlands further in Chapter 3. It is also our objective to assess the relationship between these methodologies and the objective of functional success (Kentula 2000). Functional success corresponds to the degree to which a restored ecosystem displays stable ecosystem functioning analogous to that which would be observed in a similar natural ecosystem. Functional success may be a more useful objective for restoration relative to approaches that seek to produce sites that mimic reference sites in physical or chemical parameters of ecosystem quality (Lougheed et al. 2007).

E. Experimental System: The Jones Farm Wetlands

My research took place between June 2004 and August 2007 at the restored wetlands on the south side of the George Jones Memorial Farm ("the Jones Farm") in New Russia Township, located in Lorain County, Ohio. The Jones Farm is owned by Oberlin College and administered by the New Agrarian Center, a non-profit organization that promotes sustainable agriculture, ecological design, and food security in northeast Ohio. (The NAC, previously the Ecological Design Innovation Center (EDIC), maintains a website at http://www.gotthenac.org/.) The NAC has developed the Jones Farm as a model for sustainable, mixed land use. The Jones wetlands is thus one component in a larger landscape that integrates agriculture, education, small-scale industry, community space, and nature preservation as well as ecological research (Masi 2000). I will draw heavily from Smith's (2006) work in my description of the experimental setup of the Jones wetlands.

In July, 2003 personnel from Oberlin and EDIC established the research site on a field at the southern edge of the Jones Farm that had previously been used for conventional corn and soybean agriculture. The tile drainage system in the old field was disrupted and earthmovers were used to create six individual marshes ("wetlands") in a row. Individual wetlands are referred to as Cells 1-6; Cell 1 is the westernmost cell (Fig. 1). The wetlands were designed to be as uniform as possible in basin morphology since the development of upland, emergent, floating-leaved, or submerged plant communities

is integrally linked to wetland depth (Keddy 2000). Each wetland is 60 meters long from North to South and 30 meters wide from West to East. Each is approximately .18 ha (.44 acres) in area. Wetlands are bordered by earthen walls on their northern, eastern, and western edges and consist of deep (roughly 1.5 meters) northern ends that rise fairly sharply before sloping gently toward seasonally wet meadows on the southern ends (Fig. 2). Control boxes with adjustable weirs located in each wetland are used to control water levels, which have stabilized at relatively uniform depth with the exception of one wetland (Cell 4) that is generally 4-5 inches shallower than the others. The site is located in the Plum Creek Basin of the Black River Watershed, which drains into Lake Erie (Smith 2006). Each cell at the Jones wetlands is hydrologically isolated and all six cells together comprise a "closed" ecosystem, which does not allow water to leave the system. Relative to "open" systems that receive input from a large landscape and/or lose effluent to the landscape, the Jones wetlands can be easily manipulated through differential restoration treatments and management.

There is a permanent rebar grid laid across each of the six wetlands. This grid is used to reference specific points within cells (Fig. 3). The grid conforms to a standard Cartesian (x,y) coordinate system with seven rebar points spanning the West-East axis and seven spanning the North-South axis. Rebar columns are lettered A-G from West to East and rows are numbered one through seven from North to South. Points A-G are spaced five meters apart and points 1-7 are spaced 10 meters apart (Smith 2006). In July 2006, several rebar in each wetland were marked with orange and yellow foam buoys. A transit was used to locate all buoys within an individual cell at the same distance above the water. Buoys are used as reference points in photographs of the research site.

On 29 October 2003, four of the six wetland cells were planted with vegetative vegetative propagules and seeds (Smith [2006] offers greater detail). Cells 1 and 4 were left as the unplanted (UP) control. Vegetative vegetative propagules of 11 species and seeds of 11 species were introduced to wetlands 2, 3, 5, and 6 in a consistent planting scheme (Table A). The planted cells constitute the planted (PL) treatment group. Surveys of vegetation in all cells were conducted in the summers of 2004-2007. In the fall of 2004-6, vegetative propagules were planted at all locations within cells 2 and 5 where previously planted vegetative propagules did not survive. Cells 2 and 5 were thus replanted three times and comprise the high-intensity management (HI) treatment group while cells 3 and 6 (which were only planted once) constitute the low-intensity (LI) treatment groups. Planting treatments at the Jones wetlands were designed in order to facilitate comparisons between different restoration strategies. Natural recruitment of seeds and the innate potential of ecosystems for "self-organization" influence vegetation composition in both PL and UP cells (Odum 1989). While the "self-design" (UP) strategy limits human control over which species are introduced in a restored ecosystem, the "designer" (PL) strategy entails pre-restoration selection and post-restoration promotion of communities consisting of desired species (Mitsch 1993, Mitsch and Wilson 1996). The HI and LI treatments are designer strategies of two intensities. Replanting of vegetative propagules in HI cells took over 100 person-hours while LI cells were not replanted.

Two exotic plant taxa were targeted for removal during the study period (Ch. 4). Though exotic, non-invasive species may be useful components of ecological restoration (Ewel and Putz 2004), successful restoration of native plant communities is often

impossible without considerable control of invasive exotics and natives (Mulhouse and Galatowitsch 2003, Zedler 2003). I also trapped muskrats at the Jones wetlands during the study period in order to limit their capacity to tunnel between cells and to introduce non-experimental effects on the abundance and diversity of vegetation. Both of these perturbations would have interfered with the project's experimental design.

Chapter 2: Water Quality

Since June 2004, we have monitored water quality at the Jones wetlands, including dissolved oxygen (DO), turbidity, dissolved nitrate and nitrite ($NO_{2,3}$) and dissolved phosphate (PO_4).

A. Why Measure Water Quality?

Water quality monitoring permits the evaluation of the critical functional effects of restoration efforts. Natural wetlands display characteristic biogeochemical processes (Mitsch and Gosselink 2000). These processes are of interest to restorationists as indicators of functional success (Costanza and Daly 1992, Kentula 2000). Prevailing restoration techniques often aim to restore measurable patterns of ecosystem structure that give rise to function. In order to do this, restorationists place constructed wetlands at appropriate places in the watershed, restore historical hydrology and substrate, and replant desirable vegetation (Erwin 1990). Yet the degree to which these restoration methodologies can restore ecosystems that function like natural ecosystems is uncertain (Zedler and Kercher 2005). Water quality monitoring in restored wetlands allows for the appraisal of functional restoration success and quantification of the levels of ecosystem services that restored wetlands supply. Additionally, water quality monitoring is often included in post-restoration monitoring, especially of projects that seek to improve water quality in contaminated watersheds (Kentula et al. 1992, NRC 1992). Policymakers and site managers can use water quality data to evaluate the effectiveness of restoration and advocate for future projects. Wetland water quality monitoring thus provides critical information about restoration success and restored ecosystem function.

Water quality research conducted at restored sites is also of broader theoretical significance. An understanding of the dynamics present in restored systems can help answer questions of general ecological interest by providing information about the biogeochemical dynamics characterizing wetlands (Ehrenfeld and Toth 1997). Indeed, the application of findings from restored systems to questions of ecological importance demonstrates the role of these systems as potential testing grounds for ecological theory (Bradshaw 1987). Restoration sites may be especially useful given the greater degree to which it is possible and acceptable to exercise experimental manipulation in restored ecosystems than is the case in less human-modified ecosystems. The persistent deficit of research addressing the causal factors and dynamics of wetland ecosystem function in both natural and modified systems constitutes a strong argument for restored wetlands as sites both of basic ecological research and of specific inquiries involving questions of restoration methods and outcomes (Kusler and Kentula 1990).

B. Study Objectives

Water quality research at the Jones wetlands is designed to address basic questions about wetland ecosystem function, strategies for monitoring water quality, and the relative effectiveness of different restoration methods in providing for functionally successful restoration. These two goals are made possible through weekly analysis of dissolved oxygen, turbidity, dissolved nitrogen, and dissolved phosphorus (as described below) during the growing season and through the experimental design of the Jones wetlands.

Research at the Jones wetlands addresses basic questions about biogeochemical and energetic dynamics in hydrologically closed, herbaceous marshes. To characterize these process in marshes like those at the Jones Farm, it is necessary to measure community metabolism and turbidity as well as to identify which nutrients are taken up or released in the system, how they are transformed between organic and inorganic forms, how they are stored or lost, and at what rates theses processes occur. Answering these questions allows for an assessment of the types and magnitudes of ecological processes occurring in a wetland. The Jones wetlands is hydrologically isolated, meaning that influx and efflux of water and nutrients is minimal. This facilitates detailed study of the autochthonous processes of productivity, respiration, and nutrient cycling occurring in the wetlands.

It is also our objective to assess the validity of the methods used for monitoring water quality. Manuals of methods for water quality analysis often provide specific technical instructions for laboratory analyses of water samples and general discussions of water quality sampling methods, but fail to address specific patterns or schemes for practical sampling within wetlands (Meybeck et al. 1996, Clesceri et al. 1998). Though restoration handbooks highlight the need for post-restoration water quality monitoring and may describe parameters to be measured, they very rarely discuss sampling extent, intensity, or duration (NRC 1992). As a result, there is a need for formal analysis of the efficacy and consistency of the sampling strategies and experimental design employed in studies of water quality. Studies such as those performed by Sherwani and Moreau (1975), Pearson et al. (1987), and Hirsch (1988) provide useful insights for restorationists who are attempting to plan projects in a way that is conducive to post-restoration monitoring. Analysis of the consistency and interpretive potential of water quality data collected at the Jones wetlands will provide validation for current monitoring methodologies.

Finally, as noted in Chapter 1, the Jones wetlands project is designed to characterize the relationship between restoration methodologies and ecosystem function. Biogeochemical processes and community metabolism are forms of ecological function that can be assessed through water quality monitoring. Understanding the effects of different restoration treatments on ecological functioning will allow restorationists to make more informed choices about what restoration strategies to pursue. Therefore, water quality monitoring at the Jones wetlands and the experimental design of the system have provided the opportunity to improve our understanding of how to achieve functional restoration success.

C. Parameters of Water Quality

Measurements of dissolved oxygen (DO), turbidity, dissolved nitrate and nitrite ($NO_{2,3}$), and dissolved phosphate (PO_4) were conducted within and among cells and experimental treatments during the study period.

Dissolved Oxygen. Point measurements of dissolved oxygen in wetlands indicate a balance between several ecosystem processes and implicitly inform an understanding of the role of local emergent and submerged vegetation in structuring these processes (McKenna 2003). A single DO measurement represents the relative magnitudes of processes that consumer oxygen versus processes that return oxygen to the water column (Chimney et al. 2006). DO also is generally vertically stratified in marshes and other stillwater wetlands; it is highest at the surface and gradually declines with depth. Wetland soils are often hypoxic or anoxic (Chimney et al. 2006). Though internal biogeochemical processes often determine short-term DO dynamics, allochthonous DO entering wetland ecosystems, either from the atmosphere or from inflowing waters of different oxygen concentration can also influence the amount of oxygen in the water column. Materials within inflowing waters are also critical to oxygen dynamics. Influent rich in dissolved nutrients can lead to eutrophication, resulting in anoxia. DO measurements from hydrologically “open” wetlands should therefore be interpreted in a way that considers allochthonous nutrient inputs. Most superficially, then, single DO measurements in closed bodies of water reveal the relative importance of oxygen-liberating and -disseminating processes (photosynthesis, diffusion) relative to oxygen-binding (respiration, oxidation) processes (Mitsch and Gosselink 2000). High levels of DO may indicate a net autotrophic system and low levels indicate a net heterotrophic system. Ultimately, fluctuations in DO are constrained by the availability of the basic substrates of metabolism: energy (in the form of sunlight), nutrients, and molecular oxygen. The availability of these substrates controls periodic change in wetland DO levels.

DO levels vary in consistent ways in wetlands over diel and seasonal periods. DO generally increases from sunrise through late afternoon, peeking as the sun goes down. Oxygen is consumed at night, and DO reaches its lowest level shortly before sunrise. The capacity of water to hold DO also varies with salinity, pressure, and temperature. When the first two variables are constant, the capacity of water to hold DO decreases as temperature increases. Thus climate and weather partially influence seasonal and diel patterns of DO. These patterns of DO and other water quality parameters may also change for biological reasons. For instance, DO levels may change with vegetation patterns, either those associated with naturally induced succession or with the maturation of a restored site. Dampening of seasonal fluctuations in DO, for instance, suggests greater system stability (Mitsch et al. 2005). Expectations of how DO dynamics fluctuate with biological and physical conditions in wetlands thus provide one context for interpreting point DO measurements.

Consideration of the floristic setting of point DO measurements allows for additional interpretation. It is possible to assess the relative contributions of metabolic processes in controlling DO levels either through diel and seasonal comparisons of or through single point measurements placed in the context of vegetation patterns. Often, plant communities in marshes are characterized by a mix of species with emergent, floating-leaved, and submerged growth habits as well as algae (Hamilton et al. 1995,

Chimney et al. 2006, Rose and Crumpton 2006). These communities are subject to complex DO dynamics. Emergent plants exchange oxygen directly with the atmosphere while still contributing organic carbon inputs to their surroundings. Stands of highly productive, emergent vegetation thus produce lower DO levels in surrounding water by limiting light availability and mixing of the water column and by providing a litter substrate for high levels of microbial decomposition (Chimney et al. 2006). Conversely, communities characterized by submerged macrophytes and algae directly introduce oxygen into the water column, leading to DO saturation (Rose and Crumpton 1996). Floating-leaved plants are intermediate between submerged and emergent macrophytes; they exchange gases with the atmosphere but also limit light availability and mixing. Floating-leaved plants do not deplete DO in the water column as much as emergent plants do and submerged macrophytes and algae exert the strongest saturating effect. Thus it appears that the net effect of vegetation pattern on metabolic processes that produce or consume oxygen may play a dominant role in generating the patterns observed through either instantaneous or continuous sampling of DO.

DO dynamics also affect the biogeochemical cycling of key nutrients by altering the redox potential in wetland water columns and soils. Anoxic (reducing) conditions favor the denitrification of soluble, inorganic forms of nitrogen into atmospheric nitrogen, which is released to the atmosphere. Likewise, phosphorus-containing compounds are more accessible to macrophytes under reducing conditions; when oxidized, these compounds become insoluble through precipitation with ferric iron, calcium, and aluminum (Mitsch and Gosselink 2000). Understanding DO flux in wetlands therefore complements monitoring of key wetland nutrients

Turbidity. Measurements of turbidity quantify the degree to which suspended particles in a given water sample scatter or absorb light (Harris et al. 2007). Turbidity is fairly easy to measure and can serve as a proxy for the more laborious process of measuring total suspended solids or chlorophyll content of water samples (Mitsch et al. 2005, Harris et al. 2007). Turbidity in wetland water samples is the result of several ecosystem processes. Bioturbation by fish or muskrats suspends solids and can lead to high turbidity, while vegetation can either raise turbidity through increased deposition of decomposed organic matter or lower it by shielding the water column from turbation and fostering deposition of suspended sediments that enter (Harter and Mitsch 2003, Fink and Mitsch 2007). Abiotic turbation processes such as high influent velocity, intense rain, and wave action can likewise increase turbidity in natural and constructed or restored wetlands (Bachmann et al. 2001, Harris et al. 2007). Turbidity in wetlands located downstream from heavily human-modified landscapes is also dependent on management decisions made further upstream in the watershed (Harris et al. 2007). Turbidity readings are therefore the product of a number of biotic and abiotic ecosystem processes.

Because decreasing turbidity (and thereby increasing water clarity) may be a goal for restored or constructed treatment wetlands, wetland managers are often concerned with the role of wetlands in altering the turbidity of influent. Many treatment sites are designed to remove suspended particles from influent as it flows through surficial or sub-surface constructed wetlands (Mitsch and Gosselink 2000). Wetlands can be effective at sequestering solids if bioturbation within the constructed wetlands and high nutrient loading do not result in increasing turbidity (Gu et al. 2006, Fink and Mitsch 2007). However, in the absence of long-term accretion of organic matter, accumulation of

sediments over time can reduce the effectiveness of mature wetlands at performing this service (Mitsch et al. 2005).

Dissolved Inorganic Nitrogen. Monitoring of dissolved inorganic nitrogenous species provides information about several of the biogeochemical processes that characterize wetland ecosystems. Though phosphorus is more likely to limit vegetative growth in freshwater marshes, nitrogen, depending on its oxidation state, solubility, and availability to plants, plays a significant role in several ecosystem processes (Valiela et al. 1997). Organic nitrogen in wetlands, whether imported from allochthonous sources or liberated from autochthonous sources, is mineralized through ammonification and nitrification into nitrates and nitrites, which are fairly soluble because they are negatively charged and thus less likely than ammonium to adhere to negatively charged soil particles (Mitsch and Gosselink 2000). Nitrates and nitrites, along with some ammonium, constitute the bulk of dissolved inorganic nitrogen (DIN) in a wetland water quality sample (Gu et al. 2006). Because preliminary analysis of ammonia concentrations suggested that ammonium levels are quite low, I will treat measurements of dissolved $NO_{2,3}$ as interpretively comparable to measures of total DIN presented in the literature. DIN is removed from the water column through biological uptake by plants or by bacterially mediated denitrification (Poe et al. 2003). The latter process results in removal of nitrogen from the ecosystem, while the former only constitutes cycling of nitrogen within the system. Measurements of DIN, then, reflect relative rates of processes creating soluble nitrogen species (allochthonous nitrogen inflow, nitrogen mineralization from organic matter) and removing nitrogen (integration into plant tissues, incorporation into sediment, denitrification).

As is the case with other water quality parameters, the ecosystem processes controlling DIN levels are themselves the result of structural patterns. Bacterial communities mediate the key processes of ammonification, nitrification, and denitrification; the activity of these communities depends on a wide range of biotic and abiotic factors. Poe and colleagues (2003) found that the microbial communities responsible for denitrificaiton in a constructed treatment wetland were sensitive to variables such as temperature and nitrogen loading. They also noted considerable spatial heterogeneity in rates of nitrogen removal through denitrification. DIN removed from the water column but retained within the wetland ecosystem is frequently stored in sediments or as organic matter in vegetation (Heath 1992). As such, vegetated wetlands are temporary nitrogen sinks. They seasonally cycle nitrogen through inorganic and organic forms and exhibit considerable potential for storage of nitrogen introduced from the wider landscape (Heath 1992, Sartoris et al. 2000, Mitsch et al. 2005, Fink and Mitsch 2007). Yet uptake by aquatic macrophytes constitutes only temporary nitrogen storage. While nitrogen lost to denitrification or runoff is permanently removed from wetland ecosystems, nitrogen incorporated into organic tissues is only seasonally removed from the water column. Permanent removal of nitrogen by wetland systems is dependent on patterns of hydraulic mixing and microbial community development rather than on macrophyte productivity (Sirivedhin and Gray 2006). DIN measured from water quality samples collected longitudinally suggest the relative magnitude of the processes structuring such cycles.

Levels of DIN are important in the monitoring of ecosystems designed to process high-nutrient effluent from municipalities and agricultural sites (Romero et al. 1999,

Sartoris et al. 2000). Organic nitrogen is a major component of the waste matter and agricultural runoff that are fed into treatment wetlands (Mitsch and Gosselink 2000, Zedler 2003). Promoting the transformation of this organic nitrogen into DIN and its subsequent sequestration or dispersion through denitrification is one of the central challenges in the creation of treatment wetlands. The discharge of nitrogen-rich waste into the landscape can result in ammonia toxicity, nitrate contamination, and the generation of hypoxic dead zones (Mitsch and Gosselink 2000, Sartoris et al. 2000, Zedler and Kercher 2005). Management of DIN, especially in systems subjected to intentional or unintentional nitrogen loading, is therefore central to wetland restoration and mitigation efforts.

Dissolved Inorganic Phosphorus. Soluble reactive phosphorus (SRP) is the most comprehensive measure of phosphorus in an aquatic system. SRP is an aggregate measure of levels of inorganic orthophosphates and organic, phosphorus-containing species. Yet most SRP assays report levels that are determined mostly by the presence of orthophosphates rather than by organic phosphorus (Clesceri et al. 1998). Therefore, I will equate PO_4 with dissolved inorganic orthophosphates (DIP).

Measures of DIP are important because phosphorus is frequently the growth-limiting nutrient in wetland systems; this is especially the case in isolated freshwater marshes (Mitsch and Gosselink 2000). DIP can enter the water column through either autochthonous or allochthonous processes. DIP is produced as the product of bacterial decomposition of organic phosphorus introduced into wetlands either from allochthonous or autochthonous sources. Phosphorus can also enter wetlands as DIP in allochthonous influent. Levels of DIP can be interpreted as an aggregate representation of the rates of allochthonous phosphorus influx and autochthonous conversion of organic phosphorus into orthophosphates relative to the rates of processes that transform these phosphates into other compounds or render them insoluble. Phosphorus recently liberated from organic or sedimentary sources as orthophosphate can be transformed into more refractory forms in several ways. Uptake by periphyton or macrophytes constitutes a short-term storage in organic material which structures annual patterns of intra-system phosphorus cycling in wetlands (Heath 1992, Vaithiyanathan and Richardson 1997, Noe et al. 2001). More important to long-term phosphorus dynamics is storage of phosphorus in soil substrates through deposition of sediments, sorption of phosphates, and (in peat bogs) peat accretion (Noe et al. 2001, Bruland and Richardson 2006). Measurements of DIP in wetlands, if taken during the growing season, are especially relevant as measures of the ecosystem patterns giving rise to short-term storage of phosphorus.

Because it is a nutrient introduced in large concentrations across agricultural landscapes, control of phosphorus dynamics is often an objective of wetland restoration and construction projects. Yet few studies describe the ecosystem processes that sequester phosphorus in freshwater wetlands (Richardson and Craft 1993). Constructed wetlands are designed to maximize phosphorus retention by facilitating both long-term storage in soils (via settling and sorption) and short-term storage in macrophytes and periphyton (via uptake of DIP and soil phosphorus) of dissolved inorganic and organic phosphates (Noe et al. 2001, Zedler and Kercher 2005). Constructed wetlands can serve as effective sinks of phosphorus, but their capacity to do so may be reduced as they reach phosphorus saturation over time and begin exporting phosphorus stored over the short term in decomposing sediments (Mitsch et al. 2005, Gu et al. 2006, Fink and Mitsch

2007). Because supply of phosphorus limits the growth of macrophytes, constructed wetlands with low or no influx of phosphorus species are expected to show low DIP.

D. Methods of Water Quality Analysis

Routine Weekly Water Quality Sampling. Since June 2004, Oberlin students have collected weekly water quality data at the Jones wetlands during the growing season. Though starting and ending dates of collection have varied from year to year, we have consistent weekly water quality data covering the months of June to October from 2004 to 2007. Parameters of water quality measured weekly are temperature, DO, turbidity, and concentrations of dissolved chloride, nitrate, nitrite, phosphate, and sulfate. All weekly monitoring occurs at a single point in each cell: the "sampling point." The sampling point is located at a similar location in each cell (in each cell's most northeast or northwest corner) (Fig. 3). The methods used for assessment of water quality are based on the American Public Health Association's *Standard Methods for the Examination of Water and Wastewater* (20th ed., 1998) and Petersen's Methods of Aquatic Ecosystem Analysis (2007).

Temperature and DO were measured using a model 550A YSI probe *in situ* in the permanently filled northern portion of each wetland basin (YSI Inc.; Yellow Springs, OH). Research personnel calibrated the probe before use per the manufacturer's instructions. Personnel moved the probe roughly ½ ft. per second over a patch of water about six inches in diameter at the sampling point. The probe was submerged beneath the water's surface, but remained in the upper 5 inches of the water column. The probe was allowed to reach a stable reading (usually in 30 seconds) and DO and temperature were recorded. Temperature was measured in degrees Celsius and DO was measured in mg of dissolved oxygen per liter of water. The probe's membrane was changed three to four times a year. Measurements of DO were generally taken either in the midmorning (~10:00) or mid-afternoon (~18:00), but varied during the study period.

Water samples were collected at the wetlands and transported to John Petersen's laboratory at Oberlin College for assessment of turbidity and dissolved ion concentrations. Prior to sampling, research personnel transported acid-washed 0.5 L Nalgene Labware sampling bottles to the Jones wetlands (Thermo Fisher Scientific, Inc.; Waltham, MA). Bottles were rinsed twice with wetland water before they were filled with water at the sampling point and capped. Bottles were attached to a plastic pole, which allowed for sampling at the sample site with minimal disturbance and induction of turbidity. All water samples were filtered and either assayed or frozen in 25 mL vials as soon as possible. Samples that were processed more than two hours after collection were refrigerated until processing could occur. Samples were filtered through 47 mm glass microfibre filters (North Central Laboratories; Birnamwood, WI). Filtered samples were either loaded directly into the ion chromatograph or frozen.

Turbidity assays were conducted using an 890 nm infrared light turbidity sensor (Vernier Software and Technology; Beaverton, OR), which reported turbidity in Nephelometric Turbidity Units (NTU). NTU ranged from 0-100 for most wetland water samples, with 0 representing the level of light scattering and absorption typical of distilled, deionized water. The turbidity sensor was calibrated before every use with distilled water and a 100 NTU StablCal formazin standard. Calibration and analysis were

conducted per the manufacturer's instructions. Solids in water samples were resuspended through four gentle inversions of the sample bottle prior to turbidity assay.

Concentrations of dissolved ions were measured using an ASRS-II Suppressed Conductivity Ion Chromatograph (IC) (Dionex Corporation; Sunnyvale, CA). The column used was an IONPAC AS9-HC Analytical (4 by 250 nm) anion column. Eluent flow was at 1.0 mL/min and a 9.0 mM Sodium Bicarbonate eluent was used. The procedure followed that described in the Petersen lab's "Basic Operation of the Dionex Ion Chromatograph (IC)," which was modeled after the operation instructions provided by Dionex (Petersen 2007). Output data from the Chromeleon 6.80 program included an export summary with "peaks" for all recognized ions (Dionex Corp.). I reviewed peaks and manually corrected them when the Chromeleon software failed to recognize or misread a peak. Chromeleon integrated the area under each peak and presented nutrient concentrations in mg of dissolved nutrient per liter of water. Every run of the IC included a set of five standards, which calibrated the machine to appropriate levels of ion concentration. The IC was calibrated to measure dissolved chloride, nitrate, nitrite, phosphate, and sulfate. Standards used were 180.0 mg/L chloride, 2.0 mg/L nitrite, 120.0 mg/L nitrate, 20.0 mg/L phosphate, and 180.0 mg/L sulfate. Samples were not diluted.

A second set of filtered water samples was also frozen for future ammonia assays using a Thermo Orion 720 ammonia probe (Thermo Fisher Scientific, Inc.). Standards used contained ammonia concentrations of 0.2, 1.0, 5.0, 10.0, and 20.0 mg/L. I unfroze and assayed several dozen weeks' collections of water samples taken between 2005 and 2006 years for ammonia and found very low levels of ammonia in most samples. As a result, I did not assess ammonia level in all collected samples. Additionally, frozen samples were not kept for ammonia analysis following numerous collections in 2004 and 2005.

One-Time assessment of the validity of water quality sampling procedure. In order to assess spatial heterogeneity in water quality at the site, I collected water quality data and water samples at 48 points in the Jones wetlands on the mornings of 16-17 July 2007. Additionally, I hoped to validate our sampling methodology through intensive sampling in a small area. I collected data for the same parameters assessed on a weekly basis since June 2004.

A colleague assisted me in collecting samples on the first day of this assessment project. We collected water quality data and samples on 16 July 2007 from 36 points at the Jones wetlands. We sampled at six points in each of the six cells. Point designations refer to the Cartesian coordinate system mapped out in each cell by rebar. Within each cell, we collected one surface sample and one depth sample at three locations (giving us six collection points). Each of the three locations was in the middle of one of the sampling transects described by Smith (2006) and used annually in the biodiversity survey (Smith 2006). Specifically, we sampled from the center of quadrats 1, 2, and 3 (Fig 4a). For each of these quadrats, we located the center of the quadrat by eye and recorded data and took samples from the water's surface and from the deepest point possible without hitting sediments. We took the depth of each point using a t-shaped PVC device designed to allow for a consistent measure of depth in a soft-bottomed water column. We pressed the lateral bar as far into the sediments as we could and measured the amount of the vertical bar submerged with a tape measure. We took dissolved oxygen and temperature readings at surface and depth using the YSI probe. We took

water samples by hand at the surface and used a hand-operated pump to get depth samples. To obtain depth samples and to lower the YSI probe to depth, we used a PVC pipe to extend the probe and pump inflow tubing as deep as possible without hitting sediments.

I collected water quality data and samples on 17 July from 12 points within cell 2. Because data collected from a single cell is spatially autocorrelated to some degree, tests for intra-cell variation may be of "pseudoreplicated" rather than replicated data points (Hurlbert 1984). Given this concern, I distributed sampling points across the deepwater portion of each cell and have limited the degree to which I subsequently interpreted the findings generated by these water quality data. I only collected samples and water quality data from surface points. Once again, I sampled from the NW, NC, and NE quadrats. However, I sampled from four points from within each quadrat and only from one cell in order to provide for more intensive sampling. The four points were located at the same four relative positions in each quadrat. For each quadrat, I went to each of the rebar points delineating the quadrat, measured 6 feet "in" to the quadrat (East/West) and collected data and water from that point. Thus, for the NW quadrat, I collected data from 6 ft. east of A1, 6 ft. west of C1, 6 ft. east of A2, and 6 ft. west of C2 (Fig 4b). I did this for all three northern quadrat in Cell 2. I named each of my sample points according to its quadrat and location relative to the nearest corner. The four points just described would be known as NWNW, NWNE, NWSW, and NWSE respectively. I used the YSI probe to collect DO and temperature data from the surface, collected surface water samples by hand, and measured depth as on 16 July.

I filtered all water samples collected on 16-17 July and ran them through the IC within 12 hours of collection. I measured turbidity of all samples within 48 hours of collection. Samples not being actively processed were refrigerated. Otherwise, I adhered to the same laboratory methods used to process samples collected on a weekly basis.

Data Analysis. I conducted one-factor ANOVAs using Microsoft Excel to test for significant differences: among points within the same cell, mean or median measures from different cells, mean or median measures from different treatments, mean seasonal measures, and mean annual measures. To assess relative seasonal (intra-annual) change in DO, I calculated the mean May to August drop in DO for each cell by averaging together the drop from each of year of the study. I then aggregated these mean values into PL and UP groups and compared them using an ANOVA. This process provided low statistical power (d.f. = 1,4) but avoided pseudoreplication. To see whether patterns in seasonal DO levels were due to biological or physical factors, I calculated the expected saturation point for DO for each month based on the assumption of constant salinity and pressure and using average temperature recorded for each month (following Benson and Krause [1984]). To assess levels of $NO_{2,3}$ in the entire system (all six cells), I used an ANOVA to compare $NO_{2,3}$ values from each of the four years of study period, treating mean values from each cell as replicates (d,f = 3,20).

For all significance tests described below, I note the F-value, between- and within-group degrees of freedom (respectively), and p-value. The alpha significance value for all tests was .05. I produced all regressions and graphical figures using Microsoft Excel and edited some graphs using Microsoft Powerpoint.

E. Results

Weekly Water Quality Sampling. Temperature varied seasonally across all wetlands, but did not differ significantly among cells or treatments. I did not perform analysis of dynamics in concentrations of dissolved chloride or sulfate. Trends in DO, turbidity, dissolved nitrate and nitrite, and dissolved phosphate are discussed below.

Dissolved oxygen ranged from less than one mgL^{-1} to over 14 mgL^{-1} and did not show long-term directional change in any of the cells or treatments between Summer 2004 and Fall 2007. Rather, levels of DO were characterized by an annual cycle that repeated itself at a slightly different magnitude every year. Figure 6 represents a typical annual cycle of DO. DO was generally highest in May and June, when sampling began. It fell to its lowest point in late August and September and generally rose to levels characteristic of May/June as the growing season ended. Median percentage DO saturation for all six cells followed the same pattern (Fig. 7). The average May to August drop displayed by Planted (PL) cells during every year of the study period significantly exceeded the drop in unplanted (UP) cells ($F = 11.09$; d.f. = 1,4; $p = .03$)

Turbidity did not show directional or regular change during the entire course of the study period; measured turbidity ranged from 0 to well over 100. Turbidity levels were unusually high in cell 5 and were generally highest in cells 5 and 6. Turbidity was not significantly higher in PL cells than in UP cells (in 2007: $F = .77$; d.f. = 1,4; $p = .43$); difference in turbidity between treatments shrunk even further when cells 5 and 6 were excluded (in 2007: $F = .26$; d.f. = 1,2; $p = .66$). These results suggest that consistently high turbidity in cell 5 may distort analyses of turbidity level that aggregate cells by treatment. Comparisons of turbidity between treatments are displayed in Figure 8.

Dissolved nitrate and nitrite ($NO_{2,3}$) varied from year to year (Fig. 9). In the first year following restoration (2004), dissolved $NO_{2,3}$ was low, averaging .05 mgL^{-1} across all cells. In 2005, dissolved $NO_{2,3}$ increased to an average of .13 mgL^{-1} before subsiding in 2006-7 to frequently undetectable levels. Differences between years, with annual averages for each cell treated as replicates, were highly significant ($F = 120.79$; d.f. = 3,20; $p = 5 \times 10^{-13}$).

Dissolved phosphates (PO_4) concentrations were low (never exceeding .07 mgL^{-1}) and erratic – probably near detection capacity. Cells 5 and 6 showed moderately higher levels of dissolved PO_4, but statistically meaningful treatment differences across entire seasons or years were not detectable.

One-Time Water Quality Sampling. Comparison of parameters measured on the same day from all wetlands (with measurements from different locations in the same cell treated as replicates) revealed no significant difference between cells in turbidity and dissolved $NO_{2,3}$. Cells were significantly different in DO ($F = 2.99$; d.f. = 5,30; $p = .03$) and dissolved PO_4 ($F = 9.49$; d.f. = 5,30; $p = 1.7 \times 10^{-5}$). Treatment groups did not differ significantly in dissolved PO_4, but mean DO readings were significantly higher in UP than in PL cells ($F = 9.53$; d.f. = 1,4; $p = .04$). Dissolved nutrient levels ($NO_{2,3}$ and PO_4) were not significantly different between surface and depth samples with all samples treated as replicates. However, DO was significantly higher ($F = 10.34$; d.f. = 1,10; $p = .009$) in surface samples than in depth samples while turbidity was significantly lower at surface relative to depth ($F = 11.19$; d.f. = 1,10; $p = .007$). The data used in the hypothesis tests described in this paragraph are pseudoreplicated to some degree. As noted above, the design of the one-day sampling study was organized to provide

information about uniformity within and among cells on a single day. Within-cell sampling was stratified to minimize pseudoreplication.

Comparison of parameters measured intensively in cell 2 with all 12 sampling points treated as replicates (d.f. = 2,9) revealed no significant difference in DO ($F = .90$, $p = .44$), turbidity ($F = 1.78$, $p = .22$), dissolved $NO_{2,3}$ ($F = .31$, $p = .74$), or dissolved PO_4 ($F = 2.78$, $p = .11$).

F. Discussion

My analysis of water quality at the Jones wetlands since June 2004 revealed several statistically significant trends, some of them related to changes over time and others reflecting differences among the experimental treatments over the course of the study period. The data collected also validate our methods for assessing water quality at the Jones wetlands.

Though DO dynamics at the Jones wetlands are probably influenced by seasonal temperature change, the data suggest that plant community dynamics also affect DO levels. Across all cells and years, DO fell during the early part of the season, bottomed out in July or August, and rose back to levels typical of May or June by September. Seasonal changes in temperature certainly influenced this trend. Water's capacity to hold dissolved oxygen is inversely related to its temperature and DO levels at the Jones wetlands hold to this relationship. DO decreased when water temperature increased and reached its lowest point just as water was warmest (Fig. 6). Yet, when I divided these DO values by the expected DO level at saturation, this trend persisted (Fig. 7). This suggests that the pattern in DO I observed is not due entirely to physical factors. Rather, phytoplankton, submerged aquatic vegetation, and macrophyte productivity and the resultant, seasonal production of organic matter may also play important roles in determining DO (Rose and Crumpton 2006). Several studies report wide variability in DO in wetlands, with highest (and most variable) values occurring in open waters, fairly high DO in beds of submerged vegetation, and hypoxia or anoxia in beds of emergent and floating-leaved vegetation when season, time of day, and weather are treated as constant. The presence of emergent vegetation may depress DO through several mechanisms. These include shading of highly productive submerged algae, prevention of mixing of the water column, provision of a substrate for heterotrophic bacteria, and release of oxygen produced through photosynthesis into the air rather than the water column (Hamilton et al. 1995, Chimney et al. 2006, Rose and Crumpton 2006). The Jones wetlands are very productive and feature dominant stands of emergent macrophytic growth. If this vegetation controls annual DO dynamics, then DO would, like temperature, inversely correlate with emergent macrophyte productivity. Though some areas where I monitored DO were dominated by algae and submerged macrophytes rather than emergent vegetation, the annual trend in DO that I observed tracks the productivity of the emergent community, reaching its lowest point in the middle of the growing season and rising as senescence begins to occur. Thus, it appears that DO may be determined by both physical and biological factors.

Future work should attempt to assess the relative contributions of biotic mechanisms in shaping DO dynamics at the Jones wetlands. *In situ* assays of biological oxygen demand (BOD) using light and dark bottles would measure rates of organic respiration in the water column, allowing for assessment of the levels of reactive carbon

in the water column (as opposed to the sediments). These assays could be carried out at different temperatures in order to determine the relationship between temperature and rate of respiration. Increased BOD associated with higher temperatures would suggest that DO dynamics at the Jones wetlands are influenced by plankton community metabolism in the water column. Additionally, comparisons of DO and BOD from wetland zones dominated primarily by emergent, floating-leaved, and submerged macrophytes could be used to assess the relationship between plant community growth form and system metabolism.

The deepwater zone of PL cells dropped much more in DO than did that UP cells between May and August. This significant difference accords with the postulated role of vegetation as a factor controlling DO levels, given findings about plant community diversity in the Jones wetlands. In Chapter 3, I will show that plant community diversity is higher in PL cells relative to UP cells. A more diverse plant community may be more productive and cycle nutrients faster. High rates of emergent plant productivity in these cells could contribute to higher amounts of decomposing biomass at the start of the growing season, which would drive a more extreme drop in DO than is observed in less productive UP cells. It is therefore reasonable to expect that the more diverse assemblages of emergent macrophytes present in PL cells would deplete intra-cell DO levels to more quickly than would less diverse communities in UP cells. Shading from the more diverse and dense emergent canopy would also contribute to the observed more extreme drop in DO in PL cells. The finding that planted cells show a greater annual fluctuation in DO than do unplanted cells supports my hypothesis that more diverse cells will display higher ecosystem function. It *is* possible that UP cells are less productive than PL cells and that they therefore respire less and have higher DO. Also, my comparisons of change in DO between treatments have fairly low statistical power. Thus, they are only suggestive rather than definitive.

Turbidity appears to be heavily determined by levels of muskrat activity, and, perhaps, by planting treatment. I have consistently witnessed high levels of muskrat activity in cells 5 and 6. Visible signs of muskrat activity include massive reductions of standing emergent vegetation, “muskrat trails” in submerged and floating-leaved vegetation, extensive construction of muskrat lodges and tunnels, and sightings of animals during daylight hours. I offer these observations as an explanatory mechanism for the high levels of turbidity noted in the eastern cells.

Significantly higher turbidity in planted cells relative to the unplanted cells could be a function of experimental planting treatment or of a non-experimental prevalence of muskrat activity. Planted cells are more diverse than the unplanted cells, and so may show higher rates of primary productivity and decomposition of organic matter or may harbor species that are preferred by muskrats. Harter and Mitsch (2003) note that dense macrophytic cover may either reduce or increase turbidity. At the same time, higher levels of turbidity observed in PL cells may be a function of non-experimental muskrat activity in two of these cells (and low muskrat activity in both UP cells). High levels of muskrat-induced bioturbation constitute a non-experimental effect on ecosystem structure that may confound statistical analysis of turbidity levels between treatments.

Dissolved $NO_{2,3}$ and PO_4 levels suggest that the planting treatments have not had much of an effect in what remain nutrient-poor (oligotrophic) systems. Each cell receives influent from a small watershed of only a few hundred square meters and none of these

watersheds support crops. Levels of dissolved $NO_{2,3}$ and PO_4 are consistently comparable with measured nutrient levels in restored wetlands similar to the Jones wetlands in geographic location and/or hydrology (Heath 1992, Poe et al. 2003, Gu et al. 2006, Fink and Mitsch 2007). In the growing season following restoration (2004), cells showed low levels of dissolved $NO_{2,3}$. Given the paucity of sources of allochthonous nitrogen, the dissolved $NO_{2,3}$ that was present may have been generated through decomposition of the modest vegetation that populated the wetlands during the first season after restoration. Newly developed macrophyte communities would have made more autochthonous organic nitrogen available through decomposition at the start of the growing season in 2005 than in 2004, when a full season had not yet elapsed since restoration was initiated. It is possible that the plant communities developing at the Jones wetlands were unable to assimilate mineralized nitrogen, perhaps due to phosphorus limitation. This pattern could account for the high levels of dissolved $NO_{2,3}$ observed during year two. The low levels of dissolved $NO_{2,3}$ detected in subsequent years suggest that nitrogen assimilation capacities of the plant communities have increased with maturity, that the Jones wetlands are now capable of a greater degree of denitrification, or that the nitrogen is trapped in living or dead organic matter.

The variable and generally low levels of dissolved PO_4 in water quality samples collected at the Jones wetlands over the last four years suggest either that the wetlands are phosphorus limited or that the ion chromatography methods used to measure them are not sufficiently sensitive. Dissolved PO_4 levels are slightly higher in cells 5-6. As with turbidity, this may be an artifact resulting from non-experimental processes. Pre-restoration analysis of the substrate at the wetlands site shows that the soils underlying cells 5 and 6 may have been enriched with nutrients (John Petersen, *personal communication*). This could account for higher levels of dissolved PO_4 in these cells relative to the four western cells.

Data collected during one-day sampling suggest vertical water column stratification across all cells in the Jones wetlands. As would be expected, DO decreases with depth, sometimes resulting in hypoxia or anoxia at depth (Chimney et al. 2006, Rose and Crumpton 2006). Turbidity is highest at depth, where turbation induced by abiotic or biotic factors including, perhaps, water quality collection, is more likely to suspend sediments in the water column. Vertical stratification of DO, turbidity, and other (potentially stratified) parameters is currently not measured during weekly water quality assessment; the single-point "grab" approach is not sensitive to variation along gradients of depth (Pearson et al. 1987).

Single-day testing also validated the current method for measuring water quality at the Jones wetlands. Currently, we collect and analyze water samples from a single point in each cell and interpret water quality data produced in this way as representative of entire cells rather than single collection points. Water samples taken at 12 points within wetland 2 were not very chemically different. This suggests that it is appropriate to treat data from a single point as representative of water quality within a cell, rather than as representative of only the microenvironment from which it is collected.

Findings related to water quality at the Jones wetland address my study objectives of characterizing ecosystem processes in a marsh, assessing water quality monitoring methods at the wetlands, and characterizing the relationship between restoration treatment and ecological functioning. I was able to successfully characterize a seasonal

pattern in DO dynamics and to trace inter-annual change in dissolved $NO_{2,3}$ at the Jones wetlands. Overall, nutrient levels were too low to permit for much characterization of seasonal cycling of nitrogen or phosphorus. Muskrat activity may also have confounded my attempts to describe turbidity dynamics at the Jones wetlands. Based on an analysis of water quality monitoring techniques, I was able to confirm the meaningfulness of water quality data collected at the wetlands over the last four years. Finally, in conjunction with data presented in the following chapter, water quality data suggested but did not confirm that planting of wetland macrophytes during restoration may facilitate changes in ecological functioning relative to those that occur in self-designing restored ecosystems.

Chapter 3: Plant Community Diversity

From 2004-2007, Oberlin College personnel have conducted annual biodiversity surveys of the plant communities at the Jones wetlands.

A. Why Measure Diversity?

The magnitude and consistency of contemporary rates of biodiversity loss have prompted observers to describe the modern era as one of historically unprecedented anthropogenic extinction (Ehrlich and Wilson 1991). Biodiversity loss facilitated directly by human activity will only be augmented in the future by the spread of invasive species, rising rates of human resource exploitation, and the intensification of global climate change (Pressey et al. 2007). Yet it is necessary to emphasize urgency of biodiversity loss relative to other environmental problems such as food and water scarcity, the depletion of non-renewable resources, and the social and economic impacts of climate change. Attempts to justify research on and response to biodiversity loss must quantify the importance of biodiversity in both humanistic and economic terms.

Humanistic Value. The humanistic value of biodiversity is generally framed in cultural and psychological terms. Managed, diverse landscapes, though of some economic importance, may be closely associated with national identity and opportunities for recreation and personal restoration (Stenseke 2006). Organismal diversity, even below the species level, may provide societies with valuable, traditional foods that play an important role in both cuisine and cosmology (Caillon and Degeorges 2007). Biodiverse ecosystems may also plan an important, universal role in sustaining human psychological health. Wilson (1995) posits the existence of "biophilia," as "the innately emotional affiliation of human beings to other living organisms." If biophilia is, as Wilson claims, a central part of the human psyche, then loss of organismal diversity will universally inhibit human beings from fulfilling a basic psychological need. Social psychologists at Oberlin have operationalized some of the factors that may be associated with biophilia in their Connectedness to Nature Scale (CNS). The CNS quantifies those aspects of "experiential sense of oneness with the natural world" that may lead individuals to adopt more sustainable attitudes toward the environment (Mayer and Frantz 2004, Frantz et al. 2005). An improved understanding of the cultural and psychological importance of biodiversity and, more generally, of nature, may provide a limited warrant for biodiversity protection. Yet justifications of the protection of biodiversity premised on its humanistic importance often compare unfavorably with

demands for economic development or for allocation of scarce resources to projects focusing on other aspects of sustainability.

Economic Value. Biodiversity is both directly and indirectly a source of economic value. Biodiverse ecosystems provide a greater range of agricultural and medicinal resources that that humans can directly harvest and utilize. Yet the indirect economic importance of biodiversity is an equally essential and currently understudied phenomenon. Many recent economic assessments of biodiversity turn on the alleged relationship between biodiversity and ecosystem functioning – the BEF perspective (Naeem 2002, Naeem 2006). BEF theory asserts that biodiversity bolsters or maintains the ecosystem processes that comprise ecological functioning and that sufficiently robust ecological functioning provides humans with valuable ecosystem services (Millennium Ecosystem Assessment 2005) (Loreau et al. 2001, Marcot 2007). Ecosystem services provide an under-recognized supply of natural capital; recent scholarship has attempted to describe specific ecosystem services and the role that diversity plays in maintaining them (Hawken et al. 2000). Some workers have criticized the experimental design of studies linking higher levels of biodiversity to higher levels of ecosystem functioning and supply of ecosystem services (Huston 1997, Allison 1999, Kaiser 2000), but recent advances in the design of BEF experiments have addressed many of the field's recalcitrant problems (Naeem and Wright 2003).

The economically valuable services provided by biodiverse ecosystems take many forms. The edible primary productivity that both unmanaged and managed systems provide for humans constitutes one of the most widely exploited ecosystem services. Numerous long-term grassland studies have described increases in rates of primary productivity associated with diverse plant communities (Tilman et al. 2001a, Tilman et al. 2001b, Robertson and Swinton 2005, Spehn et al. 2005). Biodiverse, managed ecosystems interspersed with urban and rural landscapes also enhance food output by providing habitat and sustenance for the pollinators and dispersers that underpin the modern agricultural industry (Albrecht et al. 2007, Andersson et al. 2007). Functionally and taxonomically diverse systems are also more resilient to change (Elmqvist et al. 2003). Because landscape stability will be increasingly rare due to accelerating global climate change, the stabilizing effect of diversity on ecosystems constitutes a service. Biodiverse ecosystems may also reduce the prevalence and intensity of emerging infectious diseases, especially of those diseases with multiple-host lifecycles (Pongsiri and Roman 2007). Because highly diverse and species rich communities often contain high densities of poor hosts for a given pathogen, the rate of transmission will be lower in these communities for many specialist diseases (Zhu et al. 2000). This tendency of diversity to reduce disease transmission is called the "dilution effect" (Schmidt and Ostfeld 2001). Given the importance of the ecosystem services that provide humans with food, environmental stability, protection against disease, and other ecosystem services (as noted by Costanza and Daily 1992; MEA 2005; Costanza, *et al.* 2006; Naeem 2006), research that addresses the relationship between biodiversity and ecological functioning is of prime social importance.

B. Study Objectives

It was my objective to address several basic gaps in knowledge in my analysis of plant community diversity at the Jones wetlands. These gaps are related both to

treatment-induced change in and to the spatial organization of biodiversity at the Jones wetlands. My broader objective in addressing these issues was to be better able to describe the impacts of various techniques used in wetland restoration and to characterize the relationship between restoration treatment and plant community development.

I was interested foremost in describing patterns of biodiversity at the wetlands. As noted in Ch. 1, three different restoration strategies were employed at the Jones wetlands: high-intensity planting, low-intensity planting, and self-design. An early objective of this study was to determine whether or not these three treatments produced wetlands that differed in biodiversity. And, because all wetland cells were subjected to some uniform treatments (e.g. invasive control and hydrological restoration) as well as experimental treatments (e.g. different types of planting), it is was necessary to assess whether the observed biodiversity dynamics at the wetlands were driven by experimental planting treatments or by the general restoration treatment. Given that differences did exist between treatments, it was also my goal to quantify the effects of these treatments on biodiversity at the wetlands. Finally, I assessed biodiversity data to track the proliferation of exotic species at the wetlands and to ascertain whether treatment-driven changes in biodiversity were a product of the development of the native, wetland plant communities that the restoration project was designed to engender.

My other biodiversity-related study objectives had to do with quantifying the spatial heterogeneity of biodiversity at the Jones wetlands. Specifically, I wanted to know whether plant communities within each wetland were homogenously diverse or whether certain sectors, perhaps coinciding with ecotones, showed different levels of diversity. I was also interested in investigating the usefulness of various measures of beta-diversity in measuring heterogeneity in patterns of biodiversity.

C. Methods

Biodiversity Surveys. In August 2004 and 2005 and July 2006 and 2007, Oberlin and OSU personnel conducted biodiversity surveys of vascular plants in all six cells at the Jones Wetlands using methods based upon those used by the Ohio EPA, as adapted from the North Carolina Vegetation Survey (Peet et al. 1998, Smith 2006). We selected nine 10m by 10m quadrats in each cell to be surveyed, yielding a total of 54 quadrats per year. Each quadrat contained two of the 5m by 10m rectangles described previously (Fig. 2). Quadrats were established to be uniform in relative location from cell to cell (Fig. 10). For each quadrat, we identified all vascular plants within the quadrat's boundaries to the lowest taxonomic level possible, collecting vouchered specimens when we were uncertain of an identification. After identification of a species present in the quadrat, each individual surveying would approximate the percentage of the quadrat covered by that species. We would then agree on and record a cover class for the species (Table B). We rarely accorded cover class values above 6 (10%-25% coverage), therefore, the system we used, which utilized more classes at lower levels of coverage and fewer at higher levels, allowed for better resolution than a system with equally sized cover classes. Smith (2006) describes survey techniques in greater detail.

Biodiversity Indices. We electronically recorded plant community data from biodiversity surveys in Excel spreadsheets. I have given each recorded species a "natural number" that stays consistent from year to year in order to limit confusion over changes in the identification of previously misidentified species. Each natural number is

associated with the scientific and a common name of its species as well as several attributes (coefficient of conservativism, wetland status, native species status) drawn from Andreas (2004). Species were classified as either native or non-native and as either obligate wetland, facultative wetland, facultative, facultative upland, or upland. Coefficient of conservativism (C) values were also assigned to all identifiable taxa. C values are assigned by trained ecologists and reflect a taxon's relative autecological properties. Endemic species with narrow ranges are given high values and exotic or wide-ranging species are given low values (Andreas et al. 2004).

I used cover data from our biodiversity surveys to calculate five indices of biodiversity for sectors, entire cells, and entire treatments: species richness (SR), Shannon-Weaver diversity (SWD), the Floristic Quality Assessment Index (FQAI), Whittaker's Beta (B_w), and Routledge's Beta (B_r). The first three measures are alpha-diversity measures, which measure diversity in spatially inexplicitly. SR is what Magurran (1988) refers to as a richness-based measure; it represents only species composition and not relative abundance of a surveyed community. SR is simple and widely used, but because it does not incorporate abundance data, it can misrepresent patterns of species diversity in a community. This can limit its usefulness as a source of information for management (Jennings et al. 2008). SWD and FQAI are "heterogeneity measures" that incorporate relative abundance data as well as composition data (Magurran 2005). These measures are useful in that they represent multiple aspects of diversity. However, both are non-parameterized, and thus can reflect different information depending on the size and evenness of sampled communities (Magurran 1988). Magurran (1988) notes some objections to the use of SWD, but classifies it as a moderately useful measurement of diversity. SWD may also be one of the few measures that can be reliably decomposed into alpha- and beta-diversity units (Jost 2007). FQAI is also limited as an index; it is only applicable when C values are available and also fails to account for the impact of exotic species on diversity (Ervin et al. 2006). SR and SWD have been used for decades, while FQAI has recently become more popular and has been widely applied in studies of wetland plant communities (Lopez and Fennessy 2002). B_w and B_r are measures of beta-diversity, or spatially explicit diversity (Magurran 1988). Both of these indices are derived from compositional data and represent heterogeneity in the distribution of species within a larger area (Wiersma and Urban 2005, Reilly et al. 2006). Neither beta value was well characterized in the current literature, so I employed both in order to see if they measured equivalent qualities of biodiversity.

SR is simply the total number of species found in a discrete area. I calculated SR values for individual cells using two methods. The "whole-cell" method entails counting the total number of species recorded in all of the quadrats of a cell. The "average-quadrat" method involves averaging the SR of all of the quadrats in a cell. The latter method produces lower SR values, but both methods show similar trends in SR over time.

SWD balances a count of the species found in a discrete area (SR) with the input of relative abundance data. I used the Multivariate Statistical Package produced by Kovach Computing Services to calculate SWD (Kovach 1999, Krebs 1999). I calculated SWD for individual quadrats using a natural log transformation and averaged SWD scores from individual quadrats to get SWD scores for entire cells or treatments. The traditional SWD formula is:

$$SWD = -\Sigma p_i \ln p_i$$

where p_i is the proportion of the species found in a survey found of the "i_{th}" species (Magurran 1988). This formula calls for p_i to be calculated as the percentage of individuals identified of species "i." The SWD calculator in MVSP, however, has been used to calculate SWD by treating p_i as the cover class in the surveyed area attributable to species "i"(Smith 2006).

I used Excel to produce FQAI scores for individual quadrats and cells following Andreas, *et al.* (2004) and Smith (2006). As with SR, I was able to produce whole-cell FQAI (by calculating FQAI for entire cells) and average-sector FQAI (by averaging FQAI scores from all quadrats within a cell). Andreas and colleagues (2004) give the formula for FQAI as:

$$FQAI = \Sigma[(C_i)/(S)^{.5}]$$

where C is the coefficient of conservativism value for species I and S is the total species richness of the site being evaluated. C is 0 for exotic or invasive species, so these are automatically excluded from calculations of FQAI.

I used Excel to calculate B_w and B_r for whole cells and for sectors, following Magurran (1988) such that

$$B_w = S/\alpha - 1$$
$$B_r = [(S^2)/(2r+S)] - 1$$

where S is the total number of species found in the whole system, α is the average diversity (SR) of all of the subunits within the system, and r is the number of species with distributions overlapping in one or more subunits.

I calculated biodiversity indices for sectors (each composed of multiple quadrats) as well as for quadrats and whole cells. The sectors that I assessed and the quadrats that comprise them are: Edge (comprised of quadrats 1, 3, 4, 6, 7, and 9), Central (5 and 8), Wet (1-6) and Shallow (7-9) (Fig. 10). I excluded data from quadrat 2 in analyses of the Central sector. This quadrat often included an open, deep-water microenvironment with only a few species of macrophytes. As such, its inclusion skewed assessments of what was otherwise a more homogeneous central corridor through each cell. To get SR, SWD, and FQAI values for these sectors, I averaged constituent quadrat values. To calculate B_w and B_r for these sectors, I treated constituent quadrats (the parts) as the units comprising the sector (the whole).

I also calculated biodiversity indices based on modified datasets from which I had removed species associated with specific ecological qualities. To calculate wetland, native, and wetland & native SWD and average-quadrat SR, I excluded species from the original datasets based on Ohio EPA standards and carried out new SR and SW analyses (Andreas et al. 2004) (Table C). I also calculated average-quadrat SR for a dataset comprised only of exotic species.

Statistical Analysis. To test for significant difference between populations (sectors, cells, or treatments), I conducted one-factor ANOVAs using Microsoft Excel (Microsoft Corporation 1999). For all significance tests described below, I note the F-

value, between- and within-group degrees of freedom (respectively), and p-value. The alpha significance value for all tests was .05. In cases where I treated quadrats from a single cell as replicates, I adjusted my interpretations to account for possible pseudoreplication in the results. I produced all graphical figures (excluding graphs displaying the results of multivariate analyses) using Microsoft Excel and edited all graphs using Microsoft Powerpoint.

I performed Principal Components Analysis (PCA) using Kovach Computing Systems Multivariate Statistical Program (MVSP) (Kovach 1999). PCA has been widely used for ordination of plant communities and phytosociological data can be assessed using PCA in order to search for the existence of "natural," recurring plant communities (Kovach 1999, Kuzelova and Chytry 2004). The analyses were conducted on cover class data for all six cells from the 2007 biodiversity survey; data were untransformed and were centered on 0. I considered variable loadings above .4 significant. I used MVSP to create graphical scatterplots of the first two principal components for each analysis and modified these graphs in Microsoft Powerpoint.

D. Results

D1. Biodiversity Indices Calculated for All Identified Species

Species Richness. We recorded SR values ranging from 2 to 38 species in a single quadrat during the study period. The average SR of all 54 quadrats surveyed was 13.7 in 2004, 15.3 in 2005, 15.4 in 2006, and 16.4 in 2007. Averaging together all quadrat SR values for a cell produced an average-quadrat SR value for each cell; these ranged from 8.6 to 22.4 during the study period and all cells showed an increase in average SR (data not shown; Fig 11). Total SR measured in a single cell ranged from 19 to 57 and all cells showed an increase in total SR (Fig. 12; Appendix 1). Throughout the study period, PL cells were significantly more species rich than UP cells (Fig. 13; Appendix 2, a-d). The higher numbers of species recorded in PL cells was consistent at the whole-cell level and in analyses of only shallow and edge sectors (Appendix 2, e-l). Analyses of SR in wet and central sectors did not show consistently statistically significant difference in SR. In both UP and PL cells, central sectors and edge sectors were not significantly different in SR. However, dry and wet sectors differed significantly in SR over time in both UP and PL treatments (Appendix 2, o-v). Initially, dry sectors were more species rich than wet sectors in UP cells; in the second half of the study period, these differences were no longer significant. In PL cells, the reverse was true; by 2007, dry cells were significantly more diverse than wet cells. An analysis of all quadrats showed dry sectors becoming significantly more species rich than wet sectors during the study period.

Though all cells were more species rich at the end of the study period than they were when first surveyed, rates of change were not always uniform (Table D). Initially, the UP cells both lost species on the level of the whole cell, and then gained species for the rest of the study period. At the same time, whole-cell SR in the PL cells increased steadily during the study period. From 2004-2005 (the first year of the study period), the PL cells gained significantly more species than UP cells ($F = 8.65$; d.f. = 1,4; $p = .04$). However, in subsequent years, differences in change in SR were not significant (Appendix 2, m-n).

We identified 98 species at the Jones wetlands during the study period. Of these, 17 species comprised over 25% of the vegetative cover in any one quadrat surveyed

during the study period. These species were: *Sparganium americanum, Juncus effuses, Scirpus validus, Sagittaria latifolia, Nymphaea orodata, Eleocharis robustum, Najas flexilis, Ludwigia sp., Leersia oryzoides, Penthorum sedoides, Alisma subcordatum, Echinochloa crusgalli, Potomogeton nodosus, Potomogeton crispus, Elodea canadensis, Certatophyllum demersum,* and *Utricularia vulgaris*. Of these, the first five were planted in PL cells and the rest either grew from the seedbank or were otherwise naturally dispersed to the site. These species include upland, floating-leaved, emergent, and submersed plants; none of them are exotic.

Shannon-Weaver Diversity. I calculated SW values for individual quadrats ranging from 0.5 to 3.5 during the study period. The average SW of all 54 quadrats surveyed was 2.3 in 2004, 2.4 in 2005, 2.4 in 2006, and 2.5 in 2007. Average SW measured in a single cell ranged from 1.9 to 2.8 during the study period (Appendix 1). Cells 1, 3, 4, 5, and 6 increased in average SW diversity during the study period while Cell 2 became slightly lower in average SW diversity (Fig. 14). Throughout the study period, PL cells were significantly more diverse than UP cells (Fig. 15; Appendix 3, a-d). This difference was consistent at the whole-cell level and in analyses of edge sectors (Appendix 3, e-h). Along the shallow sector, PL cells were consistently more diverse than UP cells, but only significantly so for the second through fourth years of the study period (Appendix 3, i-l). By the fourth year of the study period initially small significant differences in SW in the central sector had lost significance (Appendix 3, m-p). Though initially significantly more diverse than the wet sector in UP cells, the wet sector in PL cells was, by the third year of the study, no longer significantly more diverse (Appendix 3, q-t). An analysis of all quadrats showed dry sectors becoming significantly more diverse than wet sectors during the study period; differences were significant in the second through fourth years (Appendix 3, u-x). Analysis of relative diversity in all central vs. all edge quadrats revealed no significant difference.

Though most cells were more diverse at the end of the study period than they were when first surveyed, rates of change were not always uniform (Table D). The UP cells initially became less diverse, and then gained species for the rest of the study period. At the same time, SW diversity in the PL cells increased during the study period. In the second through fourth years, UP had an average increase in diversity larger than that in PL cells. In the first year, PL diversity increased more than UP diversity. Differences in rate of change of diversity between UP and PL cells were never significant.

FQAI. In surveying individual quadrats, we recorded FQAI values ranging from 2.0 to 15.5 during the study period (Appendix 1). The average FQAI of all 54 quadrats surveyed was 8.6 in 2004, 9.7 in 2005, 10.1 in 2006, and 9.7 in 2007. Average-quadrat FQAI measured in a single cell ranged from 3.2 to 12.7 during the study period and total FQAI measured in a single cell ranged from 7.3 to 18.7 (Table E). Cells 1, 3, and 4 showed an increase in both average sector and whole-cell FQAI while cells 2, 5, and 6 dropped in FQAI during the study period (Figs. 16,17). Throughout the study period, PL cells had significantly higher FQAI than UP cells (Fig. 18; Appendix 4, a-d). This difference was consistent at the whole-cell level and in analyses of only shallow, wet, and edge sectors (Appendix 4, e-p). FQAI was significantly higher in the central sector of PL cells in the first three years of the study period, but this difference became insignificant in 2007 (Appendix 4, q-t). In both UP and PL cells, central sectors and edge sectors were not significantly different in FQAI. An analysis of all quadrats showed no significant

difference between wet and dry quadrats in FQAI in 2004, 2006, and 2007; in 2005, wet quadrats had significantly higher FQAI ($F = 4.17$; d.f. = 1, 52; $p = .05$). Analysis of relative diversity in all central vs. all edge quadrats revealed no significant difference.

Though PL cells had significantly higher FQAI than UP cells during the study period, rates of change in FQAI were different (Table D). Year-to-year increaes in both average-sector and whole-cell FQAI were higher for UP cells than PL cells (with the exception of average-sector FQAI in 2005). However, UP cells increased in FQAI almost significantly more than PL cells between the beginning and end of the study period ($F = 6.63$; d.f. = 1,4; $p = .06$).

Beta Diversity – B_w. I calculated B_w values for sectors and whole cells ranging from .21 to 2.36 during the study period (Appendix 5). Whole-cell B_w ranged from 1.00 to 2.36 across the study period. Some cells and sectors showed increases in B_w during the study period, while others decreased in B_w. Within a given cell, B_w did follow consistent trends: central and dry sectors had the lowest B_w, while wet and edge B_w were highest (Fig. 19). B_w for central and dry sectors in 2007 averaged across all six cells was .54 and .66 respectively; B_w for wet and edge sectors was 1.10 and 1.07. With few exceptions, central/dry and wet/edge/whole-cell values of B_w stratified into these patterns in all cells and across all years. B_w for whole cells, averaged across all cells, was 1.64 in 2007. With one exception (Cell 6 in 2005), whole-cell B_w was higher than all sector values. There were no consistent treatment differences in B_w.

Beta Diversity – B_r. I calculated sector B_r values for sectors and whole cells ranging from 3.26 to 26.77 during the study period (Appendix 6). Whole-cell B_r ranged from 6.68 to 23.07 across the study period. Some cells and sectors showed increases in B_r across the study period, while others decreased in B_r. There were no consistent trends in B_r within individual cells. However, PL cells had higher B_r than UP cells across the entire study period (fig. 20).

D2. Indices Calculated for Specific Groups of Species

Wetland Species. We recorded average-quadrat SR values ranging from 2 to 22 wetland species in a single quadrat during the study period (Fig. 21). The average Wetland-SR (WSR) of all 54 quadrats surveyed was 9.9 in 2004, 11.6 in 2005, 12.4 in 2006, and 12.8 in 2007. Average-quadrat WSR measured in a single cell ranged from 5.1 to 16.0 during the study period and all cells showed an increase in average WSR (Appendix 1). Throughout the study period, PL cells had significantly more wetland species than UP cells (Appendix 7, a-d).

Wetland-SW (WSW) values ranged from .5 to 2.9 in a single quadrat during the study period (Fig. 22). The average WSW of all 54 quadrats surveyed was 2.0 in 2004, 2.2 in 2005, 2.3 in 2006, and 2.3 in 2007. Average WSW measured in a single cell ranged from 1.5 to 2.6 during the study period and all cells either showed an increase or no change in average WSW (Appendix 1). Throughout the study period, PL cells had significantly higher WSW than UP cells (Appendix 7, e-h).

Native Species. We recorded SR values ranging from 2 to 24 native species in a single quadrat during the study period (Fig. 21). The average Native-SR (NSR) of all 54 quadrats surveyed was 9.9 in 2004, 11.3 in 2005, 12.2 in 2006, and 12.7 in 2007. Average NSR measured in a single cell ranged from 5.7 to 16.3 during the study period

and all cells showed an increase in average NSR (Appendix 1). Throughout the study period, PL cells had significantly more native species than UP cells (Appendix 7, i-l).

Native-SWD (NSW) values ranged from .5 to 3.0 in a single quadrat during the study period (Fig. 22). The average NSW of all 54 quadrats surveyed was 2.0 in 2004, 2.2 in 2005, 2.2 in 2006, and 2.3 in 2007. Average NSW measured in a single cell ranged from 1.5 to 2.6 during the study period and all cells either showed an increase or no change in average NSW (Appendix 1). Throughout the study period, PL cells had significantly higher NSW than UP cells (Appendix 7, m-p).

Wetland and Native Species. We recorded SR values ranging from 2 to 21 native wetland species in a single quadrat during the study period (Fig. 21). The average Native and Wetland SR (NWSR) of all 54 quadrats surveyed was 9.5 in 2004, 10.9 in 2005, 11.8 in 2006, and 12.1 in 2007. Average NWSR measured in a single cell ranged from 5.1 to 15.2 during the study period and all cells showed an increase in average NWSR (Appendix 1). Throughout the study period, PL cells had significantly more native wetland species than UP cells (Appendix 7, q-t).

Native and Wetland-SW (NWSW) values ranged from .5 to 3.1 in a single quadrat during the study period (Fig. 22). The average NWSW of all 54 quadrats surveyed was 2.0 in 2004, 2.1 in 2005, 2.3 in 2006, and 2.3 in 2007. Average NWSW measured in a single cell ranged from 1.3 to 2.6 during the study period and all cells either showed an increase or no change in average NWSW (Appendix 1). Throughout the study period, PL cells had significantly higher NWSW than UP cells (Appendix 7, u-x).

Exotic Species. We recorded SR values ranging from 0 to 18 exotic species in a single quadrat during the study period (Fig. 28). The average Exotic SR (ESR) of all 54 quadrats surveyed was 3.8 in 2004, 4.0 in 2005, 3.2 in 2006, and 3.8 in 2007. Average ESR measured in a single cell ranged from .67 to 7.0 during the study period. Some cells increased in average ESR while other cells decreased (Appendix 1). Though both UP cells decreased in average ESR during the study period, PL cells did not display a consistent trend.

D3. Principal Components Analysis

I performed three PCA (PCA1, 2, and 3) on cover class data from the 2007 biodiversity survey (Table F, Fig 17-19). PCA1 and PCA3 included all quadrats surveyed in 2007 (54) and cover class data for 98 species. The first two principal components accounted for over 50% of variation between quadrats Component 1 loaded significantly on *Elodea canadensis* and Component 2 loaded significantly on *Sagittaria latifolia* and *Sparganium americanum*. PCA2 included all quadrats except the north central quadrats (48) and cover class data for 98 species. Results were similar to those for PCA1 and PCA3 (Table F).

E. Discussion

My objectives in this portion of my research were to characterize both treatment-related change in and the degree of heterogeneity of biodiversity at the Jones wetlands. Although it is difficult to generalize results emerging from diverse study systems and methods of data collection, the SR, SWD, and FQAI values for the Jones wetlands fell within the range of values collected at other restored wetlands in the American Midwest

and East. Table G displays some biodiversity results collected from wetland restoration projects similar to the Jones wetlands. Patterns of biodiversity at the Jones wetlands were relatively similar between cells and individual cells appeared to show some degree of heterogeneity in the patterns of biodiversity that emerged.

It became evident in my own preliminary analysis and in Smith's (2006) assessment of biodiversity at the Jones wetlands that there were minimal differences in diversity cells restored with high-intensity and low-intensity planting treatments (Fig. 12). At the same time, HI and LI cells, when considered part of the same quadruplicated PL treatment, showed significantly higher SR, SWD, and FQAI than UP cells (Figs.13, 15, and 18). My p values for ANOVAs that tested for significant differences in diversity by treatment grew smaller by several orders of magnitude when I treated HI and LI cells as components of a single group relative to UP cells. Additionally, UP cells clustered together in multivariate ordinations of 2007 community data (Figs. 23-25). In each ordination I performed, UP quadrats were distinct from PL quadrats because they lacked *Sagittaria latifolia* and *Sparganium eurycarpum*, two species that were planted in and subsequently came to characterize PL cells. Therefore, when considering biodiversity, I decided to treat all HI and LI cells as a single PL treatment in future analyses. The similarity of plant communities in HI and LI cells and the equivalent levels of biodiversity I measured in these cells also suggest that the low- and high-intensity planting treatments we used produced similarly structured plant communities. Re-plantings, therefore, may not facilitate more effective ecological restoration. Between 2004 and 2007, Oberlin personnel spent over 100 hours replanting the HI wetlands (Table H). In future restoration of herbaceous wetlands where engendering stable, native plant communities is an objective, single plantings may be effective. Instead of planting wetlands multiple times, personnel can devote resources to other practices that increase or maintain native biodiversity, such as control of invasive species and other forms of post-restoration management (Mulhouse and Galatowitsch 2003).

Across all four years of the study period and for all three measures of composition-abundance data I assessed, planted cells had significantly higher biodiversity than unplanted cells (Figs. 13, 15, and 18; in all cases, $p < .02$). These differences held for both whole-cell and average-sector SR and FQAI values (Figs. 11 and 13, 16-18) and were also significant for both SR and SW when these values were calculated for datasets containing only native, only wetland, or only native and wetland species (Fig. 21,22). Finally, B_r values for PL cells were consistently higher than those for UP cells, indicating that PL cells have more spatially heterogeneous species distributions (Fig. 20). These data suggest that planting with wetland species, either once or repeatedly, engenders plant communities that are more diverse and spatially heterogeneous (Mitsch et al. 1998, Balcombe et al. 2005, Spieles et al. 2006). Analysis of diversity in "sectors" designed to include homogeneous quadrats (e.g. a wet sector) presented a more equivocal picture. Shallow and edge sectors generally were significantly more species rich and diverse and had significantly higher FQAIs in PL than in UP cells. For central and wet sectors, differences between UP and PL cells were sometimes significant and were sometimes not. This suggests that planting treatment may have had its most dramatic impacts on shallow and edge quadrats. I don't think this is a plausible interpretation because the edge and shallow sectors both contain large amounts of upland habitat while all of the plant species included in the PL treatment were wetland plants. If experimental treatment

were to make any sectors more diverse, it would be the wet and central sectors which had more modest gains in biodiversity. Rather, I think that the high levels of biodiversity in wet and edge sectors are the result of the location of these sectors on ecotones – borders between two different ecosystems. Many of the edge sectors contain deepwater, shallow, and upland habitats while the shallow sectors generally fluctuate between flooded and dry conditions during the year. As sites of both spatial and temporal hydrological variability, they may provide suitable habitat for a wider range of plants, boosting sector-level biodiversity (Keddy 2000, Seabloom and Van der Valk 2003). Furthermore, quadrats that include at least some upland habitat tend to show higher diversity because they can support a wide range of facultative wetland *and* upland plants, whereas fully hydric habitats can only support wetland plants (Thompson et al. 2007).

Cell-wide trajectories of biodiversity also suggested that the global increases in biodiversity in individual cells might not have been driven by specific planting treatments, but rather by the general restoration treatment employed at the site. All cells either increased or stayed the same for all measures of composition-abundance biodiversity between the beginning of the study period and the 2007 biodiversity survey (Figs. 11, 14, 16, and 17). UP and PL cells sometimes increased in biodiversity at significantly different rates from year to year, but these dynamics were never consistent and did not display consistent trends across biodiversity indices (Spieles et al. 2006). Some restoration studies have reported sustained differences between biodiversity in planted and unplanted ecosystems, while others have reported that unplanted systems catch up to planted ones in a few years (Mitsch et al. 1998, Mitsch et al. 2005, Spieles 2005, Hartzell et al. 2007, Matthews and Endress 2008). It will be necessary to continue studying plant community assembly at the Jones wetlands over the coming years to assess trends that may emerge (Seabloom and Van der Valk 2003, Spieles et al. 2006).

UP and PL cells showed divergent patterns of spatial organization during the study period. As noted above, B_r was consistently higher for PL cells, although the gap between B_r for PL and UP cells appears to be closing (Fig. 20). B_w did not differ consistently between treatments. In assessments of all quadrats in each year of the study period, central and edge sectors did not differ significantly in diversity within either UP or PL cells. Similar assessments of wet and dry sectors showed some differences between treatments, but these were not consistent from year to year and did not display consistent trends among indices. Principal components analysis of community data from 2007 revealed 2 axes that accounted for a total of between 50% and 52% of variation among quadrats (Table F). The first principal component (PC1) loaded significantly on the presence of *Elodea canadensis*, a submerged wetland macrophyte that was present in deepwater cells. Deepwater quadrats, regardless of treatment, scored high on PC1 when it was positive and low on it when it was negative. The second principal component, which loaded on *S. americanum* and *S. latifolia*, appears to be associated with treatment. With a few exceptions of recent spread into UP cells, these species are confined to PL cells. Thus, PL cells scored high when PC2 was positive and low when it was negative (Figs. 23-25). Outliers represent either PL quadrats with little *S. americanum* or *S. latiffolia* in them or UP quadrats that do not support these species. Though the data on differences in spatial heterogeneity in Pl relative to UP cells is more equivocal than data on biodiversity differences between treatments, they still suggest that planting treatment has made wetland cells, in some ways, more spatially diverse. Restored wetlands of

various treatment types have been shown to differ from analogous reference wetlands in spatial organization (Seabloom and Van der Valk 2003); it is possible that self-designing restored wetlands and planted restored wetlands also display different spatial patterns.

Measurements of B_w across all four years showed a trend that was uniform for all cells and that did not differ according to treatment. Specifically, B_w values were highest for whole cells, intermediate for shallow and edge sectors, and low for central and wet sectors (Fig. 19). High B_w values for whole-cell quadrat sets were unsurprising because individual wetland cells contain numerous microenvironments defined by gradients in basin depth and annual hydrological patterns. Comparison of B_w values for sectors suggests that edge and shallow quadrats (even if to a lesser extent than the whole cell) also cover a number of environmental gradients while central and wet sectors are more physically homogeneous. This inference agrees with my anecdotal observations: edge quadrats often encompassed deepwater, shallow, and upland habitat while central cells were almost always comprised of shallow water. Likewise, shallow quadrats were saturated with water for only part of the year while wet quadrats were generally saturated year-round. It may be fair to say, then, that the quadrats comprising the edge and dry sectors are environmental ecotones characterized by high biological and physical heterogeneity and that B_w measures the biodiversity aspect of this heterogeneity (Keddy 2000). These findings also argue for the use of B_w as a measure sensitive to diversity arising along gradients such as wetland depth (Koleff et al. 2003).

The significant differences in SR and SWD between PL and UP cells when only subsets of all species surveyed are considered suggest that the restoration treatment used in PL cells was successful at establishing communities of native wetland plants. Individual assessments and meta-analyses of wetland restoration often find that increases in richness and diversity in restored wetlands are driven by increases in the abundance and composition of non-native species (Mulhouse and Galatowitsch 2003, Seabloom and Van der Valk 2003, Spieles 2005, Zedler and Kercher 2005). Significant differences in NSR and NSW in PL and UP cells in native-species datasets suggest that this is not the case at the Jones wetlands (Figs. 21,22). The persistence of this trend when I considered only wetland species also leads me to believe that higher diversity in PL cells was not largely a function of higher diversity of non-wetland plants that were not introduced as part of the PL planting treatments (Appendix 2, u-bb). Assessment of a dataset comprised of only exotic species suggests that the planting treatment employed does not have a consistent effect on the presence of exotic species (Fig. 28). The robustness of my findings of higher diversity in PL cells across datasets including only wetland and native plants suggests that at the Jones wetland, restoration treatment has engendered the development of native, wetland plant communities. Further monitoring, especially of UP cells, will be necessary to ascertain that this trend is permanent (Spieles et al. 2006).

Chapter 4: Invasive Species Management

Managing invasive, exotic plant species is one of the central challenges of any restoration project. Exotic species are those that were introduced artificially into a landscape (*sensu* Sax, *et al.* [2007]), while invasive species are those that aggressively out-compete non-invasives (*sensu* Marris [1995]). The two terms are not necessarily synonymous. For the purposes of this chapter, I will use the term "invasive species" to

refer to exotic, invasive species. This definition is analogous to that of "biotic invaders" (*sensu* Mack, *et al.* [2000]).

While biological invasions are a historical problem, contemporary increases in the connectivity of global economies have intensified the introduction of exotic species to ecosystems across the world; some such exotic species introductions result in the establishment of invasive populations (Mack et al. 2000, Marris 2005, Hobbs et al. 2006). Whereas the domestic cost of controlling and compensating for the impact of invasive species was roughly 97 billion dollars between 1906 and 1991, twenty-first century costs of invasive species are estimated to be near 120 billion dollars per year (Pimentel et al. 2005). Predictions of the impact of new introductions place the cost of a new biological invasion in the tens of millions of dollars (Cook et al. 2007). Invasive species are also a consistent obstruction to the restoration of native wetland plant communities (Boers et al. 2007, Wilcox et al. 2007, Adams and Galatowitsch 2008).

In this chapter, I will describe the challenges to restoration posed by invasive species and review the progress of invasive species control at the Jones wetlands. My objectives in this portion of my research were to identify which species have interfered with restoration at the Jones wetlands and to assess the effectiveness of methods used to control them.

A. Invasive Species and Wetland Restoration

Invasive species are a major obstacle to the effective restoration of wetland plant communities in freshwater wetlands (Brinson and Malvarez 2002). Restored wetlands are by nature "novel ecosystems" that arise out of human planning and management; as such, the effectiveness and type of invasive species control utilized determines the composition of the plant communities that dominate these ecosystems (Hobbs et al. 2006). Yet invasive species management is not only a matter of importance when the goal of restoration is to engender communities that include certain desired native species. The effects of invasive species on restored communities are important at the functional level. Biological invasions can alter ecosystems' trophic structure, hydrology, disturbance regime, productivity, successional trajectory, and biogeochemical cycling (Mack et al. 2000, Levin et al. 2006, Sax et al. 2007). When restorationists attempt to control any of these processes or to determine compositional diversity, managing invasive species is unavoidable.

Management at the Jones wetlands has centered on the intentional introduction of a range of species through planting and seeding and the removal of two native taxa to become established during the study period, *Phalaris Arundinacea* (reed canarygrass) and *Typha* sp. (cattail). Fortunately, we have never recorded the presence of the common wetland invasives *Phragmites australis* and *Lythrum salicaria* (purple loosestrife) at the Jones wetlands. These four taxa are persistent barriers to successful restoration at similar wetland sites (Adams and Galatowitsch 2005, 2007, Boers et al. 2007, Frieswykt and Zedler 2007, Wilcox et al. 2007, Adams and Galatowitsch 2008). Most significantly, this is because they can form dense, monotypic stands that exclude other wetland plants and alter nutrient cycling and both ecosystem composition and function. Reed canarygrass and cattail have threatened to form such stands at the Jones farm.

Reed canarygrass is a rapidly growing grass that reproduces both through aggressive vegetative spread and high-density seed distribution. Populations of reed

canarygrass easily outcompete existing populations of native species and preempt the establishment of interspecific competitors (Adams and Galatowitsch 2005). Reed canarygrass is highly developmentally plastic and can tolerate a number of hydrological conditions; it thrives in human-disturbed and newly restored landscapes (Wilcox et al. 2007). The presence of reed canarygrass suppresses native species diversity in restored wetlands and many wetland restoration projects, especially those that are not heavily managed for reed canarygrass post-restoration, develop plant communities dominated by the grass (Mulhouse and Galatowitsch 2003, Adams and Galatowitsch 2008). Furthermore, reed canarygrass recolonizes experimental plots following removal and is highly resistant to competition from sympatric natives, burning, application of herbicides, and hand-pulling (Adams and Galatowitsch 2007, Wilcox et al. 2007). Eradication of reed canarygrass prior to the establishment of a stable population and seedbank and simultaneous efforts to reestablish native species are critical to the development of diverse, native wetland communities (Adams and Galatowitsch 2008).

Invasive cattail populations are often dominated by either the invasive *Typha angustifolia* (narrow-leaved cattail) or the native-invasive hybrid *Typha x. glauca* (hybrid cattail). Hybrid cattail is an F1 hybrid of *T. angustifiolia* and native *T. latifolia* (broad-leaved cattail) and is more aggressive and phenotypically plastic than either of its parents. Though sterile, *T. x. glauca* can produce quickly through clonal, vegetative growth (Boers et al. 2007). By crowding out native species and covering the seedlings of competitors with abundant litter, both invasive cattails suppress native species and expand into dense, monotypic stands (Frieswykt and Zedler 2007). Like reed canarygrass, cattail suppresses native species richness, is difficult to remove, and rapidly reinvades wetlands after its extirpation (Boers et al. 2007).

B. Management at the Jones Wetlands

Invasive species management at the Jones wetlands has largely taken place through weeding from June through August on an as-needed basis. Cattail and reed canarygrass were both controlled through hand-pulling, which consists of the removal of whole individuals, including (ideally) all rhizomes, by hand. Round-up was applied to the shoots of reed canarygrass growing in upland areas using a hand-powered sprayer in 2005 and 2006; emergent stands were not sprayed so as to avoid damage to other aquatic plants. In 2006 and 2007, the above-ground growth of reed canarygrass was removed using hand-shears and a weed-whacker. This method did not disrupt rhizomataceous growth. Invasive management was not focused evenly on each wetland cell. Rather, personnel devoted more time to those cells with more entrenched invasive populations. Cells 1, 5, and 6 required two to three times as much invasive control as cells 2, 3, and 4 (Fig. 26). Additionally, the bulk of invasive species management targeted cattail during the first year of the study period. In subsequent years, control focused increasingly on reed canarygrass (Fig 27).

The strategies we employed have been so unsystematic in part because the challenges posed by invasive species at the Jones wetlands changed considerably during the study period. In the summer of 2004, the first season following restoration, both reed canarygrass and cattail were present at the Jones wetlands. Oberlin personnel spent thirty-eight person-hours removing the two species, mostly through hand pulling. We do not have records of what proportion of this time was spent pulling cattail versus reed

canarygrass, but informal notes and John Petersen's personal recollections suggest that cattail populations were well-established during the summer of 2004 and that most management targeted cattail. In 2005, personnel spent 6.4 person-hours hand-pulling cattail and 14.6 hours either weeding reed canarygrass by hand or controlling it using Round-up and a steel-blade weed-whacker. In 2006, only a few stalks of cattail were recorded at the wetlands. Their inflorescences were removed, but they were allowed to remain to determine the degree to which cattails spread once a wetland is initially established. reed canarygrass was still well-established in all cells (especially cells 1, 5, and 6). Personnel spent a total of 13.5 person-hours removing reed canarygrass using hand-pulling, Round-up application, and removal of above-ground growth using a weed-whacker and hedge shears. In 2007, cattails were very rare and were not targeted for management. reed canarygrass was controlled through hand-pulling and growth removal with a weed-whacker and hedge shears. Control of reed canarygrass totaled only 4.2 hours in 2007. During the study period, the grassy berms that comprise the watersheds of most of the cells were occasionally mowed with a riding mower. These berms were frequently overgrown with reed canarygrass, so mowing was in part an attempt to decrease reed canarygrass propagule pressure. Mowing was especially intensive in the watershed corresponding to cell 1.

The distribution of person-hours devoted to invasive species management at the Jones wetlands suggests that selective removal of invasive reed canarygrass and cattail may control the spread of these taxa and enhanced the development of native plant communities. The yearly decline in hours spent removing cattail has apparently culminated in the reduction of an originally widespread population of cattail to only a few individuals (Fig. 27). And, though reed canarygrass was still prevalent in all cells during the 2007 biodiversity survey, the person-hours spent removing reed canarygrass dwindled from 14.6 hours to only 4.2 hours between 2005 and 2007. Apparent suppression of the spread of cattail at the Jones wetlands suggests that intensive, long-term control of this invasive species shortly after restoration enhanced the development of diverse native plant communities (Boers et al. 2007). Control of reed canarygrass has been less successful. The presence of reed canarygrass in several quadrats in all cells in 2006 and 2007 suggests that the reed canarygrass invasion at the Jones wetland has not been fully suppressed (data not shown). Given the tendency for unmanaged or lightly managed plant communities restored wetlands to succumb to reed canarygrass invasion during the first decade of restoration, continued aggressive control of reed canarygrass at the Jones wetlands is optimal (Mulhouse and Galatowitsch 2003, Adams and Galatowitsch 2007). Finally, the disproportionate amount of invasive species control required in cells 1 and 6 (and, to a lesser extent, cell 5) suggests that cells on the edges of the study area may be subject to higher invasive propagule pressure than inner cells, leading to more entrenched invasive communities (Adams and Galatowitsch 2008).

The fairly stable SR of exotic species in individual cells, the control of cattail across the study system, and the declining number of person-hours required to manage invasives during the study period suggest that restoration at the Jones wetlands has probably been successful at fostering plant community development in the face of invasive pressure. Periodic reevaluation of management procedures as the threat of invasive species changes will consolidate these gains in the future.

Chapter 5: Conclusion

A. Study Objectives

To date, research at the Jones wetlands has addressed basic research questions in restoration ecology and characterized water quality dynamics, plant community diversity, and invasive species management at a restored, herbaceous marsh. To conclude my discussion of the portion of this research conducted between 2004 and 2007, I will revisit my study objectives from previous chapters and provide recommendations for future management at the Jones wetlands.

Chapter 1: Basic Questions. On the most general level, data collected at the Jones farms contributes to the conversation in the field of restoration ecology about what ecosystem attributes to restore, what methods to use in restoration and post-restoration monitoring, and whether there is a relationship between ecosystem structure and functioning in restored wetlands.

Restoration at the Jones wetlands has focused on the restoration of the following ecosystem attributes: macrophyte diversity, water quality and nutrient cycling, community metabolism resistance to invasion, stability, and typical wetland hydrology. The global increases in biodiversity that occurred in all restored wetlands at the Jones farm during the study period suggest that it is reasonable for restorationists to select diverse wetland plant communities as an ecosystem attribute for restoration (Figs. 11, 14, 16, and 17). The relative dominance of native, wetland taxa (Fig. 21) relative to exotic taxa (Fig. 28) and the near- or total-exclusion of cattail, purple loosestrife, and *P. australis* from the wetlands also further indicate that restoration of desirable plant communities is a realistic goal. Restoration of water quality, nutrient cycling, and community metabolism has been harder to assess. As noted in Ch. 2, the Jones wetlands currently appear to be oligotrophic with respect to dissolved nitrogen and phosphorus (Fig. 9). The wetlands may also still be maturing in terms of their ability to incorporate and store inorganic nutrients; further studies may address this aspect of ecological functioning. The trends in seasonal dissolved oxygen flux that I noted (Fig. 5-7) appear to be biologically driven, suggesting that regular patterns of community metabolism may have been re-established through restoration. Assessment of invasive species richness and the amount of time necessary to control invasive species at the wetlands suggests that the system has been successfully restored as one resistant to invasion. Finally, the consistent patterns of DO cycling and of macrophyte diversity (especially of dominant species) suggest that restoration at the Jones farm has produced six hydrologically and biologically stable wetlands.

Research at the Jones wetlands has also attempted to assess the role of restoration treatments in engendering different ecological structure and function and to evaluate post-restoration monitoring strategies. As discussed in Chs. 2 and 3, there were no significant differences in water quality or plant community diversity between the high- and low-intensity planting treatments. There were, however, a number of differences corresponding to planted (PL) versus unplanted (UP) wetlands. PL wetlands showed significantly higher drops in dissolved oxygen levels during the summer and also were significantly more species rich and diverse than UP cells during the study period. These findings lead me to cautiously suggest that both planting treatments used on PL cells have engendered wetlands that are more compositionally diverse and ecologically functional

relative to self-designing wetlands. However, continued monitoring will be necessary to further consolidate this claim. Assessments of ecological restoration, especially, often suffer from a paucity of post-restoration monitoring. Therefore, monitoring of water quality, plant community diversity, and invasive species management at the Jones wetlands should continue into the future. Long-term monitoring will be necessary to confirm the validity of my treatment-level findings related to DO and nutrient dynamics, compositional biodiversity, and community invasibility.

Finally, the co-occurrence of both functional (dissolved oxygen-related) and compositional (species diversity-related) differences between planted and unplanted corroborates the proposed relationship between biodiversity and ecosystem function. My interpretations, however, are still only tenuous in this respect.

Chapter 2: Water Quality. Water quality research at the Jones wetlands has been focused on characterizing seasonal and inter-annual dynamics of biogeochemical cycling and community productivity and on assessing current strategies for measuring water quality. I assessed four variables of water quality: dissolved oxygen, turbidity, dissolved nitrogen, and dissolved phosphorus. I found a consistent pattern of seasonal change in DO (noted above), high levels of turbidity in two adjacent cells, a sharp drop in dissolved nitrogen after 2 years, and very low levels of dissolved phosphorus. Turbidity differences between cells currently seem to be controlled by non-experimental muskrat activity, which has also confounded analysis based on differences in planting treatment. I have interpreted the nitrogen patterns that I observed at the wetlands (Fig. 9) as the possible result of plant community development. A more diverse and abundant plant community may have provided for the incorporation of all inorganic nitrogen in the system into biomass. Low levels of observed dissolved phosphorus may be due to low levels of allochthonous input or to insufficient resolution in our detection methods.

My assessment of water quality monitoring methodologies employed at the Jones wetlands also suggests that the current system of weekly point sampling from each wetland sufficiently characterizes water quality dynamics at the whole-cell level.

Chapter 3: Plant Community Diversity. In my analysis of plant community diversity data, I attempted to characterize changes in the magnitude of and patterns of heterogeneity in biodiversity at the Jones wetlands. In short, I found consistent increases in biodiversity in all cells. This trend was consistent when measured by species richness, Shannon-Weaver diversity, and the Floristic Quality Assessment Index (Fig. 22). Increases in biodiversity were also not driven by the proliferation of exotic or non-wetland species; the general patterns observed corresponded to increases in native, wetland plant diversity (Fig. 21). As discussed above, planted wetland cells were also significantly more diverse than unplanted wetland cells (Figs. 13, 15, 18). Planted and unplanted cells also clustered distinctly in principal components analysis (Fig. 25). One measure of beta-diversity, B_r, also suggested that planted cells had a more heterogeneous arrangement of species (Fig. 20). Finally, exotic species were present at the Jones wetland system, but did not show definite trends of increasing species richness and did not show compositional difference between treatments (Fig. 28).

Plant diversity at the Jones wetlands was also organized heterogeneously, with ecotonal “edge” and “wet” sectors showing higher values of B_w (Fig. 19). This interpretation was supported by principal components analysis, which showed some clustering of wet, dry, and central quadrats (Figs. 23, 24).

Chapter 4: Invasive Species Management. Cattail (*Typha* sp.) and *Phalaris arundinacea* emerged as the two most important invasive species at the Jones wetlands. The presence of common wetland invaders *Phragmites australis* and purple loosestrife (*Lythrum salicaria)* was not recorded during the study period. The focus of invasive management at the wetlands shifted from a focus on cattail to a focus on reed canarygrass as cattail was largely excluded by 2006. Reed canarygrass continues to be a troublesome species at the Jones wetlands. However, every year during the study period, the number of person-hours required to control the two species decreased, suggesting that current methods may ultimately result in the stabilization of both populations at low levels (Fig. 27). The amount of management required may be a function of propagule pressure as cells closer to the edges of the study area require more management than cells on the inside of the study area (Fig. 26).

B. Recommendations

Based on the foregoing analysis of the objectives of research at the Jones wetlands and my findings to date, I recommend the following:

B1. Water Quality

1. The parameters currently included in water quality monitoring are suggestive of functional dynamics in the Jones wetlands. However, it would be useful for personnel to perform more short-term studies of water quality that quantify ecosystem processes that are currently unstudied at the Jones wetlands. Studies could include assessments of biological oxygen demand, diel variation in DO, SRP, total dissolved nitrogen (including organic nitrogen), and of the differences in DO levels near submerged and floating-leaved versus emergent vegetation.
2. Assessments of spatial diversity in water quality (as noted above) will be informative, but the results of single-day water quality monitoring suggest that the current methods of weekly monitoring provide information representative of each whole wetland rather than of single sampling points. This methodology should be maintained for weekly, long-term water quality monitoring.
3. Low levels of dissolved ions suggest that the Jones wetlands are currently oligotrophic. It may be beneficial to widen the current research agenda at the wetlands by altering the current hydrology to allow for the inflow of nutrient-rich agricultural waste into some or all wetlands. The same effect could be accomplished by applying fertilizer to some or all wetlands. This could provide data for a study of nutrient retention in restored herbaceous marshes.

B2. Plant Community Diversity

4. Comparisons of diversity in HI, LI, and UP treatment groups suggest that yearly replantings of native vegetative propagules have little effect on plant community diversity at the Jones wetlands (Fig. 12). Replanting did not occur in Fall 2007. I recommend that no further replantings occur.
5. Annual biodiversity surveys should continue. Over the next several decades, data from these studies will demonstrate the robustness of current differences in diversity between PL and UP wetlands, provide feedback on management decisions, and alert managers to the incursion of invasive species (Mulhouse and Galatowitsch 2003).

6. During the summers of 2006 and 2007, I noted that several individuals of *S. latifolia*, *Pontederia cordata*, and *S. americanum* (three planted species) had dispersed into UP wetlands. I took GPS readings of the locations of these colonizers. Future managers should carefully track the spread of these species in UP cells, especially since *S. latifolia* and *S. americanum* are major components of the vegetation that dominates PL cells.

B3. Invasive Species Management

7. Future invasive management should focus aggressively on populations of *Phalaris arundinacea* (reed canarygrass) in all cells. If biomass removal using hedge shears and a weed whacker is not sufficient, additional methods such as herbicide applications and burning should be employed (Adams and Galatowitsch 2007, Wilcox et al. 2007). If populations of *Typha* sp. recover, they should also be subject to aggressive monitoring and control.

B4. Data Analysis

8. Analyses of the data presented in this thesis were severely constrained by my own limited knowledge of statistics and of the many indices used to measure water quality and plant community diversity. Future studies could benefit from the assessment and use of more sophisticated univariate and multivariate statistics and of different conventions for representing diversity. Furthermore, pseudoreplication frequently limited the types of statistical analyses I could carry out. It is possible that different forms of data analysis could mitigate this problem.

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Figures

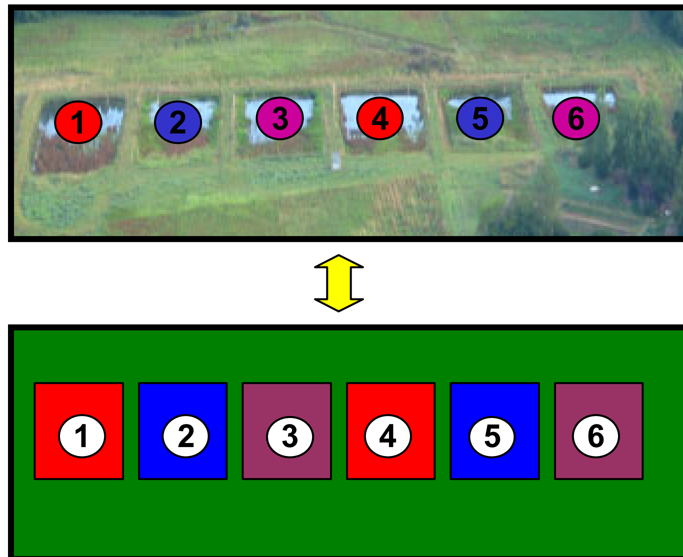

Fig. 1. A photographic (top) and schematic depiction of the Jones wetlands.

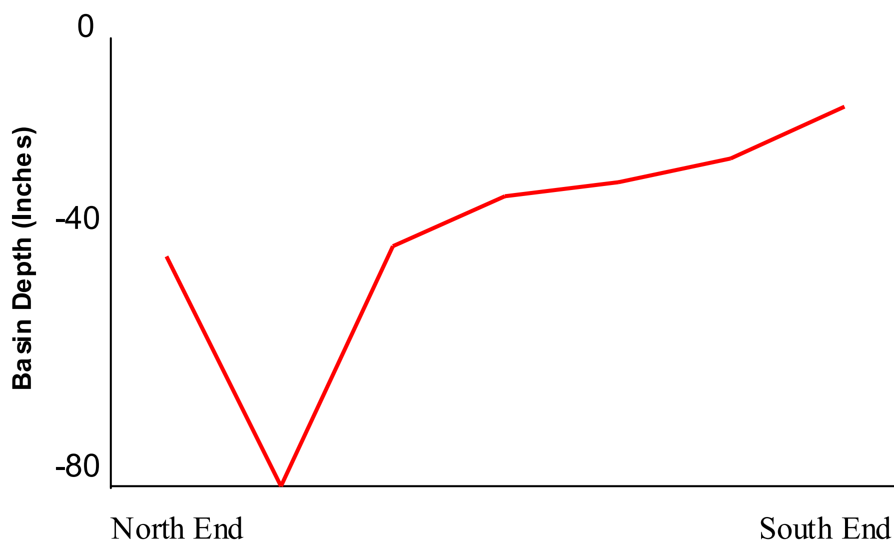

Fig. 2. Median basin depth for Cell 2, which is characteristic of other cells, from North to South. The deepest point occurs at row 2. Adapted from Smith (2006).

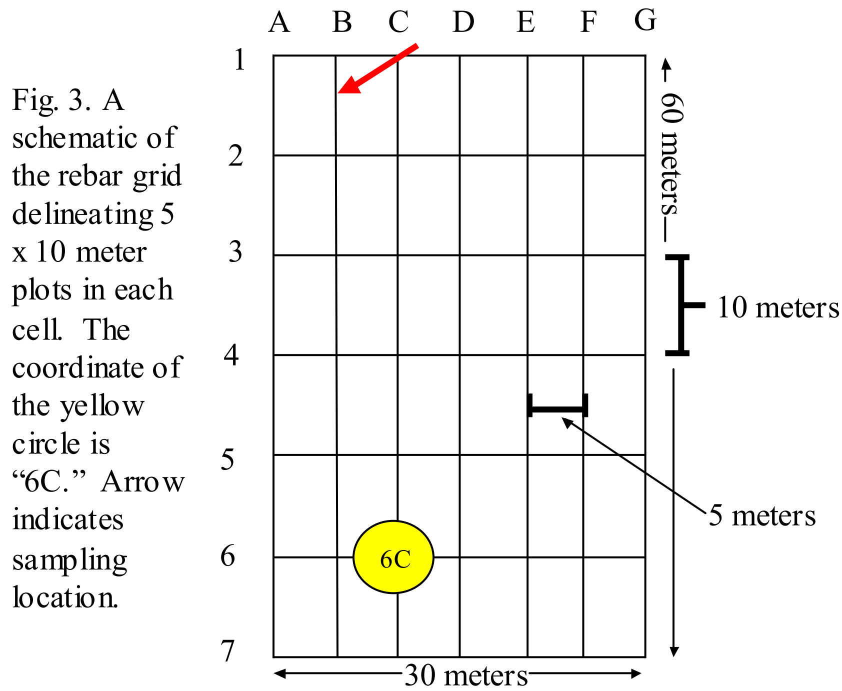

Fig. 3. A schematic of the rebar grid delineating 5 x 10 meter plots in each cell. The coordinate of the yellow circle is "6C." Arrow indicates sampling location.

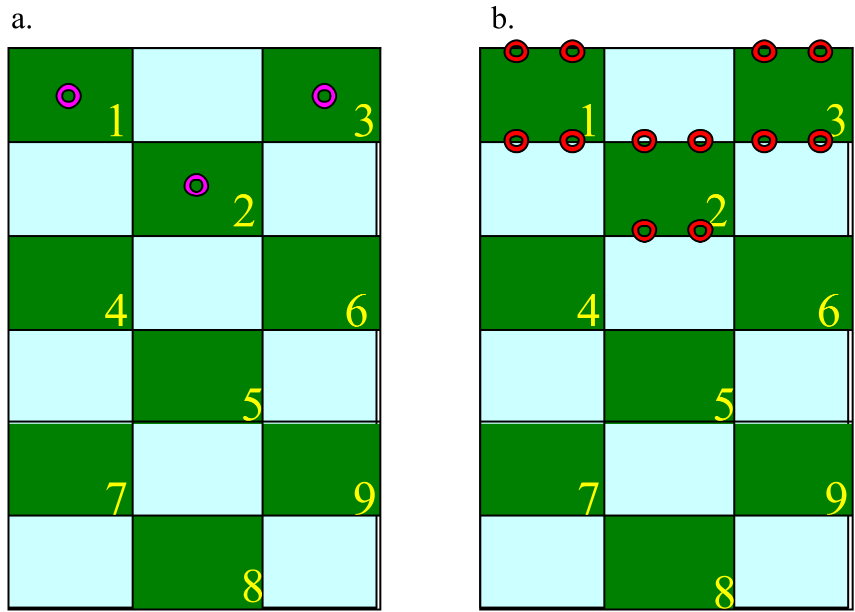

Fig. 4. Circles indicate sampling locations for a) day 1 and b) day 2 of one-time sampling. Red circles designate surface sampling; pink circles designate surface and depth sampling.

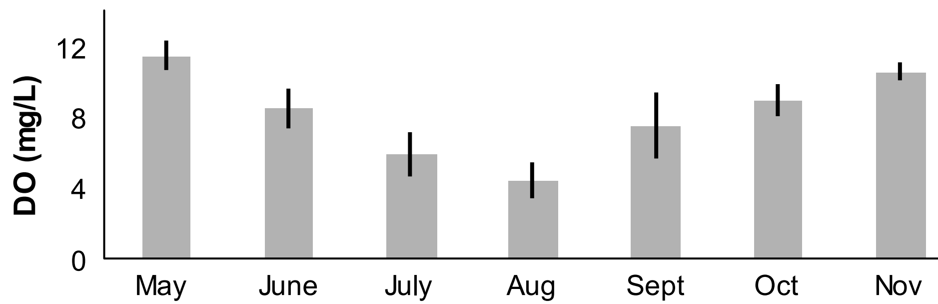

Fig. 5. Mean monthly DO of all wetlands during the study period. Error bars indicate variation between years.

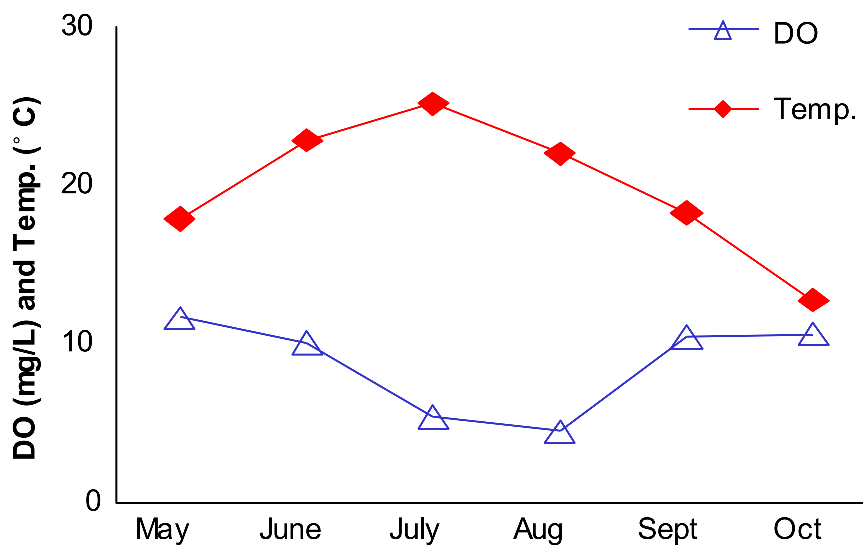

Fig. 6. Median monthly DO across all cells for 2006, shown with average water temperature.

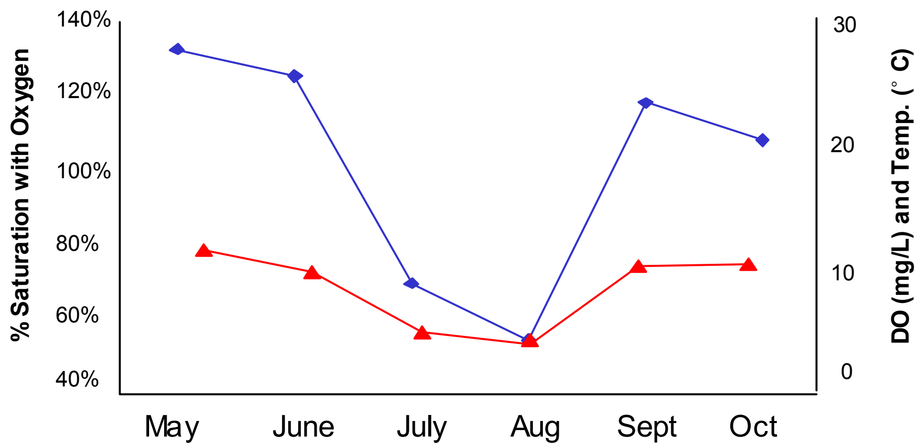

Fig. 7. Comparison of median dissolved oxygen (DO) in all cells by month in 2006 (red) and % saturation with oxygen that these concentrations represent (blue). % Saturation is calculated by dividing measured DO for a given month by the value for DO calculation corresponding to 0% salinity, atmospheric pressure, and mean monthly temperature for all readings (calculated as in Benson and Krause [1984]). The similar shape of the two curves suggests that seasonal changes in DO are due to biological rather than physical factors.

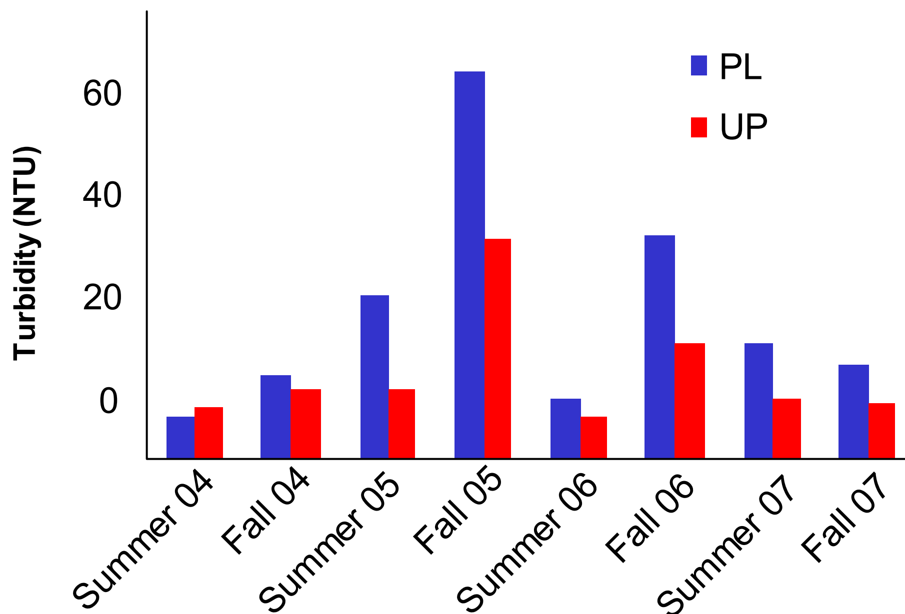

Fig. 8. Turbidity averaged across season and group of cells. Differences are not significant.

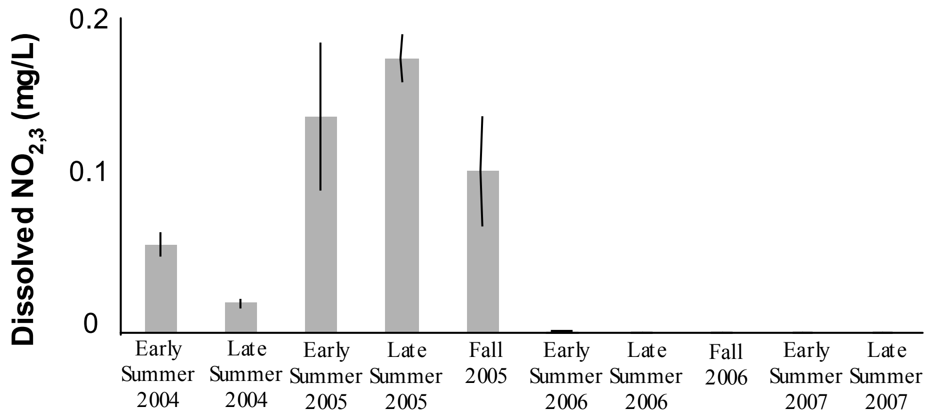

Fig. 9. Median dissolved DIN averaged within season. Error bars are standard deviation measuring variance between cells.

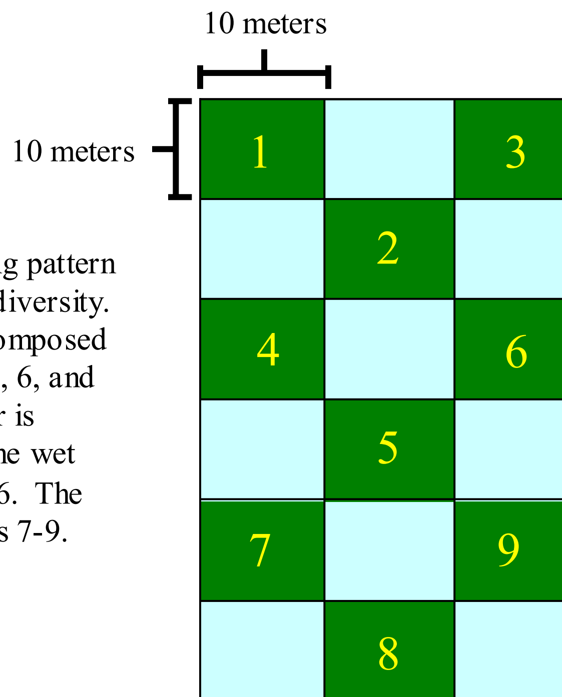

Fig. 10. The sampling pattern used to quantify biodiversity. The edge sector is composed of quadrats 1, 4, 7, 3, 6, and 9. The central sector is quadrats 5 and 8. The wet sector is quadrats 1-6. The dry sector is quadrats 7-9.

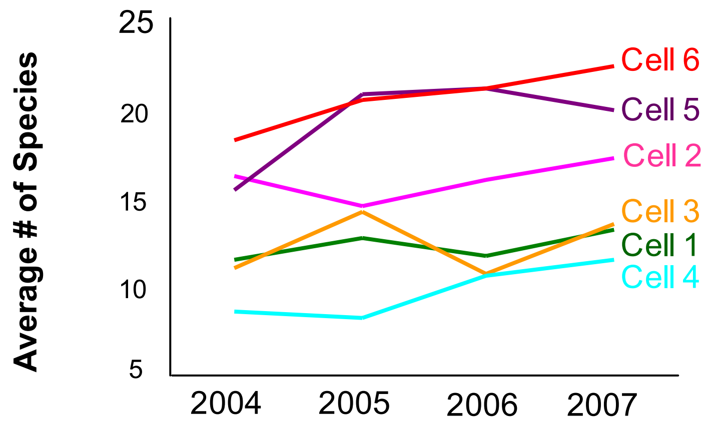

Fig. 11. Average-quadrat Species Richness was calculated by averaging SR values from all 9 of a cell's quadrats for a given year.

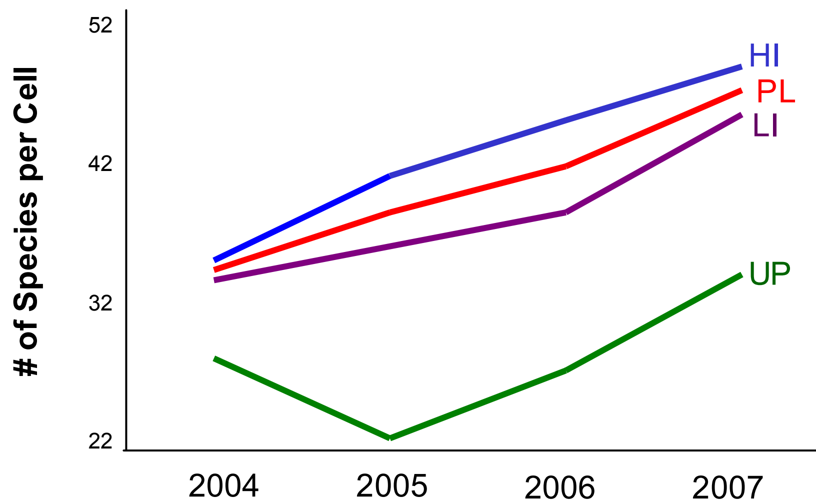

Fig. 12. The average whole-cell Species Richness for High Intensity (HI), Low Intensity (LI), Planted (PL), and Unplanted (UP) cells.

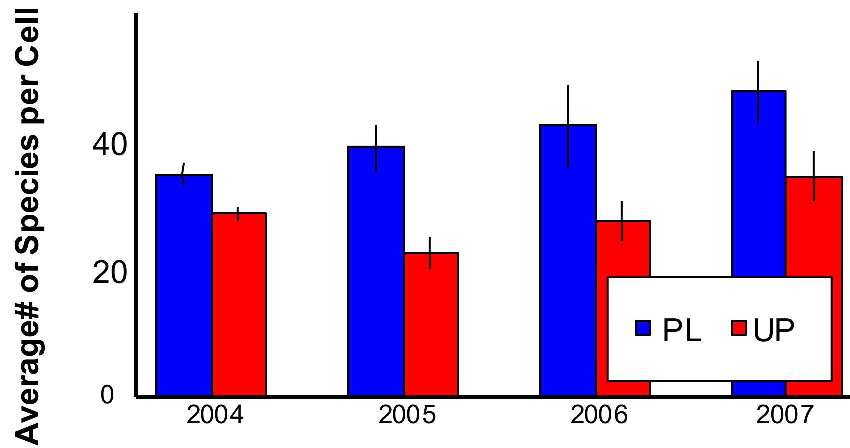

Fig. 13. Comparison of averaged whole-cell Species Richness in Planted (**PL**) and Unplanted (**UP**) cells. All differences are significant ($p<.05$). Error Bars are SD

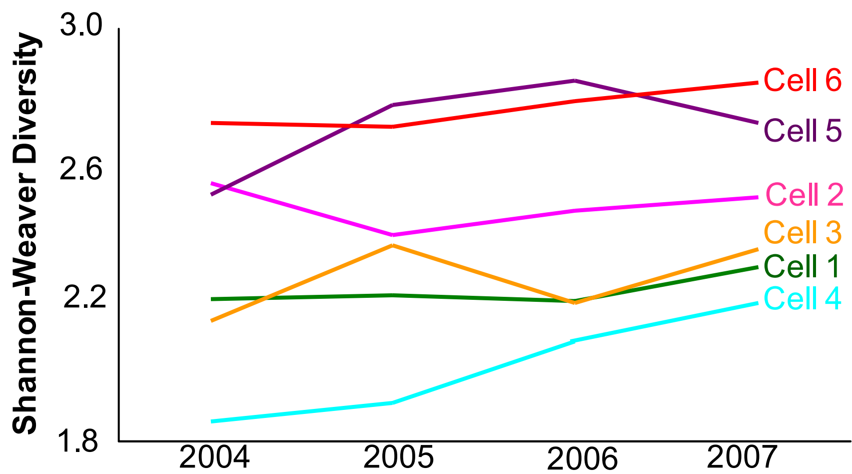

Fig. 14. Shannon-Weaver values for each cell were calculated by averaging values from all 9 of a cell's quadrats for a given year.

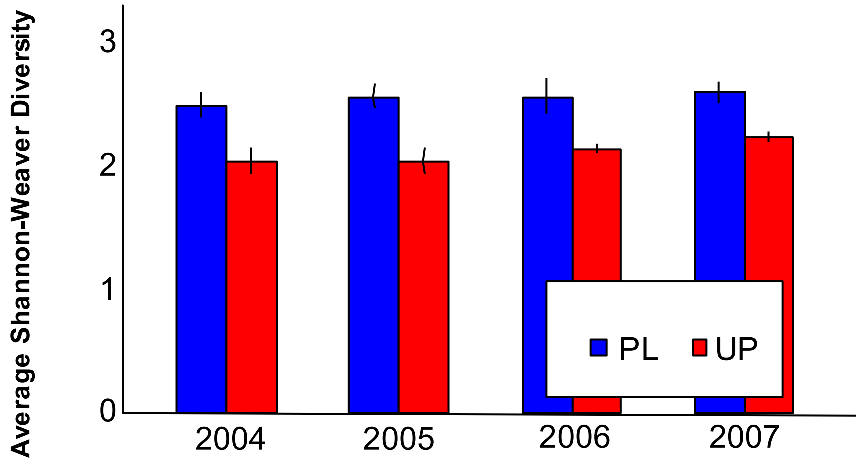

Fig. 15. Comparison of Shannon-Weaver Diversity in Planted (**PL**) and Unplanted (**UP**) cells. All differences are significant (p<.05). Error bars are SD.

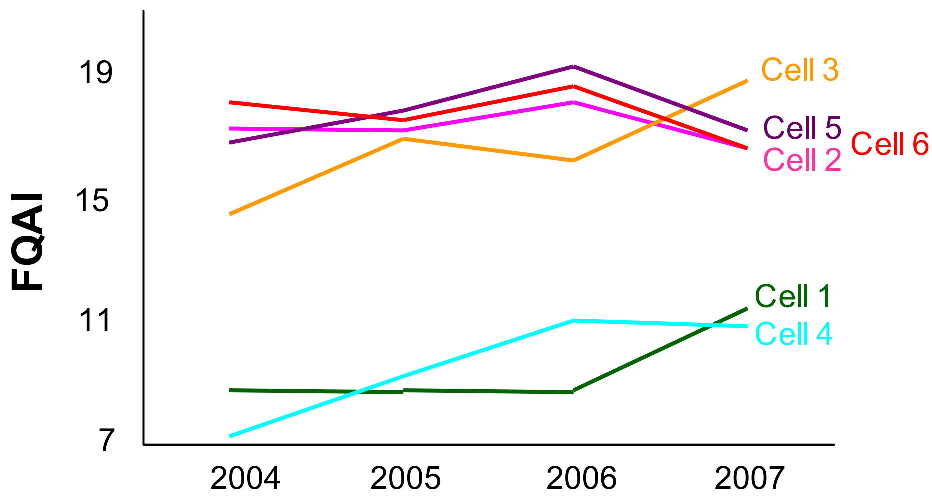

Fig. 16. Whole-cell FQAI from 2004-2007

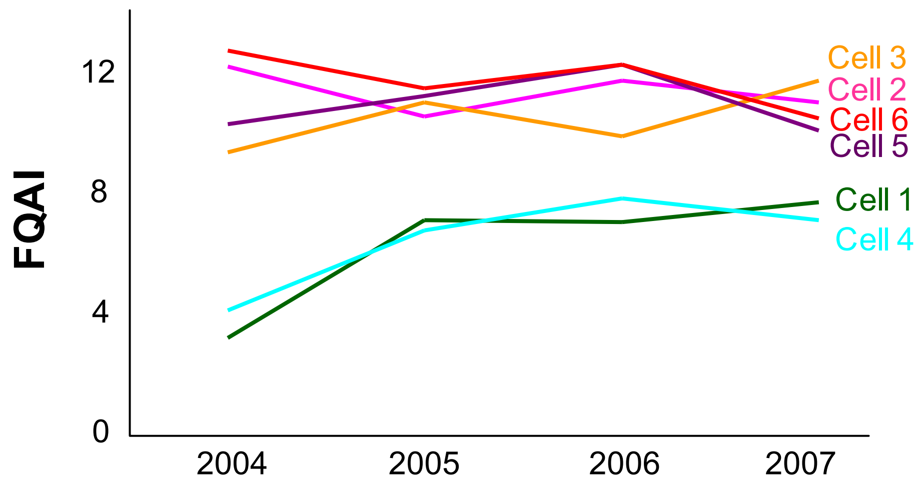

Fig. 17. Average-quadrat FQAI from 2004-2007

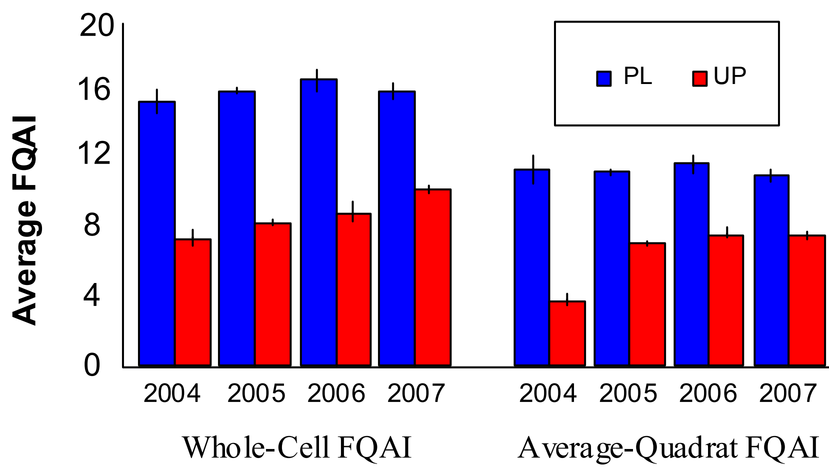

Fig. 18. Comparison of UP and PL values of FQAI for whole cells and for averages of all quadrats within a cell. Error Bars are SD

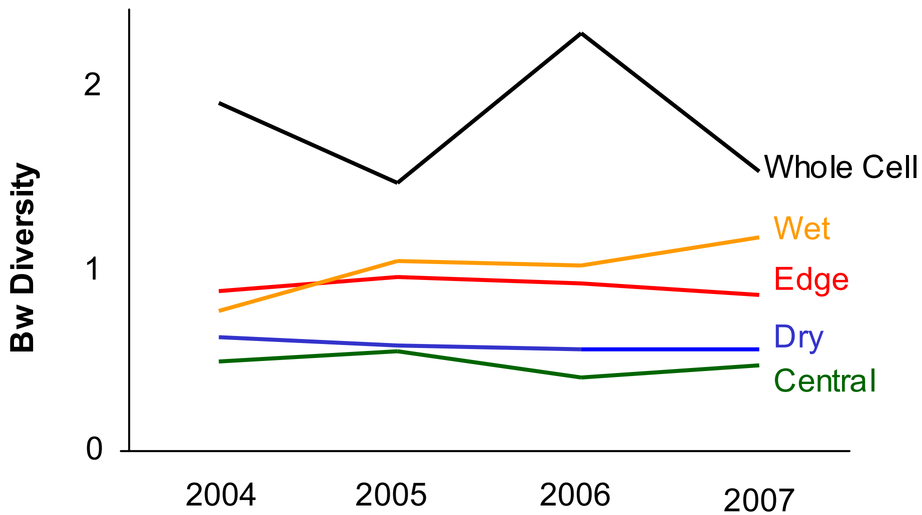

Fig. 19. Bw Diversity at the whole-cell level and in wet, edge, dry, and central sectors in Cell 2. Bw patterns in Cell 2 are representative of all six cells.

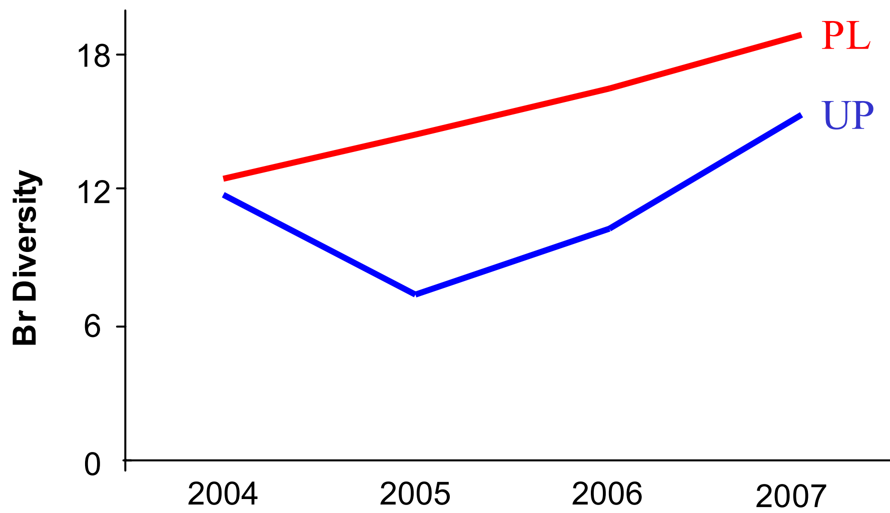

Fig. 20. Br Diversity with cell values averaged to produce Unplanted (**UP**) and Planted (**PL**) treatment values

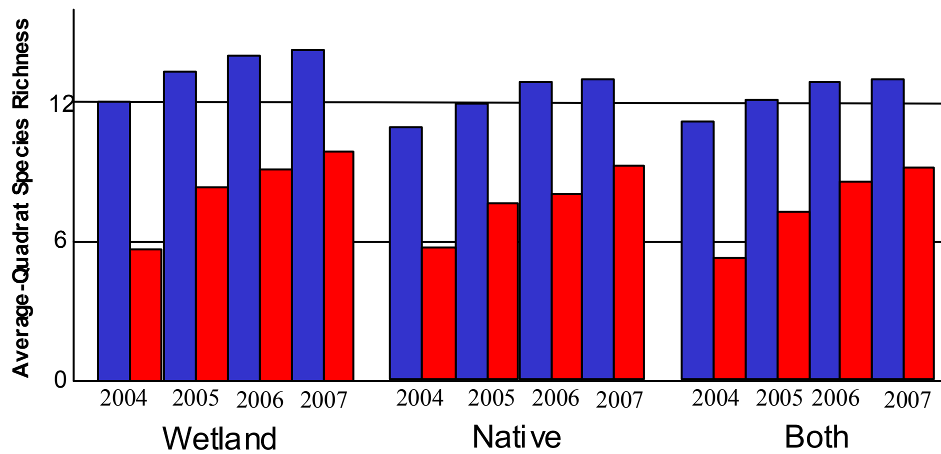

Fig. 21. Comparison of Average-Quadrat Species Richness for Wetland Species, Native Species, and Wetland & Native Species Data for Planted (PL) and Unplanted (UP) cells. All treatment differences are significant.

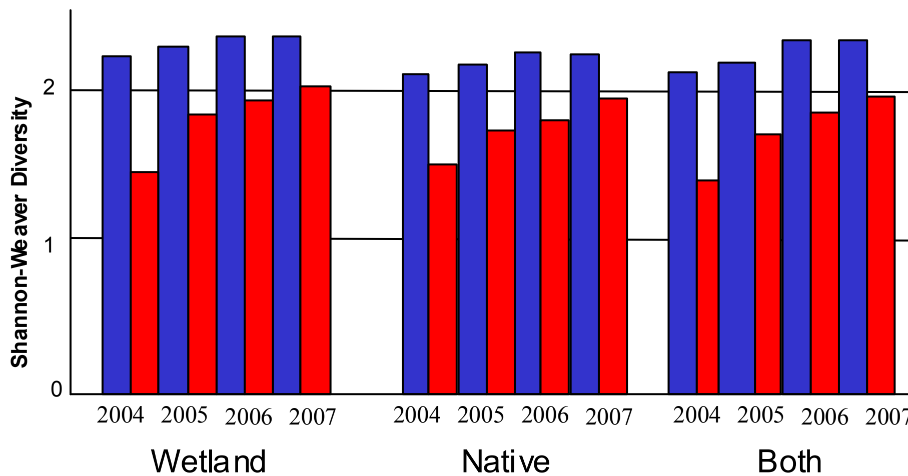

Fig. 22. Comparison of Shannon-Weaver Diversity for Wetland Species, Native Species, and Wetland & Native Species Data for Planted (PL) and Unplanted (UP) cells. All treatment differences are significant.

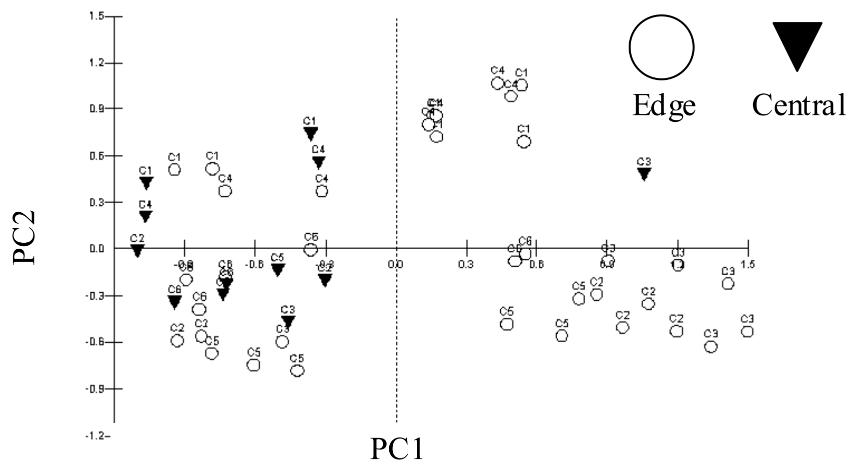

Fig. 23. Principal Components Analysis of 2007 plant diversity data. Points are labeled according to cell ("C1" is Cell 1). Central quadrats generally have lower values for PC1.

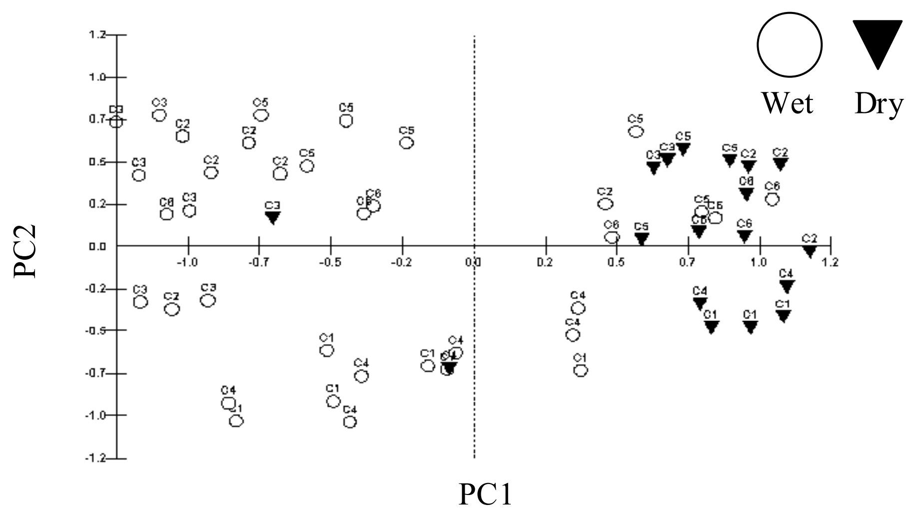

Fig. 24. Principal Components Analysis of 2007 plant diversity data. Points are labeled according to cell ("C1" is Cell 1). Dry quadrats generally have higher values for PC1.

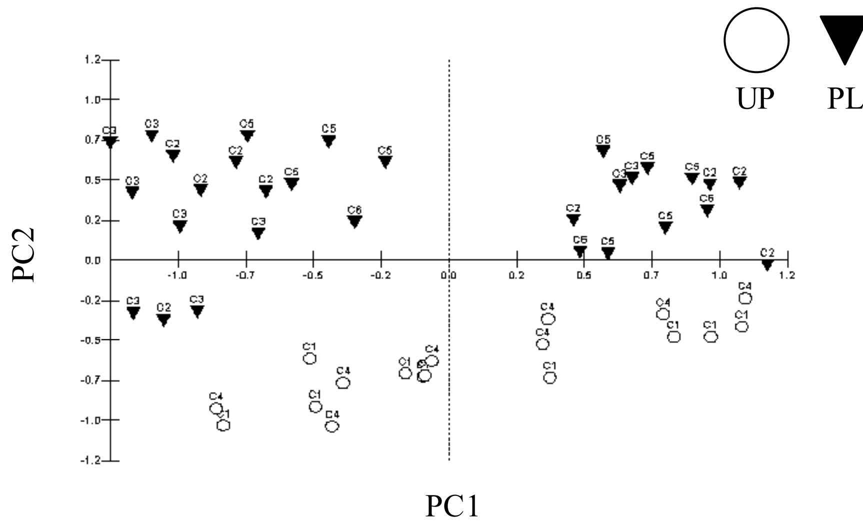

Fig. 25. Principal Components Analysis of 2007 plant diversity data. Points are labeled according to cell ("C1" is Cell 1). PL quadrats generally have higher values for PC2.

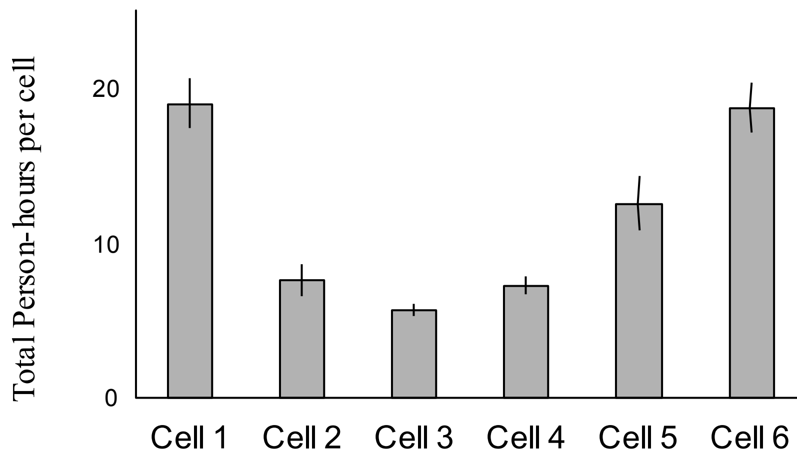

Fig. 26. Total Person-hours spent managing invasive species during the study period. Error bars are standard deviation between annual means.

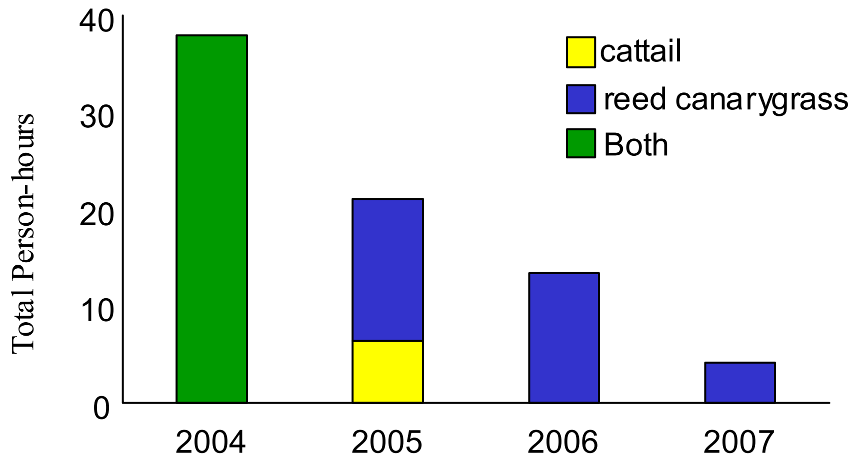

Fig. 27. Total Person-hours spent managing invasive species during the study period. Data from 2004 does not differentiate between time spent controlling cattail and reed canarygrass.

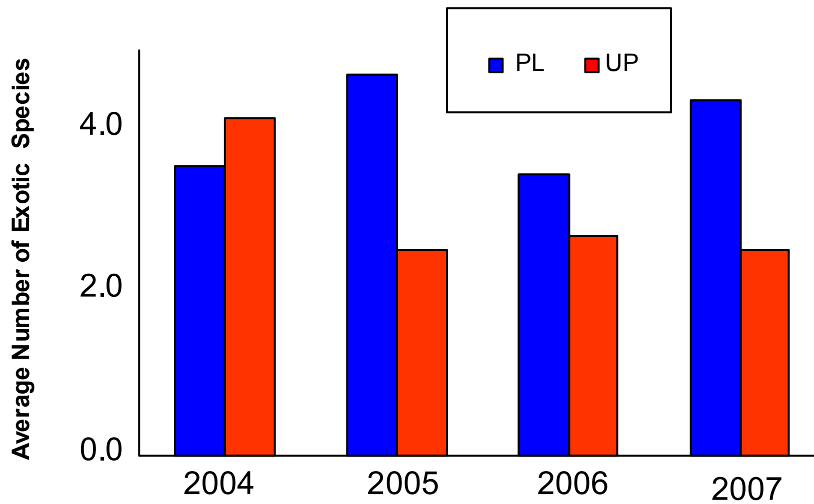

Fig. 28. Comparison of average-quadrat exotic species richness of Planted (PL) and Unplanted (UP) cells. Cells did not show consistent treatment differences in exotic species richness

Tables

Table A. Species planted in PL cells in Fall 2003. Species planted originally as vegetative propagules were replanted in HI cells in 2004, 2005, and 2006. Adapted from Smith (2006).

Species	Planting Method	Wetland Status	Source
Saururus cernuus	Vegetative	OBL	Local
Peltrandra virginica	Vegetative	OBL	Local
Acorus americanus	Vegetative	OBL	Local
Carex stricta	Vegetative	OBL	Nursery
Spartina pectinata	Vegetative	OBL	Nursery
Sagittaria latifolia	Vegetative	OBL	Local
Sagittaria latifolia	Vegetative	OBL	Nursery
Pontederia cordata	Vegetative	OBL	Local
Iris versicolor	Vegetative	OBL	Nursery
Sparganium americanum	Vegetative	OBL	Local
Nymphaea odorata	Vegetative	OBL	Local
Carex frankii	Seed	OBS	Nursery
Carex vulpinoidea	Seed	OBS	Nursery
Schoenoplectus tabernaemontani	Seed	OBS	Nursery
Juncus torreyi	Seed	FACW	Nursery
Juncus effusus	Seed	FACW	Nursery
Hibiscus mocsheutos	Seed	OBL	Local
Ascelpias incarnata	Seed	OBL	Local
Decodon verticillatus	Seed	OBL	Local
Rosa paustris	Seed	OBL	Local
Lobelia cardinalis	Seed	FACW	Local
Cephalanthus occidentalis	Seed	OBL	Local
Peltandra virginica	Seed	OBL	Local

Table B. Cover classes (adapted from Peet, *et al.* [1998]) used to quantify percent cover during diversity surveys. Adapted from Smith (2006).

COVER CLASSES:
1 = single individual or very few individuals
2 = 0-1%
3 = 1-2%
4 = 2-5%
5 = 5-10%
6 = 10-25%
7 = 25-50%
8 = 50-75%
9 = 75-95%
10 = 95-99%

Table C. Criteria for breaking entire dataset into wetland, native native & wetland, and exotic datasets.

Dataset	Criteria	Number
All	All species recorded at the wetlands	98
Wetland	All species designated as obligate or facultative wetland plants by Andreas, et al. (2004)	51
Native	All species designated as native by Andreas, et al. (2004)	55
Native and Wetland	All species designated as both native and either obligate or facultative wetland by Andreas, et al. (2004)	44
Exotic	All species designated as non-native by Anreas, et al. (2004)	42

Table D. Average rates of change in average-quadrat SR, SWD, and average-quadrat FQAI.

	2004/2005	2005/2006	2006/2007	2004/2007
SR				
UP/SD	-6.0	5.0	7.0	6.0
PL	4.3	3.3	5.5	13.0
HI	6.0	4.0	4.0	14.0
LI	2.5	2.5	7.0	12.0
SW				
UP/SD	0.03	0.08	0.10	0.22
PL	0.08	0.01	0.03	0.12
HI	0.06	0.07	-0.04	0.08
LI	0.11	-0.05	0.11	0.16
FQAI				
UP/SD	1.0	0.9	1.3	3.1
PL	0.7	0.6	-0.8	0.6
HI	0.5	1.2	-1.8	-0.1
LI	1.0	0.1	0.3	1.4

Table E. Comparison of average-quadrat and whole-cell FQAI for cells and treatments.

	2004	2005	2006	2007
Average-Quadrat FQAI				
Cell 1	3.2	7.1	7.0	7.7
Cell 2	12.2	10.5	11.7	11.0
Cell 3	9.3	11.0	9.9	11.7
Cell 4	4.2	6.8	7.8	7.1
Cell 5	10.3	11.1	12.2	10.1
Cell 6	12.7	11.4	12.2	10.4
UP	3.7	6.9	7.4	7.4
PL	11.1	11.0	11.5	10.8
Whole-Cell FQAI				
Cell 1	8.7	8.7	8.7	11.4
Cell 2	17.2	17.1	18.0	16.6
Cell 3	14.4	16.9	16.1	18.7
Cell 4	7.3	9.2	11.0	10.8
Cell 5	16.7	17.8	19.2	17.1
Cell 6	18.0	17.5	18.5	16.5
UP	9.9	9.9	10.1	11.1
PL	15.8	17.0	17.1	17.6

Table F. Eigenvalues and Factor Loadings for Principal Components Analyses 1-3. Values for the first two principal components per analysis are shown.

	PCA1		**PCA2**		**PCA3**	
Eigenvalues						
	PC1	**PC2**	**PC1**	**PC2**	**PC1**	**PC2**
Eigenvalues	30.275	14.123	28.345	13.17	30.275	14.123
Percentage	35.61	16.612	34.688	16.117	35.61	16.612
Cum. Percentage	35.61	52.222	34.688	50.806	35.61	52.222
Significant Factor Loadings						
Sparganium americanum		0.497		-0.533		0.487
Sagittaria latifolia		0.427		-0.418		0.427
Elodea canadensis	-0.619		0.634		-0.619	

Table G. Biodiversity findings from comparable wetland restorations. Study systems differ from Jones wetlands in wetland type, restoration treatment, age, data collection methods, and size of system.

Study	Findings
Callaway, *et al*. 2003	SR per sq. meter ranged from 5.29 to 7.75. SWD per sq. meter ranged from 1.38 to 1.82.
DeSteven, *et al.* 2006	mean whole-wetland SR ranged from 16.7 to 35.5 (wet year).
Spieles, *et al.* 2006	mean whole-wetland FQAI ranged from 16.1 to 31.6.
Hartzell, *et al.* 2007	mean Taxa Richness was 6.86/6.92 and mean FQAI was 7.44/4.85 for natural/restored wetlands.
Thompson, *et al.* 2007	SR per sq. meter ranged from 4.62 to 11.00. SWD per sq. meter ranged from 0.90 to 1.89.
Matthews and Endress 2008	Mean FQAI in year 1 was 14.7; mean FQAI in year 4 was 19.4.

Table H. Person-hours spent replanting Cells 2 and 5 (HI treatment)

Date	Hours
7/16/2004	28
7/19/2004	1
7/19/2004	4
7/20/2004	3
9/2/2005	27.5
9/10/2005	13
9/9/2006	10
9/10/2006	16
Total	102.5

Appendices

Appendix 1. Species richness and Shannon-Weaver diversity of cells and treatments.

Species Richness

	2004	2005	2006	2007
All Species (Whole Cell)				
Cell 1	30	26	32	40
Cell 2	35	37	40	45
Cell 3	30	30	25	35
Cell 4	27	19	23	29
Cell 5	36	46	51	54
Cell 6	38	43	53	57
UP/SD	29	23	28	35
PL	35	39	42	48
Wetland Species (Avg. Quadrat)				
Cell 1	6	9	9	10
Cell 2	14	12	14	15
Cell 3	9	13	11	13
Cell 4	5	8	9	10
Cell 5	11	14	16	15
Cell 6	14	14	15	15
UP/SD	6	8	9	10
PL	12	13	14	14
Native Species (Avg. Quadrat)				
Cell 1	7	9	9	11
Cell 2	14	12	14	15
Cell 3	9	13	10	12
Cell 4	6	7	8	9
Cell 5	10	13	16	14
Cell 6	14	14	15	15
UP/SD	6	8	9	10
PL	12	13	14	14
Wetland and Native Species (Avg. Quadrat)				
Cell 1	6	8	9	10
Cell 2	13	12	14	14
Cell 3	9	13	10	12
Cell 4	5	7	9	9
Cell 5	10	13	15	14
Cell 6	13	13	14	14
UP/SD	6	8	9	10
PL	12	13	15	15
Exotic Species (Avg. Quadrat)				
Cell 1	5	4	3	3
Cell 2	2	3	2	3
Cell 3	2	2	1	1
Cell 4	3	2	2	2
Cell 5	5	8	5	6
Cell 6	5	7	6	8
UP/SD	4	3	3	3
PL	4	5	4	4

Shannon-Weaver

All Species				
Cell 1	1.7	1.9	2.0	2.1
Cell 2	2.4	2.2	2.4	2.4
Cell 3	2.0	2.3	2.2	2.3
Cell 4	1.3	1.8	1.9	2.0
Cell 5	2.2	2.4	2.6	2.5
Cell 6	2.5	2.4	2.5	2.5
UP/SD	1.5	1.9	2.0	2.1
PL	2.3	2.3	2.4	2.4
Wetland Species				
Cell 1	1.7	2.0	2.0	2.1
Cell 2	2.4	2.2	2.4	2.4
Cell 3	2.0	2.3	2.2	2.3
Cell 4	1.5	1.7	1.9	2.0
Cell 5	2.1	2.4	2.6	2.4
Cell 6	2.4	2.4	2.5	2.5
UP/SD	1.6	1.9	1.9	2.1
PL	2.2	2.3	2.4	2.4
Native Species				
Cell 1	1.7	2.0	2.0	2.1
Cell 2	2.4	2.2	2.4	2.4
Cell 3	2.0	2.3	2.2	2.3
Cell 4	1.5	1.7	1.9	2.0
Cell 5	2.1	2.4	2.6	2.4
Cell 6	2.4	2.4	2.5	2.5
UP/SD	1.6	1.9	1.9	2.1
PL	2.2	2.3	2.4	2.4
Wetland and Native Species				
Cell 1	1.6	1.8	1.9	2.0
Cell 2	2.4	2.2	2.5	2.4
Cell 3	1.9	2.3	2.2	2.4
Cell 4	1.3	1.7	1.9	2.1
Cell 5	2.1	2.3	2.6	2.5
Cell 6	2.4	2.3	2.5	2.5
UP/SD	1.5	1.8	1.9	2.0
PL	2.2	2.3	2.4	2.4

Appendix 2. ANOVA Statistics for Species Richness

	Index	Year	Area	Average UP	Average PL	Count (UP/PL)	F	Significance
a	SR	2004	Whole Cell	10.3	15.4	18,36	13.2	y**
b	SR	2005	Whole Cell	10.8	17.6	18,36	16.9	y**
c	SR	2006	Whole Cell	11.4	17.4	18,36	9.4	y*
d	SR	2007	Whole Cell	12.6	18.4	18,36	8.9	y*
e	SR	2004	Shallow	13.8	15.7	6,12	0.9	n
f	SR	2005	Shallow	14.3	20.5	6,12	5.6	y
g	SR	2006	Shallow	12.7	29.8	6,12	7.2	y
h	SR	2007	Shallow	14.7	23.1	6,12	8.6	y*
I	SR	2004	Edge	11.4	16.7	12,24	14.0	y**
j	SR	2005	Edge	12.0	18.7	12,24	21.7	y**
k	SR	2006	Edge	13.0	17.75	12,24	7.4	y
l	SR	2007	Edge	14.3	18.9	12,24	7.5	y*
m	SR	2005/6	Whole Cell	5.0	3.3	2,4	0.1	n
n	SR	2006/7	Whole Cell	7.0	5.5	2,4	0.4	n

	Index	Year	Area	Average Wet	Average Dry	Count (Wet/Dry)	F	Significance
o	SR	2004	Dry vs. Wet	8.6	13.8	12,6	7.7	y
p	SR	2005	Dry vs. Wet	9.0	14.3	12,6	6.7	y
q	SR	2006	Dry vs. Wet	10.8	12.7	12,6	0.7	n
r	SR	2007	Dry vs. Wet	12.0	13.7	12,6	0.5	n
s	SR	2004	Dry vs. Wet	15.3	15.7	24,12	0.1	n
t	SR	2005	Dry vs. Wet	16.2	20.5	24,12	4.3	y
u	SR	2006	Dry vs. Wet	15.8	20.8	24,12	3.7	n
v	SR	2007	Dry vs. Wet	16.0	23.1	24,12	8.4	y*

Appendix 3. ANOVA Statistics for Shannon-Weaver Diversity

	Index	Year	Area	Average UP	Average PL	Count (UP/PL)	F	Significance
a	SW	2004	Whole Cell	2.04	2.49	18,36	12.4	y**
b	SW	2005	Whole Cell	2.07	2.57	18,36	13.0	y**
c	SW	2006	Whole Cell	2.14	2.58	18,36	6.6	y
d	SW	2007	Whole Cell	2.25	2.61	18,36	6.5	y
e	SW	2004	Edge	2.20	2.61	12,24	15.2	y**
f	SW	2005	Edge	2.23	2.69	12,24	33.0	y**
g	SW	2006	Edge	2.38	2.66	12,24	5.9	y
h	SW	2007	Edge	2.44	2.70	12,24	7.0	y
I	SW	2004	Shallow	2.34	2.50	6,12	1.1	n
j	SW	2005	Shallow	2.37	2.78	6,12	7.8	y
k	SW	2006	Shallow	2.36	2.81	6,12	11.2	y*
l	SW	2007	Shallow	2.40	2.89	6,12	11.9	y*
m	SW	2004	Central	1.93	2.51	4,8	4.8	0.053
n	SW	2005	Central	2.14	2.71	4,8	5.9	y
o	SW	2006	Central	2.28	2.75	4,8	5.3	y
p	SW	2007	Central	2.29	2.81	4,8	3.7	n
q	SW	2006	Wet	1.89	2.48	12,24	12.3	y*
r	SW	2007	Wet	1.92	2.47	12,24	8.9	y*
s	SW	2004	Wet	2.05	2.46	12,24	3.2	n
t	SW	2005	Wet	2.18	2.47	12,24	2.4	n

	Index	Year	Area	Average Wet	Average Dry	Count (Wet/Dry)	F	Significance
u	SW	2004	Dry vs. Wet	2.28	2.44	36,18	1.3	n
v	SW	2005	Dry vs. Wet	2.29	2.64	36,18	5.7	y
w	SW	2006	Dry vs. Wet	2.32	2.67	36,18	4.0	y
x	SW	2007	Dry vs. Wet	2.37	2.73	36,18	6.5	y

Appendix 4. ANOVA statistics for the Floristic Quality Assessment Index

	Index	Year	Area	Average UP	Average PL	Count (UP/PL)	F	Significance
a	FQAI	2004	Whole Cell	3.7	11.1	18,36	99.1	y**
b	FQAI	2005	Whole Cell	6.9	11.0	18,36	56.6	y**
c	FQAI	2006	Whole Cell	7.4	11.5	18,36	49.9	y**
d	FQAI	2007	Whole Cell	7.4	10.8	18,36	38.4	y**
e	FQAI	2004	Shallow	2.9	9.2	6,12	34.9	y**
f	FQAI	2005	Shallow	7.0	9.5	6,12	13.0	y*
g	FQAI	2006	Shallow	8.3	11.9	6,12	12.3	y*
h	FQAI	2007	Shallow	7.5	10.0	6,12	8.5	y
I	FQAI	2004	Wet	4.1	12.1	12,24	86.0	y**
j	FQAI	2005	Wet	6.9	11.8	12,24	59.2	y**
k	FQAI	2006	Wet	7.0	11.3	12,24	38.4	y**
l	FQAI	2007	Wet	7.3	11.2	12,24	31.1	y**
m	FQAI	2004	Edge	4.3	12.0	12,24	93.8	y**
n	FQAI	2005	Edge	7.2	11.5	12,24	33.5	y**
o	FQAI	2006	Edge	7.6	11.6	12,24	38.0	y**
p	FQAI	2007	Edge	7.5	11.1	12,24	32.1	y**
q	FQAI	2006	Central	2.3	10.1	4,8	20.0	y*
r	FQAI	2007	Central	6.4	10.5	4,8	107.9	y**
s	FQAI	2004	Central	7.6	12.7	4,8	21.6	y**
t	FQAI	2005	Central	7.4	9.9	4,8	3.3	n

Appendix 5. Whittaker's Beta values

Whittaker's Beta for Cells

	Cell 1				Cell 2			
	2004	2005	2006	2007	2004	2005	2006	2007
Central	0.65	0.53	0.30	0.60	0.50	0.56	0.42	0.49
Wet	1.14	0.90	1.26	1.48	0.79	1.07	1.05	1.21
Dry	0.38	0.21	0.50	0.63	0.64	0.59	0.59	0.59
Edge	1.15	0.84	1.27	1.28	0.90	0.98	0.96	0.88
Whole Cell	1.55	1.00	1.69	1.98	1.97	1.52	2.36	1.58

	Cell 3				Cell 4			
	2004	2005	2006	2007	2004	2005	2006	2007
Central	0.76	0.44	0.46	0.60	0.33	0.29	0.36	0.36
Wet	0.85	0.77	0.89	0.81	1.17	1.13	0.90	1.17
Dry	0.95	0.53	0.47	0.62	0.82	0.45	0.34	0.67
Edge	0.89	0.68	0.65	1.09	1.54	0.86	0.86	1.10
Whole Cell	1.65	1.09	1.27	1.56	2.04	1.22	1.11	1.49

	Cell 5				Cell 6			
	2004	2005	2006	2007	2004	2005	2006	2007
Central	0.79	0.57	0.31	0.69	0.41	0.43	0.49	0.52
Wet	0.86	0.98	1.13	1.04	0.91	1.11	1.13	0.91
Dry	0.81	0.52	0.58	0.82	0.64	0.44	0.67	0.64
Edge	0.67	0.81	0.92	1.08	0.80	0.89	1.06	0.96
Whole Cell	1.31	1.20	1.40	1.70	1.07	1.09	1.50	1.54

Whittaker's Beta for Treatments

	UP				PL			
	2004	2005	2006	2007	2004	2005	2006	2007
Central	0.49	0.41	0.33	0.48	0.58	0.43	0.48	0.56
Wet	1.16	1.01	1.08	1.33	0.88	0.94	1.01	0.86
Dry	0.60	0.33	0.42	0.65	0.79	0.49	0.57	0.63
Edge	1.35	0.85	1.07	1.19	0.85	0.78	0.86	1.03
Whole Cell	1.79	1.11	1.40	1.73	1.36	1.09	1.39	1.55

Appendix 6. Routledge's Beta values

Routledge's Beta for Cells

	Cell 1				Cell 2			
	2004	2005	2006	2007	2004	2005	2006	2007
Central	5.00	13.30	6.26	12.33	15.20	15.03	16.66	21.56
Wet	6.05	6.37	10.27	13.78	10.20	11.65	9.90	13.22
Dry	9.00	7.40	7.80	11.25	11.55	12.25	16.78	16.78
Edge	11.76	8.94	12.93	16.25	12.78	12.84	12.47	11.19
Whole Cell	11.79	8.14	12.84	19.00	12.17	15.11	15.67	16.78

	Cell 3				Cell 4			
	2004	2005	2006	2007	2004	2005	2006	2007
Central	16.29	11.90	9.94	15.00	5.00	3.26	6.76	6.76
Wet	6.69	7.97	6.81	8.31	6.05	6.81	6.08	9.59
Dry	12.09	11.07	8.68	10.95	9.00	5.53	6.05	9.00
Edge	7.97	9.90	6.08	12.64	11.76	6.04	7.97	11.25
Whole Cell	12.24	11.16	9.25	13.41	11.79	6.68	7.67	11.55

	Cell 5				Cell 6			
	2004	2005	2006	2007	2004	2005	2006	2007
Central	19.16	22.14	9.26	26.77	13.08	18.44	22.68	24.09
Wet	9.71	15.67	20.63	14.38	12.17	15.79	15.71	14.74
Dry	12.00	14.21	11.65	21.83	13.06	14.38	20.77	21.31
Edge	9.13	14.06	15.64	17.31	12.22	15.98	18.29	18.96
Whole Cell	13.40	17.24	20.15	22.52	12.37	14.54	20.78	23.07

Routledge's Beta for Treatments

	UP				PL			
	2004	2005	2006	2007	2004	2005	2006	2007
Central	5.00	8.28	6.51	9.55	15.93	16.88	14.63	21.86
Wet	6.05	6.59	8.17	11.69	9.69	12.77	13.26	12.66
Dry	9.00	6.47	6.93	10.13	12.18	12.98	14.47	17.72
Edge	11.76	7.49	10.45	13.75	10.53	13.20	13.12	15.02
Whole Cell	11.79	7.41	10.25	15.28	12.54	14.51	16.46	18.94

Appendix 7. ANOVA statistics for Restricted Datasets

	Index	Year	Dataset	Average UP	Average PL	Count (UP/PL)	F	Significant
a	SR	2004	Wetland	5.7	12.1	18,36	44.9	y**
b	SR	2005	Wetland	8.3	13.3	18,36	26.3	y**
c	SR	2006	Wetland	9.1	13.8	18,36	17.4	y**
d	SR	2007	Wetland	9.4	14.3	18,36	15.7	y**
e	SW	2004	Wetland	1.50	2.26	18,36	46.6	y**
f	SW	2005	Wetland	1.88	2.33	18,36	17.2	y**
g	SW	2006	Wetland	1.97	2.41	18,36	10.5	y*
h	SW	2007	Wetland	2.07	2.41	18,36	10.1	y*
I	SR	2004	Native	6.2	11.8	18,36	38.0	y**
j	SR	2005	Native	8.2	12.9	18,36	23.2	y**
k	SR	2006	Native	8.7	13.9	18,36	19.1	y**
l	SR	2007	Native	10.0	14.0	18,36	12.3	y**
m	SW	2004	Native	1.61	2.24	18,36	38.5	y**
n	SW	2005	Native	1.85	2.31	18,36	16.9	y**
o	SW	2006	Native	1.93	2.40	18,36	12.5	y**
p	SW	2007	Native	2.07	2.39	18,36	8.5	y*
q	SR	2004	W&N	5.5	11.5	18,36	41.2	y**
r	SR	2005	W&N	7.5	12.6	18,36	32.2	y**
s	SR	2006	W&N	8.8	13.3	18,36	17.2	y**
t	SR	2007	W&N	9.5	13.4	18,36	15.4	y**
u	SW	2004	W&N	1.48	2.21	18,36	43.2	y**
v	SW	2005	W&N	1.79	2.29	18,36	21.9	y**
w	SW	2006	W&N	1.94	2.44	18,36	13.3	y**
x	SW	2007	W&N	2.05	2.44	18,36	14.3	y**