A comparison of techniques for establishing Nebraska sedge and hardstem bulrush

by Jeffrey M Klausmann

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in Land Rehabilitation
Montana State University

Abstract:
Land development has degraded or destroyed a large proportion of U.S. wetlands. Public recognition of the beneficial functions performed by wetlands has resulted in laws to protect wetlands. The regulatory permitting program under Section 404 of the Clean Water Act requires compensation for most wetland alteration or destruction. Wetland creation or restoration is frequently required to mitigate for unavoidable wetland losses. Because vegetation affects site stability, biogeochemical processes, and wildlife and fisheries habitat, vegetation establishment is one of the primary objectives of wetland creation or restoration projects.

Selection of effective wetland revegetation techniques is critical, but information comparing different techniques is often lacking. In many cases managers have relied on a passive approach to wetland revegetation, which assumes that a representative plant community will colonize naturally once a wetland with suitable hydrologic conditions has been established. Weed infestation, site isolation, and permit requirements often exclude use of passive revegetation. Active revegetation relies on human introduction of plant propagules to ensure relatively rapid plant establishment and to direct future plant community composition. However, active revegetation is costly and little is known about the relative effectiveness of various active revegetation techniques.

I conducted a two-year greenhouse experiment to evaluate five techniques for revegetating Nebraska sedge (Carex nebrascensis) and hardstem bulrush (Scirpus acutus). Survival and growth were compared for wild plug transplants, greenhouse-propagated containerized plants, greenhouse-propagated bareroot plants, stratified commercial seed, all of which may be used in active revegetation, and a control that relied on passive revegetation from the soil seed bank. Benchtop germination trials were used to verify commercial seed viability and the vendor's germination estimates.

Plant survival for Nebraska sedge (NS) and hardstem bulrush (HB) was greater than 85% for all active planting treatments except seed. In the greenhouse, establishment from seed was limited by germination, which was 9% and 22% for NS and HB, respectively. Greenhouse-propagated, containerized plants generally outperformed other treatments for both species through two years. All planted treatments, regardless of species, performed better than controls after two years. Plant treatment effects were generally more pronounced for NS than for HB in both years. Growth differences between planting treatments that were significant after one year diminished after two years. Greenhouse-propagated plants performed significantly better than wild plug transplants.

If these findings apply in the field, planting these species may be more effective than seeding or passive recruitment from the seed bank. At some sites it may also be more cost-effective to plant greenhouse-propagated stock than to collect wild plant material.

A COMPARISON OF TECHNIQUES FOR ESTABLISHING NEBRASKA SEDGE AND HARDSTEM BULRUSH

by

Jeffrey M. Klausmann

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

of

Master of Science

in

Land Rehabilitation

MONTANA STATE UNIVERSITY
Bozeman, Montana

May 1998

APPROVAL

of a thesis submitted by

Jeffrey M. Klausmann

This thesis has been read by each member of the thesis committee and has been found to be satisfactory regarding content, English usage, format, citations, bibliographic style, and consistency, and is ready for submission to the College of Graduate Studies.

24 April, 1998
Date

Chairperson, Graduate Committee

Approved for the Major Department

4-24-98
Date

Head, Major Department

Approved for the College of Graduate Studies

4/28/98
Date

Graduate Dean

STATEMENT OF PERMISSION TO USE

In presenting this thesis in partial fulfillment of the requirements for the master's degree at Montana State University-Bozeman, I agree that the Library shall make it available to borrowers under the rules of the Library.

If I have indicated my intention to copyright this thesis by including a copyright notice page, copying is allowed only for scholarly purposes, consistent with "fair-use" as prescribed in the U.S. Copyright Law. Requests for permission for extended quotation from or reproduction of this thesis in whole or in parts may be granted only by the copyright holder.

Signature Jeffrey Hinsman

Date 4/24/98

ACKNOWLEDGMENTS

I would like to thank my graduate advisor Dr. Paul Hook for his guidance throughout this project and my graduate education. Additionally I would like to thank the other members of my graduate committee Drs. Frank Munshower, Gary Pierce, Otto Stein, and Clayton Marlow for their technical expertise and editing. Thanks to Owens Excavation, Bitteroot Native Growers, Granite Seed, and Aquatic and Wetland Nurseries for their technical support and gracious donation of materials. Further thanks to the staff at the NRCS Plant Materials Center in Aberdeen, Idaho and Cathy Seibert at the MSU Herbarium for their advice and encouragement.

I would particularly like to thank my wife Darcy for her continual moral support and my family for their moral and financial support throughout my academic career.

TABLE OF CONTENTS

TABLE OF CONTENTS – Continued

LIST OF TABLES

LIST OF FIGURES

Figure

ABSTRACT

Land development has degraded or destroyed a large proportion of U.S. wetlands. Public recognition of the beneficial functions performed by wetlands has resulted in laws to protect wetlands. The regulatory permitting program under Section 404 of the Clean Water Act requires compensation for most wetland alteration or destruction. Wetland creation or restoration is frequently required to mitigate for unavoidable wetland losses. Because vegetation affects site stability, biogeochemical processes, and wildlife and fisheries habitat, vegetation establishment is one of the primary objectives of wetland creation or restoration projects.

Selection of effective wetland revegetation techniques is critical, but information comparing different techniques is often lacking. In many cases managers have relied on a passive approach to wetland revegetation, which assumes that a representative plant community will colonize naturally once a wetland with suitable hydrologic conditions has been established. Weed infestation, site isolation, and permit requirements often exclude use of passive revegetation. Active revegetation relies on human introduction of plant propagules to ensure relatively rapid plant establishment and to direct future plant community composition. However, active revegetation is costly and little is known about the relative effectiveness of various active revegetation techniques.

I conducted a two-year greenhouse experiment to evaluate five techniques for revegetating Nebraska sedge (*Carex nebrascensis*) and hardstem bulrush (*Scirpus acutus*). Survival and growth were compared for wild plug transplants, greenhouse-propagated containerized plants, greenhouse-propagated bareroot plants, stratified commercial seed, all of which may be used in active revegetation, and a control that relied on passive revegetation from the soil seed bank. Benchtop germination trials were used to verify commercial seed viability and the vendor's germination estimates.

Plant survival for Nebraska sedge (NS) and hardstem bulrush (HB) was greater than 85% for all active planting treatments except seed. In the greenhouse, establishment from seed was limited by germination, which was 9% and 22% for NS and HB, respectively. Greenhouse-propagated, containerized plants generally outperformed other treatments for both species through two years. All planted treatments, regardless of species, performed better than controls after two years. Plant treatment effects were generally more pronounced for NS than for HB in both years. Growth differences between planting treatments that were significant after one year diminished after two years. Greenhouse-propagated plants performed significantly better than wild plug transplants.

If these findings apply in the field, planting these species may be more effective than seeding or passive recruitment from the seed bank. At some sites it may also be more cost-effective to plant greenhouse-propagated stock than to collect wild plant material.

INTRODUCTION

When the first European settlers arrived, wetlands covered about 9 % (90 million hectares) of what is now the conterminous United States (Dahl and Johnson 1991). Historically, wetlands were viewed as wastelands and were systematically "reclaimed" by draining or filling for agricultural crop production, land development, and disease control. It is estimated that over half of the original wetlands in the U.S. have been lost since that time (Mitsch and Gosselink 1993). Between the mid-1970s and mid-1980s 1.2 million hectares of wetlands were lost in the conterminous United States; freshwater wetlands accounted for 98% of the total loss (Dahl and Johnson 1991). Current wetland losses are from agricultural production, highway construction, urban development, coastal dredging, and stream channelization.

In recent decades changing values and concern about wetland losses created political support for wetland protection. It is now recognized that wetlands are important for fish and wildlife habitat, environmental benefits such as pollution filtration and sediment control, and other socioeconomic values (Tiner 1984). Federal regulation of wetlands began to take effect on a large scale in the 1970s and now encompasses virtually all wetlands (NRC 1995). Federal wetland protection has relied primarily on the Section 404 dredge-and-fill permit program of the Clean Water Act (CWA), executive orders, including a "no net loss" policy, wetland protection programs in agricultural legislation, and the development of wetlands delineation procedures (Mitsch and Gosselink 1993). Increased efforts to protect wetlands have heightened interest in wetland creation to compensate for unavoidable losses. Restoration and creation can help maintain the benefits of wetland ecosystems, and at the same time accommodate the human need for development (Kentula 1996).

Strict enforcement of Section 404 of the Clean Water Act by the U.S. Army Corps of Engineers commonly requires wetland construction or enhancement to mitigate wetland loss due to land development. Mitigation replaces existing wetlands or their functions by creating, restoring, enhancing or preserving wetlands (Votteler and Muir 1993). Mitigation usually involves the creation of new wetlands in the same vicinity to provide similar functions as the wetlands lost (Confer and Niering 1992). The new wetlands are designed to be low-maintenance, self-regulating systems, and the objective of mitigation projects is often limited to satisfying permit requirements for size and vegetative cover (Mitsch and Cronk 1992). Nationwide thousands of wetlands are created for mitigation.

Wetlands are also constructed to treat wastewater from a variety of sources and to enhance wildlife and fisheries habitat. Federal incentives through programs funded under the U.S. Department of Agriculture Conservation Reserve Program and Federal Abandoned Mine Lands Act have resulted in the voluntary restoration and creation of thousands of wetlands by both public and private entities in recent years (Galatowitsch 1993, McKinstry and Anderson 1994). Wetlands have also been created and restored by private sportsmen's and conservation organizations such as Ducks Unlimited and The Nature Conservancy and by state and federal wildlife and fisheries bureaus.

The primary goal of many wetland projects is to provide wildlife habitat and encourage the establishment of diverse, native vegetation (Reinartz and Warne 1993). Regulatory guidelines frequently require establishment of diverse native plant communities in created or restored wetlands. Diversity is usually evaluated by comparing plant species composition and abundance at the project site to a relatively natural reference community. Restoration techniques that increase the likelihood of establishing diverse and desirable wetland plant communities are needed, therefore. Successful revegetation of created or restored wetlands generally involves four primary steps (Fig. 1), any of which may act as a bottleneck that impedes revegetation and results in poor success or outright failure. The process of establishing vegetation begins with a source of propagules, which must reach the project site. Environmental conditions must be suitable for plant establishment and survival. Finally, continued plant growth and reproduction accelerates revegetation and ensures continued community development.

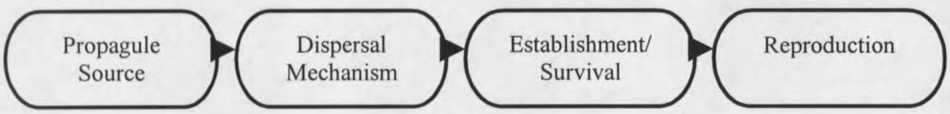


Figure 1. Major components of wetland revegetation (see text for details).

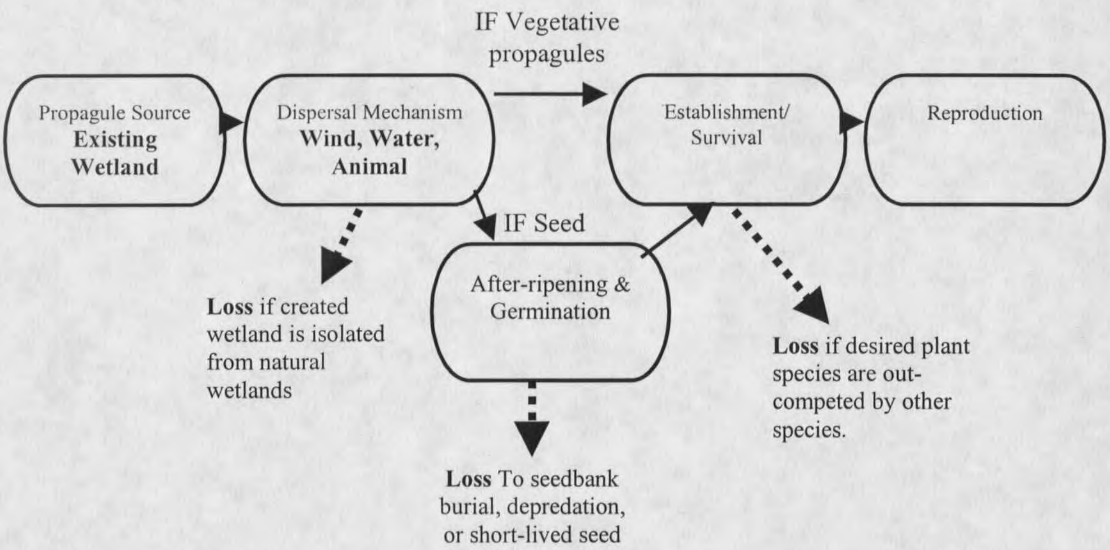
In some cases stands of wetland vegetation become established by natural selection without any plantings at all. This is referred to as passive revegetation (Fig 2a). LaGrange and Dinsmore (1989) suggested that adding water to drained basins in the mid-western U.S. would result in wetland plant species establishment and rapid colonization by a representative community of wetland animals. This is possible when the project site has suitable soils and hydrology for wetland plants and a seedbank exists or plants from adjacent areas supply propagules. These conditions are most likely at previously drained wetland sites restored by reestablishing wetland hydrology. In passive revegetation the propagule source is often an existing natural wetland near the project site. Revegetation depends on the propagule types that are most abundant and easily dispersed, which usually gives an advantage to rapidly colonizing, wind-dispersed species such as cattail (*Typha sp.*). If seed is the primary propagule type available, revegetation is affected by seed after-ripening and germination requirements. Isolation can affect species composition at created wetlands by hindering colonization by species that rely on water for dispersal. Without surface water connections to other wetlands, important species are often underrepresented or absent at a project site.

Because passive revegetation is largely out of human control and at the mercy of natural forces in many created and restored wetlands passive revegetation is unlikely to result in a diverse native plant community. Passive revegetation is particularly dubious when sites lack vestigial wetland seedbanks, are subject to prolonged drainage, are geographically isolated from propagule sources or are next to wetlands dominated by weedy species. Success is also less likely when mitigation projects are required to replace wetland vegetation "in-kind" and the preexisting community has rigid establishment requirements (e.g. sedge meadows). Objectives for mitigation wetlands are often limited to satisfying regulatory requirements for vegetative cover over periods of 3-5 years. Passive approaches usually are not permitted at wetland mitigation projects because regulatory agencies have no way of knowing in advance whether site conditions will allow success (Shank 1997). Consequently permits usually require active revegetation.

Active revegetation attempts to eliminate potential bottlenecks through human intervention at various points along the revegetation pathway (Fig. 2b). Typically humans supply and disperse propagules. Several methods are used to actively revegetate restored or created wetlands. Wetlands can be seeded, substrates such as salvaged soil that contain propagules can be transplanted, or plants can be purchased from nurseries, collected in the wild, or grown for a specific project. Each method has distinct advantages and disadvantages regarding quality of material, availability of plants, and cost for acquisition and planting (Hammer 1989).

Effective revegetation requires information about which revegetation methods will achieve plant coverage rapidly (Zedler and Weller 1989). The cost to revegetate wetlands can be high and the "bottom line" often dictates the method used regardless of its potential for success. Comparative costs, however, cannot be ascertained without determining the relative success of available wetland revegetation techniques or the risk of failure and additional expenditures. Success must be evaluated in terms of project objectives and performance standards as well as plant performance. If the revegetation goal is site stability, then rapid plant establishment may be important. In this case, percent plant survival, growth rates, or productivity may be used to measure success, and revegetation supplemented by low cost seeding with easy to establish species may be appropriate. If plant diversity and establishment of particular species are required, similarity indices and measures of species composition or abundance of undesirable plants may be used to judge success. In this case, a more deliberate revegetation approach may be needed, and practices such as of hand transplanting desired species and aggressive weed control may be required.

a) Passive Revegetation.



b) Active Revegetation

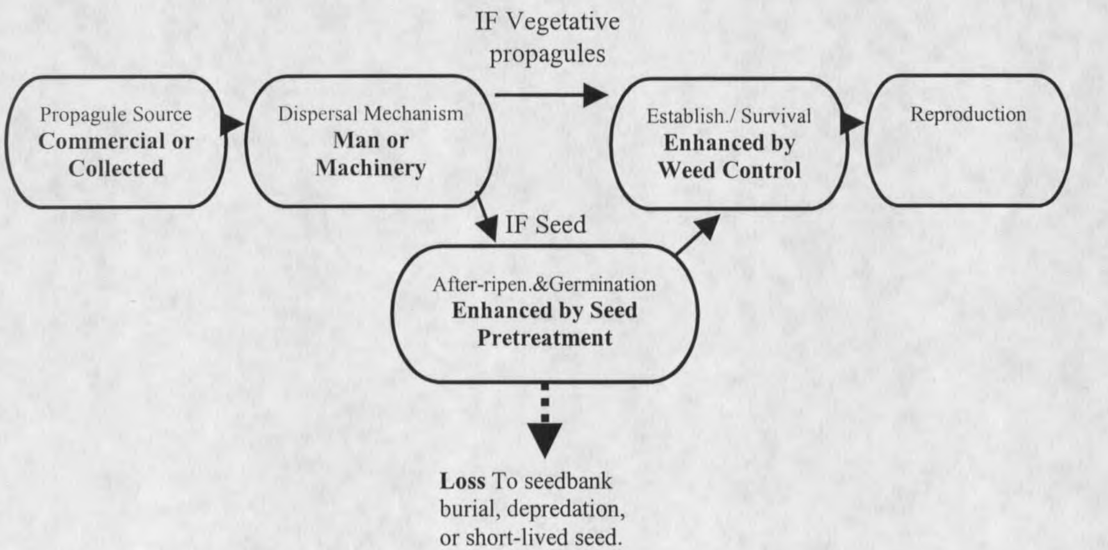


Figure 2. a) Components of passive revegetation. b) Components of active revegetation and human efforts to ensure success.

Research Objective

Few studies have systematically compared the efficacy of active revegetation techniques for establishing wetland plants. The objective of this project was to compare several techniques for actively establishing two species representative of wetmeadows and emergent marshes, Nebraska sedge (*Carex nebrascensis*) and hardstem bulrush (*Scirpus acutus*). Plants were propagated from wild-collected plug transplants, greenhouse-propagated containerized transplants, greenhouse-propagated bareroot transplants, stratified commercial seed, and from the soil's vestigial seedbank. I hypothesized that the amount of effort initially invested, expressed as relative planting unit size or growth stage, would affect the performance (survival and growth) of plants established by active revegetation techniques. For example "started" plants would be expected to survive and perform better than plants recruited from the soil seedbank or broadcast seeding due to a developmental "headstart". My prediction was that overall performance would follow the sequence: wild transplants > containerized transplants > bareroot transplants > stratified seed > passive recruitment from the soil's vestigial seedbank (control). This sequence loosely follows a developmental and economical gradient beginning with wild transplants being most fully developed and requiring the highest initial effort and monetary investment and ending with the control, which requires no initial revegetation effort or monetary investment.

A secondary objective was to evaluate if these plant species are suitable for revegetating sandy barrow pit substrates with relatively poor fertility and physical structure. Borrow pits offer a good model for testing active revegetation techniques because they are ubiquitous in western U.S. landscapes and typically respond poorly to passive revegetation.

LITERATURE REVIEW AND BACKGROUND

Wetland Plant Community Establishment

The establishment of diverse and productive plant communities is a fundamental goal of wetland creation. Diverse and productive plant communities provide for functions related to ecology and also meet regulatory requirements for vegetation coverage and species composition (Weihe and Mitsch 1995). Vegetation establishment affects wildlife use, sedimentation, substrate stability, microbial activity, biogeochemistry, and hydrology. Rapid plant establishment and high ground cover are important (Garbisch 1986) because the time before a site is fully revegetated can be considered a wetland loss (Kentula et al. 1992). One critical gap in our knowledge of wetland creation and restoration is how to efficiently revegetate wetland projects (Kusler and Kentula 1990). Wetland revegetation methodologies are not well developed in the inter-mountain region of the western U.S. The published literature on wetland planting efforts in this region is limited and much of the work was completed with little attention to research design and dissemination of results. Successful wetland creation and restoration depends on understanding the processes involved in vegetation change (Niering 1989).

Traditionally plant establishment at created or restored wetlands has been attempted using two general methods. One is natural colonization from air and waterborne seeds, or vegetative invasion from adjacent areas. The other technique, artificial establishment, includes seeding and transplanting whole plants, shoots, rhizomes, or tubers (Levine and Willard 1989). The first technique is referred to as passive revegetation and the later as active. An intermediate approach is to rely on seedbanks within substrates transplanted from other wetland sites. Substrate is relocated actively, but vegetation establishment is passive.

Mitsch and Cronk (1992) proposed the concepts of "self-design" and "designer" wetlands to describe the amount of human intervention used to steer development of the desired wetland plant community. Self-design wetlands combine active and passive approaches by allowing natural development with some initial seeding and planting. In designer wetlands continuous horticultural selection for desired plants is imposed. Mitsch (1993) advocates self-design because it emphasizes the self-organization capacity of nature to recruit species on its own and to make "choices" from those species introduced by humans. Others have pointed out potential limitations of self-design. A self-design strategy should be adequate when establishment requirements for the desired plant community are flexible, conditions within the wetland are sufficiently varied to accommodate plant requirements, and the weed community does not have the capacity to displace the desired future plant community (Budelsky et al. 1996). Unfortunately, these conditions typically are not met in many created or restored wetlands.

More recently the "efficient-community" hypothesis has been presented and tested (Galatowitsch and van der Valk 1996a). This hypothesis holds that all species that can become established and survive under the environmental conditions found at the site will eventually be found growing there or will be found in its seedbank. In the prairie pothole region some species (e.g.-wet meadow and wet prairie species) are absent from the plant community and the seedbanks of restored wetlands (Galatowitsch and van der Valk 1996a). This was related to dispersal ability and indicated that the efficient community hypothesis cannot be completely accepted as a basis for restorations.

Predicting vegetation change in wetlands is precarious. Wetland creation and restoration practices draw heavily on models of plant community dynamics. Models have been used to predict the future vegetation community using a passive revegetation approach. For example Fennessy et al. (1994) used van der Valk's (1981) qualitative model of succession to accurately predict vegetation changes at the Des Plaines mitigation wetlands (Ohio, USA). The model predicts which species should become established on exposed mudflats during drawdown. Predictions are based on three key life history traits: life span, propagule longevity, and propagule establishment requirements. The physical environment behaves like a sieve, allowing the persistence of species adapted to the conditions at hand. On any given site the species that colonize will depend on the seedbank (van der Valk 1981), relative rates of dispersal, and growth habit and rate of the colonizer in relation to species already present (Kadlec and Wentz 1974).

The systems theory of ecology predicts that ecosystem development is a function of initial conditions and forcing functions acting on the system. Composition of the macrophyte community and seedbank (initial conditions) and the water depth that results from the hydrology (forcing function) influence primary production and species composition and richness (Fennessy et al. 1994). At created wetland sites initial conditions and forcing functions are usually the result of design and construction specifications with constraints imposed by inherent site conditions.

The success of passive revegetation depends on two primary factors. Suitable environmental conditions must exist to enable establishment of seeds or propagules, and seeds or propagules must be present in the soil seedbank or dispersed to the site via wind, water, or animals. Assuming that the proper hydrologic conditions can be established, the focus becomes one of seed source, landscape position, dispersal, species composition, and weed competition.

Passive Revegetation

Freshwater wetland seedbanks have been studied extensively to determine their role in vegetation dynamics (van der Valk and Davis 1978) and marsh habitat management (Kadlec 1962). Successful revegetation from the seedbank depends primarily on seedbank viability and environmental conditions for seed germination. In the prairie pothole region of the United States hydrophytic revegetation at restoration sites relies exclusively on natural recolonization from the remnant wetland seedbank (Galatowitsch and van der Valk 1996b).

At wetland restoration sites duration of drainage may affect seedbank viability. Seedbanks will not contribute to the wetland plant community if they have become depleted through long-term disturbance of the former wetland site or if upland plants have occupied the site for long periods. Seed distribution in the seedbank is also a concern because it may yield unpredictable, patchy vegetation (Kobringer et al. 1983).

Some species (e.g. shallow emergents and sedge-meadow) do not store long-lived propagules in the seedbank and are not typically present in the seedbanks of drained wetlands (Weinhold and van der Valk 1989, Galatowitsch and van der Valk, 1996).

Hydrology, soil chemical and physical properties, photoperiod, and temperature are all critical environmental variables that affect seed germination and establishment. Many wetland plants require specific environmental conditions to trigger physiological responses for reproduction. For example many emergent and mudflat annual species store long-lived seeds in the wetland seedbank. An absolute light requirement for germination contributes to the longevity of buried seeds (Baskin et al. 1989). These species do not become established from seed within stands of existing revegetation because shading from the plant canopy reduces temperature fluctuations and prevents seed from being exposed to direct sunlight (van der Valk 1981).

There is usually only a narrow window of environmental conditions in which a seed can successfully germinate and mature (Salvaggio 1996). Light and temperature are the environmental conditions most frequently used to induce seed germination (van der Valk and Davis 1978). Soil microtopography exerts its effect largely by modifying seed water relationships (Keddy and Constabel 1986). The lack of organic matter in newly constructed systems may limit the establishment of desirable taxa like *Carex* while favoring weedy taxa like *Typha* (Confer and Niering 1992). Seed size can also affect rates of germination.

Seed germination depends strongly on hydroperiod and water chemistry. Seedling emergence is highest when soil moisture is high (van der Valk et al. 1992), temperature is moderate, and soil electrical conductivity is low (Welling 1988). Surface soil moisture levels in constructed and restored wetlands are strongly affected by soil particle size (Keddy and Constabel 1986) and the organic matter content (van der Valk and Pederson 1989). Plants tend to become established in shallow margins (Fennessy et al. 1994) because standing water strongly limits which seeds germinate (van der Valk and Davis 1978, Smith and Kadlec 1985, Willis and Mitsch 1995, Brown and Bedford 1997). Many emergent and annual species require oxygen to germinate and during the marsh drawdown phase these species quickly revegetate exposed mudflats (van der Valk and Davis 1978). The success or failure of a project can depend as much on environmental conditions as on the composition of the seedbank. For these reasons the seedbank alone will not be sufficient to re-establish all the species formerly present (van der Valk et al. 1992).

Because recruitment from the seedbank is unpredictable, seed dispersal may play a pivotal role in the revegetation of wetlands using a passive approach. This is particularly important at created sites where a viable wetland seedbank is usually lacking. If the proper environment exists, a created or restored wetland can become quickly revegetated by volunteer species (Stauffer and Brooks 1997). At the Olentangy Wetland Research Park in Ohio, 28 naturally colonizing macrophytes were observed at a created wetland site two years following construction (Weihe and Mitsch 1995).

Many wetland plants have the ability to float for long periods of time making them particularly adapted to dispersal by water (Morton and Hogg 1989) others are dispersed by wind or animals. If the constructed wetland is located downstream, adjacent to, or near to an existing wetland, it is likely that the project will be able to revegetate passively by plant invasion alone (Kentula et al. 1992). As distance to the nearest wetland seed source increases, the diversity and number of native wetland species decreases significantly (Reinartz and Warne 1993).

As natural plant communities become increasingly fragmented by human activities, the ability of the isolated wetlands to revegetate themselves beyond early successional stages may be hampered by the limited diversity of proximate seed sources (Odum et al. 1983). Because many mitigation wetlands are isolated and unconnected to natural wetlands, dispersal of many species' seeds to these wetlands may no longer occur, particularly for species with seeds that are neither wind nor waterfowl dispersed (Galatowitsch and van der Valk 1996b). The result in many cases is the absence of characteristic wetland community types. In a study of restored prairie pothole wetlands Galatowitsch (1993) found that after three years, wet prairie and sedge meadow zones had not reestablished in most wetlands. Sedge meadows and fens are important for a number of reasons including their importance as regional water storage and purification systems, their value as habitats for wildfowl and furbearers, and their production (Bernard and Soukupova 1988).

A final problem often associated with passive revegetation is weed competition and infestation. Surrounding vegetation strongly affects the species composition of disturbed sites (Borgegard 1990). Because the dispersability of propagules of different species varies significantly, the rate of revegetation of different species in a restored wetland will also vary significantly (Galatowitsch and van der Valk 1996a). There is often little control over what species will enter new wetland sites; those that do may not be the most effective in achieving the desired plant composition. Some species of marsh and aquatic plants such as cattail (*Typha sp.*), with K-selected adult traits and R-selected juvenile traits are well adapted to disturbed sites and tend to take over an area and exclude other more desirable species for long periods of time (Shipley et al. 1989). Cattails are widespread in the landscape and tend to form large, nearly pure stands with low structural diversity and wildlife food value (Rogers 1992, Brown and Bedford 1997). In a survey of created and natural wetlands in Connecticut a cattail monoculture was the characteristic emergent vegetation at created sites whereas a more diverse mosaic of emergent wetland species was often associated with cattail at natural sites (Confer and Niering 1992).

The long term result of wetland weed infestations may be an environment that has limited functional value for wildlife habitat and is less aesthetically pleasing than originally planned. Introducing plant material to unvegetated wetland project sites may be effective in controlling undesirable weedy species and directing succession to a desired end.

Active Revegetation

The need to stabilize substrates, regulatory requirements dictating rapid plant establishment, and the desire to enhance vegetation diversity often encourage active revegetation. Planting provides relatively rapid vegetative cover, which could inhibit the germination of cattails (van der Valk and Davis 1978) and enhance long-term vegetative diversity in created wetlands (Reinartz and Warne 1993).

Wetland plant material can be introduced in several different ways. Selecting a planting technique depends on a host of considerations, but the most significant are plant availability and the overall cost associated with material collection or purchase, handling time, and installation and initial care. Even though active revegetation techniques increase the initial cost of construction, they may prevent the need for more costly, intensive management techniques to control weeds or repeated plantings if initial efforts fail.

The most common active revegetation techniques include seeding, transplanting wild or commercial plants or plant parts, and transplanting salvaged marsh surface substrate. Each technique has distinct advantages and disadvantages. Erwin (1989) has suggested two points to consider if planting is the preferred method of revegetation. First, will the species of plants be available in the numbers required when needed during the construction period? Second, can all the materials be planted in a reasonable time to improve chances for survival? The size of the planting site may also influence the choice of propagule type and establishment technique.

Wetland establishment by seeding is the most economical approach, but success is least predictable. Seed propagation has the potential to provide many plants quickly and is most suitable when large areas need to be established (Chambers and McComb 1994). Similar to natural recruitment from the wetland seedbank, artificial seeding is usually germination limited. Consequently, seeding can be unreliable for many species.

Relatively little is known about the germination requirements of most aquatic and wetland plants (Garbisch and McIninch 1992). Superficially similar wetland species with the same growth form often do not have the same or even similar seed germination requirements (Galinato and van der Valk 1986). In addition, many aquatic plant species' seeds require stratification for enhanced germination. Storage requirements for seed vary greatly and many aquatic species must be stored in water and at cool temperatures if seeds are to retain maximum viability (Muenscher 1936).

Small-seeded marsh species are most susceptible to poor germination by burial in the sediment and require high light intensity and fluctuating temperatures for enhanced germination (Galinato and van der Valk 1986). Seed floatation can be another problem. Because of the tendency of wetland plant seeds to float for long periods before sinking, seeding of flooded areas is questionable (Garbisch 1996). Finally the timing for seeding may be critical. In order to capitalize on the full growing season seeding must be accomplished early on. Poorly developed seedlings may not over winter in some regions of the country and they are most vulnerable to animal depredation (Garbisch 1986). Environmental and biological constraints associated with seeding wetland plants may limit the use of this method at all but the very largest project sites where other methods are not economically feasible. Pierce (1994) found that over the course of ten years of wetland plantings, that the most efficient and economical method to insure presence of most emergent species is to plant transplants and not attempt to seed them.

The most successful as well as the most expensive method of wetland establishment is transplanting plugs, sprigs, and dormant underground plant parts

(Garbisch 1986). Transplant material is essentially available at all times of the year; seed and certain vegetative structures are available only during certain seasons. These plant materials have either the top growth or the stored energy to emerge from the water and sustain growth after planting. Hardiness is another major advantage; transplants are more resistant to physical stresses such as siltation, erosion, or current and wave action making this method more successful then seeding (Woodhouse et al. 1972). Transplants can be collected in the wild or acquired from commercial nurseries as tubelings or bareroot planting units.

Wild collections have the advantage of local genetic adaptation. In most cases the planting of propagules from similar or nearby habitats will have a greater probability of success than plants from distant sources (Kadlec and Wentz 1974). In Ohio at the Olentangy Wetland Research Park, 2400 plant propagules of 14 species typical of local marshes were introduced to a constructed wetland. It is noteworthy that the only species planted with local stock also had by far the greatest survival (Weihe and Mitsch 1995). Transplanting plugs or cores from existing wetlands has the added benefit of bringing seeds, shoots and roots of a variety of wetland plants along with associated microfauna to the newly constructed site. Disadvantages of local collections are: 1) weedy species may be inadvertently included; 2) logistics or collecting conditions may increase costs; 3) plant supply may be limited due to local regulations or access restrictions; and 4) plants may be unavailable early or late in the growing season (Hammer 1989). Establishment, however, is relatively rapid and planting intervals of 0.5 –1.0 meters for herbaceous perennials have successfully filled areas within one to two growing seasons (Willard et al. 1989). On a hectare basis, harvesting and replanting at this spacing interval would require 134 man-hrs. /hectare (Woodhouse et al. 1972). This estimate does not include logistical costs associated with the distance between the plant source and the created wetland project site. The use of commercial plants eliminates this concern but may be costly due to shipping and plant storage.

Commercial plants are usually supplied as vegetative propagules such as rhizomes and tubers or as transplants. Transplants are grown from seed or propagated as vegetative divisions; they are usually shipped potted or bareroot. Rhizomes and roots are generally collected in the wild and shipped in a dormant condition. Special concerns associated with commercial plant supply are environmental or genetic adaptation, limited species selection and quantity, and storage and shipping. Plants should be shipped by express service to ensure survival, which may add to their cost. Storage prior to planting is also critical and plants should be kept moist, shaded and cool if possible. The major advantages of commercial plant acquisition are that plants are available at the planting location in predicted quantities and at the desired time, and cost is controllable. Containerized plants may be planted at anytime during the year – in a growing condition or in a dormant one.

Species Selection

Before plants are established actively it is necessary to decide which species are most likely to grow at the site and at the same time valuable for desired wetland functions. Physiological, ecological and economic criteria are usually considered in wetland plant species selection in this regard. Newly created wetlands vary in their physical and chemical characteristics, and it is essential to consider them when assessing potential species for selection. Above all plants selected for created sites must be adapted to the expected hydrologic regime of that site. A common pitfall in wetland planting plans is the grouping of species without regard to their adaptive strategies and the environments in which they are best able to utilize those strategies for survival and interspecific competition (Pierce 1994).

Emergent wetland plants are often chosen for planting in created wetlands because of their desirable attributes. Many emergent wetland species are ecologically important for wildlife food and cover, site stability, and wetland biogeochemistry. Knowledge of wild propagation is economically valuable because ponds with an emergent fringe are the most commonly created freshwater wetland in the country (Kentula 1996). Generally, marsh emergents are considered either "sprouters", which are readily established from rhizome transplants but not by seed and "seeders", which can be readily established from seed but not from rhizome transplants. Seeder species occur in large populations after disturbance in natural systems; sprouter species are often more desirable in the long term, but sufficient source material is sometimes difficult to obtain (Chambers and McComb 1994). Rhizomatous species have the added benefit of relatively rapid colonization due to extensive vegetative growth, which also promotes substrate stability. Emergent plants often possess a robust growth habit further aiding in site stability by dampening the erosive forces of wind and water.

Many emergent species support abundant invertebrate populations that are commonly used by wetland avifauna. Rhizomes and tubers are also utilized as forage by many wildlife species (Payne 1992). More important to wildlife is the cover and production of emergents, which provide critical shelter from weather and predators. In many cases the biomass contribution of emergent species dominates the total biomass of the wetland (Fennessy et al. 1994). Attributes of high stem density and production have been long been recognized by managers at many created wetlands (eg. sewage-treatment lagoons and sediment filter strips) to enhance wetland functions and emergent vegetation provides these attributes. Aside from the ecological attributes of wetland plant selection, economic factors also play a dominant role.

Increasingly wetland revegetation plans specify using only regionally native wetland plants. This specification may drastically limit commercial availability depending on the geographical location of the project site. Limited consumer demand usually prohibits larger nurseries from supplying localized species with narrow geographical ranges. Therefore, wetland species that are regionally important but occupy relatively narrow ranges must often be grown on contract, which may add to overall project costs.

In this study selection was restricted to native wetland plants, which significantly limited the pool of potential candidate species for selection. Additional selection criteria included: regional abundance; high likelihood for survival in a disturbed setting such as an abandoned gravel pit; commercial availability in several different propagule types (e.g. seed, bareroot & containerized units); and access to a wild population for collection of plug material. Several types of planting material were required to test potential growth differences between commonly used revegetation techniques.

Nebraska sedge and Hardstem bulrush

Hardstem bulrush (*Scirpus acutus* Muhl.) and Nebraska sedge (*Carex nebrascensis* Dewey) occupy relatively wet and dry areas of many western emergent marsh communities, respectively.

Nebraska sedge is a native obligate wetland plant that occurs at low to mid elevations (1000 to 2050 m) throughout the western US. It is common and widely distributed, east of the Cascade Mountains from British Columbia south to California and east to New Mexico, Kansas and South Dakota (Hitchcock and Cronquist 1973).

All members of the genus *Carex* exhibit a more or less uniform graminoid growth form, reproduction is by rhizomes or other asexual means to form clones (Bernard 1989). A genet is the total biologic or genotypic individual; an individual is composed of several shoots, including shoot clumps and a genet like a shoot passes through stages of development and eventually dies (Ratliff and Westfall 1992). Estimates of *Carex* genet lifespans differ over a wide range; theoretically clones of many species may survive for hundreds of years by growing at one end and dying at the other (Bernard 1989).

Nebraska sedge is a moist soil species and will normally be found in a fluctuating water regime. Common habitats throughout its range include wet meadows, swamps, and ditches, often in alkaline soils. In California the species is found on level to nearly level sites where water flows over the surface in spring but does not pond; nearly pure stands grow where overflow water is about 10-cm deep (Ratliff and Westfall 1992). In Montana Nebraska sedge communities occur on slightly drier sites than the hardstem bulrush (*Scirpus acutus*) and beaked sedge (*Carex rostrata*) habitat types (Hansen et al. 1995).

In Montana the Nebraska sedge community type is not extensive and represents a grazing disclimax according to Hansen et al. (1995). The community is typical of early to mid-seral communities. Nebraska sedge is highly rhizomatous with high below-ground biomass, while very palatable to livestock and wildlife, it appears to withstand moderate to heavy grazing pressure or defoliation (Hansen et al. 1995, Ratliff 1983). Nebraska sedge communities, aside from providing valuable forage, have high root length densities, which suggests superior site stabilizing characteristics (Manning et al. 1989). Auclair (1982) has suggested that extensive root growth in *Carex* meadows may result from nutrient limitations. These qualities make the species an excellent candidate for riparian and wetland rehabilitation particularly when grazing is a desired component of the post-rehabilitation landuse.

Hardstem bulrush is also a native obligate hydrophyte that occurs at low to mid elevations (665 to 2025m) in temperate North America and is most common throughout the western U.S. (Hitchcock and Cronquist 1973). Hardstem bulrush is a robust rhizomatous perennial, mostly 1-3m tall, and usually found on permanently saturated soils that may be inundated for long periods. It can grow at water depths up to 1m. Typical habitats include marshes, ditches, and muddy shores of lakes and streams. Hardstem bulrush forms nearly pure stands and is known for its wide-ranging physical and chemical tolerances that include diverse substrate type, hydrology, salinity, alkalinity and pH. It is an early colonizer of suitable habitats and is able to persist under wet conditions.

In Montana the hardstem bulrush habitat type typically occurs as a fringe along the margins of ponds and also occupies basins where water tables remain relatively high but may drop below the soil surface later in the growing season (Hansen et al. 1995). This habitat type provides valuable nesting habitat for a variety of songbirds and waterfowl. Stems, tubers, and seeds are consumed by a variety of wildlife including geese and muskrats.

MATERIALS AND METHODS

A greenhouse experiment was used to compare relative survival and growth of plants established by different active revegetation techniques for Nebraska sedge and hardstem bulrush. Wetland hydrologic conditions were maintained by placing plants in a stock tank with a constant water level. The statistical design was completely randomized with five plant material treatments including a control. Eight replicates were used.

Greenhouse Experimental Setup

Plants were grown in containers made from perforated polyvinyl chloride (PVC) pipe. Each container was 38.50 cm tall and 10.25 cm in diameter. Containers were cut lengthwise down opposite sides and then clamped back together using a screw-clamp so that belowground biomass could be sampled more easily at the end of the study. Four treatments for each species and the eight control containers resulted in a total of 72 plant containers. Two 100-gallon plastic stock-watering tanks were used to control water levels. The bottom of each tank was filled with approximately 15 cm of clean pea gravel to anchor sample containers. Containers were pushed upright into the gravel and leveled to the same elevation. Each stock tank held 36 containers.

Clean mineral sand was packed into each container to a depth of 20 cm from the top of the containers. Heat-pasteurized, mixed borrow pit coversoil was added to all containers except the eight control containers. Soil used for control containers was identical except that it had not been pasteurized. A depth of 7.5 cm of soil was added over the sand and allowed to settle.

Stock tanks were filled with water until soil in all containers was covered by 1-2 mm of standing water. After standing overnight, soil was added to containers that had settled to establish uniform levels. To simulate favorable field hydrologic conditions, in which newly planted material is not subject to drying, the water level was held static at the soil surface. The stock tank water level was maintained by supplying a constant trickle of water to the tank and allowing water to drain through an outlet set at the same elevation as the soil surface. This resulted in saturated conditions during the growing season.

Plant material treatments consisted of greenhouse-propagated bareroot and containerized plants, stratified commercial seed, and wild plug transplants. An unplanted control was used to test passive seedling recruitment from the soil seedbank. Sixteen plants were used for each treatment. Extra plants were obtained to ensure uniform initial size within each treatment; I discarded exceptionally large and small plants and kept plants of similar size. From this pool a random sample of 16 plants was selected. Eight plants were randomly selected for destructive initial biomass sampling and eight were planted in containers for each of the treatments.

Treatments were assigned to containers in a completely randomized design. Containers were planted on June 20, 1996 and were checked daily for a period of four weeks to record seed germination for seeded treatments. At the end of October 1996, tanks were drained to simulate annual groundwater drawdown and stored at 4° C in a moist, dark, controlled environment room for a period of approximately four months. In May, 1997, the tanks were removed from cold storage and returned to the greenhouse for