EXCERPT FROM:

Development of a Statewide Freshwater Wetland Restoration Strategy

Site Identification and Prioritization Methods

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Task C. Generic review of the types of freshwater wetland restoration opportunities that exist in Rhode Island, both in terms of impacts that would have to be removed and the probability of restoration success for individual freshwater wetland types.

Section 1: Impacts to Rhode Island Freshwater Wetlands

We identified nine categories of impacts that might be removed during restoration of freshwater wetlands in Rhode Island. Those impacts, and the restoration activities that may remedy them, are discussed in the following paragraphs in relation to specific wetland functions. Although additional impact categories exist (e.g., nonpoint-source pollution and conversion to deepwater habitat), they are either considered to be beyond the scope of this project or there are other factors—ecological or social—which impede restoration in those cases. Information in this chapter contributed heavily to the development of prioritization methods, which are presented under Task F. A summary of the information presented in this chapter is provided in Table C1. Recommendations for the relative priority that each impact type should receive are presented in Table C2.

Filling

Impact description and restoration required. A wide variety of materials (e.g., sand, gravel, rock, tree stumps, and construction and demolition debris) have been dumped in wetlands throughout Rhode Island. Filling may destroy an entire wetland or a portion of one. Although the vast majority of wetland fill sites have been built upon, some areas remain undeveloped; these are often excellent candidates for restoration. Restoration activities would primarily involve removal of fill material and a return to original wetland elevations.

Effects of impact removal on wetland functions. Removal of fill material may be more effective than any other restoration activity at increasing the functional capacity of altered

Table C1. Probable effects of restoration activities on wetland functions. Symbols are defined as follows: "+" = positive effect on function, "-" = negative effect, "+/-" = positive and negative effects, "n/a" = no effect. Parentheses indicate uncertainty or minor effects.

	Restoration activity									
Long-term function	**Remove fill**	**Remove sediment**	**Recreate natural channel**	**Plug ditches**	**Remove trash**	**Reestablish wetland vegetation**	**Reestablish upland vegetation**	**Enhance surface flow**	**Remove invasives**	**Effective restoration activities**
Flood abatement	+	+	+	+	+	+	(+)	-	n/a	7
Water quality improvement	+	+	+	+	+	+	+	+/-	n/a	7+
Groundwater recharge	(+)	(+)	+	-	(+)	(-)	(-)	(+)	n/a	5
Groundwater discharge	+	+	(+)	+	n/a	(-)	(-)	-	n/a	4
Wildlife habitat	+	+	+	+	+	+	+	+/-	+	9
Fish habitat	+	+	+	+	+	+	+	+	n/a	8
Heritage	+	+	+	+	+	+	+	+/-	+	8+
Functions benefited	7	7	7	6	6	5	5	2+	2	

Table C2. Restoration priority of impacts to Rhode Island freshwater wetlands.

Type of impact	*Priority*[1]
Filling	High
Draining	High
Removal of adjacent upland vegetation	High
Impedance of surface flow	Moderate
Removal of wetland vegetation	Moderate
Trash dumping	Moderate/Low
Stream channelization	Moderate/Low
Invasive species	Low/Moderate
Sedimentation	Low

[1]See Task C, Section 1 text for rationale.

wetlands. This activity results in an increase in the abundance and size of wetlands, and therefore has the potential to positively influence each of seven wetland functions (Table C1). Removal of fill increases the total volume of floodwater that a wetland has the potential to store. By increasing the abundance or size of wetlands, surface water is "filtered" through a greater wetland area, resulting in increased nutrient transformation, sediment trapping, and pollutant removal capabilities. Groundwater discharge would be more likely to occur because fill removal lowers the ground surface to a point closer to the groundwater table. Groundwater recharge would be more likely to occur where fill removal restores basin wetlands that could collect and hold surface runoff or streamflow for extended periods. Wetland-dependent wildlife would benefit from an increase in the size or abundance of wetland habitat. Fish would directly benefit from an increase in open water or marsh habitat, and may indirectly benefit from restoration of other adjacent habitat types due to the potential for improved water quality. Aesthetics, educational opportunities, biodiversity, open space, and recreational potential all can be expected to increase with greater wetland area.

Recommendations. Because filling has resulted in extensive wetland loss in this State, and because removal of fill material has the potential to positively influence all seven of the wetland functions of interest, wetlands that have been filled should receive high priority during restoration planning.

Sedimentation

Impact description and restoration required. Sedimentation occurs most often where roads—which are sanded during winter—cross wetlands. Other sources of sediment may include unpaved road surfaces, cultivated fields, active construction sites, and gravel mining operations. The effects of sedimentation are similar to those of filling (see above). Portions of wetlands may be converted to upland, or wetland surface waters may become turbid or shallower due to sedimentation.

Effects of impact removal on wetland functions. As with fill removal, removal of sediment has the potential to enhance all seven wetland functions.

Recommendations. In most instances, sedimentation is an ongoing process; simple removal of sediments may not result in restoration of sustainable wetland functions and it may contribute to the establishment of invasive species. For these reasons, sediment removal should not be a primary focus of wetland restoration efforts.

Stream Channelization

Impact description & restoration required. In Rhode Island, many small streams have been channelized, often due to historic farming practices. Channelized streams are usually straighter, deeper, and wider than natural streams. As a result, the water table in the surrounding land is lowered and local flooding problems may be reduced. However, channelization can cause local wetland loss and greater flooding problems downstream. Channelization may be partly remedied through reconstruction of the stream channel, but full restoration often is not feasible due to development in adjacent areas and the increased threat of flooding.

Effects of impact removal on wetland functions. Where re-creation of a natural channel is possible, all seven of the functions of interest may be enhanced. Floodwaters are slowed by the meandering of a natural channel and by floodplain vegetation during overbank flow; these processes reduce flood levels and delay the flood crest downstream. The same

processes cause greater sediment deposition and increased interactions among water, substrates, and vegetation, resulting in improved water quality downstream. Water that has topped the banks of a re-created stream channel may percolate down through the substrate of the bordering floodplain and recharge the local groundwater system. Stream channelization usually lowers the local groundwater table; re-creation of a natural channel could raise the local groundwater table to a point where more groundwater discharge occurs seasonally. A return to a natural stream bottom, decreased water velocities, and resulting increases in the abundance of in-stream vegetation would greatly improve fish habitat. Wildlife would also benefit from re-creation of natural stream channels due to increased microhabitat diversity and (for piscivorous species) increased prey abundance. Natural stream channels are more visually complex and aesthetically appealing than channelized streams; restoration would increase all of the heritage functions.

Recommendations. Although re-creation of natural channels has the potential to positively influence each of the seven functions, this impact type probably should not be emphasized during the restoration planning process. As already noted, the restoration process may increase the flooding threat to adjacent areas that were developed after channelization took place.

Drainage

Impact description and restoration required. In the past, many wetlands of the Woonasquatucket watershed study area were drained—via ditching—and then converted to agricultural fields. Wetlands in other areas of Rhode Island also have been ditched for agriculture or mosquito control. Ditching may result in partial drainage (i.e., conversion to a drier wetland water regime) or complete drainage (i.e., conversion to upland). Many completely drained areas have been built upon, while others have naturally revegetated. In some areas, ditches have filled in naturally, promoting a return to wetland conditions. The potential for restoration exists where ditches continue to drain undeveloped areas. Restoration activities would include plugging or filling of the ditches. Former wetlands

drained by open ditches may be among the easiest to restore; they already have hydric soils that contain a seedbank of hydrophytic species (Tiner 1995).

Effects of impact removal on wetland functions. In cases where wetland has been completely drained, restoration activities would cause an increase in wetland abundance, as with fill removal. Regardless of whether drainage has resulted in wetland loss or degradation, plugging or filling of ditches has the potential to restore six of the functions of interest (Table C1). After ditches have been plugged or filled, water may more readily disperse across the wetland surface; this will increase floodwater retention time. It will also promote increased interactions among water, substrates, and vegetation, resulting in nutrient uptake, adsorption, and transformation and improved water quality. Because drainage increases soil oxygen levels, former wetland organic soil horizons may decompose more rapidly and the surface may subside. After restoration, the groundwater table would be closer to the surface than prior to the ditching, and discharge would be more likely. At the same time, significant groundwater recharge would be less likely because the restored wetland would have a higher water table for longer periods of time. Ditch removal activities that increase wetland extent provide more habitat for wetland-dependent wildlife. If the plugging of ditches results in increased marsh or open water, fish would benefit. Restoration of ditched wetlands may create additional recreational opportunities, enhance regional biodiversity, and otherwise improve the heritage function.

Recommendations. Plugging ditches is a relatively inexpensive endeavor, when compared to fill removal. Although there are few of these opportunities in Rhode Island, unlike in states where agriculture has been a principal cause of wetland alteration, these opportunities should be ranked high during the restoration planning process.

Trash Dumping

Impact description and restoration required. Wetlands often serve as dumping grounds for broken appliances, junked cars, and other trash. When significant amounts of trash have been deposited, the impacted area may no longer perform certain functions and the

wetland area has essentially been lost. Therefore, the issues associated with this type of impact—and its remedy, trash removal—are similar to those associated with filling. Certain types of dumped material (e.g., junked cars, oil drums) may leak oil and toxic chemicals.

Effects of impact removal on wetland functions. Trash could be considered a type of fill; see the section on fill removal for information about how trash removal might influence specific wetland functions, with the following exception. Trash removal probably has no effect on groundwater discharge; groundwater can continue to discharge in and around mounds of trash.

Recommendations. Wetlands that have been subjected to significant amounts of dumping should be targeted for restoration and ranked relatively high. Removal of minor amounts of trash and litter should be the focus of local volunteer efforts.

Removal of Wetland Vegetation

Impact description and restoration required. Freshwater wetland vegetation may be removed for a number of reasons, for example, to obtain fuelwood or to improve a viewshed. If limited to a single occurrence, removal of vegetation is a temporary form of wetland degradation. As long as there has not been an accompanying alteration of hydrology, most wetlands will revegetate naturally. Planting could enhance and expedite this process. However, certain plants such as Atlantic white cedar (*Chamaecyparis thyoides*) might be difficult to reestablish.

Effects of impact removal on wetland functions. Reestablishment of wetland vegetation has the potential to positively influence five of the seven functions of interest (Table C1). The presence of emergent vegetation (especially dense, persistent vegetation) plays a key role in reducing the velocity of floodflow in wetlands; this increases the probability that wetlands may reduce the severity of downstream flooding. Reduction of water velocity also causes sediments to drop out of suspension. Combined with the nutrient uptake capabilities of emergent plants, the reestablishment of wetland vegetation can therefore

positively influence water quality. The groundwater discharge function of wetlands might be reduced by reestablishing vegetation because plants would take up and evapotranspire water that might otherwise be discharged. Reestablished vegetation could have a negative effect on recharge as well, by reducing the amount of water that might reach the groundwater table. Vegetation is a key component of wildlife habitat; reestablishment of vegetation that has been removed would do much to reinstate lost habitat values. Reestablishment of submergents, floating-leaved plants, or emergents could provide microhabitat for fish. Other wetland vegetation types could provide shade and water quality improvement functions that would also benefit fish. There is little doubt that a naturally vegetated wetland is more aesthetically pleasing than a denuded wetland. Revegetation would enhance heritage functions.

Recommendations. Although reestablishment of wetland vegetation may positively influence most of the functions, this impact type should probably not receive priority during the restoration planning process. If the disturbance that caused the vegetation removal is not ongoing, the wetland will most likely revegetate naturally from its seedbank and by seed dispersal from other wetlands. However, planting may grant more desirable species a competitive edge in areas where there is a threat of invasion by the common reed (*Phragmites australis)* or purple loosestrife (*Lythrum salicaria*).

Removal of Adjacent Upland Vegetation

Impact description and restoration required. Removal of adjacent upland vegetation was, by far, the most common type of impact observed in this study (see Task E chapter). In cases where parking lots or yards now occupy the denuded areas, restoration is unlikely. The best restoration opportunities exist where cleared areas remain free of development and are unused. The primary restoration activity for this type of impact would involve reestablishment of adjacent upland vegetation.

Effects of impact removal on wetland functions. Reestablishment of adjacent upland vegetation would have minimal influence on flood abatement in the wetland; however, dense vegetation in the adjacent upland may slow surface runoff into a wetland and

promote infiltration in upland soils, thereby prolonging the period before floodwaters exceed the storage capacity of the wetland. The presence of dense, persistent, upland vegetation would positively influence the long-term ability of a wetland to improve water quality. Upland vegetation surrounding wetlands can remove sediments, nutrients, and other pollutants from surface water, decreasing the likelihood that a wetland will become saturated with these pollutants. Planting vegetation in uplands surrounding wetlands has no direct impact on wetland groundwater functions; however, theoretically, high transpiration rates in the adjacent upland might lower wetland groundwater levels and reduce the probability or duration of groundwater discharge. Similarly, slowing surface runoff into wetlands and increased infiltration in adjacent uplands might reduce total water available for groundwater recharge in the wetland. Vegetated uplands adjacent to wetlands screen out noise and filter water-borne pollution that would otherwise adversely influence wetland-dependent wildlife and fish. Vegetation in the adjacent upland may also satisfy key habitat requirements (e.g., nest sites, roosting sites, foraging areas) for wetland wildlife. Reestablishment of upland vegetation adjacent to wetlands can increase the effective size of natural areas and positively influence the heritage function, especially aesthetics.

Recommendations. Although reestablishment of adjacent upland vegetation might have major positive effects on only four of the seven functions of interest (Table C1), this restoration activity should receive high priority during restoration planning. Planting upland vegetation is non-invasive (i.e., there is minimal disturbance of existing wetland soils or wetland vegetation) and it is inexpensive relative to restoration efforts that require heavy machinery.

Impedance of Surface Flow

Impact description and restoration required. Culverts that have been blocked by sediments or other material and those that have been installed at inappropriately high elevations often cause surface water to impound. These impoundments can create wetter water regimes, and they may result in a change in wetland type. The restoration activities required to amend these impacts include removing obstructions to flow or installing new

culverts at more appropriate elevations; these activities may both be categorized as enhancing surface water flow.

Effects of impact removal on wetland functions. Enhancing surface water flow positively influences at least three of the seven functions (Table C1). Flood abatement would be negatively impacted because restoration would remove constrictions to wetland outlets; constricted outlets enhance the floodwater storage capability of wetlands. Water would flow more quickly through the wetland after restoration and sediments would be less likely to drop out of suspension; in this regard, water quality would be negatively influenced. However, if flow enhancement replaced open water with dense, persistent emergent vegetation, nutrient uptake would be improved and sediments might be trapped by the dense vegetation; in this case, water quality would be positively influenced. Groundwater recharge might be enhanced if improved surface water flow resulted in a return to a temporarily or seasonally flooded water regime instead of longer periods of inundation. If water were allowed to move more quickly through a wetland, the local water table might drop farther below the surface and the probability of groundwater discharge would decline. Fish and wildlife that had inhabited the site prior to impoundment would benefit from a return to original habitat types, while species favoring the wetter conditions would decline. The heritage function would not necessarily be directly influenced, since restoration would simply result in a change in water regime or conversion from one wetland type to another. However, impoundment might threaten survival of highly regarded plant species or communities. On the other hand, water-based recreational opportunities might decline after flow enhancement.

Recommendations. Enhancement of surface flows may be accomplished relatively inexpensively if sediment removal is the only issue. These situations should receive moderate priority in the restoration planning process. In cases where the elevation (invert) of the culvert is too high, restoration would be more costly. Under both scenarios, restoration could result in temporary flooding problems downstream.

Invasive Species

Impact description and restoration required. Invasion of wetlands by *Phragmites* or purple loosestrife is a growing problem in Rhode Island. Both species have the potential to produce monotypic stands covering large expanses of wetland. In such extreme situations, control can only be achieved through intensive, continuous management (i.e., cutting and application of herbicides). Where *Phragmites* has only begun to colonize a wetland, hand removal and treatment with herbicides may check its advance; however, continual maintenance will probably be required. Results from recent experiments suggest that biological control of purple loosestrife may be achieved by introducing an exotic beetle to impacted wetlands.

Effects of impact removal on wetland functions. Regardless of the technique employed, removal of invasive species has the potential to positively influence only two of the seven functions. Wildlife habitat would be improved because many wetland-dependent species cannot make use of monotypic stands of *Phragmites* or purple loosestrife. These monotypic stands also threaten regional biodiversity. For this reason, removal of purple loosestrife or *Phragmites* would improve the heritage function of a wetland.

Recommendations. This restoration activity should receive low priority during the restoration planning process because removal of invasive plant species requires continual, intensive maintenance and the techniques used are not without risk (e.g., application of herbicides, introduction of exotic beetles). In addition, very few wetland functions are enhanced via this removal. Efforts should be restricted to particularly valuable wetlands where there is some hope of stemming the tide of invasives or where maintenance of high-quality wildlife habitat is a major objective.

Section 2: Restorability of Major Wetland Types

In Section 1, we showed that the nature of the impact to a wetland can determine which functions have the potential to be restored. The restorability and projected functions of a given site also depend on the wetland type that is targeted for restoration. Kusler and Kentula (1990) noted that the majority of "restoration" activities actually

involve wetland enhancement (e.g., impoundment of existing wetland to enhance waterfowl habitat) and wetland creation (e.g., using dredged material to create marshes along rivers or in bays). Although literature related to such projects has been consulted, the following discussion hinges on our narrower definition of restoration: re-creation or rehabilitation of wetland ecosystems whose natural functions have been destroyed or impaired. In this instance, the goal is to re-create the type of wetland that existed before alteration, and its functions, insofar as possible.

Different wetland types are often quite distinct in structure and certain functions. Some wetland types are easier to rehabilitate or re-create than others. The National Research Council (1992) stated that "controversy exists as to whether or not certain wetland systems can be restored." Much of this controversy exists because, although there have been many attempts to restore wetlands, very few of those attempts have been critically evaluated (Kusler and Kentula 1990). The following paragraphs present information about the relative restorability of major wetland types, based on restoration projects that have been evaluated. A summary of this information is provided in Table C3.

Table C3. Restorability of Rhode Island freshwater wetland types, based on the scientific literature.

Wetland type	***Rating***[1]
Ponds	High
Marshes	High
Wet meadows	Moderate
Streams	Moderate
Vernal pools	Moderate
Shrub swamps	Moderate
Forested swamps	Low/Moderate
Fens	Low
Bogs	Low

[1]See Task C, Section 2 text for rationale.

Ponds

Ponds are among the easiest of wetland types to create. Tiner (1995) noted that "ponds have been successfully created by many cultures throughout the course of human

history." In fact, open water bodies are restored and created disproportionately to the frequency with which they occur in nature (Kentula 1993). But this bias may not be due entirely to the ease of restoring ponds; pond creation often is unintentional. Galatowitsch and van der Valk (1996) found that, in recently restored prairie pothole wetlands, the water regime was often wetter than planned, resulting in more open water habitat than what occurred historically. This overabundance of open water was at the expense of other, less common wetland types such as sedge meadow. Organic substrates drained for agriculture subside due to enhanced decomposition rates; subsidence results in lower soil elevations and wetter water regimes after hydrologic restoration. Additional anthropogenic causes of pond formation include gravel mining, rock quarrying, crop irrigation or livestock watering, floodwater detention, and road construction (Hollands 1990). Filled sites that are restored as a result of enforcement orders are sometimes excavated to below the original wetland grade in an attempt to ensure wetland hydrology (S. Tyrell, RIDEM, pers. comm.). Ponds may also be disproportionately represented because of their aesthetic and recreational properties. Numerous ponds have been created in Rhode Island through the efforts of the Natural Resource Conservation Service—and other agencies and groups—to restore or enhance wildlife habitat or the heritage function. Although the total number and acreage of Rhode Island ponds has undoubtedly increased as a result of human activities, additional pond restorations are justifiable in an attempt to recreate natural habitats that have been destroyed or degraded.

The success of pond restoration will depend on the nature of the impact and, therefore, the restoration technique employed. Removal of fill material, sediment, or trash simply requires excavation to a depth sufficient to maintain the desired amount of water. Excavation to a certain water depth may also effectively control invasive plant species such as *Lythrum salicaria* (Weiher et al. 1996) and *Phragmites australis*. Water quality impacts are more difficult to rectify. Open water bodies act as sinks for nonpoint-source runoff of silt, nutrients, and pesticides (National Research Council 1992). Although revegetation of adjacent uplands may help to filter some of these pollutants (Fennessy and Cronk 1997), significant water quality improvement would require reduction of pollutants at the source—a task well beyond the scope of onsite wetland restoration projects. Aquatic bed plant communities of ponds (e.g., *Potamogeton* spp., *Nymphaea*

odorata, *Nuphar luteum*) may be difficult to restore if water quality is poor. Successful aquatic bed restoration requires water that is clean, clear, permanent, and shallow (Tiner 1995). Turbidity can limit reestablishment of submersed species (Mitsch and Gosselink 1993). Galatowitsch and van der Valk (1994) found that submergent plant species readily recolonized prairie potholes of the Midwest that were drained for agriculture and subsequently restored. Pond drainage appears to be rare in Rhode Island; no drained ponds were detected in the Woonasquatucket study area.

Vernal Pools

Restoration of vernal pools has been largely overlooked. Some have been created inadvertently as a result of excavation for other purposes (e.g., borrow pits, fire protection). Restoration or creation techniques and issues for vernal pools are similar to those for permanent ponds (see above). However, the hydrology of vernal pools is much more dynamic than that of permanent ponds, and it would likely be much more difficult to replicate natural water regimes. Research over the last 10 years has demonstrated that pond hydroperiod is a key determinant of amphibian community composition, species richness, and reproductive success (Semlitsch 2000). A solid understanding of vernal pool hydrology—and the relationship between hydrology, surficial geology, and other site factors—is required for restoration efforts to succeed. Because of their small size, destruction of entire pools is likely; therefore, most restoration efforts would involve vernal pool creation. Schiller et al. (2000) reported success in restoring populations of an endangered plant species (*Pogogyne abramsii*) by creating vernal pools in California. Many vernal pool-breeding amphibians require extensive tracts of upland forest contiguous with their breeding pools. To successfully recreate habitat for these species, vernal pool restorations should occur in large forested areas not threatened by development.

Marshes

Reintroduction of proper hydrology is critical to restoration of a specific wetland plant community (Lowry 1990). Fortunately, most marsh plants can tolerate relatively large fluctuations in water level (Kusler and Kentula 1990) and, for that reason, marshes

are “among the easiest wetlands to restore” (Tiner 1995). The simple structure and rapid maturation of marsh vegetation communities and the presence of native seed stocks also contribute to quick and relatively successful restorations (Kusler and Kentula 1990). However, in a paper discussing the trajectories and time requirements of wetland restorations, Zedler and Calloway (1999) warned that even simple communities like cattail marshes require 5 to 10 years for restoration of most functions. More diverse, complex communities have greater time requirements. Galatowitsch and van der Valk (1996) found that tile-drained prairie pothole marshes could be restored; emergent plant species quickly recolonized restoration sites (Galatowitsch and van der Valk 1994). Marshes are often disproportionately represented in restoration and creation efforts for the same reasons that ponds are over-represented (see above). In addition, other wetland types (e.g., forested wetland) are often converted to marsh for purposes of wildlife habitat “enhancement” (Golet 1986); this is at the expense of wildlife dependent on the former wetland type.

Marshes provide habitat for a diversity of wetland-dependent and wetland-associated wildlife species; they are often targeted for restoration to benefit migratory waterfowl. After 3 years of monitoring avian communities of New York marshes restored from drained agricultural sites, Brown and Smith (1998) concluded that, although restored wetlands provided adequate habitat for wetland birds, they did not function quite as well as natural reference marshes. They predicted, however, that further succession of the restored vegetative community might cause bird communities in restored and natural marshes to become more similar.

Former marshes that have been drained and cultivated are relatively easy to restore, particularly if they are small and drained by open ditches (Tiner 1995). Because natural seedbanks may be viable for centuries, planting is not required; restoration of such sites simply involves plugging the ditches. Tile-drained marshes may be more difficult and costly to restore. Fortunately, marshes that have been drained for agriculture in Rhode Island generally have been drained using open ditches, but such restoration opportunities are scarce here. Wetland basins with large watersheds are also more difficult to restore because they may require water- and erosion-control measures to prevent washouts (Tiner 1995). Tiner also recommended tilling sites before restoring

hydrology to hasten recolonization by hydrophytes in the existing seed bank. However, Brown (1999) found that restored sites with disturbed substrates ended up as monotypic stands of cattail, providing less value for wetland birds.

Wet Meadows

In comparison to marshes, there is little published information on restoration of wet meadows. Tiner (1995) grouped wet meadows with marshes in regard to restoration difficulty, perhaps because, like marshes, they represent an early stage of hydrarch succession. Although the lack of vegetative structural complexity suggests that restoration should be relatively simple, the hydrologic regime required to maintain wet meadows—and to prevent rapid invasion by woody plants—is more difficult to re-create. Wet meadow vegetation requires prolonged soil saturation, often with temporary or seasonal flooding. Galatowitsch and van der Valk (1994) stated that sedge meadow zones were rarely established successfully following restoration of hydrology to drained prairie potholes in the Midwest. This probably can be attributed to two factors: restored water regimes were wetter than original water regimes (Galatowitsch and van der Valk 1996) and *Carex* species typical of wet meadow communities have poor seed set and low long-term seed viability (Budelsky and Galatowitsch 1999). The absence of certain breeding bird species in restored prairie potholes, when compared to natural potholes, has been attributed to the lack of a wet meadow zone in restored sites (Delphey and Dinsmore 1993, VanRees-Siewart and Dinsmore 1996).

Larson (1999) stated that attempts to create mature sedge-meadow communities in a short period of time will result in failure; he said that, instead, we should attempt to mimic the natural successional pathways of wetland communities. Larson has achieved this in restored Wisconsin sedge meadows by planting woolgrass (*Scirpus cyperinus*) at restoration sites. Woolgrass forms tussocks upon which other sedges can establish, and also inhibits invasion by exotics. Later-successional plants can be propagated in “waves,” or succession can be allowed to occur naturally.

Tiner (1995) recommended placing fences around restored wet meadows to eliminate grazing problems. In many areas of the Northeast, however, wet meadows typically succeed to shrub swamp and, eventually, forested wetland in the absence of

continued disturbance (Golet et al. 1993). Disturbance regimes such as grazing, cutting, or mowing may need to be applied to ensure continued dominance of wet meadow vegetation after restoration. However, this need for long-term maintenance could make the restoration of wet meadow habitat controversial. Many restoration scientists (e.g., Zedler 1988, Kentula 1993) assert that the probability of long-term success is significantly reduced if restored wetlands are not designed to be self-maintaining. If a "hands off" approach is taken, restored wet meadow can be expected to succeed to forested wetland. As Zedler (1988) pointed out, "it is hard to plan for a system that is naturally dynamic."

Forested Swamps

Although there have been numerous attempts to restore bottomland forests in the South (Tiner 1995), forested wetland restoration has rarely been attempted in the glaciated Northeast (Lowry 1990). In their assessment of the status of restoration science, Kusler and Kentula (1990) concluded that forested wetlands are much more difficult to restore than earlier-successional wetlands such as marshes. At that time, Clewell and Lea (1990) stated that it was too early to evaluate the success of forested wetland restorations conducted in the southeastern United States because forests are complex ecosystems that require long periods of time to fully develop. Clewell (1999) later reported success in creating forested wetland within 11 years on phosphate-mined land in Florida. After restoration this site contained over 200 species of trees, shrubs, vines, ferns, grasses, and forbs; the canopy had reached 85% coverage and some trees had attained a height of 12.5 meters. Tiner (1995) suggested that it may take 50 years before it is possible to assess success because trees require decades to reach maturity. Although some functions of forested wetlands (e.g., flood abatement, groundwater functions) may be effective despite the lack of a mature forest canopy, restoration sites presumably would not be suitable for forested wetland-dependent wildlife for several decades.

The lengthy time requirement for ecosystem maturation and for evaluation of success is not the only factor that makes restoration of forested wetlands difficult. The restoration of appropriate hydrologic conditions may be the most critical factor in forested wetland restoration (Clewell and Lea 1990, Tiner 1995). McLeod et al. (2000)

reported that slight differences in elevation, and therefore hydrology, can substantially influence the survival and health of trees planted in swamps. This sensitivity to hydrologic regimes is long-term (Kusler and Kentula 1990); even mature forest vegetation can be damaged by wide-ranging hydrologic conditions. In an attempt to create forested wetland in New Hampshire, Barry et al. (1996) contended with this hydrologic sensitivity of woody species by mimicking the mound and pool microtopography found in natural wetlands. The rationale was that mounds provide a wide variety of water regimes (see Golet et al. 1993) and therefore may increase the probability that planted trees can survive prolonged periods of excessive inundation; i.e., there is more room for error. However, this technique may only be appropriate for the creation of swamps on non-organic substrates. Barry et al. (1996) cited an attempt by Crispin and Randall (1990) to restore microrelief in former forested wetland of southeastern Massachusetts; the attempt was unsuccessful because heavy equipment became mired.

Shrub Swamps

Few restoration or creation attempts have targeted shrub swamp communities. In terms of restorability, shrub swamps are intermediate between forested wetlands and marshes. Shrubs reach maturity more quickly than trees. Tiner (1995) stated that many types of shrub wetlands may be as easy to establish as marshes because of similar hydrology. This may be true for swamps containing buttonbush (*Cephalanthus occidentalis*), which can tolerate prolonged flooding. However, many shrub species are less tolerant of inundation; in these cases, many of the difficulties cited for forested wetland restoration (see above) would also characterize shrub swamp restoration. Hollands (1990) reported that most of the shrub vegetation in a created shrub swamp in Massachusetts was killed off during one summer with unusually high water levels. Marsh vegetation at the same site survived the excessive inundation.

Fens

The lack of scientific literature available regarding fen restoration suggests that it has rarely been attempted. In 1990, Lowry questioned the feasibility of fen restoration

and stated that it had not been attempted in the glaciated northeastern United States. Perhaps the unique hydrology and water chemistry of fens have precluded such attempts. Crosson et al. (1999) removed invasive alien shrub species from a Wisconsin fen and reported some success in planting native fen species. Continued success in this fen, however, may be dependent on prescribed burns. Some research has been conducted in the Netherlands to compare restoration methods for fens impacted by acidification and eutrophication (Beltman et al. 1996). In the fens that were studied, a surplus of acid rainwater had formed a lens over calcareous groundwater, promoting dominance by mosses (especially *Sphagnum* spp.). The authors reported that a combination of drainage and sod removal was successful in removing the symptoms of this impact. There are no studies that have addressed restoration of fens impacted by filling or draining, the impacts that are most likely to have occurred in fens of Rhode Island.

Zedler and Callaway (1999) stated that wetland systems with unique water-quality requirements (such as fens) are likely to take much longer to restore than other wetland types. Full functionality may not be achievable. Zedler (1988) presented a strong argument for preservation of such ecosystems by stating that restoration may not be a realistic option.

Bogs

Tiner (1995) considered bogs with ericaceous shrubs to be the most difficult wetland type to establish, due to their unique chemistry and deep organic soils. If restoration is even possible—and that has been questioned (Lowry 1990)—it would require an extremely long period of time (Tiner 1995). Kusler and Kentula (1990) asserted that isolated freshwater wetlands supplied by groundwater are the most difficult types to restore; Rhode Island bogs clearly fit this description. No bog restorations have been attempted in the glaciated northeastern United States (Lowry 1990). Bog restoration attempts in Europe have met with some success. Grosvernier et al. (1997) reported that Swiss bogs that have been drained "undergo a strong chemical disturbance, which, in turn, [negatively] affects the growth of *Sphagnum* mosses." However, Buttler et al. (1996) claimed that cutover bogs of Switzerland can be rapidly and fully restored via natural development of secondary peat-forming vegetation. These Swiss bogs, however,

are not kettle-hole bogs; they have formed via paludification and the same process restores them after peat harvest. Paludification does not occur under Rhode Island's climate; therefore, the results of these Swiss studies are not directly applicable to Rhode Island bogs. In addition, peat harvest has rarely occurred in Rhode Island bogs. Bogs of this region are more likely to be impacted by filling or by changes in nutrient and water inputs resulting from development of surrounding uplands.

Highly acidic, nutrient-poor wetlands, such as bogs, are relatively scarce when compared to other wetland types, and they often support rare and endangered plant species. Because they are uncommon, and also because they are difficult or, perhaps, impossible to restore, emphasis should be placed on preservation of these wetlands instead of restoration (Moore et al. 1989, Weiher et al. 1996).

Streams

The feasibility of stream restoration is a function of both the current land use surrounding the stream and the nature of the impact to the stream. Mitsch and Gosselink (1993) stated that "the restoration of entire rivers has been shown to be an elusive goal in many parts of the world." They attribute this failure to channelization, subsequent development in floodplains, and increases in sediment loads and other nonpoint pollutants. Channelization reduces flooding problems locally, and therefore encourages development of adjacent floodplains. Restoration of meanders and natural stream morphology within those floodplains is not feasible after the areas have been urbanized. Many smaller streams occurring in urbanized areas of Rhode Island have been filled and built upon; therefore, these streams have little chance of being restored. Where buried streams have not been built upon, and thus could be restored, vast improvements in fish and wildlife habitat, flood abatement, and water quality improvement functions might result. Whether channelized, buried, or unaltered, many streams have been impacted through removal of adjacent upland vegetation (i.e., riparian vegetation). The National Research Council (1992) stressed that vegetated riparian habitats are essential to the natural ecological functioning of streams and rivers. See the following section ("Vegetation of adjacent upland") for more details about upland riparian vegetation.

Restoration projects that have manipulated in-stream characteristics have met with mixed results. Fennessy & Cronk (1997) reported that restoration of natural channel morphology is an important means to safeguard water quality in agricultural landscapes. In Finland, Laasonen et al. (1998) compared channelized streams with "near-pristine" streams and streams that had been restored from 0 to 16 years previously using boulder dams, flow deflectors, excavations, and channel enlargements. They found that biotic and abiotic characteristics of restored streams were intermediate between characteristics of near-pristine and channelized streams. Although conditions had been improved through restoration, the authors concluded that there was no indication that communities would reach pristine conditions with a longer recovery period (i.e., there was no trajectory, after restoration, toward natural conditions). Measurements of macroinvertebrate species richness and abundance were used to reach these conclusions; success in stream restoration is often measured in terms of macroinvertebrate communities. Gortz (1998) measured trout and macroinvertebrate populations in a stream restored to improve trout habitat. Gravel, boulders, and stream concentrators were used in the restoration. Invertebrate populations were different from pre-restoration conditions, and five times as many spawning trout occurred in restored areas relative to non-restored areas. However, trout in the restored areas experienced low spawning success. The authors concluded that trout production was not enhanced. Perhaps this restoration actually created a sink habitat for trout, attracting the fish away from more productive areas. Powell (1997) stressed that stream restoration projects that have targeted fish habitat as a goal will fail if water quality is poor.

Vegetation of Adjacent Upland

Vegetated uplands bordering wetlands enhance the ability of a wetland to provide many functions. In the following discussion, vegetated upland riparian zones are included in this category. Hollands (1990) asserted that establishment of a vegetated upland buffer zone should be a part of all restoration projects. From a wildlife habitat perspective, adjacent upland vegetation can provide a barrier or screen for wetland wildlife from human activity and provide habitat for wetland-dependent species (e.g., salamanders) that use both wetlands and adjacent uplands during different stages of their life cycle

(Hollands 1990). Vegetation surrounding wetlands can also control nonpoint-source pollution (Fenessy and Cronk 1997, Lowrance 1998), increase the aesthetic appeal of wetlands (Holland 1990), maintain wetland water temperatures, filter sediments from surface runoff, and slow down surface runoff, thereby enhancing the wetland's flood abatement capacity.

Restoration of adjacent upland vegetation should be a simple process relative to wetland restoration. Difficulties in restoring hydrology are the most common cause of wetland restoration failure, but hydrology is not nearly as great an issue in the restoration of upland vegetation. Forested buffer zones are the most structurally complex and would probably be the most effective at protecting and enhancing wetland functions. However, as with forested wetlands, it may take decades for the forest ecosystem to reach maturity. Hawkins et al. (1997) compared the distribution of riparian vegetation at points along a California river before and after a large flood. They found that the amount of riparian vegetation destroyed by the flood ranged from 0 to 40%; this loss was negatively correlated with the total area of riparian vegetation and positively correlated with the amount of development nearby. The authors concluded that, in order to increase the probability of success, riparian revegetation should be pursued in areas that already have extensive riparian vegetation and that are distant from urban development. Unfortunately, re-establishment of riparian vegetation is often needed most in heavily developed areas.

Conclusions

In any restoration attempt, we should be realistic about what it is possible to accomplish (Ehrenfeld 2000). Even if certain wetland types are deemed "easy" to restore relative to other types, it should never be assumed that *any* type can be restored to completely natural conditions. Duplication of naturally occurring wetlands over short time frames is impossible; at best, we can attempt to approximate systems and restore individual wetland functions (Kusler and Kentula 1990). Berger (1990) stated that "…no restoration can ever be perfect; it is impossible to replicate the biogeochemical and climatological sequence of events over geological time that led to the creation and placement of even one particle of soil, much less to exactly reproduce an entire ecosystem. Therefore, all restorations are exercises in approximation and in the

reconstruction of naturalistic rather than natural assemblages of plants and animals with their physical environments." Due to our inability to duplicate natural wetlands, and to the uncertainties involved, many authors have concluded that preservation of existing functional wetlands is a much better option than attempting to restore degraded or destroyed systems (Tiner 1995, Ehrenfeld 2000). This is clearly the case for rare or uncommon wetlands that are difficult, or perhaps impossible, to restore (e.g., fens, bogs, cedar swamps). Many of these arguments have been made within the context of wetland mitigation, where permitted wetland impacts are often erroneously assumed to be offset via wetland restoration. However, within the context of a proactive restoration program, attempts to restore wetland communities can strengthen the functional capacity of our watersheds and landscapes. In instances where rare wetland types have been destroyed or where restoration to the original type is not possible, proactive restoration attempts may require a shift in focus from restoration of a specific wetland type to re-creation of certain wetland functions. In such cases, there may be no choice but to substitute wetland types that can be created with a greater probability of success.

Section 3: Additional Factors That Influence Restorability

Galatowitsch and van der Valk (1994) stressed that restoration success depends on different factors for each individual wetland. Although the nature of the impact and the wetland type targeted both greatly influence the probability of restoration success, there are additional factors. The following paragraphs address those factors.

Urban vs. Rural Context

Much of the Rhode Island landscape has been urbanized. The highly developed northeastern and coastal portions of the State contrast sharply with the rural and extensively forested western area. Wetlands surrounded by urban development have properties that differ from those in less disturbed contexts (Erwin 1990). Urban wetlands often have altered hydrology, increased chemical and nutrient inputs, and increased sediment inputs, and are surrounded by destroyed or fragmented upland habitats. The fact that urban wetlands receive greater inputs of surface water runoff and pollutants—coupled with the fact that neighboring wetlands have also likely been degraded or

destroyed—increases the importance of those wetland functions that directly impact human health and welfare. The water quality improvement capabilities, flood abatement, aesthetic properties, and open space value of wetlands have great social significance in urban areas. These facts argue for an emphasis on restoration of urban wetlands in restoration planning. However, urban areas are "highly stressed environments" where restoration failures often are a result of the degraded status of the landscape in which they occur (Zedler and Callaway 1999). Restoration efforts in such settings may be subject to modified hydrology, exotic species invasions, and the effects of feral animals (e.g., bird predation by domestic cats). The National Research Council (1992) predicted quicker and more successful restorations in landscapes that are still intact. Kusler and Kentula (1990) claimed that excessive sedimentation can be a serious problem for many restored wetlands in urban areas. In addition, natural plant recolonization may be less likely in urban areas where nearby wetlands have also been destroyed.

Urban impacts may affect the various functions of wetlands differently. Restored urban wetlands may contribute more to watershed flood abatement and water quality improvement than rural wetlands because they have greater opportunity to perform these functions. However, increased inputs of runoff and pollutants will negatively influence the ability of a restored wetland to provide fish and wildlife habitat. Tiner (1995) noted that it may be impossible to restore healthy, viable fish and wildlife populations in urban areas with poor water quality. Helfield and Diamond (1997) asserted that wetland restorations often cannot serve the dual purpose of water quality improvement and fish and wildlife habitat; they referred specifically to highly urbanized sites that receive metallic, organic, or other contaminants. Wetlands tend to collect and concentrate contaminants; bioconcentration and biomagnification of these contaminants could occur throughout the aquatic community. If habitat is constructed to benefit fish or wildlife, but other aspects of the habitat are deleterious, these areas may form habitat "sinks" in the landscape; i.e., places where wildlife reside but do not reproduce successfully. Some research has investigated whether restored wetlands function as well as reference wetlands in providing habitat (e.g., Delphey and Dinsmore 1993, Brown and Smith 1998). However, Brown (1999) pointed out that even reference wetlands are rarely free of impacts, and "while we can restore wetlands similar to those that now exist in this

landscape, we may be recreating systems that are significantly less valuable as wildlife habitat than the pristine wetlands that once existed in this area." Wildlife that depends on both wetland and upland for habitat (e.g., certain frogs and salamanders) will not benefit from wetland habitat restoration in urban areas where upland habitat is either extremely fragmented or nonexistent.

Wetland restoration efforts may meet with public resistance in highly urbanized landscapes due to perceived threats to humans. Wetlands are often viewed as the source of mosquitoes and mosquito-borne diseases, such West Nile Virus and Eastern Equine Encephalitis. Hydrologic changes associated with certain wetland restoration activities (e.g., dam removal, re-creation of a natural stream channel) could damage property or otherwise inconvenience nearby residents.

Kentula (1993) stated that "…activities surrounding a wetland can disrupt the functions that cause the wetland to exist." In selecting appropriate restoration sites, she recommended considering not only the current condition of site contexts, but also the projected condition of the surrounding upland 20 years after restoration. Our ability to re-create *sustainable* wetlands may be influenced greatly by landscape context.

Hydrogeomorphic Setting

Many authors have suggested that restoration success is largely based on the ability to restore appropriate hydrology (Golet 1986, Lowry 1990, Mitsch and Gosselink 1993). Kusler and Kentula (1990) suggested that the hydrology of restoration sites located near lakes and rivers is easier to predict than that of isolated wetlands. In other words, the probability of restoration success may be a function of hydrogeomorphic setting. Kusler and Kentula ranked sites according to their restorability, based on hydrogeomorphic setting. Estuarine and coastal marshes were considered to be among the easiest to restore. Freshwater wetlands along lakes and streams were considered only slightly more difficult because it is relatively easy to obtain surface water elevations from adjacent water bodies. In addition, such water bodies may have long-term data available from gaging records. Forested wetlands along lakes and streams were considered much more difficult to restore because woody vegetation has a much narrower range of hydrologic tolerance than most nonwoody vegetation. They placed isolated wetlands

supplied predominantly by surface water in the next category of difficulty, and claimed that it is very difficult to attain the correct hydrology unless water-control structures are used. Isolated freshwater wetlands supplied predominantly by groundwater were considered the most difficult to restore.

Surficial geology may also influence restoration success. Lowry (1990) stated that there is a lack of knowledge regarding the relationship between restoration success and hydrogeologic setting. He pondered whether the potential for success was greater in stratified sand and gravel, where the local water table could be intersected, or on low-permeability till deposits, where surface water would be the major input. Tiner (1995) listed important considerations for site selection; they included local topography, hydrology, soil properties, degree of exposure to wave action, slope, site elevations, and proximity to other wetlands.

Monitoring and Maintenance Requirements

Monitoring, although it rarely occurs, is necessary to determine the success of restoration (Kusler and Kentula 1990). Monitoring of restored wetlands also allows researchers to determine which techniques have been most useful or cost-effective; such information can improve the feasibility and success of future restoration attempts. But monitoring can also contribute to the success of ongoing restoration efforts. Tiner (1995) stated that most problems with wetland restorations arise within the first 2 years of completion; monitoring is therefore imperative for at least 2 years. Simpler restorations (e.g., marshes, wet meadows, and shrub swamps) should be monitored for up to 5 years following restoration; forested wetlands and bogs should be monitored for at least a decade. During these monitoring efforts, problems with vegetation, hydrology, or other aspects of the restoration can be detected and rectified. Odum (1988) argued for long-term monitoring; he claimed that restoration success is often determined within 2 years of completion, but that "dramatic, unanticipated changes may occur over the ensuing years." Clewell (1999) attributed the successful creation of forested wetland in Florida to—among other things—continual maintenance, mid-course corrections, and follow-up work. These activities are often referred to as "adaptive management." Monitoring enables early detection of problems and development of site-specific solutions such as

stabilizing eroding soils, replacing non-surviving trees, and removal of exotics and nuisance vegetation. Successful restoration of wetland communities requires effort and resources well beyond the construction and revegetation stages.

The spread of invasive plant species may be one of the strongest justifications for restoration monitoring and adaptive management. Certain plants (e.g., *Phragmites australis*, *Lythrum salicaria, Typha* spp.) have the potential to invade and eventually dominate freshwater wetlands (Odum 1988, Levine and Willard 1990, Weiher et al. 1996). These species are especially likely to invade degraded wetlands, particularly if bare substrates have been exposed. Unfortunately, most restoration efforts create ideal conditions for invasion by these species. Dominance by invasive plants often results in lower plant species diversity (Weiher et al. 1996) and reductions in wetland functions such as wildlife habitat, nutrient processing, and aesthetics (Odum 1988).

Once invasive species become established, the need for continual control is likely. Spurr and Niering (2000) reported the need to manually remove *Lythrum salicaria* from a recently created marsh; despite these efforts, the species continued to spread. Odum (1988) painted a bleak picture, saying "in many freshwater wetland sites it may be an expensive waste of time to plant species which are of high value to wildlife…. It may be wiser to simply accept the establishment of disturbance species as a cheaper although somewhat less attractive solution." Research has been conducted to establish techniques for the prevention or removal of invasive species. Weiher et al. (1996) found that it is possible to inhibit *Lythrum salicaria* establishment by flooding substrates to a depth of 5 or more centimeters. Gabor et al. (1996) found that treating *Lythrum salicaria* with herbicides can give native plants a competitive edge; herbicide treatment combined with biological control (i.e., introduction of exotic *Lythrum*-eating beetles) may be particularly effective. Although biological control has been proposed, its application is controversial, particularly for native species such as *Phragmites australis* (for more details, refer to the opposing viewpoints presented by Blossey and McCauley [2000] and Rooth and Windham [2000]).

Long-term commitments may be required, in particular, for restorations that rely on artificially maintained hydrology (i.e., water-control structures such as levees, irrigation pipes, pumps, dams, or weirs). Because the long-term maintenance of such

technological fixes is never fully assured, Zedler (1988) concluded that they are "destined to malfunction." She therefore contended that the probability of long-term restoration success is significantly reduced if wetlands are not designed to be self-maintaining. Kentula (1993) echoed this sentiment, stating that "a system dependent on significant input from man is likely to fail."

Time Requirement

Most natural wetlands originated thousands of years ago. During these millennia, they have evolved into highly complex ecosystems (Berger 1990). It may not be realistic to expect that restoration attempts can provide an expeditious return to such a complex state. Recent research supports this assertion. Kentula (1993) stated that functional replacement in wetlands has not been demonstrated, mainly because the vast majority of restoration sites are ecologically young (i.e., it is too early to determine success). VanRees-Siewart and Dinsmore (1996) studied wetlands that had been restored up to 4 years previously and concluded that more time was needed to regain overall bird diversity. Street (1998) investigated whether functional replacement had occurred at 3-year-old Maryland mitigation sites and determined that it was too soon to tell. In at least some cases, success may not improve with time. According to Zedler and Callaway (1999), ecosystems do not necessarily follow a trajectory back to natural conditions after restoration takes place. A study of macroinvertebrate populations in restored Finnish streams—ranging in age from 0 to 16 years—supported this claim. Laasonen et al. (1998) found that, although macroinvertebrate populations were improved in restored streams relative to channelized streams, there was no indication that macroinvertebrate communities would approximate those in natural streams given a longer recovery period. Despite this, many restorations have been deemed successes. For example, Clewell (1999) reported successful restoration of a diverse forested wetland community in only 11 years. However, Kusler and Kentula (1990) maintained that even after a restoration has been deemed successful, it may be subjected to unanticipated threats (e.g., drought, flood, pollution, erosion) that could destroy the restoration site. Success is not only an elusive goal, but also an elusive concept; its very definition is often at issue. Success is usually measured by assessing the establishment of vegetation. However, Lowry (1990)

questioned the validity of these measures, and asked "is there a direct correlation between vegetative composition and structure and the presence or degree of other wetland functions?"

Current Land Use

Many wetlands have been drained for agriculture. In some cases where the resulting cropland or pasture is "marginal," wetland restoration may be both popular and successful. Entire state and Federal programs have been built around such activities (e.g., see the program profile of Illinois' Conservation Reserve Enhancement Program in the Task A section of this report). Where drained wetlands have been converted to productive croplands, there is less incentive for restoration. Filled wetlands were usually built upon for construction purposes; consequently, such areas usually are not available for restoration. The greatest opportunities for wetland restoration occur where filled or drained areas are currently unused.

Degraded—but not destroyed—wetlands still may perform valuable functions. Kentula (1993) considered restoration of degraded wetlands to be risky. She suggested that, before attempting such restorations, we ask ourselves "can we afford to lose this system?" To illustrate this issue, Kentula stated that some highly degraded wetlands of California provide important habitat for endangered species. Any actions taken to restore this habitat could produce results harmful to the endangered species; their habitat could be destroyed inadvertently.

Size of the Restoration Site

It is intuitive that the size of a wetland influences the magnitude of that wetland's functions. Large wetlands have the capacity to store a greater volume of floodwater than smaller wetlands. It has also been shown that larger patches of habitat often support a greater diversity of wildlife, and may provide the only habitat for certain area-sensitive species. Martine (1999) investigated the influence of size, classification, age, average depth, shoreline sinuosity, vegetative diversity, and percentage of surrounding cover on total bird species richness, breeding-bird species richness, waterfowl presence, and waterfowl diversity in 13 restored wetlands. He concluded that the size of the restoration

site was the most important factor influencing each of the bird community characteristics. Restoration of large areas of wetland is likely to be more beneficial than restoration of small areas.