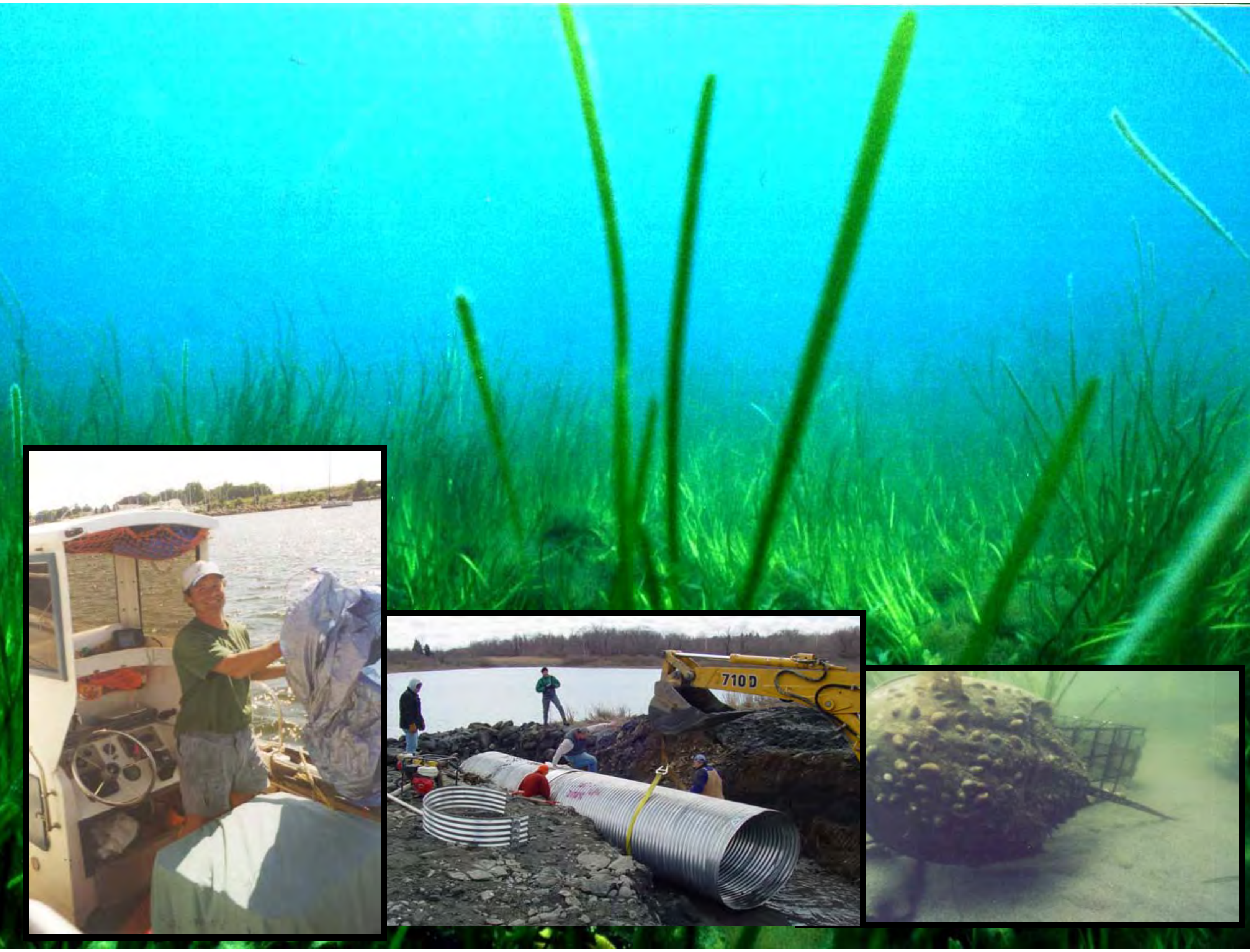
2006 Rhode Island NRCS Fish & Wildlife Action Plan

Prepared by
Andrew Lipsky
Biologist
USDA-NRCS



NRCS Natural Resources Conservation Service

RI State Fish and Widlife Action Plan Purpose and Format:

The purpose of the RI Fish and Wildlife Action Plan is to serve as the basis for allocation of Farm Bill funds to the NRCS State Office. The state plan will ensure that resources are targeted to the needs of the highest priority wildlife habitats. The objective of Rhode Island's WHIP and WRP programs is to enhance and restore native wildlife habitats that have been degraded, altered, or eliminated as a result of agriculture, urban & residential development, and changes in land use. This Fish and Wildife Action Plan will serve as RI NRCS's State WHIP Plan. Additionally, this plan will support the National WHIP program objectives and National Priorities for 2006.

WHIP National Program Objective

The objective of WHIP is to help participants protect, restore, develop, or enhance habitat for upland wildlife, wetland wildlife, threatened and endangered species, fisheries, and other types of wildlife.

WHIP National Priorities

In order to provide direction to the State and local levels for implementing WHIP to achieve its objective, NRCS has established the following national priorities:

(i) Promote the restoration of declining or important native wildlife habitats.
(ii) Protect, restore, develop or enhance wildlife habitat of at-risk species (candidate species, and State and Federally listed threatened and endangered species).
(iii) Reduce the impacts of invasive species on wildlife habitats.
(iv) Protect, restore, develop or enhance declining or important aquatic wildlife species' habitats

The focus of fish and wildlife projects supported by RI Farm Bill programs will be to restore habitat types that have been identified by existing local, state, and federal restoration planning initiatives and NRCS partnerships. For instance, the Rhode Island Habitat Restoration Team and RI-NRCS have focused coastal restoration activities on eelgrass, diadromous fish habitat (fish runs), and salt marsh habitats.

NRCS will support existing efforts by providing technical assistance and implementation resources to restore native habitats to a close approximation of their condition prior to disturbance. NRCS will focus habitat restoration actions using techniques that have proven to be effective in improving fish and wildlife habitat, as reported in the most current scientific literature.

The format of this strategic plan was based upon input from the State WHIP Technical Team meeting in September 2002. The Team recommended that the following categories be addressed for each of the priority habitats:

1. Overview of Priority Habitats (Baseline Conditions, functions& values)
2. Degradation and Impacts to the Priority habitat
3. Goals and Approach for Restoration (what is restorable based upon current technology, identify restoration techniques, and identification of priority projects)

Habitat break out meetings were held from September- November 2002 with scientists, practitioners, and other expertise to provide the necessary information and overview for this strategic plan. Additionally, habitat break out teams conducted a “Needs assessment” for each of the priority habitat types in order to identify critical tasks, gaps in the science, and recommended strategies to improve the restoration of these priority habitats. Needs assessments identified activities to be carried out by NRCS as well as partner organizations and agencies. **The Habitat Needs Assessments can be found in the Appendix A. of this RI State Plan.**

The WHIP/WRP strategic plan is broken down by priority habitat type as identified by NRCS and the WHIP/WRP Technical Team members.

Habitat Type Priorities for NRCS:

1. **Coastal Habitats**
   - **Eelgrass beds**
   - **Salt marshes**
   - **Anadromous/Catadromous Fish Runs**
2. **Freshwater Wetlands & Riparian Habitats (including urban riverways and floodplain buffers)**
3. **Upland Habitats of state significance (native grasslands and oak pitch pine barrens)**

NRCS WHIP/WRP Coastal Habitat Priorities:

Coastal Habitat Overview:

Rhode Island is home to an array of coastal habitats, including salt marshes, seagrass beds, and river systems. Coastal habitats support a wide variety of fish and wildlife, contribute greatly to the state's biological integrity and diversity, and help support the state's economy. These habitats help to support a significant amount of annual capital for the state of Rhode Island: 75 million dollars in commercial fishery landings; a recreational fishery valued at 150 million dollars; and a tourism and outdoor recreation industry valued at two billion dollars from Narragansett Bay alone. Despite their exceptional importance and value, Rhode Island's coastal habitats have suffered from several hundred years of human impacts – development and agricultural activities that have destroyed or degraded many habitats. Salt marshes have been diked, ditched, and filled. More than 500 dams have been built on RI Rivers and streams. Underwater eelgrass beds have succumbed to coastal development and declines in water quality. USDA Farm bill programs, working together with an established and effective state, local, and federal partnership, are now uniquely positioned to positively impact these valuable but declining habitats.

RI-NRCS WHIP/WRP programs have been closely aligned with the Rhode Island Habitat Restoration Team (RIHRT). RIHRT has been responsible for coordinating statewide restoration planning since its inception in 1998. RI-NRCS has been a founding member of this local, state, and federal partnership whose mission is to promote the restoration of damaged coastal habitats by coordinating restoration planning, projects, and information at the state level. RIHRT is responsible for developing a state coastal habitat restoration plan as mandated by the RI "Coastal and Estuary Habitat Restoration Program and Trust Fund which was established into law in spring 2002.

The RI WHIP technical team and NRCS staff have recommended that NRCS continue to target WHIP/WRP programs to the priority coastal habitats identified by the RIHRT and to support applicable projects identified and prioritized using the RI Habitat Restoration Plan and information system (http://www.csc.noaa.gov/lcr/rhodeisland/index.htm).

The Rhode Island Habitat Restoration Portal is a web based tool that provides data and information about habitat restoration in Rhode Island to the public, Federal and State agencies, and nonprofit groups. The Rhode Island Habitat Restoration Portal is the result of a partnership between RI Coastal Resources Management Council (CRMC), the RI Department of Environmental Management Narragansett Bay Estuary Program (NBEP) and Save The Bay, Inc., working with the Rhode Island Habitat Restoration Team (RIHRT) and the University of Rhode Island Environmental Data Center (URI EDC). The project is funded through the National Oceanic and Atmospheric Administration (NOAA) Coastal Services Center.

NRCS biologist Andrew Lipsky assisted in the development of the RI Restoration Information System over the past 4 yours in his previous capacity of co-leader of the RIHRT and co-author of much of the text in the portal. Because the NRCS coastal habitat restoration strategy will support local and state priorities as identified by RIHRT, the information presented in RI-NRCS Coastal Habitat Restoration Section has been adapted from the RI Coastal Habitat Restoration Information System by permission.

RI Habitat Restoration Team

Coastal Resources Management Council
Megan Higgins
4808 Tower Hill Road
Wakefield, RI 02879
401-783-3370

Narragansett Bay Estuary Program
Tom Ardito
URI Narragansett Bay Campus
Narragansett, RI 02882
401-874-6492

Save The Bay
Wenley Ferguson
434 Smith Street
Providence, RI 02908
401-272-3540

Neighborhood Associations

Common Fence Point Improvement Association
Mil Kinsella-Sullivan
401-683-4549

Allin's Cove Neighborhood Association
Sandra Wyatt
28 Byway Road
Barrington, RI 02806
401-246-0747

Land Trusts Aquidneck Island Land Trust
Anne Garnett, Executive Director
790 Aquidneck Avenue, Middletown, RI 02842
401-849-2799 TEL
401-851-8998 FAX

Federal Agencies

US Fish and Wildlife Service
Tom Halevik
Rt. 1A, Shoreline Plaza
PO Box 307 Charlestown, RI 02813
401-364-9124 PH

EPA Ocean and Coastal Protection Division
Rhode Island State Program Unit
1 Congress Street, Suite 1100
Boston, MA 02114-2023

NOAA Fisheries, Northeast Fisheries Science Center
Habitat Restoration Center
James G. Turek
28 Tarzwell Drive
Narragansett, RI 02882
401-782-3338 PH
401-782-3201 FAX

State Agencies
DEM Division of Agriculture, Mosquito Abatement Coordination Office
Al Gettman
401-789-6280 PH

RI Department of Transportation
M. Emilie Holland,
Senior Environmental Scientist
2 Capitol Hill - Room 229
Providence, RI 02903-1124
401-222-2023, ext. 4051
401-222-3006 (f)
DEM Division of Fish and Wildlife
Chris Powell, Coastal Fisheries
401-294-4524
Phil Edwards, Freshwater Fisheries
401-789-0281
Arthur Ganz, Shellfish
401-783-2304

Narragansett Bay National Estuarine Research Reserve
Roger Greene
Ken Raposa
National Estuarine Research Reserve
55 South Reserve Drive
Prudence Island, RI 02872
401-683-6780 PH

Watershed Programs
Woonasquatucket River Greenway Project
RI Watershed Partnership

Salt Ponds Coalition
Vic Dvorak
401-322-3068 PH
401-322-1092 FAX

Ten Mile River and Mt. Hope/ Narragansett Bay Watersheds
Andrea D. Langhauser, Team Leader
20 Riverside Drive, Lakeville MA 02347
508-946-2878 PH
508-947-6557 FAX
The Ten Mile River Watershed Team
The Narragansett/Mt.Hope Bay Watershed Team
The Massachusetts Watershed Initiative

Corporate Programs
The Rhode Island Corporate Wetlands Restoration Program

Academic Programs

University of Rhode Island
Coastal Resources Center
Rhode Island Sea Grant
University of Rhode Island
Narragansett Bay Campus
Narragansett, RI 02882
401-874-6842

Roger Williams University
Center for Economic and Environmental Development
One Old Ferry Road
Bristol, RI 02809
401-254-3563
Timothy M. Scott, Ph.D.

NRCS WHIP Strategy for Eelgrass Habitat Restoration

Overview:

Eelgrass, *Zostera marina*, is a submerged aquatic plant that was once widespread under the waters of Narragansett Bay. Less than a century ago, vast eelgrass meadows sustained one of the most significant bay scallop fisheries in New England and large populations of migratory waterfowl, such as the Atlantic brandt. Hurricane damage and disease outbreaks of the 1930's followed by water quality degradation over the past 70 years, have caused wide spread loss of eelgrass in Rhode Island and the North Atlantic (Short et al. 1987, 1988). Recent studies in Rhode Island indicate that less than 100 acres of eelgrass remain in Narragansett Bay (Cottrell et al.,1999). Eelgrass once thrived though out the coves of Narragansett Bay with populations documented as far North as the Providence River and Mt Hope Bay. Today the Bay's scallop fishery, which in 1925 would have been worth $33 million at today's wholesale prices, is extirpated (RI Commissioners Report on Shellfisheries, 1926). Other creatures that rely on eelgrass, like tautog, winter flounder, and Atlantic brandt, have declined precipitously (McKay and Mulvaney, 2001; NMFS, 2003).

After nearly a decade of research and restoration trials, an ambitious program to restore eelgrass populations in Rhode Island is now underway. USDA Farm bill programs, working together with an established and effective state, local, and federal partnership, are now uniquely positioned to positively impact this valuable but declining habitat. Significant opportunities now exist to increase the scale of eelgrass restoration in RI due to major improvements in water quality achieved since passage of the Clean Water Act, and scientific advances in eelgrass restoration techniques.

Rhode Island's primary seagrass is eelgrass, *Zostera marina*. Eelgrass provides many ecologically valuable functions. It produces organic material that becomes part of the marine food web; helps cycle nutrients; stabilizes marine sediments; and provides important habitat. Eelgrass can form large meadows or small separate beds, which range in size from many acres to just a yard across. Found in depths up to 20 feet in some areas, eelgrass growth and survival is dependent on clear water to provide light for photosynthesis.

As new growth replaces older eelgrass leaves, the dead leaves decay, becoming a valuable source of organic matter for microorganisms at the base of the food chain (NOAA Damage Assessment and Restoration Program, 2001). Eelgrass reduces shoreline erosion caused by storms and wave energetics thus protecting adjacent coastal properties. Eelgrass meadows can stabilize sediments and filter nutrients from the water column. Eelgrass also provides a unique habitat for recreational SCUBA divers and snorklers to explore (Chesapeake Bay Program, 2000).

Though poor water quality conditions is attributed to the loss of eelgrass population in RI, recent studies demonstrate that a measurable increase in water clarity has occurred in the waters of Narragansett Bay in the past decade (Li and

Smayda 1998, Borkman and Smayda, 1998). Areas where eelgrass once thrived and was lost may once again support eelgrass populations. Natural recruitment of eelgrass to these areas can to take on the order of hundreds to thousands of years. Consequently, significant opportunities now exist to use transplant and seed restoration methods in order to speed the recovery of eelgrass to historic levels.

Fish & Wildlife

Many species of fish and wildlife depend on eelgrass. Several studies have shown the significance of eelgrass habitat in promoting and maintaining species diversity (Orth et. al*.,* 1984; Bell and Pollard, 1989; Heck et al*.,* 1989; Howard *et. al.,* 1989) in marine systems. Eelgrass beds provide protection for bay scallops, quahogs, blue crabs and lobsters. Waterfowl such as Atlantic brandt feed on eelgrass. In Waquoit Bay, Massachusetts, researchers found the loss of eelgrass to be correlated with reduction in fish community ecological integrity (10 of 13 species observed were found in maximum abundance in habitats with high eelgrass complexity, (Hughes, et. al., 2002).

Eelgrass also functions as nursery habitat for commercially valuable species such as tautog, winter flounder, and lobster. Because of its structural complexity and cover, eelgrass has been shown to reduce mortality rates of juvenile fish and increase food availability (Tolan et. al., 1997). But the predators know this, so eelgrass beds are often visited by striped bass, bluefish, weakfish, and scup who take advantage of open patches and canopy edges to ambush small prey. As dead eelgrass leaves decay they become a source of organic matter for microorganisms at the base of the food chain.

Species rich plant and animal communities attach themselves to the surface of eelgrass leaves; and are known as eelgrass “epiphytes.” Epiphytes that use eelgrass as essential habitat include the lacuna snail, blue mussel, bay scallop and oyster. Below the sediment surface the eelgrass root system pumps oxygen into an otherwise anaerobic soil. In this “root-soil jungle” dozens of species of polychaete worms, crustaceans, shellfish, bacteria, and other denizens can be found in numbers greater than in unvegetated bottom habitats. This rich and diverse fauna form the base of the estuarine food web, providing the necessary food resources to support commercial and recreationally valuable finfish and shellfish fisheries

Social & Economic

Eelgrass reduces shoreline erosion caused by storms and wave energetics thus protecting adjacent coastal properties (Ginsberg and Lowenstam 1958, Taylor and Lewis 1970, Den Hartog 1971, Harlin et al. 1982, Fonseca et al. 1983, Fonseca 1996). By buffering the energy of the ocean’s waves, eelgrass beds stabilize sediments and filter nutrients from the water column, creating clearer water. Eelgrass also can reduce shoreline erosion caused by storms and waves, protecting coastal property (Ginsberg and Lowenstam 1958, Taylor and Lewis 1970, Den Hartog 1971, Harlin et al. 1982, Fonseca et al. 1983, Fonseca 1996).

Eelgrass also provides a unique habitat for recreational SCUBA diving and snorkeling. On Mexico's Sea of Cortez, people of the Seri culture harvest eelgrass seed for use as food. While there is no recorded use of eelgrass as grain in the U.S., New Englanders used eelgrass for packing fish prior to the development of refrigeration, and farmers and Native Americans used eelgrass as compost for crops.

Eelgrass Habitat is strongly linked to healthy productive fisheries. Along the Greenwich Cove shoreline of Rhode Island, was a site historically known as "Scalloptown." This now extinct fishing port was comprised of rows of scallop shucking houses that processed the renowned bay scallop, harvested from the eelgrass beds of Greenwich Bay. During this era, Narragansett Bay's eelgrass beds produced scallop harvests in the hundreds of thousands of bushels. Today, the only remaining and viable bay scallop fisheries occur along coastlines and estuaries that still support eelgrass populations. Blue mussel aquaculture in the Gulf of Maine rely on spat collected from eelgrass beds.

Restoration Benefits

Restoration projects implemented since 2001, have resulted in increasing biodiversity at restoration sites through out Narragansett Bay. During September, 2002, Save The Bay in collaboration with the University of Rhode Island conducted a comprehensive field survey of benthic infuana and epifauna inhabiting the 2002 restored eelgrass beds, unrestored areas and natural eelgrass beds. Benthic Infauna and epifauna form the base of the estuarine food web, providing the necessary food resources to support commercial and recreationally valuable finfish and shellfish fisheries. Preliminary results indicate that there is a statistically significant difference between species diversity inside and outside of restoration transplants. Species diversity was also higher in transplanted beds (Shanon removal (June – July 2002) and shoot counts (August 2002). All species were identified in the field.

Species Common in Restoring Eelgrass Beds (Save The Bay, unpublished data)

Vertebrates		**Invertebrates**	
Common Name	**Scientific**	**Common Name**	**Scientific**
Seahorse	*Hippocampus spp*	Blue Crab	*Callinectes sapidus*
Pipefish	*Syngnathus fuscus*	Lady Crab	*Ovalipes ocellatus*
Flounder	*Pleuronectes americanus*	Spider Crab	*Libinia emarginata*
Scup	*Stenotomus chrysops*	Rock Crab	*Cancer irroratus*
Cunner	*Tautogolabrus adspersus*	Jonah Crab	*Cancer borealis*
Tautog	*Tautoga onitis*	Hermit Crab	*Pagurus spp*.
		Shrimp	*Crangon sp*
		Chinc Snail	*Lacuna vincta*
		Tunicate	*Botlyoides sp*
		Golden Star	*Botryllus Schlosseri*
		Tube worm	*Spirobis* sp.

Habitat Degradation and Impacts

Historic evidence indicate that the majority of eelgrass habitat in Narragansett Bay have been lost (Doherty et al.,1995). Though hurricane damage and the wasting disease decimated eelgrass populations in the 1930's, poor water quality conditions remain the leading cause of eelgrass decline in Rhode Island as well as other regions of the Mid-Atlantic. The principal culprit is nutrient pollution from waste water treatment plants, septic systems, lawn fertilizers, agriculture, and other sources. The pollutants fertilize the growth of algae, from tiny plankton plants to large seaweeds. The algae reduce the amount of light available for photosynthesis by eelgrass plants, destroying the beds. The good news is that recent efforts to reduce nutrient pollution are beginning to have an effect on Narragansett Bay. Positive changes to slow or stop the nutrient loading from both point and nonpoint sources are now beginning to have an effect on Narragansett Bay. This is evident by measurable changes in water quality and clarity in Narragansett Bay that have been observed by URI researchers (Li and Smayda 1998, Borkman and Smayda, 1998). Areas that once supported large populations of eelgrass may again be able to sustain eelgrass.

Eelgrass was widespread in Narragansett Bay as late as the 1860s. Historical accounts record eelgrass beds in the lower Providence River, at the head of the Bay. During the 1930s wasting disease, a widespread infection partly attributed to the slime mold *Labryinthula zosterae* decimated Atlantic coast eelgrass populations (Short et al. 1987, 1988). Some recovery was documented up until the 1960's. Since 1960, there has been an estimated 40% decline in

Narragansett Bay's eelgrass beds (NOAA, 2002). Approximately 100 acres of eelgrass remain in Narragansett Bay today (Cottrell et al 1999).

Eelgrass beds are susceptible to destruction and degradation by increased turbidity, increased nutrient loading from urban runoff, and destruction by boat propellers or invasive predators (Fonseca et al., 1998). The most significant threats to the remaining eelgrass beds in Narragansett Bay, and a deterrent to their long-term recovery, are nutrient pollution from sewage and polluted runoff from the land. Specific sources of these nutrients are septic systems, fertilizer runoff from lawns, and wastewater treatment plant discharges. Increases in surface water & groundwater nutrient loads result in phytoplankton blooms and excessive growth of macroalgae, which shades eelgrass beds, and inhibits plant growth and colonization. Declining water quality also increases the opportunity for wasting disease, which is caused by a marine slime mold that thins eelgrass beds and makes them more vulnerable to environmental stresses. The chronic presence of wasting disease has been tied to increases in water temperature and salinity.

Nine coastal ponds located along Rhode Island's South Shore are managed by the CRMC through a Special Area Management Plan (SAMP). The ponds were historically maintained as brackish systems through natural, seasonal breaches in the barrier beaches, which separate them from the open ocean. Permanent breachways were constructed at 5 of the 9 ponds (Point Judith, Ninigret, Winnapaug, Quonochontaug and Green Hill) during the 1950s and early 1960s. Construction of the permanent breaches has resulted in significant changes to the ecology of the ponds. Salinity has increased changing the ponds from a fresh/seasonally brackish system dominated by widgeon grass (*Ruppia maritima*) to a marine system--dominated by eelgrass. Faunal changes have accompanied the changes in the submerged aquatic vegetation community and salinity regime. Sedimentation has increased as a result of the permanent breachways. Flood tidal shoals are expanding within the ponds, encroaching upon eelgrass and shellfish bed habitat. Attempts to dredge the tidal shoals have only been partly successful in reducing the rate of encroachment upon the existing eelgrass beds (USACE-NED 2002).

Goals and Approach:

The RIHRT has not established any specific long term goals for the restoration of eelgrass populations in RI. However, a restoration partnership between USDA-NRCS, Save The Bay, and the University of Rhode Island Graduate Shool of Oceanography (URI-GSO) have identified 284 acres of Narragansett Bay as moderate and high potential for eelgrass restoration. 115 acres have been targeted for restoration by these partners over the next five years. Additionally, the Army Corps Of Engineers (ACOE) South Shore Restoration Project has identified 180 acres of eelgrass restoration habitat as part of their flood tide delta shoal excavation project along the South Shore Salt Ponds. Habitat models now under development have identified 1000’s of acres of potential eelgrass restoration in state waters.

In developing eelgrass restoration projects, site selection is critical, as restoration failures have been attributed to transplanting in inappropriate environments that do not provide minimum habitat requirements. Over the past six years, Geographic Information System (GIS) based spatial models have been employed to plan and design eelgrass restoration projects. The Narragansett Bay Eelgrass Restoration Site Selection Model (NBERSSM) is now being used to site future eelgrass restoration projects in Rhode Island. This decision support tool builds upon model approaches developed by Short et al. (2002). **SEE APPENDIX D for more information on NBERSSM.**

In Narragansett Bay 4400 acres have been identified as suitable for eelgrass restoration using the PTSI. Based upon the results of the 2001 FTSI model output, 270 acres have been identified as having moderate to high full scale restoration potential - 115 acres of which are targeted by Save The Bay, URI-GSO, and NRCS. NRCS and its RI partners will use the NBERSSM and a statewide site selection model now under development by NOAA CSC to identify and prioritize eelgrass restoration sites through out Rhode Island coastal waters. The NOAA Coastal Services Center is currently developing site selection models RI Salt Ponds.

Current RI Eelgrass Restoration technologies:

The first attempts to restore eelgrass in Rhode Island were undertaken in 1996 by the Narragansett Bay Estuary Program. Since that time, a number of organizations have undertaken seagrass restoration projects in Rhode Island waters, using a variety of restoration techniques. Eelgrass restoration methods to date have focused primarily on transplantation of either small clusters of plants or denser, sod-like sections. The transplants may be grown in aquaria or taken from healthy donor beds. In Narragansett Bay, researchers are restoring eelgrass using the Transplanting Eelgrass Remotely with Frames (TERF^{tm}) method, developed by Dr. Fred Short from the University of New Hampshire, in which clusters of plants are temporarily tied with degradable crepe paper to a weighted frame of wire mesh. Once the plants have become established, the frame is taken off the bottom for re-use elsewhere. The use of seeding techniques is also a restoration technique that has the potential to increase the scale of eelgrass restoration projects and eliminate the need to collect whole plant material from naturally occurring donor locations. **For a full analysis of eelgrass restoration seeding technology development please review Appendix G in this document.**

URI GSO has recently designed and constructed an innovative eelgrass seeding machine. It is used to sow hundreds of thousands of eelgrass seeds under water. The seeding machine acts as an underwater planting device where seeds are injected in a nutritive gelatinous matrix, pumped into the tines of the planting sled- and injected just below the sediment surface. The seeding machine uses the same technology the food industry employs to inject jelly into donuts. Additionally, whole eelgrass plants, collected from designated donor sites, can also being planted within the restoration areas to provide positive feedback mechanisms for establishing seedlings- protection from predators, wave dissipation, and sediment stabilization.

A three-pronged effort is now being implemented to maximize restoration success using techniques that are proving to be effective in Southern New England:

- Eelgrass transplants using Transplanting Eelgrass Remotely with Frames (TERF™) technique, developed by Dr. Fred Short of UNH, with harvested whole plants collected from eelgrass donor beds. This technique uses modified lobster pot frames, which are used to temporarily anchor attached eelgrass plants to the bottom of the seabed. Sites restored in 2001 that were sited with the assistance of the Narragansett Bay Eelgrass Restoration Site Selection Model and planted with the TERF™ method are achieving a mean survival of 80%, one of the highest success rates reported in Southern New England.

- Underwater seeding using a mechanized gel injection seed drill developed by URI GSO. Millions of eelgrass seeds can be harvested from flowering eelgrass shoots with little to no impact on extant eelgrass beds. This unique approach avoids the harvesting of adult shoots from existing beds thereby reducing any potential negative impacts to those sites. The underwater seed drill is used to sow hundreds of thousands of eelgrass seeds under water. The drill acts as an underwater planting device where seeds are injected in a nutritive gelatinous matrix, pumped into the tines of a planting sled and injected just below the sediment surface. The seeding machine uses the same technology the food industry employs to inject jelly into donuts. This method has the benefit of increasing the scale of restoration and reducing time & labor costs.

- Transplants using seed grown eelgrass plants mericultured in flowing seawater tanks at URI GSO. This technique takes advantage of the efficiencies of seeding with the demonstrated success of transplanting whole shoots using either TERF™ or hand planting. Research partners from URI GSO over eight years of experience rearing seedlings in flow through seawater tanks. With this technique optimum conditions for seed germination and seedling growth can be maintained, yielding a healthy crop of adult shoots within 9-10 months of seed planting. Through Efforts from Save the Bay volunteers and Staff the raised shoots are harvested and transplanted to restoration sites using TERF™ or other anchoring methods.

The techniques that are being implemented are increasing restoration success and acreage while reducing costs and minimizing disturbance to natural eelgrass beds, which act as donor sites. By growing eelgrass for restoration from seed; restoration efforts will be sustained in the future without negatively impacting natural beds.

Partners and Restoration Collaborators:
NBEP Watershed and Coastal Funds
U.S. EPA Watershed Initiative
CICEET –University of New Hampshire
USDA Small Watershed Program
USACOE/CRMC Coastal Pond Study
Development of Marine Aquaculture/Mericulture Projects
CRMC-State Coastal Habitat Restoration Trust Fund
NOAA-Restoration Center

References

Bell, J.D. and Pollard,D.A. 1989. Ecology of fish assemblages and fisheries associated with seagrasses p. 565-609. In Biology of Seagrasses. Elsevier, NY

Borkman G.D. and T.J., Smayda. 1998. Long-term trends in water clarity revealed by secchi-disk measurements in lower Narragansett Bay. Journal of Marine Science. 55:668-679.

Chesapeake Bay Program. 2000. (http://www.chesapeakebay.net/baygras.htm). Annapolis, Maryland.

Den Hartog, C. 1970. The dynamic aspects in the ecology of seagrass communities. Thalassia Jugosl. 7:101-112.

Fonseca, M.S., J.C. Zieman, G.W. Thayer, J.S. Fisher. 1983. The role of current velocity in structuring eelgrass (*Zostera marina*) meadows. Estuar. Coast. Shelf. Sci. 17:367-380.

Fonseca, M.S. 1996. The role of seagrasses in nearshore sedimentary processes: A review. Pp. 261-286. In: Roman, C. and K, Nordstrom. (eds.) Estuarine Shores: Hydrological, Geomorpological and Ecological Interactions. Blackwell, Boston, MA

Harlin, M.M., T.M. Boyce, J.C. Boothroyd. 1982. Seagrass- sediment dynamics of a flood-tidal delta in Rhode Island (U.S.A.). Aqua. Bot. 14:127-138.

Heck, K.L Jr., Able, K.W., Roman, C.T., Fahay, M.P. 1995. Composition, abundance, biomass, and production of macrofauna in a New England estuary: comparisons among eelgrass meadows and other nursery habitats. Estuaries 18(2) 379-389

Howard, R.K., Edgar, G.J., Hutchings, P.A. 1989. Faunal assemblages and seagrass beds , p. 536-564, in . In Biology of Seagrasses. Elsevier, NY

Hughes, E.H., Deegan, L.A., Wyda, J.C., Weaver, M.J, Wright, A. 2002. The effects of eelgrass habitat loss on estuarine fish communities of southern New England. Estuaries 25(2) 235-249

Li, Y. and T.J., Smayda. 1998. Temporal variability of chlorophyll in Narragansett Bay, 1973-1990. Journal of Marine Science. 55:661-667.

Fonseca, M.S., et al. 1998. Guidelines for the Conservation and Restoration of Seagrasses in the United States and Adjacent Waters. NOAA Coastal Ocean Program Decision Analysis Series No.12. NOM Coastal Ocean Office, Silver Spring, MD. 222pp.

Ginsberg, R.N. and H.A. Lowenstam. 1958. The influence of marine bottom communities on the depositional environment of sediments. J. Geol. 66:310-318.

Lipsky, A.A. 2003. Development of an Eelgrass Restoration Site Selection Model for Narragansett Bay. Appendix D. U.S. Department of Agriculture Natural Resources Conservation Service Technical Guidance Document, Warwick, Rhode Island. 109 pp.

NOAA Restoration Center. 2001. NOAA Restoration Center Web site: Are restored habitats as productive as the previously unperturbed habitats were? (http://www.nmfs.noaa.gov/habitat/restoration).

National Marine Fisheries Service. 2003. Annual Commercial Landing Statistics. (http://www.st.nmfs.gov/st1/comercial/landings/annual_landings.html)

Rhode Island Commissioners Report on Shellfisheries. 1926. Annual Report to the Rhode Island General Assembly for 1925.

Save The Bay. 2002. Restoration projects throughout the Narragansett Bay watershed. (http://www.savebay.org/bayissues/restoreprojects.htm).

Save The Bay, Inc., People for Narragansett Bay. 2001. Save The Bay Web site: What is Habitat Restoration? (http://www.savebay.org/).

Save the Sound, Inc. 1998. "The Long Island Sound Conservation Blueprint: Building the Case for Habitat Restoration In and Around the Sound." Stamford, CT. Save the Sound, Inc.

Short F.T., Davis, R.C., Kopp, B.S., Short, C.A., Burdick, D.M. 2002. Site-selection model for optimal transplantation of eelgrass *Zostera marina* in the northeastern US. Mar Ecol Prog Ser 227: 253-267.

Short, F.T., L.K. Muehlstein, and D. Porter. 1987. Eelgrass wasting disease: Cause and recurrence of a marine epidemic. *Biological Bulletin* 173:557-62.

Short, F.T., B.W. Ibelings, and C. Den Hartog. 1988. Comparison of a current eelgrass disease to the wasting disease of the 1930's. *Aquatic Botany* 30:295-304.

Taylor, J.D. and M.S. Lewis. 1970. The flora and fauna and sediments of the marine grass beds of Mahe, Seychelles. J. Nat. Hist. 4:199-220.

Thayer G.W., M.S. Fonseca, and W.J. Kenworthy. 1985 Restoration of seagrass meadows for enhancement of nearshore productivity. Proceedings of the international symposium on utilization of coastal ecosystems: Planning, pollution and productivity, 21-27 Nov. 1982, Rio Grande, Brazil. Volume 1: 259-278pp.

USACE-NED. 2002. Rhode Island South Shore Habitat Restoration Feasibility Report and Environmental Assessment (Draft). U.S. Army Corps of Engineers, New England

WHIP/WRP Habitat Restoration Strategy: Salt Marsh Restoration*
Overview:

Rhode Island salt marshes are found along the shores of salt ponds, the Narragansett Bay estuary, small embayments and estuarine rivers (such as the Narrow River estuary). RI salt marshes provide nursery grounds and foraging habitat for hundreds of species of fish, shellfish, birds, and mammals. Fish of all sizes, from mummichogs to striped bass, hunt in creeks and ponds. Quahogs and oysters live beneath the surface, while mussels, fiddler crabs, and snails occupy intertidal areas. Many kinds of birds visit the marsh to feed on the fish and invertebrates: osprey and herons, waterfowl, and mosquito-eating sparrows that nest in the marsh.

In addition to their habitat value, salt marshes serve as natural pollution treatment systems by filtering out pollutants before they reach coastal waters. The location of salt marshes between developed coastal communities and the waters of the state also provides a buffer during storms and flooding. Seventy-five percent of commercial fish species depend on estuaries for their primary habitat, spawning grounds, and nursery areas. The sweeping vistas afforded by the low lying salt marsh landscape contribute to the beauty and serenity of Rhode Island's coastline, as well as our tourism and outdoor recreation industry.

Salt marshes contain a complex of specialized plants and animals living in a low-energy, intertidal environment. The dominant vegetation in a New England low marsh is usually smooth cordgrass, *Spartina alterniflora*. Ribbed mussels and fiddler crabs are some of the more common animals. In New England, the dominant high marsh vegetation is generally salt hay grass (*Spartina patens*), black grass (*Juncus gerardii*), and spike grass (*Distichlis spicata*). Common animals include deer and a wide variety of migratory waterfowl, shorebirds, raptors, and songbirds.

As early as the 1950s, studies had shown that changes to the tidal flushing, or hydrology, of a salt marsh would degrade its quality as habitat, and that the restoration of historic hydrology could restore the habitat value of the marsh. In New England, where many salt marshes have been hydrologically altered by road and dike construction, dredged material disposal, and ditching, salt marshes have been a major focus of coastal habitat restoration efforts. Looking back on these projects after a decade or more, many of them have been very successful. The science and technology of salt marsh restoration are becoming fairly well established.

The simplest form of salt marsh restoration involves re-establishing historic hydrology where it has been lost. In New England, roads and railways have often been built through marshes. Many times, culverts were placed in the tide creeks at the road crossings. The culverts may have been too small to begin with, or may have collapsed over time. The resulting tidal restrictions prevent most tides from reaching the marsh inland of the road. The marsh becomes a brackish, rather than tidal system; *Phragmites*, replaces the salt marsh vegetation; and the marsh habitat becomes severely degraded. In these instances, the installation of a culvert or culverts large enough to allow the full tidal range to return may be all that is needed to restore the marsh. Restoration of tidal hydrology leads to increased salinity, sulfide toxicity, and increased flood duration in wetland soils which negatively affect the vigor and expansion of *Phragmites*. Over time, salt marsh vegetation and fauna will return in response to the elimination of Phragmites.

In other cases, the marsh itself may have been filled, often as a disposal site for marine sediments dredged from navigational channels. In these instances, it is not enough merely to restore hydrology because the surface of the marsh is too high to allow the tide to flood it. In these instances, marsh elevations must be re-established in order to restore a salt marsh ecosystem. These types of restoration projects tend to be more difficult and expensive as more construction is required, and the logistics of earth-moving in a marsh can be challenging. Nevertheless, some very successful projects of this type have been accomplished, such as the Galilee Salt Marsh Restoration in Narragansett, and the Allen Harbor Restoration in North Kingstown, Rhode Island.

Habitat Degradation and Impacts

It is estimated that 50% of Rhode Island's historic salt marshes have been filled (Save The Bay 2002). Consider that downtown Providence was once known as the Great Salt Cove, prior to filling and conversion to uplands. Marshes can be completely filled or they can be partially filled, altering the tidal exchange of water, and impacting vegetation communities that rely on twice-daily flooding. Often the result of such changes in elevation and flooding is the invasion by undesirable species such as *Phragmites australis* (common reed). *Phragmites* is very tolerant of disturbed sites, and can rapidly overtake such areas.

Construction of dikes, roads and rail crossings has resulted in the degradation of many marshes in Rhode Island. Restriction of tidal flow by installation of small culverts or drainage pipes under roads and rail beds leads to changes in salinity and alteration of the natural vegetation community due to a reduction in duration and frequency of tidal flooding. *Phragmites*, which is tolerant of these altered conditions, especially reduced salinity, often invades rapidly in areas that have been tide restricted. *Phragmites* out-competes the native short grass marsh community, and can reduce local plant, invertebrate, bird, and nekton biodiversity. Some 1200 of the existing 3700 acres of salt marsh in Narragansett Bay are impacted by *Phragmites* and other invasive plant species (Save The Bay 2002). These types of impacts to a salt marsh result in lower biodiversity, a decrease in flood abatement and erosion benefits, and provide potential mosquito breeding habitats.

Fish communities of salt marshes also suffer from road/rail infrastructure, as they rely on the natural tidal cycle to maintain populations in salt marshes. Marsh resident fish species, such as killifish (*Fundulus* spp.) spawn in concert with the tidal cycle, timing their spawning activity to coincide with the highest Spring tides, to ensure deposition of eggs in the high marsh portions of the marsh (Taylor et.

al. 1979). When natural tidal cycles are interrupted, or reduced, killifish spawning success is impaired. Tidal restrictions can reduce the amount of habitat available for estuarine-dependent fish that travel up into tidal creeks in search of food.

Mosquito ditching has impacted many marshes in Rhode Island. Mosquito ditches are very straight, narrow channels that were dug to drain the high marsh meadow for agricultural purposes. Historically, it was believed that ditching marshes would control populations of mosquitoes that breed there. It is now known that ditching, in fact, drains standing water which support populations of mosquito-eating fish (*e.g.*, killifish), leading to possible increases in mosquito populations. These fish are an important prey item for wading birds (herons and egrets), as well as larger, predatory fish species. Mosquito ditching alters natural patterns of groundwater drainage, which alters plant community composition, and nutrient cycling.

Polluted runoff from adjacent uplands can degrade salt marshes. Runoff from roads and other paved surfaces, and nutrient-rich runoff from fertilized lawns, agricultural areas, and septic systems can degrade marshes by encouraging growth of *Phragmites* and other invasive species. Forested buffer zones between populated areas and salt marshes have diminished as population growth in coastal areas increases. Approximately 58% of Narragansett Bay's marshes are impacted by polluted runoff. Some 30% of the Bay's marshes have inadequate or non-existent buffer zones (Save The Bay, 2002).

Summary Statistics for Narragansett Bay Marshes,
based on data from Tiner et al. (2003)

- 4,021 acres in need of restoration
- 900 acres of converted coastal wetlands (saltwater to freshwater, wetland to upland)
- 2/3 of restoration sites occur on private land
- 50% of the restoration sites are impacted by off site stresses
- 80% of the restoration sites are smaller than 1 acre
- 56% of coastal wetlands do not have adequate buffer zones
  - 33% of Buffer Zone is Single Family homes/Lawn
  - 22% of Buffer Zone is Forest
  - 15% of Buffer Zone is Rangeland
  - 8% of Buffer Zone is Commercial
  - 6% of Buffer Zone is Industrial

- 48% of Restoration Sites are Tidally Restricted

Goals and Approach:

According to Tiner et al. (2003) 65% of the remaining coastal wetlands in Narragansett Bay are impacted by human activities and potentially restorable. Over 236 restoration sites, representing 4,026 acres were identified in this report. Tiner et al. (2003) ***An Inventory of Coastal Wetlands, Potential Restoration***

***Sites, Wetland Buffers, and Hardened Shorelines for the Narragansett Bay Estuary* is found in Appendix F of this State Plan.** This document, although limited in scope to the 223,000 acre Narragansett Bay, serves the RI WHIP program as a baseline inventory of coastal wetland restoration sites. NRCS Coastal Wetland Restoration projects will use the priorities set forth by the RI Coastal Habitat Restoration Plan as established by RIHRT. SEE APPENDIX C for current projects identified by RIHRT . Priority projects are evaluated and prioritized on a yearly basis. Data from the NOAA Coastal Services Center salt marsh restoration site selection model will be used to identify and prioritize coastal wetland restoration sites on a statewide basis.

The restorability of brackish and tidal freshwater wetland restoration projects remain a gap in the science of wetland restoration ecology. Efforts to restore brackish wetlands have only recently been conducted in the Northeastern U.S with few published studies documenting successful restoration techniques. Sites that have been identified by various entities in RI that are considered degraded brackish/freshwater tidal wetland systems include: Briggs Pond, Long Pond, Old Mill Creek, and Coastal Lagoons in South County. Many of these systems are impacted by the monotypic dominance of *Phragmites australis*. For this reason, brackish Marsh/Freshwater tidal restoration projects should proceed with extreme caution due to the uncertain nature of achieving restoration success in these habitats. Brackish marsh and freshwater tidal wetland restoration projects that meet WRP/WHIP criteria should include well articulated restoration performance goals and multi-year monitoring plans utilizing standardized methods.

References

Save The Bay. 2002. Restoration projects throughout the Narragansett Bay watershed. (http://www.savebay.org/bayissues/restoreprojects.htm).

Save The Bay, Inc., People for Narragansett Bay. 2001. Save The Bay Web site: What is Habitat Restoration? (http://www.savebay.org/).

Taylor M.H., G.J. Leach, L. DiMichele, W.M. Levitan, and W.F. Jacob. 1979. Lunar spawning cycle in the mummichog, *Fundulus heteroclitus* (Pisces: Cyprinodontidae). *Copeia* 1979:291-297.
Salt Marsh Restoration

Adapted from the RI Coastal Habitat Restoration Portal

WHIP/WRP Habitat Restoration Strategy: Diadromous Fish Habitat (Anadromous & Catadromous Fish Species)

Overview:

Anadromous fish runs in Rhode Island occur in rivers, streams, and adjacent areas that drain into coastal ponds, Narragansett Bay, and Block Island Sound. These systems are used by migratory fish to feed and reproduce. River herring, Atlantic salmon, rainbow smelt, sturgeon, and American shad depend on passage upstream for survival. These anadromous fish spawn in fresh water, and mature and spend most of their lives in salt water. Conversely, American eels are catadromous fish, living in lakes and ponds as adults. They migrate downstream and eventually far out into the Atlantic, where they spawn and die in the Sargasso Sea. Their newly born young, less than an inch long, travel on ocean currents back to Rhode Island's rivers and streams.

Many of Rhode Island's rivers are blocked or obstructed by dams, weirs, tide gates, and other water-control structures. In addition to unobstructed passage through the water, migratory fish need healthy riparian areas whose vegetation provides cover, bank stabilization, and temperature regulation. Riparian vegetation also provides detritus (leaf litter, wood, etc.), which forms the base of the riverine food chain. Recreational and commercial fisheries benefit when river corridors remain healthy and passable to migratory fish (Save the Sound, Inc. 1998).

Habitat Degradation and Impacts

Rhode Island once supported lucrative fisheries for Atlantic salmon, shad, and river herring (alewife and blueback herring). Prior to European colonization, Native Americans depended on the spawning runs of herring and salmon as staples. Accounts by Roger Williams, Verrazano, and other explorers and colonists describe the astounding productivity of the Bay's tributaries.

During colonial times, dams were constructed throughout Rhode Island to harness water power. The advent of the Industrial Revolution in the late 18th century resulted in an increased number of larger dams. By the early 20th century, over 500 dams had been constructed in Rhode Island streams and rivers, with disastrous effects on anadromous fish runs. The Atlantic salmon fishery was lost by 1870. The river herring harvest was significantly depleted by 1930. Although commercial fisheries for these species are not currently viable, some runs still persist (*e.g.*, Gilbert Stuart Brook and Annaquatuck River in North Kingstown).

Dams change stream flow patterns, encourage upstream siltation and physically prevent fish from reaching upstream spawning habitat. Many rivers and streams that flow through urban, residential, and farmed areas are subject to industrial and agricultural pollution, both from point and non-point runoff. Rivers and streams, which have been straightened and channelized (often with concrete beds), are lacking in potential fish spawning habitat. Additional impediments to spawning success may include blockage of migratory pathways by debris (*e.g.*, construction materials, trash, brush piles, logs, etc.), blockage of smaller waterways by vegetation, culverts which may drain waterways or divert flow into rivers and ponds, and poor water quality. Water quality parameters critical to the successful movement of anadromous fish upstream include temperature, salinity, pH and dissolved oxygen (Durkas 1992)

In a healthy riverine system, fish migrate upstream to lay their eggs, and the eggs remain there until they develop into juveniles. In the fall, triggered by a decrease in water temperature and change in daylight, most of the juveniles begin their downstream run into more brackish water. Naturally functioning, stable stream systems promote the diversity and availability of habitats. Sinuous streams with slightly undercut banks, fallen logs, boulders and riffle/pool sequences provide some of the most diverse habitats for aquatic animals.

When a river or stream is blocked or altered, it will change the flow levels of the river or stream, sometimes allowing more sand and silt to build up on the bottom of the channel, covering previously used habitat for these fish species. Pollutants will accumulate in this sediment at the base of the obstructions. Obstructions can alter the water flow significantly, and in effect, they can change the bottom contours of the water body both upstream and downstream of the obstruction.

Riverine systems often run through urban and agricultural areas, and are often degraded by point and nonpoint source runoff when excess sediments, nutrients, and other pollutants clog streams and poison fish and wildlife. Stream banks that have been channelized and whose banks have been stripped of natural vegetation cannot provide the habitat necessary for the living resources of the water body (Chesapeake Bay Program 2000).

Goals and Approach:

It has been determined by RIDEM that there are at least 41 streams with potential for fish run restoration in Rhode Island and Massachusetts. Currently 18 streams support herring runs in Rhode Island but most are impaired to some degree and in need of restoration. Historically, at least 45 runs existed in the Narragansett Bay watershed. The most significant of these are the Taunton, Blackstone, Pawtuxet, Wood-Pawcatuck and Ten Mile rivers. As of October 2003, RIDEM Fish and Wildlife has finalized an anadromous fish restoration plan for Rhode Island. This plan will be used to assist NRCS in determining funding priorities and identification of potential projects. **This document is included in Appendix E of the State WHIP Plan.** GIS data, documenting restoration potential that includes target species, dam information, river mile and open water obstruction areas, and other pertinent information, is found off the Restoration Portal Website, as previously discussed.

In Rhode Island, the emphasis for restoration is on the herring family, particularly American shad, alewife, and blueback herring. The R.I. Department of Environmental Management (DEM) runs an Atlantic salmon restoration program on the Wood-Pawcatuck river system; however, most of the state's other rivers no longer have high enough water and habitat quality for salmon. Nevertheless, fish passage facilities can also benefit instream species such as brook and brown trout.

The most common fish passage facilities are fish ladders. Fish ladders can be built of concrete, timber, or aluminum. Some of the most common types are Alaskan steep-pass, denil, and pool-and-weir. Each is suitable for a particular species, stream size, or project cost. If a fish run restoration project is aimed at several target species, the ladders should generally be designed for the weakest swimmer.

Where practicable, dam removal is a better option than ladder construction because it restores the natural hydrology of the river, has the potential for many habitat and water-quality benefits, and because some anadromous fish such as rainbow smelt and sturgeon do not climb fish ladders. Federal, state, and non-profit agencies have begun promoting this approach to river restoration over the past several years, and several dams have been removed or breached in New England.

In Rhode Island, the legacies of industrialization and urbanization present special problems for dam removal. For example, on the Woonasquatucket River in Providence, the discovery of dioxin-contaminated sediments has spurred the reconstruction of an obsolete dam to prevent the release of contaminants into the river. Nevertheless, among the hundreds of dams in Rhode Island, most of which no longer serve their original purpose, there are many dams that it would be beneficial to remove.

Dam removal increases fish spawning habitat upstream of the obstruction. Where dam removal is not an option, fish ladders can be used. These structures are designed to enable anadromous fish to bypass these blockages and return upstream to spawn. Blockages can also be removed, notched, or breached, particularly if the dam is small or in disrepair**.**

Fish Passage Projects identified and prioritized by the RIHRT as state priorities (as identified in *9-30-2003*):

Blackstone River
Phase 1. Lower Four Dams
Wood Pawcatuck River
Phase 1. Feasibility and Conceptual Designs
Pawtuxet River
Phase 1. Pawtuxet Falls Dam
Woonasquatucket River
Multiple small dams on lower stem
See Freshwater Wetlands Strategy
Ten Mile River
Phase 1. First three dams

NATURAL RESOURCES CONSERVATION SERVICE
Rhode Island Anadromous Fish Habitat Restoration
Special Project Proposal
Fiscal Year 2005

Pawtuxet River Falls Dam, Warwick & Cranston RI

NRCS Natural Resources Conservation Service

Statement of Need:

Rhode Island once supported lucrative fisheries for anadromous Atlantic salmon *(Salmo salar)*, American shad (Alosa sapidissima), and river herring - alewife *(Alosa pseudoharengus)*and blueback herring (Alosa aestavalis). These “anadromous“ species spawn in fresh water, and mature and spend most of their adult lives in salt water. Because most of Rhode Island's rivers are blocked or obstructed by dams, weirs, tide gates, or other water-control structures; anadromous fish populations in Rhode Island have been severely impacted. Although commercial fisheries for these species are not currently viable, some fish runs still persist today (*e.g.*, Gilbert Stuart –North Kingstown and Nonquit in Tiverton). USDA NRCS Farm bill programs, working together with an established and effective state, local, and federal partnership, are now uniquely positioned to positively impact these valuable fish runs. Significant opportunities now exist to increase the scale of fish passage restoration in RI. Hundreds of restoration opportunities have been evaluated and identified by the Rhode Island Department of Environmental Management (RIDEM) Division of Fish and Wildlife’s *Strategic Plan for the Restoration of Anadromous Fishes to RI Coastal Streams*. Based upon a number of State Watershed Restoration Planning Meetings conducted in 2004, the highest priority river basin projects have been selected as part of this NRCS Special Project request. NRCS is requesting $4,313,750 in financial assistance to restore over 3559 acres of anadromous fish habitat to RI coastal and inland communities. This will result in far reaching ecological, social, and economic benefits to the state of Rhode Island.

Fish Run Restoration Overview

Anadromous fish runs in Rhode Island occur in rivers, streams, and adjacent areas that drain into coastal ponds, Narragansett Bay, and Block Island Sound. These systems are used by migratory fish to reproduce and provide nursery habitat for juveniles. River herring, Atlantic salmon, rainbow smelt, sturgeon, and American shad depend on passage upstream for survival. American eels are a catadromous species, living in lakes and ponds as adults. They migrate downstream and eventually far out into the Atlantic, where they are believed to spawn and die in the Sargasso Sea. Their newly born young, less than an inch long, travel on ocean currents back to Rhode Island's rivers and streams. Proposed fish passage restoration efforts include provisions for passing both catadromous and anadromous species. In addition to unobstructed passage through the water, anadromous fish need healthy riparian areas whose vegetation provides cover, bank stabilization, and temperature regulation. Riparian vegetation also provides detritus (leaf litter, wood, etc.), which forms the base of the riverine food chain. Recreational and commercial fisheries benefit when riparian corridors remain intact and passable to migratory fish.

Fish Run Habitat Degradation and Impacts

Prior to European colonization, Native Americans depended on the spawning runs of herring and salmon as staples. Accounts by Roger Williams, Verrazano, and other explorers and colonists describe the astounding productivity of the

Bay's tributaries. During colonial times, dams were constructed throughout Rhode Island to harness water power. The advent of the Industrial Revolution in the late 18th century resulted in an increased number of larger dams. By the early 20th century, over 500 dams had been constructed in Rhode Island streams and rivers, with disastrous effects on anadromous fish runs. The Atlantic salmon fishery was lost by 1870. The river herring harvest was significantly depleted by 1930.

Dams change stream flow patterns, encourage upstream siltation and physically prevent fish from reaching upstream spawning habitat. Many rivers and streams that flow through urban, residential, and farmed areas are subject to industrial and agricultural pollution, both from point and non-point runoff. Rivers and streams, which have been straightened and channelized (often with concrete beds), are lacking in potential fish spawning habitat. Additional impediments to spawning success may include blockage of migratory pathways by debris (*e.g.*, construction materials, trash, brush piles, logs, etc.), blockage of smaller waterways by vegetation, culverts which may drain waterways or divert flow into rivers and ponds, and poor water quality. Water quality parameters critical to the successful movement of anadromous fish upstream include temperature, salinity, pH and dissolved oxygen (Durkas, 1992).

In an unaltered riverine system, fish migrate upstream to lay their eggs, and the eggs remain there until they develop into juveniles. In the fall, triggered by a decrease in water temperature and change in daylight, most of the juveniles begin their downstream run into more brackish water. Naturally functioning, stable stream systems promote the diversity and availability of habitats necessary to support anadromous fish habitat requirements. Sinuous streams with intact streamside vegetation, undercut banks, fallen logs, boulders and riffle/pool sequences provide some of the most diverse habitats for aquatic organisms.

When a river or stream is blocked or altered, it will change the flow levels of the river or stream, sometimes allowing more sedimentation to build up in the channel upstream of the dam, covering previously used habitat for these fish species. Pollutants can accumulate in these sediments to levels of concern for humans and the ecosystem. Obstructions can alter the water flow significantly, and in effect, they can change the bottom contours of the water body both upstream and downstream of the obstruction.

Fish runs often run through urban and agricultural areas, and are often degraded by point and nonpoint source runoff when excess sediments, nutrients, and other pollutants clog streams and poison fish and wildlife. Stream banks that have been channelized and whose banks have been stripped of stream-side vegetative cover cannot provide the habitat necessary for the living resources of the water body.

Unimpacted Anadromous Fish Habitat	Degraded Anadromous Fish Habitat
• Forested or thickly vegetated riparian zone bordering river or stream • Presence of fallen logs or boulders that provide habitat structure • Indicators of good water quality, such as diverse benthic community • Valuable in-stream species such as native brook trout • Vegetated banks • Gravelly or sandy sediments	• Presence of an obstruction to fish passage • Channelized streambank • Unvegetated or undercut banks • Paved banks or riparian areas • Presence of floodwalls • Trash in the river or along the banks • Erosional areas along banks • Poor water quality • Presence of fish species representative of degraded habitats, such as carp • Contaminated sediments • Mucky sediments

Anadromous Fish Restoration Techniques:

The most common fish passage facilities are fish ladders. Fish ladders can be built of concrete, timber, or aluminum. Some of the most common types are Alaskan steep-pass, Denil, rock-ramp, bypass channels, and pool-and-weir. Each is suitable for a particular species, stream size, project cost, or specific site condition. If a fish run restoration project is aimed at several target species, the ladders should generally be designed for the weakest swimmer.

Where practicable, dam removal is a better option than ladder construction because it restores the natural hydrology of the river, has the potential for many habitat and water-quality benefits, and because some anadromous fish such as rainbow smelt and sturgeon do not climb fish ladders. Federal, state, and non-profit agencies have begun promoting this approach to river restoration over the past several years, and several dams have been removed or breached in New England. In Rhode Island, the legacies of industrialization and urbanization present special problems for dam removal. For example, on the Woonasquatucket River in Providence, the discovery of dioxin-contaminated sediments has spurred the reconstruction of an obsolete dam to prevent the release of contaminants into the river. Nevertheless, among the hundreds of dams in Rhode Island, most of which no longer serve their original purpose, there are many dams that it would be beneficial to remove. Dam removal increases fish spawning habitat upstream of the obstruction. Where dam removal is not an option, fish ladders can be used. These structures are designed to enable anadromous fish to bypass these blockages and return upstream to spawn. Blockages can also be removed, notched, or breached, particularly if the dam is small or in disrepair.

Proposed Restoration Goals

It has been determined by RIDEM that there are at least 41 streams with potential for fish run restoration in Rhode Island and Massachusetts. Currently 18 streams support herring runs in Rhode Island but most are impaired to some degree and in need of restoration. Historically, at least 45 runs existed in the Narragansett Bay watershed. RIDEM's Anadromous Fish Restoration Plan along with priority projects identified by the Rhode Island Habitat Restoration Team (RIHRT) have been used to determine funding priorities and identification of potential projects. RIHRT has been responsible for coordinating statewide restoration planning since its inception in 1998. RIHRT consists of a local, state, and federal partnership whose mission is to promote the restoration of damaged coastal habitats by coordinating restoration planning, projects, and information at the state level. RIHRT is responsible for developing a state coastal habitat restoration plan as mandated by the RI "Coastal and Estuary Habitat Restoration Program and Trust Fund which was established into law in Spring 2002. NRCS is an active partner on the RIHRT

<u>The following River Basin Projects have been identified and prioritized by the RIHRT and RIDEM Anadromous Restoration Plan as part of this NRCS special project proposal:</u>

Almy Creek Watershed
Almy Creek has an existing river herring run into Nonquit Pond. A significant amount of habitat is currently inaccessible upstream of Watson Reservoir. Funding for engineering, design, permitting, and construction will be required to restore over 380 acres of anadromous fish habitat. Anticipated restoration action will be the installation of a denil type fishway.

Kickemuit River
The Kickemuit Reservoir Dam, located at the head of tide, currently prevents the passage of migrating river herring and other fish species. The proposed project includes the installation of a Denil fish ladder and plunge pool to allow both upstream fish access to the reservoir during the spring adult migration, as well as out-migration by adults and juveniles in the summer and early fall. This fish ladder will allow river herring access to spawning and nursery habitat in the Kickemuit Reservoir and is an excellent opportunity to restore a historic river herring run to Narragansett Bay. Engineering and Permitting for this project is already completed. Restoration will result in the restoration of over 40 acres of habitat in Rhode Island and Massachusetts.

Pawcatuck River Watershed
The Pawcatuck river, one of the three largest and most pristine rivers in Rhode Island, is located in the southwestern portion of RI and southeastern Connecticut (Erkin, 2002). The 308 sq. mile watershed includes the Rhode Island towns of Westerly, Charlestown, South Kingstown, Richmond, Hopkinton, Exeter, and West Greenwich; and in Connecticut-Stonington, North Stonington, Voluntown, and Sterling. Aquatic habitats range from warm-water impoundments and flowing water to freestone streams in the Wood River and Usquapaug River watersheds.

Warm water areas are predominated by submerged aquatic vegetation, and have a habitat suitable for American shad, alewives, and blueback herring. Coldwater sections are suitable for Atlantic salmon and trout species. This is the only watershed identified by RIDEM with significant salmon habitat. A total of fourteen major restrictions to fish passage will be addressed by this proposal, resulting in the restoration of over 1750 acres of fish habitat. Conceptual restoration designs have already been completed for four of these projects.

Lower Pawtuxet River

The Pawtuxet River is Narragansett Bay's third largest tributary. Anadromous fish have been prevented from spawning in the Pawtuxet River due to a dam at the mouth of the river where it confluences with Narragansett Bay. Today, river herring can be found at the base of the dam where the freshwater discharge of the river provides sufficient attraction for fish migrating through Narragansett Bay. Providing fish passage to the Pawtuxet River would enhance the marine fishery in Pawtuxet Cove and Narragansett Bay and the freshwater fishery in the river. Restoration design and planning is currently underway for this critical project which has the potential to not only restore 54 acres of anadromous fish habitat but also freshwater tidal and brackish wetland habitats, the rarest wetland types in Rhode Island.

Woonasquatucket River

The Woonasquatucket River extends from its headwaters in North Smithfield, Rhode Island to the City of Providence. In Providence, the River creates "Waterplace Park" a centerpiece for Providence's downtown revival, then merges with the Moshassuck River and flows into Narragansett Bay. The watershed is approximately 52 square miles in area and drops more than 200 feet in elevation along its 19-mile length. Due to the River's significant industrial heritage, dams are prevalent. Fish passage in the lower Woonasquatucket River is currently obstructed by five abandoned mill dams. Preliminary surveys by staff of the RI Department of Environmental Management Division of Fish and Wildlife, NOAA Restoration Center, and Narragansett Bay Estuary Program have found suitable habitat and conditions for river blueback herring and alewife species with possible habitat for American shad in the lower river. Restoration of river herring to the Woonasquatucket River will provide ecological benefits to the river and upper Narragansett Bay by restoring historic anadromous fish spawning and rearing areas. This phase of the restoration will restore approximately 4.5 acres of upstream habitat for fish passage. Total area including downstream habitat from the river mouth to the first dam is approximately 37 acres.

The dams initially targeted for this project are within the area of the Woonasquatucket River Watershed Council's Woonasquatucket River Greenway Project (the Greenway Project), a designated Brownfield Showcase Community, and the Woonasquatucket River is a federally designated American Heritage River. Each of these designation efforts received broad community support. The master plan for the Greenway Project was developed after 18 community meetings, focusing on the restoration of abandoned public lands and Brownfields as new passive and active recreational spaces for the residents of Olneyville, a designated Federal Enterprise community neighborhood. Restoration of the river and its habitat is an important element in the plan for the Greenway Project. In addition to meetings for the master plan, neighborhood organizations and school children participated in the design of the public spaces. The Woonasquatucket River is an important natural asset in a neighborhood where 54% of the children live in poverty, the population is 78% minority, and the median family income is the lowest in the city - $19,676 (citywide-$32,058).

NRCS has entered into a cooperative agreement with U.S Fish and Wildlife Service to provide conceptual engineering designs for the lower four restoration projects. This work will be completed Fall 2004. Additionally, Owners of three dams have expressed their willingness and interest in pursuing fish habitat restoration. Struever Brothers, Eccles & Rouse, the owner of the first two dams, has applied for funding from NRCS for assistance in looking at options for fish passage at these two sites. The fourth impoundment, formerly known as Dyerville Dam, has deteriorated to a state where removal would be possible and welcomed by the owner.

Greenwich Bay Watershed-Gorton Pond

Gorton Pond is a 57.89-acre pond located near the City of Warwick's village center, Apponaug. It drains southwesterly through a culvert at Route 116 (Greenwich Avenue) into Little Gorton Pond. Little Gorton Pond drains at its southern end by a stream that follows a meander before becoming channelized under the mill complex located at the corner of Route 116 and Route 117. Somewhere along the stream reach under Little Gorton Pond, the stream joins Hardig Brook and ultimately drains to Apponaug Cove and Greenwich Bay. This is a highly urbanized drainage with intense residential, commercial and industrial land uses on the pond's shorelines and along the streams. In addition to noticeable impacts to the pond and its streams, (i.e. channelization, litter, debris), it suffers from several water quality impairments. According to the State of Rhode Island's 2002 303(d) List of Impaired Waters, Gorton Pond is a 'Group 2' water with excessive algal growth/chlorophyll A, low dissolved oxygen and high levels of phosphorous. Despite these negative conditions and impacts, several sources verify that the system supports a significant annual run of spawning alewives (*Alosa pseudoharengus*). According to RIDEM and local sources, migrating adult alewives have been observed at the outlet of Little Gorton Pond for the past several years in the spring. It has also been witnessed that anglers frequent the Gorton Pond and Little Gorton Pond outlets to catch migrating adults.

Annaquatucket River

The Annaquatucket River runs through the town of North Kingstown, R.I. on the west shore of Narragansett Bay. Four fish ladders on the Annaquatucket (Hamilton, Featherbed, Belleville Pond and Secret Lake) allow river herring to pass upstream to spawn in several large impoundments. Minor modifications and the installation of a slide gate at Hamilton dam (first ladder on system) will restore over 200 acres of spawning habitat for river herring.

Restoration Partners:

RIDEM Division of Fish and Wildlife
4808 Tower Hill Road
Wakefield, RI 02879**Coastal Resources Management Council**
Megan Higgins
4808 Tower Hill Road
Wakefield, RI 02879

Narragansett Bay Estuary Program
Tom Ardito
URI Narragansett Bay Campus
Narragansett, RI 02882

Save The Bay
Wenley Ferguson
434 Smith Street
Providence, RI 02908

US Fish and Wildlife Service
Tom Halevik
Rt. 1A, Shoreline Plaza
PO Box 307 Charlestown, RI 02813

EPA Ocean and Coastal Protection Division
Rhode Island State Program Unit
1 Congress Street, Suite 1100
Boston, MA 02114-2023

NOAA Fisheries, Northeast Fisheries Science Center
Habitat Restoration Center
James G. Turek
28 Tarzwell Drive
Narragansett, RI 02882

Wood-Pawcatuck River Watershed Association
203b Arcadia Road
Hope Valley, RI

RI Rivers Council
Meg Kerr
One Capital Hill
Providence, RI 02908

Kickemuit River Watershed Council
Ann Morrill
48 Laurel Lane (summer address)
Warren, RI 02885
Pawtuxet River Authority
http://www.pawtuxet.org/
Historic Pontiac Mills
334 Knight St.
Warwick, RI 02886

Buckeye Brook Coalition
Steve Insana, President
PO Box 9025
Warwick, RI 02889-9025

Woonasquatucket River Watershed Council
Jenny Pereira, Exec. Dir.
532 Kinsley Ave
Providence, RI 02909
www.woonasquatucket.org
RI Watershed Partnership

Corporate Programs
The Rhode Island Corporate Wetlands Restoration Program

Kleinschmidt Associates, Inc.
Connecticut Office
Kleinschmidt Building
35 Pratt Street, Suite 201
Essex, CT 06426

Project Budget

Basin/Project	Dam Height	Anticipated Restoration Method	Total Installation Cost	Partner Contributions-Installation	Planning & Design Costs by Partners	75% NRCS Cost share-FA	Total Habitat Acres*
Kickemuit River		denil	**$ 220,000**	**$ 140,000**	**$100,000**	**$ 80,000**	**40**
Woonasquatucket River							
Rising Sun*	5.53	denil	$ 200,000		$ 12,000	$ 150,000	1.5
Paragon*	4	removal	$ 150,000		$ 12,000	$ 112,500	3
Riverside/Atlantic Mills Dam	9.5	denil	$ 332,500		$ 12,000	$ 249,375	13
Dyerville Dam	2	removal	$ 50,000			$ 37,500	5
Manton Dam	6	denil	$ 210,000		$ 12,000	$ 157,500	7
Sub Total Watershed			**$ 942,500**		**$ 48,000**	**$ 706,875**	**31**
Lower Pawtuxet River							
Pawtuxet River Falls Dam*	9	Rock ramp	**$ 400,000**		**$150,000**	**$ 300,000**	**101**
Greenwich Bay Watershed Gorton Pond							
Upper Dam	na	obstruction removal	$ 80,000		$ 10,000	$ 60,000	58
Lower Dam	na	obstruction removal	$ 20,000		$ 10,000	$ 15,000	3
Sub Total Watershed			**$ 100,000**		**$ 20,000**	**$ 75,000**	**70**
Almy Watershed							
Watson Reservoir	34	denil	**$1,190,000**			**$ 892,500**	**606**
Pawcatuck River Basin							
Ashaway Line Pond*	4.2	steepass	$ 105,000			$ 78,750	1
Ashaway Mill Pond *	6.5	steepass	$ 162,500			$ 121,875	3
Bethel Pond*	6	steepass	$ 150,000			$ 112,500	18
Ashaway river-Chapman Pond debris clean up	****	obstruction removal	$ 10,000			$ 7,500	172
Bradford Dam	6	repair				$ -	82
Potter Hill Dam*	8	denil	$ 320,000			$ 240,000	83
Alton Pond Dam*	12	denil	$ 480,000		$ 15,000	$ 360,000	72
Woodville Pond*	7	denil	$ 245,000		$ 15,000	$ 183,750	51
Wood River	7	steepass	$ 175,000			$ 131,250	34

Project Budget Continued

Basin/Project	Dam Height	Anticipated Restoration Method	Total Installation Cost	Partner Contributions-Installation	Planning & Design Costs by Partners	75% NRCS Cost share-FA	Total Habitat Acres*
Junction*							
Route 91 gauging station*	3	slot	$ 20,000		$ 5,000	$ 15,000	22
Carolina Pond *	7	removal	$ 200,000			$ 150,000	1
Shannock Mill Pond *	7	removal	$ 245,000			$ 183,750	4
Horseshoe Falls Dam *	16	removal	$ 640,000			$ 480,000	12
Kenyon Mill Pond *	7	denil	$ 245,000			$ 183,750	1200
Sub Total Watershed			**$2,997,500**		**$ 35,000**	**$2,248,125**	**2533**
Annaquatucket River							
Hamilton Dam		Slidegate repair	**$ 15,000**			**$ 11,250**	**193**
PROJECT TOTAL			**$5,865,000**	**$ 140,000**	**$353,000**	**$4,313,750**	**3559**

*** Maximum NRCS contribution, partners will likely provide some portion of installation cost in excess of 25%**

Fish Passage Restoration Types:

Removal: Either full or partial dam removal, restoring in stream aquatic riverine habitat as well as fish passage

Denil: Artificial roughened channel used on larger river systems

Steeppass aka "Alaskan Steeppass" fishway: denil type fishway for lower flow rivers and streams

Rock Ramp: Nature Like Fishway using pool and weir concept usually run of river

Obstruction Removal: removal of in stream obstructions via mechanical methods

* Bold Acreage is the total restoration area affected by restoration actions (upstream and downstream, calculated using NOAA Anadromous Fish Site Selection Model). Unbolded acreage reported for each fish passage project is upstream area affected by each particular restriction, calculated by Erkan (2002). Therefore, unbolded numbers do not sum to subwatershed total reported in bold.

Restoration Outcomes:

Basin/Project	Upstream Habitat Acres	Potential Annual River Herring Returning Adults*
Kickemuit River	40	54,480
Woonasquatucket River		
Sub Total Watershed Cost	29.5	40,179
Lower Pawtuxet River		
Pawtuxet River Falls Dam*	54	73,548
Greenwich Bay Watershed Gorton Pond		
Sub Total Watershed Cost	61	83,082
Almy Watershed		
Watson Reservoir	380	517,560
Pawcatuck River Basin		
Sub Total Watershed Cost	1755	2,390,310
Annaquatucket River		
Hamilton Dam	216	294,192
PROJECT TOTAL	2536	3,453,351

* Based upon habitat suitability models adapted by L. Cavallaro, NOAA Restoration Center

References

Chesapeake Bay Program. 2000. (http://www.chesapeakebay.net/tribs.htm). Annapolis, Maryland.

Durkas, S.J. 1992. Impediments to the spawning success of anadromous fish in tributaries of NY/NJ Harbor Watershed. American Littoral Society, Highlands, New Jersey.

Erkan, Dennis, E. 2002. Strategic Plan for the Restoration of Anadromous Fishes to Rhode Island Coastal Streams. Rhode Island Department of Environmental Management Division of Fish and Wildlife. 81pp.

RI Habitat Restoration Plan and Information system 2003. (http://www.csc.noaa.gov/lcr/rhodeisland/index.htm).

Save the Sound, Inc. 1998. "The Long Island Sound Conservation Blueprint: Building the Case for Habitat Restoration In and Around the Sound." Stamford, CT. Save the Sound, Inc.

Coastal Habitat Restoration: Progress To Date

Projects 1999-2006	Coastal Habitat Type	Acres	NRCS Cost
Block Island	Shellfish Habitat	68.0	$ 20,000.00
	Subtotal Shellfish Habitat Restoration	**68.0**	**$ 20,000.00**
Save the Bay	SAV	50.0	$ 736,000.00
Save the Bay (Eelgrass)	SAV	1.0	$ 23,761.00
Save The Bay-Eelgrass	SAV	15.0	$ 590,000.00
	Subtotal Submerged Aquatic Vegetation	**66.0**	**$1,349,761.00**
Annaquatucket	fish habitat	193.0	$ 18,750.00
Bradford Dam	fish habitat	647.0	$ 56,000.00
BRISTOL COUNTY WATER AUTHORITY	fish habitat		$ 101,200.00
Guild	Fish Habitat	50.0	$ 51,000.00
Paragon Dam	fish habitat	3.0	$ 165,000.00
Pawtuxet River	Fish Habitat	110.0	$ 300,000.00
RICC	Fish habitat	150.0	$ 50,000.00
Rising Sun Dam	Fish Habitat	43.0	$ 169,000.00
Riverside Dam	Fish Habitat	15.0	$ 168,000.00
saugatucket/gilbert	fish habitat	450.0	$ 20,150.00
THE GRODEN CENTER	fish habitat	5.0	$ 124,167.00
	Subtotal Coastal Fish Habitat Passage	**1666.0**	**$1,223,267.00**
Alice Westervelt	Coastal Wetland	5.5	$ 17,175.00
Barrington Land Trust	Coastal Wetland	20.0	$ 12,150.00
Briggs Marsh	Coastal Wetland	244.0	$ 2,625.00
Cards Pond	Coastal Wetland	70.0	$ 36,000.00
City of Cranston (Stillhouse Cove)	Coastal Wetland	5.0	$ 9,929.00
Donald Roach	Coastal Wetland	12.8	$ 7,299.43
Duck Cove	Coastal Wetland	8.0	$

			22,000.00
Gooseneck Cove	Coastal Wetland	60.0	$ 500,000.00
Jacob Point	Coastal Wetland	40.0	$ 380,000.00
Jonson and Wales	Coastal Wetland	6.0	$ 197,000.00
Prudence Island	Coastal Wetland	10.0	$ 36,000.00
RICC	Coastal Wetland	52.0	$ 160,000.00
RIDEM (Colt State Park)	Coastal Wetland	6.0	$ 4,450.00
Save the Bay (Fields Point)	Coastal Wetland	3.0	$ 158,000.00
Town of Bristol	Coastal Wetland	15.0	$ 68,029.00
Town of Bristol (Silver Creek)	Coastal Wetland	8.0	$ 68,000.00
Walkers Farm (Town of Barrington)	Coastal Wetland	30.0	$ 69,975.00
	Subtotal Coastal Wetland Restoration	**595.3**	**1748632.4**
35 Projects	**Total Coastal Restoration in Rhode Island**	**2,395**	**$4,341,660.43**

WHIP/WRP Habitat Restoration Strategy: Freshwater Wetlands Restoration Strategy

Overview

Since July 1999, Rhode Island Department of Environmental Management's Office of Water Resources have been developing a statewide freshwater wetland restoration program.   NRCS WHIP technical team meeting, held in October 2002, concluded that NRCS freshwater wetland restoration programs would work in close partnership with RIDEM. RIDEM and URI's site identification and prioritization methods have benefited from the input and review from a wide variety of stakeholders, including: watershed associations, non governmental conservation organizations, municipalities, and representatives from State and Federal agencies. Because of the quality and widespread acceptance of the RIDEM program, the WHIP team recommended that Farm Bill programs administered by USDA-NRCS would follow the guidelines and methodology developed by RIDEM and the University of Rhode Island, as reported in Miller and Golet (2000). **This document can be found in Appendix B.**

Approach and Goals

Rhode Island Rivers provided the birthplace of the Industrial Revolution in the U.S. An unfortunate legacy of this revolution and subsequent urbanization was the destruction of important riverine habitats. The 18 mile Woonasquatucket River, now federally designated as an "American Heritage River," like many urban rivers today, has been straightened, covered over, floodplain wetlands destroyed, and its natural banks replaced by concrete flood walls. However, recent watershed studies have identified over 300 wetland and riparian restoration opportunities. Local, State and Federal partners are now realizing that significant riparian habitat restoration can occur as a part of urban revitalization efforts. Many of the river front industrial mill complexes along the Woonasquatucket River have fallen into disuse as the manufacturing sector declined in New England. These large properties offer opportunities to implement riparian restoration activities and accommodate urban redevelopment to benefit the poorest and most disadvantaged neighborhoods in our region. To assist partners in the necessary restoration efforts, NRCS was nominated as the "champion" agency to implement the Woonasquatucket River Watershed Restoration Initiative.   To kick off these efforts in 2003, the Riverside Mills Riparian Restoration Project was selected for funding through the Wildlife Habitat Incentives Program. This restoration project provides the full range of site restoration constraints: multiple uses, soil contamination, stream bank stabilization, utility relocation, and regulatory remediation requirements. Riverside Mills restoration demonstrates that our nation's most ecologically degraded and economically challenged urban rivers can be restored. "If we can do it along the Woonasquatucket, we can do it any watershed."

NRCS will continue to participate with the active wetland restoration partnership in the Woonasquatucket River Basin. The American Heritage Rivers Initiative (AHRI) Federal Interagency Working Group has identified RI-NRCS as the lead champion agency responsible for implementing 2002 Keystone Projects for the Woonasquatucket River Watershed Restoration Initiative. 2002 keystone project recommended the restoration of 5-10 riparian restoration sites to establish stronger regional and habitat corridor connections and the restoration of at least five freshwater wetlands sites within the watershed.

The Following eight sites were selected by the Watershed Team for FY 2003, dependent upon timing and funding levels:

1. Button hole and Site W6 Wetland Restoration/ Dam Removal Project
2. Johnston Wetland Restoration –22 acre site
3. Smithfield Rec. Department Floodplain Wetland Restoration
4. Deginian Riparian Restoration/Dam Removal- NRCS survey completed
5. Whipple Field Restoration (Project Funded from FY 2000 WHIP)
6. Graystone Mill Riparian Restoration
7. Providence Place Mall Riparian and Instream Aquatic Restoration
8. Olneyville Post Office Riparian Restoration

Continue to work with AHRI watershed action team for the Blackstone /Woonasquatucket river to implement priority projects selected from the their Pilot Freshwater wetland restoration plan. Thus far, over 239 potential wetland buffer restoration opportunities and 77 wetland fill sites have been identified (11 sites on public land). Contact with many landowners have been initiated at some but not all locations, feasibility studies (including cost estimates) have been conducted for a limited number of restoration sites.

Additional freshwater wetland restoration sites have not been identified using a watershed based scientific approach such has been completed for the Woonasquatucket River Basin. However, NRCS will target farm bill programs to restore freshwater wetlands in other basins on a case by case basis-with an emphasis on wetland restoration projects on working/previously worked farmlands. NRCS will work with other agencies and community groups to apply watershed based restoration planning for other basins in Rhode Island.

Partners:
RIDEM Fish and Wildlife
RIDEM Water Resources
USFWS
US EPA
Save The Bay
American Heritage Rivers Initiative
Blackstone Valley Natural Heritage Corridor
City of Providence
Woonasquatucket River Watershed Association

Rhode Island Department of Transportation
RI Cities and Towns

Riparian Restoration Cost Estimate for the Woonasquatucket River Watershed.

Golet et al. (2002) assessed the restorability of riparian buffers throughout the Woonasquatucket River Basin. Sites were evaluated for their ability to improve water quality and other wetland functions at the watershed scale. Based upon this analysis, 239 identified sites were categorized as high, medium, and low priorities for restoration, as reflected in the table below.

Woonasquatucket Riparian Buffer Water Quality
Priority Restoration Scenarios*

Scenario 1 with 50 ft Buffer

	High Priority	Medium Priority	Low Priority	Total
Number of Projects	40	103	96	239
Required Buffer (Miles)	4.8	10.6	8.4	23.7
Buffer Area (Acres)	29	64	51	144
Buffer-Linear Feet	25,400	55,746	44,183	125,329
Estimated Cost	$ 1,018,807	$ 2,236,000	$ 1,772,202	$5,027,010

Scenario 2 with 35' ft Buffer

	High Priority	Medium Priority	Low Priority	Total
Number of Projects	40	103	96	239
Required Buffer (Miles)	4.8	15.4	8.4	23.7
Buffer Area (Acres)	20	45	36	101
Buffer-Linear Feet	25,400	55,746	44,183	125,329
Estimated Cost	$ 698,880	$ 1,572,480	$ 1,257,984	$3,529,344

* **Site assessment data derived from (Golet et al. 2002)**
*** Riparian Restoration Cost/Acre=$ 34,944 (Plan, Design, Permit, Site Prep/Install)**

Cost Estimate Method:
NRCS staff generalized a typical riparian buffer restoration design scenario in order to determine an approximate riparian restoration cost for the watershed. It should be emphasized that site plans and designs will need to be carried out in order to ascertain the true cost for a particular site. Site specific characteristics may increase or decrease the actual costs of restoration at a particular site. Given these constraints, NRCS believes that this restoration cost estimate reflects an accurate cost representation of all the project components required to achieve on the ground riparian restoration. Data derived from Golet et al. (2002) and managed by Rhode Island Department of Environmental Management's Wetland Restoration Program include functional assessment attributes as well as the length of riparian buffer required by site. Therefore, NRCS staff were able to sum the required length of riparian buffer per restoration priority cateqory. NRCS assumed that the width of the potential riparian buffer would be either 35 or 50 feet, based upon ongoing projects in the watershed. Therefore two buffer restoration scenarios are depicted in the above table.

The following project components were factored in the cost estimate: Planning, Design, Permitting, Site Preparation, Planting, and Initial Plant Establishment. Operation and Maintenance (O&M) costs were not estimated due to the significant variability of O&M that may be required among restoration sites. Furthermore, Project Monitoring Costs were not included in this cost estimate. However, these cost components should be included in any restoration undertaking.

The generalized riparian buffer scenario included a typical three zone riparian area: 1. Floodplain Forest-Tree/Shrub Zone, 2. Scrub Shrub Zone, and 3. Herbaceous Zone. Cost estimates were calculated as a per acre cost for each zone. It was assumed that the typical riparian buffer would be composed of the following ratio of zones: 40% Zone 1: 40% Zone 2: and 20% Zone 3.

Installation costs for each zone were based upon the type of planting material being installed. Cost data was based upon the weighted average unit costs from 2003 Cost List provided by the Rhode Island Department of Transportation. Generally, labor, site preparation, and initial plant establishment costs are calculated by multiplying plant material unit cost by a factor of **2.25**. It was assumed that planning, design, and permitting would cost 20% of the full installation cost. Given this methodology, Total Riparian Restoration Costs were calculated at **$34,944** per acre.

Upland Wildlife Habitat Priorities
Wildlife Habitat Incentive Program (WHIP)
(*Prepared by RIDEM Division of Fish and Wildlife*)

<u>Overview</u>:

Maintaining diversity of wildlife population demands that a diversity of habitats be present on the landscape in order to meet the food, shelter and water requirements of wildlife. It is rarely possible to meet the habitat needs of every species in a single area. Habitats may then be managed to meet the needs of species that have similar habitat requirements, which are under represented on the landscape. If suitable habitat is unavailable, wildlife populations will decline or fall below an optimal level for they're continued survival. Plant succession is generally a linear process that unless disturbed, the tendency is toward mature forest communities. Since habitat is also a dynamic feature of the landscape, disturbances are periodically necessary in order to maintain conditions necessary to meet the habitat needs of several species of wildlife. Habitat dynamics are complicated by the fact that society demands that we suppress or control most natural forms of disturbance, including wild fires, flooding, or animals, such as the beavers, which create new habitats by their natural behavior. This leads to stagnation and disappearance of disturbance-dependent habitats that are needed by some wildlife species. This is further complicated by the fact that farms and commercial timber harvests have also declined to a significant point in the region and development of habitats for residential and commercial uses continues to cause losses of habitat.

Habitat Degradation and Impacts:

According to the USDA Forest Service (Alerich 2000) in Rhode Island forests cover 59 percent of the land area of the state (393,000 acres). Eighty-six percent of this acreage is classified as timberland capable of producing forest products - and wildlife. This is a decrease from 1985 when timberland acres represented 92 percent of the forest (380,000 acres). During the same period, nonforest land classes fueled by development pressure increased by 7.4 percent. Farmland acreage in Rhode Island decreased dramatically after WW Two. In 1980, total farmland acreage stood at 38,165 acres or just about 5% of the state land area (USDA-SCS 1981). In recent years, the number of farms (700) and total farmland acreage (55,000) have increased slightly mitigating trends, post WW Two, of dramatic declines in number of farms and acreage of farmlands (DEM 2001). Dairy farms, which contain a mixture of habitats beneficial to wildlife, declined in number 55 percent since in the period 1980-1990, currently representing 6791 acres statewide. The loss of farmland combined with modern farming practices on some farms which result in "clean", hedgerow free lands has caused a further loss of habitat for many upland wildlife species.

Accompanying the decline of agriculture and maturation of forests was a decline in grassland acreage, resulting in the loss of wildlife attracted to these areas.

One of the biggest concerns for wildlife is the decrease in seedling/sapling aged forest stands from 42 percent in 1953 to 6 percent in 1998. This has caused a shift in forest age class toward more mature woodlands that cannot support certain types of wildlife dependent upon so called early successional habitats for nesting, feeding and brood rearing. Several authors (Askins 2001, Trani et al. 2001, Lorimer 2001, Hunter et al. 2001, Desseker and McAuley 2001, and Litvaitis 2001) have recently discussed the conservation of woody, early successional habitats and wildlife in the Eastern United States and how declines in this habitat are adversely impacting the conservation and biological diversity of many species, most thought to be common in Rhode Island.

Restoration Benefits

Examples of bird species and mammals that inhabit Rhode Island that are associated with shrub-scrub, early successional forest or grass-herbaceous dominated conditions and have experienced declines are listed in table 1. The distribution and abundance of cottontail rabbits in southern New England have declined dramatically over the last 50-years due to the maturation of the forest and a corresponding loss of early successional woodlands. The decline in New England cottontail populations caused a corresponding decline in bobcats in the northeast (Litvaitis 2001). Rhode Island's native cottontail, the New England Cottontail, is listed as a priority species by the Northeast Nongame Technical Committee and the United States Fish and Wildlife Service has been petitioned to list this species as Threatened or Endangered under the Endangered Species Act (M. Amaral, US Fish and Wildlife Service, personal communication).

Birds and mammals associated primarily with early successional forest or grass-herbaceous conditions in Rhode Island that are in decline (table 1).

Common Name	Scientific
Ruffed grouse	*Bonasa umbellus*
American woodcock	*Scolopax minor*
Whip-poor-will	*Caprimulgus vociferus*
Blue-winged warbler	*Vermivora pinus*
Northern waterthrush	*Seiurus noveboracensis*
Praire Warbler	*Dendroica discolor*
Chestnut-sided warbler	*Dendroica pensylvanica*
Yellow-breasted chat	*Icteria virens*
American kestrel	*Falco sparverius*
Eastern wood-pewee	*Contopus virens*

Eastern bluebird	*Sialia sialis*
Worm-eating warbler	*Helmitheros vermivorus*
Wood thrush	*Hylocichla mustelina*
American Goldfinch	*Carduelis tristis*
Field sparrow	*Spizella pusilla*
Upland sandpiper	*Bartramia longicauda*
Horned lark	*Ermophila alpestris*
Vesper sparrow	*Pooecetes gramineus*
Savannah sparrow	*Passerculus sandwichensis*
Grasshopper sparrow	*Ammodramus savannarum*
Bobolink	*Dolichonyx oryzivorus*
Eastern meadowlark	*Sturnella magna*
Northern bobwhite	*Colinus virginianus*
Northern harrier	*Circus cyaneus*
New England cottontail	*Sylvilagus transitionalis*
Bobcat	*Lynx rufus*

Goals and Approach:

Priority upland wildlife habitats in Rhode Island that are of conservation concern include early successional forest, shrub-scrub dominated habitats, old fields and grass-herbaceous dominated areas. These habitats will be given most attention; however, other types of upland habitats and other habitats within the landscape (such as freshwater wetlands) will be also be considered in the overall approach to managing the landscape. **Partners in Flight Bird Conservation Plan for Southern New England highlights the importance of early sucessional habitat restoration and can be found in Appencix I of this document.**

Acreages of gravel mines and wastelands have coincidentally increased approximately 50 percent since the mid seventies to approximately 5500 acres statewide. For the most part, reclamation of these gravel mines was not required and none or very little restoration of these areas has occurred-these sites should be targeted for early sucessional habitat restoration.

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Early Successional Seedling-Sapling Forest:

1. The loss of early successional seedling sapling forest and small forest openings as the forest matures is considered to be a major factor in the decline of some birds, including the American Woodcock and many songbirds which depend on periodic disturbances in the forest. Woodcock numbers have declined over the last 27 years at a rate of 1.9% (USFWS 1996). Creation of small forest openings of 0.5 to 20 acres can help reverse the trend in habitat loss for this species. Other species in decline (see

table 1) will also respond favorably to management actions creating young forest conditions. A mosaic of forest blocks should be carefully placed on the landscape relative to other age class stands and then allowed to grow through the seedling-sapling stage into maturity. The American Woodcock Management Plan Region 5 (USFWS 1996) encourages that 20 percent of the forest be maintained in the seedling-sapling stage.

According to the 1998 survey of Rhode Island forests (Alerich 2000), sawtimber represented 54 percent of the forest (a 125 percent increase since the early 1970's), poletimber represented 40% and seedling-sapling represented 5.8% of the forest. Seedling-sapling stands decreased 633 percent in the same time period to only 6 percent of the timberland.

2. Active forest management is key to providing diverse wildlife habitat and maintaining a sustainable forestland base (Scanlon 1992). In order to provide for a diversity of wildlife species, it is recommended that forest size class distribution be maintained at approximately 20 percent seedling-sapling, 20 percent sapling pole and 60 percent saw timber. In order to meet this prescription to maintain and improve forest age class diversity in the state, 67,940 acres of early successional seedling sapling forest are needed on the landscape. Currently there are 21,300 acres of seedling sapling age class forest representing just 6% of the forest. The deficit of 46,640 acres can be addressed through outreach and projects directed towards private forest landowners. At a proposed goal of enhancing/creating 500 acres of early successional seedling sapling forest per year, correcting the deficit would take 93 years. We need to begin this process immediately to prevent further declines in species.

3. On a statewide basis, thousand of acres of forestland exist that can be managed for early successional wildlife. Even-aged silvicultural practices are most suitable where early successional wildlife is a priority and in New England hardwoods, provides habitat for more breeding bird species than does uneven-aged management (Thompson and DeGraaf 2001).   Inventory by a wildlife biologist and forester is necessary to design the proper cutting regime and layout on private non-industrial forest lands. To accomplish these objectives, even aged management must be employed on the woodlot, working to achieve the recommended forest age class composition (see #2 above). The ephemeral nature of these habitats will necessitate consideration in order to maintain habitat levels on the landscape, perhaps using a mosaic of regeneration cuts as suggested by (Thompson and DeGraaf 2001).

4. Opportunities exist for management of early successional habitats using prescribed fire. In particular, pitch pine/scrub oak habitats can be managed successfully using fire as a tool to rejuvenate the habitat values of these areas. Prescribed fire can assist in restoring these habitats by removing dead and dying fuel loads that can lead to uncontrolled wildfires and stimulate new grow of younger shrubs and saplings to improved habitat conditions.

5. The upland habitat target goal for the creation and management of seedling-sapling early successional forest is 500 acres per year statewide.

6. The cost per acre to implement the program will vary with the quality of the woodlot. It is assumed that private landowners will gain some income from the wood products (firewood, saw logs) that are removed by the cutting. The costs involved will primarily cover the technical assistance needed by the landowner to inventory and set up the even aged management program.

<u>Upland Grasslands-Herbaceous Dominated Areas:</u>

1. Large naturally occurring grassland-herbaceous dominated areas are rare in Rhode Island. There is a need to conserve and maintain these larger grassland habitats by using techniques such as fire and mechanical mowing to meet the conservation needs of species. This would include the need to identify and manage larger old-field complexes (5 acres to 100 acres) that have become overgrown with old-field shrubs and invasive shrub species such as Autumn olive (*Eleagnus umbellata*) and multiflora rose (*Rosa multiflora*).

2. Abandoned gravel mines represent an opportunity to restore considerable habitats for wildlife. Statewide there are approximately 5,500 acres of these mines. Gravel mines can be reclaimed to upland grassland habitats using organic compost (leaves, woodchips, animal manure and grass clippings) to create a suitable topsoil then seeding with warm season perennial grasses able to withstand the droughty conditions present in these habitats.

3. Grassland habitats represent preferred or utilized habitats for at least 43 species of birds and 29 mammals native to Rhode Island. A variety of wildlife utilize grassland habitats for breeding, feeding or wintering including upland sandpiper and northern harrier, both designated migratory non-game birds of management concern in the northeast by USFWS. In addition, grasslands provide habitats for a diverse assemblage of species such as American kestrel, northern bobwhite, meadowlark, wild turkey, various bats, meadow jumping mice, cottontails, and white-tailed deer.

4. The WHIP target goal for number of acres of grasslands to be managed or restored in Rhode Island is 200 acres.

5. The cost per acre to restore gravel mines by restoring grasslands is approximately $2,000 per acre. This dollar includes the approximate cost of the soil additives (compost), seed and equipment to prepare the site for planting.

<u>Field Borders:</u>

1. Field edges and borders have great potential for providing habitats for wildlife. Farms that promote abrupt, sparse or open edges between crops and woodlots are generally poor for wildlife habitat. A brushy woodland boarder combined with a low growing herbaceous zone of grasses and forbs are very productive for species preferring early successional habitats (e.g. cottontails, northern bobwhite and many songbirds).

2. Woodland boarders can be improved for wildlife by creating cutback zones of 30 feet from the edge of the existing field. This involves the removal of all overstory trees from the 30-foot zone in order to promote seedling, brush and coppice growth of woody stems. To maintain this covertype, periodic maintenance to remove growing stock is needed. This may be accomplished by selective hand cutting or with low volume EPA approved safe herbicides such as Roundup (Glyphosphate).

3. Field borders can also be improved by planting herbaceous grasses and forbs, such as mixtures of bluegrass, tall fescue, ladino clover, white clover or lespedeza. A 20-foot wide border adjacent to a 30-foot cutback zone at the woodland edge will improve the edge substantially for wildlife.

4. On a statewide basis, there exists extensive opportunity for creation of field border habitats. The WHIP target goal for creation and enhancement of field borders is 50,000 linear feet of field border or approximately 60 acres

5. The estimated cost to develop field borders is approximately $500 per acre.

<u>Native Plant Communities identified for Potential Fire Prescription</u>:

1. F. Carter Preserve (841 acres) —TNC and sizeable area of similar habitats occur on adjacent lands owned by Narragansett Tribe
2. Audubon- Epley Wildlife Refuge- grassland area reintroduction of *Agalinus acutis*
3. Block Island: Dickins Farm
4. RI Water Resources Board- West Greenwich Big River Management Area- 140 acre site for grassland restoration/ with hundreds of acres of oak and pitch pine barrens in adjacent lands

APPENDIX A. Habitat Needs Assessments

1. Eelgrass Habitat Needs Assessment (compiled from WHIP Technical Team Meeting Oct. 2002 & Appendix A.)

1. Remote sensing studies to re-map eelgrass beds in Rhode Island are necessary. Current distribution is now based upon 1996 remote sensing data. This will be necessary in order to quantify eelgrass abundance trends in the state.

2. Bathymetric mapping and re-sampling needs to be conducted on a statewide level using new technology, side scan sonar, especially focusing on near shore areas not covered by NOS surveys. This will improve the accuracy of GIS site selection models and minimize chances of planting in intertidal or areas that are too deep.

3. Light monitoring should be conducted throughout RI state waters at a fine spatial and temporal resolution. There is a significant need to have long term light data for RI coastal waters. Light monitoring can be conducted during growing season from March through November.

4. An eelgrass reference site monitoring program should be established to properly determine restoration site performance success in relation to naturally occurring eelgrass populations. This will allow resource managers to answer the question of whether restoration success is influenced by natural variability or by restoration methods?

5. Subaquaous soil mapping should be conducted. Soils are critical to the establishment of eelgrass plants and seeds. Soil mapping units may provide a better understanding of water quality conditions as well, as high organic/anoxic sediment map units will invariably covary with areas of nutrient enrichment and poor potential seagrass habitat. These areas are currently very difficult to quantitatify using remote sensing techniques.

6. Macroalgae Population Monitoring: Drift algae and sessile forms should be assessed statewide to quantify the potential for macroalgal production. This can be done either by remote sensing or population sampling techniques. Many of the restoration projects conducted in RI that have failed were caused by macroalgal smothering. Although sites may meet the light requirement thresholds for eelgrass, they may be severely impacted by macroalgae production, especially the drift algae *Ulva lactuca*. This data has the potential to vastly improve the ability of site selection models to select appropriate restoration sites.

7. Test sites need to be continually executed on a seasonal basis, ideally spring and fall test transplants should be conducted at sites over multiple years.

This will assist restoration practitioners in a number of ways: to determine appropriate sites to invest in full scale restoration, and to test whether certain locations are more suited for Fall or Spring transplanting

8. Wave Energy and Water Current Speed Data , interpolated for RI coastal waters; is a necessary data layer that can be used to further refine site selection procedures.

2. Freshwater Wetlands Restoration Needs Assessment (from WHIP/WRP Technical team meeting held October 2002 & Woonasquatucket Watershed Restoration meeting held December 18, 2002)

Partner with RIDEM, watershed organizations to develop watershed based wetland restoration plans for selected RI Watersheds

1. Site assessment and Prioritization
   Volunteer based assessments
   Scientist-based assessments
   Conduct Miller and Golet (2000) Prioritization Matrix on sites

2. Identify strategic projects with partners to implement on the ground restoration/enhancement projects coupling appropriate farm bill programs with local, state and federal resources.

3. Using the GIS based Wetland Information System, developed by Dr. Frank Golet and Dr. Peter August to identify agricultural landscapes that co-occur with important wetland complexes in Rhode Island. Prioritize agricultural lands for targeting USDA conservation planning and WRP programming.

4. South County Watershed Gap Analyses, compilation of GIS analyses looking at forest cover and resource overlap. This data can be used to overlay agricultural lands and prioritize farm bill program implementation

5. Kleinschmidt, Inc. developed a riparian buffer analysis for the mainstem Woonasquatucket River. Work with RIDEM and partners to develop restoration plan and implementation plan for these sites, if applicable for NRCS programs

6. TNC Land Acquisition analysis of Tiverton and Little Compton is a database that would assist NRCS in targeting NRCS farm bill programs. Obtain data and conduct spatial analyses.

7. TNC vernal pool mapping data for the Wood Pawcatuck River identified 1039 vernal pools that have been previously unmapped by existing wetlands coverage's. Obtain this database to assist in NRCS farm bill programming

8. Update conservation practices to include the establishment of urban buffers less than thirty feet wide.

9. NRCS and watershed partners need to develop a joint strategy to complete outreach to landowners, develop feasibility studies, and conduct field verification surveys for potential restoration sites that were not visited

Appendix B. Rhode Island Freshwater Wetland Restoration Plan for the Woonasquatucket River Watershed

Appendix C: State Estuary and Coastal Habitat Restoration Strategy

The following is a strategy ratified and adopted by the Rhode Island Habitat Restoration Team (i.e. Technical Advisory Committee) pursuant to the Coastal and Estuary Habitat Restoration Program and Trust Fund. The Trust Fund mandates that a plan be established with "comprehensive public, agency, legislative and stakeholder participation." (§ 46-23.1-5).

In so doing, the Habitat Restoration Team (comprised of public, agency, legislative and stakeholder participation) developed a plan that incorporates the following elements:

A. Description of RI's Coastal and Estuarine Habitats
B. Restoration Goals
C. Inventory of Coastal and Estuarine Projects
  1. projected comprehensive budget
  2. identification of funding sources
D. Criteria for Project Evaluation
E. Application Process

According to the plan, habitat restoration grant monies are dispersed in accordance with § 46-23.1-5(2) which allocates funding for design, planning, construction or monitoring. Eligible applicants include cities and towns; any committee, board, or commission chartered by a city or town; nonprofit corporations; civic groups, educational institutions; and state agencies.

B. Restoration Goals

Habitat restoration is necessary for a variety of reasons. Habitat restoration is being used to reintroduce locally extirpated rare plant species and to create habitat for threatened and endangered wildlife. The restoration of wetlands and riparian areas is helping to reverse long-term trends in habitat loss, which has occurred over the last century. Numerous small and large-scale projects are underway to restore the natural hydrology, soils and vegetation to habitats around Rhode Island.

Some goals of restoration may include, but are not limited to:

- The re-establishment of habitat structure, be it chemical, biological, or physical. This may include reestablishing or maintaining hydrology, whether by reestablishing river or tidal flow, restoring flood regimes, or re-establishing topography.
- Control of exotic, non-native, or invasive species of plants or animals.
- Re-vegetation through native plantings or natural succession.
- Removal of barriers or construction of fish ladders to provide passage for spawning or migrating fish.

- Controlling, reducing, or eliminating other specific adverse impacts such as controlling polluted runoff

C. Inventory of Coastal and Estuarine Projects

FY2003

PROJECT NAME	PROJECT LOCATION	CONTACT PERSON	AMOUNT REQUESTED	PROJECT TOTAL	OTHER FUNDING
1. Lonsdale Drive-In	Lincoln, RI	RIDEM James McGinn	$152,962.85	$2.7 million	$30,000+/- (Corp. Wetlands)
2. Explore the Bay/Field's Point	Providence, RI	Save The Bay Wenley Ferguson	~$25,000	$175,000	NOAA/NRCS; Johnson & Wales
3. Narragansett Bay Seagrass Restoration	Narragansett Bay	Save The Bay Wenley Ferguson	$29,773	$400,000*	WHIP
4. Stillhouse Cove	Cranston, RI	City of Cranston Jared Rhodes	$8,000	$650,000	$15,000 (WHIP)
5. Palmer Ave., Warren	Warren, RI	Warren Land Conservation Trust Dick Hallberg	$15,000	$40,000	none
6. Mussachuck Creek	Barrington, RI	RI Country Club Gary McLane	$10,000	$100,000	80/20 fed match
7. Napatree Dunes Restoration	Westerly, RI	NOAA Restoration Center Lisa Cavallaro	$7,000	$7,000	Partner with Watch Hill Fire District
8. Outreach pamphlet as insert in Narragansett Bay Journal	STATEWIDE	Tom Ardito, Editor of Narragansett Bay Journal	$7,000		

FY2004

PROJECT NAME	AMOUNT REQUESTED	PROJECT TOTAL	OTHER FUNDING
Omega Dam (Ten Mile River)	$100,000	750,000	150,000
Pawtuxet River	$50,000	150,000	35,000
Seagrass Restoration	$50,000	300,000	none secured
Gooseneck Cove	$50,000	750,000	
Wood/Pawcatuck River	$50,000	100,000	none
Town Pond/Boyds Marsh	$100,000	2.1 million	
Woonasquatucket River	$54,350	79,350	
Narragansett Bay SAV Mapping	$49,000	49,000	none
Cormorant Point, B.I.	$15,000	40,000	25,000
Water Quality and Eelgrass Habitat Restoration in Salt Ponds	$65,031	$65,031	TBD
Walker Farm	(A. Lipsky will provide #s)		
PLANNING EFFORTS			
CRMC Planning and Coordination	$93,000		
Stormwater Management/319*****	?	?	
EQUIPMENT			
Low Ground Pressure Excavator Machine	$50,000	50,000	
TOTAL STATE FUNDING	$726,381		

*Desire to have stormwater management projects encompass a restoration component.

D. Criteria for Project Evaluation

Factors to be taken into account by the technical advisory committee for the purposes of granting monies for estuary and coastal habitat restoration activities, determining the eligibility of an estuary and coastal habitat restoration projects for financial assistance, and in prioritizing the selection of estuary and coastal habitat restoration projects by the technical advisory committee (Rhode Island Habitat Restoration Team) shall include, but need not be limited to:

(1) consistency with the state estuary and coastal habitat restoration strategy, the Narragansett Bay comprehensive conservation and management plan, the state coastal nonpoint pollution control plan, the coastal resources management program, the department of environmental management regulations, the anadromous fish restoration plan, and pertinent elements of the state guide plan;

(2) the proposed timeline of the project (projects slated to begin sooner rather than later will be given greater preference);

(3) the ability of the applicant to provide adequate personnel funding, and authority to carry out and properly maintain the estuary and coastal habitat restoration activity;

(4) the proposed monitoring plan to ensure that short-term and long-term restoration goals are achieved; a final report given back to the TAC outlining what the project accomplished;

(5) the effectiveness of any nonpoint source pollution management efforts upstream and the likelihood of re-impairment;

(6) whether the estuary and coastal habitat restoration activity can be shown to improve or replace habitat losses that benefit fish and wildlife resources;

(7) potential water quality improvements;

(8) potential improvements to or replacements of fish and wildlife habitats for species which are identified as rare or endangered by the Rhode Island Natural History Survey or the federal Endangered Species Act;

(9) the level and extent of collaboration by partners (e.g., municipality, nongovernmental organization, watershed council, federal agency, etc.);

(10) potential direct economic and educational benefits to a community or the state; and

(11) ability of applicant to secure matching funds, whether the funds be NGO, state or federal dollars.

E. Application Process

Step 1:

Send a letter of inquiry before the beginning of the next fiscal year (July 1) to:
Megan Higgins, Coastal Policy Analyst
RI Coastal Resources Management Council
Oliver Stedman Government Center
4808 Tower Hill Road, Suite 3
Wakefield, RI 02879

The letter of inquiry shall include: (1) the name of the restoration project, (2) a map of the location (including street address, plat and lot) and town, (3) a detailed budget, indicating monetary amount requested from the RI Habitat Restoration Team, (4) property ownership information, (5) restoration project manager contact and contact information (phone, email address and mailing address), and (6) organization(s) responsible for the project. All contributing organizations for the project should be listed. If the project is being matched by a federal grant, the names of the grant and the granting organization should be named.

Step 2:

After the technical advisory committee has evaluated the project and the project proposal is considered for funding, send a detailed application.

The detailed application shall include the following:

(1) Cover Page

The application cover page shall include:(1) the name of the restoration project, (2) a map of the location (including street address, plat and lot) and town, (3) a detailed budget, indicating monetary amount requested from the RI Habitat Restoration Team, (4) property ownership information, (5) restoration project manager contact and contact information (phone, email address and mailing address), and (6) organization(s) responsible for the project. All contributing organizations for the project should be listed. If the project is being matched by a federal grant, the names of the grant and the granting organization should be named.

(2) Text

A description of the project shall include the type of restoration initiative that will take place, the historical impact to the site, the natural resources benefited and impacted (target species), any physical, ecological, biological, cultural/historical, geological and survey data that has been collected to date, a site map, any aerial photography and photographs of the site available, and any preliminary restoration drawings, maps and engineering plans. (refer to **Section D: Criteria for Project Evaluation** when describing project)

The text should also include proof of property owner permission for the restoration activity to take place. A list of required permits and the responsible party for obtaining the permits shall be included. (see http://www.csc.noaa.gov/lcr/rhodeisland for a complete list of necessary permits).

The narrative part of the application should not be more than six pages, double-spaced in 12-point font on one side only. Each page of the application must have a page number, a date and the project name. One signed original and two copies should be provided.
The application should not be bound in any way.

If there is any additional information that would assist in making a determination, please provide that information.

(3) Budget

A detailed budget of the costs and timeframe for the project must be included in the application. (see page 10 for a project budget template).

(4) Monitoring Plan
A monitoring plan should be established as appropriate. A portion of the funding received should be allocated for monitoring (e.g., reference monitoring).
Guides to monitoring include:

(a) “Monitoring Salt Marsh Vegetation”
(b) “Monitoring Nekton in Shallow Estuarine Habitats”
(c) “Long-Term Hydrologic Monitoring Protocol for Coastal Ecosystems”
(d) “Field Methods Manual: US Fish and Wildlife Service (Region 5) salt marsh study”

These protocols may be found on the National Park Service Inventory and Monitoring website:
http://www.nature.nps.gov/im/monitor/protocoldb.cfm

(5) Letters of Support

A letter of support from the appropriate state and/or federal resource agency is required.

(6) Submission of Applications
Megan Higgins, Coastal Policy Analyst
RI Coastal Resources Management Council
Oliver Stedman Government Center
4808 Tower Hill Road, Suite 3
Wakefield, RI 02879

Appendix D: Wildlife Habitat Incentive Program Technical Guidance Document for USDA-NRCS Eelgrass Restoration Programs in Rhode Island

Development of an Eelgrass Restoration Site Selection Model for Narragansett Bay

Prepared by
Andrew Lipsky
Department of Natural Resource Sciences
University of Rhode Island
&
USDA-NRCS-Rhode Island
May 5, 2003

ABSTRACT

Efforts to restore populations of eelgrass (*Zostera marina*) have been hampered by the lack of cost effective, rigorous, and replicable site selection tools. Site selection is considered by many to be the most important consideration for the restoration of seagrass populations. Restoration failures have been attributed to transplanting in inappropriate environments that do not provide minimum seagrass habitat requirements. The recent development of the Narragansett Bay Eelgrass Restoration Site Selection Model (NBERSSM) was intended to overcome the inadequacies of current site selection procedures and promote successful eelgrass restoration. Adapted from a recent model reported by Short et al. (2002), the NBERSSM incorporates two model components -- a GIS component and an active restoration evaluation component. A Preliminary Transplant Suitability Index (PTSI) model is performed by GIS analyses using existing data sets, representing what are believed to be the most important eelgrass habitat requirements. The PTSI model output ranked 45,000 acres of Narragansett Bay as low, moderately or highly suitable for eelgrass restoration. 4500 acres were classed by the PTSI as moderately and highly suitable for eelgrass restoration. As part of the active restoration evaluation component, restoration tests were conducted in Spring 2001 at 19 locations subsequently screened by the PTSI. Eelgrass persistence was evaluated at each site. Plant persistence is the ultimate test of suitability, as plants integrate all aspects of the physical, biological, and chemical attributes of a site. A Final Transplant Suitability Index (FTSI) model was conducted by integrating the performance of the test transplants (% survival) with subsequent site monitoring data (light, bioturbation, and temperature) to generate a final model output that ranked full-scale eelgrass restoration potential at the 19 sites. The FTSI model evaluated a total of 1700 acres, which included a range of PTSI classes, and identified over 250 acres of Narragansett Bay to be moderately and highly suitable for eelgrass restoration. In order to evaluate the performance of the NBERSSM, both the PTSI model and the FTSI model output were compared to in situ restoration success. The performance of the PTSI and FTSI model output was evaluated by testing whether sites predicted as highly suitable for restoration demonstrated restoration success. Ultimately, this is the acid test for the restoration practitioner with limited resources. ANOVA and Tukey tests performed on the survival data for the 19 test locations resulted in a significant relationship between PTSI class (unsuitable, low suitability, moderate suitability, and high suitability for restoration) and test site restoration success after one year's growth ($p < 0.0015$). ANOVA and Tukey tests were also performed on the survival data of three full scale restoration sites that were conducted in 2002, resulting in a significant relationship between restoration success and FTSI class ($p < 0.0001$). Additionally, two FTSI model input factors that were measured for the FTSI model, light and bioturbation, were evaluated for their contribution to model output by testing whether they were good indicators of transplant success. Light penetration reaching the seagrass canopy has been considered to be one of the most important factors that contribute to eelgrass survival and bioturbating organisms are known to negatively effect restoring eelgrass communities. Light data, collected during the first season's growth at restoration test sites, had a significant relationship with eelgrass survival among all locations ($p < .0001$). Stations receiving at least 20% surface light had the highest survival rates. Sites with low bioturbator abundance rankings had high survival ($p < .0030$). The 2002 sites selected from the FTSI model for full scale restoration are demonstrating a high level of restoration success with a mean survival of 68% after one season's growth.

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INTRODUCTION & OBJECTIVES

Efforts to restore seagrass habitats have been hampered by the lack of cost effective, rigorous, and replicable site selection tools. The most important consideration to re-establishing seagrass beds is the site selection process, as restoration failures have been attributed to transplanting in inappropriate environments that do not provide minimum habitat requirements (Fonseca, 1992; Short et al. 2000). Over the past six years, Geographic Information System (GIS) based spatial models have been employed to improve the site selection and planning phases of eelgrass, *Zostera marina*, restoration projects. The use of these diagnostic decision support tools could simplify and reduce the need for intensive and expensive empirical studies that have been necessary to select and predict potential seagrass restoration sites (Short et al., 2001). These GIS tools are therefore very attractive to restoration practitioners.

The complexity, pitfalls, and assumptions of spatially modeling seagrass habitat requirements are surprisingly unreported in the published literature. The first section of this paper presents important seagrass habitat requirement factors based upon a review of the published literature

The second part of this paper will report and discuss the development and testing of an eelgrass restoration site selection model for Narragansett Bay. Eelgrass transplants were conducted from May 2001 to October 2002 by Save The Bay, and were sited using a restoration site selection model adapted from Short et al. (2001). The performance of these restoration sites will be used to test whether the Narragansett Bay site selection model is a useful tool in identifying the best sites to conduct restoration. Guidance to improve future restoration site selection and potential management applications of the Narragansett Bay Eelgrass Restoration Site Selection Model are also presented.

PART I. EVALUATION OF SEAGRASS HABITAT LIFE REQUIREMENTS AND SPATIAL MODELLING CONSTRAINTS

The analysis presented in this section will draw on literature for seagrass species in North America with particular emphasis on the modeling and restoration of eelgrass, *Zostera marina*, in Chesapeake Bay and the Northeastern U.S. The lack of and non uniformity of operational definitions is apparent in review of seagrass restoration literature. Therefore, to avoid confusion definitions of key terms are presented.

Key Terms

Parameter: A quantity which is constant in a particular case considered (distinct from ordinary variables), but which varies in different cases (Addiscott 1998)

Model: representation of reality, extended hypothesis (Addiscott and Whitmore 1991)

Deterministic: model where a set events lead to a unique definable outcome

Mechanistic: *model incorporating fundamental mechanisms of process (Addiscott and Wagenet 1985)*

Decision Support Model: An applied model that provides practical guidance for management. These models may take many forms and range from quantitative, mechanistic approaches to qualitative, index-based tools

Seagrass habitat requirement: threshold values of parameters, factors, and variables established and reported in peer reviewed literature

A diversity of environmental fields have struggled with the adequacy, rigor, and applicability of deterministic models, including spatially-explicit GIS models. Key themes emerge from the literature of soil water modeling that effectively address the constraints and pitfalls of representing complex data sets in a spatial context. These themes are very applicable to a critique of seagrass restoration site selection models and are used as a foundation for this critique. The framework for this analysis will be to evaluate seagrass restoration ecology and spatial modeling against three generic modeling criteria, as identified in Addiscott (1998), Groffman and Wagenet (1994), and Wagenet (1996) considered to be important to testing the validity of environmental models:

1. Is process level information missing?

2. Does interaction of multiple factors produce unique phenomena?

3. What is the resolution and quality of environmental data?

To test the first criterion in relation to seagrass restoration models, an overview of key factors (process level information), reported in the literature of seagrass habitat requirements and restoration modelling is presented and evaluated. The overview will identify gaps in mechanistic understanding of seagrass systems, discuss issues of setting threshold limits, and relate these constraints to modeling seagrass habitat requirements. Literature on Seagrass Habitat requirements is primarily derived from two major technical syntheses: Guidelines for the Conservation and Restoration of Seagrasses in the U.S. and Adjacent Waters (1998) and Chesapeake Bay SAV Water Quality and Habitat Requirements and Restoration Targets: A Second Technical Synthesis (2001). Other supporting literature is reviewed where necessary. Based on a review of the literature and seagrass modelling to date, the following factors that constitute seagrass habitat requirements will be evaluated: **wave exposure,**

A. sediment characteristics,
B. seagrass presence (historic and current),
D. bioturbation,

E. Water quality (light, nutrients, macroalgae, epiphytic cover),

F. bathymetry (critical depth).

Tables 1-4 summarize what seagrass habitat requirements are considered most important by different restoration initiatives along the Atlantic Coast.

Table 1. Factors considered important for Chesapeake Bay SAV

Survival and growth, adapted from Batuik (2001):

Factors	**Importance**
Minimum Light Requirement	Primary

Water Column Light	Secondary
Total Suspended Solids	Secondary
Plankton Chlorophyll-a	Secondary
DIN	Secondary
DIP	Secondary
Water Movement (exposure, current speed)	Not Specified
Wave Tolerance	Not Specified
Sediments	Not Specified
Porewater Sulfide	Not Specified
Organic Matter	Not Specified

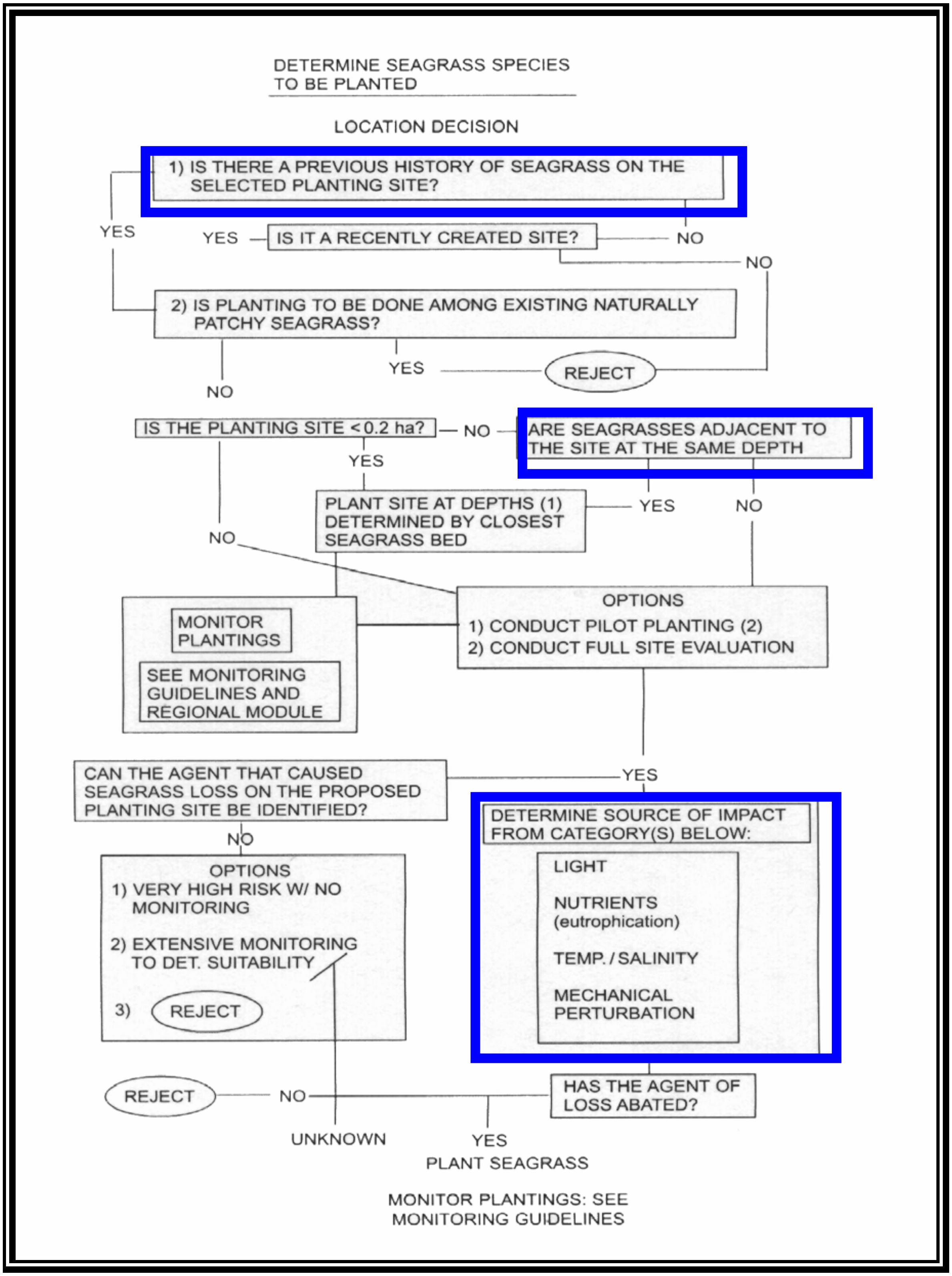
Table 2. Eelgrass Habitat Requirements: Factors Rated for Importance In Narragansett Bay

Factor	**Priority for Selection Model**
Social Use	High
Historic Distribution	Medium
Current Distribution	High
Bathymetry	High
Critical Depth	High
Water Quality	High
Salinity	low
Temperature	Medium

Light Attenuation	Very High
Nutrients	Low
Total N	High
Total P	Low
Dissolved Oxygen	Medium
CHl *a* + TSS	medium
Macroalgae-poorly flushed systems	High
Macroalgae-highly flushed systems	Medium
Humics/Color	?
Sediment Distribution	High
Wave Exposure/Tidal Energetics	Medium
Bioturbation	NA at regional scale
Proximity to natural beds	High
Plant Health	?

*This table is the result of a seagrass scientist technical team meeting that was held by Save The Bay and the NOAA Coastal Services Center. Ratings were generated by consensus opinions of a group of 15 estuarine scientists representing: NOAA, University of Rhode Island Graduate School of Oceanography, University of New Hampshire, U.S. EPA, RI Department of Environmental Management, Save The Bay, and U.S. Fish and Wildlife Service. The meeting was held at the Coastal Institute at the University of Rhode Island in Dec 5, 2000.

Table 3. Decision Support Diagnostic Model, displaying critical indicators of restoration success and suitability for eelgrass restoration (bold boxes), adapted from Fonseca *et al.* (1998).



I. Factors affecting Seagrass Restoration

A. Wave Exposure/Tidal Currents and Relative Exposure indexes:

Water energetics, which include wave exposure and tidal current velocity, have been established as an important factor influencing the distribution, density, and relative cover of *Z. marina* (Fonseca *et al*. 1998; Short *et al.* 2000, Kopp *et al.* 1995). Wave exposure is a measurement of a site's exposure to waves and their erosive forces. It is expressed as an unitless relative exposure index. The relative exposure index is calculated for a specific location by calculating the average monthly maximum wind speed, effective fetch, and the percent frequency wind occurs in a given direction. Tidal current velocity is a measure of water motion speed, and is usually measured as the peak free-stream speed at a specific location. Both metrics serve as a proxy for the potential disturbance to eelgrass plants. The erosive forces caused by water current speed and wave height have the potential to dislodge rooted plants. Research findings of the threshold limits of these variables are presented below .

Data presented in Fonseca *et al.* (1998) established relationships between seagrass coverage, relative exposure, and maximal tidal current speed. These measurements were conducted at 18 sites over a two year period in North Carolina for *Z. marina* and *Halodule wrightii*. Fonseca *et al*. (1998) recommends that restoration sites should have current velocities ranging from 15 to 50 cm/sec. Fonseca recommended that restoration sites be rejected when current velocities exceeded 50 cm/sec. However, Batiuk et al. (2001) have demonstrated in their technical synthesis paper that naturally occurring populations of *Z. marina* can survive tidal current velocities up to 180 cm/sec.

Relative exposure index value of 3 X 10^6 (unitless expression) was set as a site selection rejection limit. This is contrasted with a GIS site selection model reported by Kopp et al. (1995), using the same relative exposure model. In the case reported by Kopp et al. (1995) *Z. marina* presence occurred in areas of relative exposure ranging from 4-10 X 10^6 ,violating the relative exposure limits set by Fonseca. Fonseca *et al.* (1998) reported that relative exposure had a weak correlation with vegetative cover (r2=.45). A stronger relationship was reported when vegetative cover was correlated1 with monthly maximum tidal current speed (r2=.60). Fonseca appropriately cautions against the application of this limited data set to other regions and settings.

GIS models can be used to create high resolution relative exposure indices and have been used in spatial models by Kopp *et al.* (1995), Short *et al.* (2000), Batiuk *et al.* (2001). Intensive and expensive field data collection is necessary to accurately depict peak monthly tidal current speeds at a resolution reasonable for seagrass planning. Tidal current speeds have not been included as a model input in the site selection models reviewed in this paper. Although relative exposure Indices have been used, they do not account for subtidal geomorphological conditions and bathymetry which can affect wave and current energetics in a seagrass bed. This is one potential reason that researchers report *Z. marina* presence in sites that exceed the relative exposure tolerances (F. Short, personal communication). It is interesting to note that relative exposure indices continue to be used (Batuik et al. 2001 and Short et al. 2000) as a restoration constraint in GIS analyses of potential seagrass restoration sites, given the relatively weak correlation in the published literature.

There is a gap in our comprehension of how exposure and current speeds affect the survival of *Z. marina* and other species. No studies have examined whether there are regional differences for relative exposure and peak current velocity thresholds within seagrass species; nor has research been conducted to test whether sediment type will influence the ability of seagrass plants to withstand these energetic thresholds. As other factors are examined, I will show that wave energy is a model parameter with the potential to interact with other model inputs, such as light, epiphytic coverage, bioturbation, or sediment characteristics (i.e. sediment organic carbon content).

C. Water Quality (Light) Habitat Requirements

The primary cause of seagrass loss and decline is water quality degradation associated with the reduction of light availability (Batuik *et al*. 2001; Fonseca *et al.* 1998; Dennison *et al.* 1993). Water quality factors that affect seagrass habitats have been partitioned by seagrass researchers in a number of ways as they relate to the attenuation of downwelling light. Light habitat requirement thresholds are generally expressed by researchers as a percentage of total surface irradiance that reaches the plant surface. Most researchers measure light with photo-electric light sensors that measure Photosynthetic photon flux density, which constitute Photosynthetically Active Radiation (PAR) with light wavelengths between 400-700 nm (Short and Coles, 2001). Measurements of PAR are taken at different depths to calculate the diffuse light attenuation constant (K) for a particular location. Using the Beer-Lambert exponential decay function: $I_z = I_o e^{-kz}$ where I_z = Irradiance at Depth, I_o = Irradiance at Surface (just below water), e = natural log, k = extinction coefficient, and z = depth. Irradiance values are plotted by depth and the regression equation is calculated. The attenuation coefficient (Kd) is the absolute value of the slope of the line. Once kd is known it can be used to calculate the depth at which a certain percentage of surface irradiance would reach the depth at which seagrass is growing. For instance, when Kd=1, 10% of surface irradiance is transmitted through one meter of water.

Water column light attenuation results from a combination of TSS (total suspended solids), DOC (dissolved organic carbon), algal biomass, and water color. Concentrations of nitrogen and phosphorus (generally in dissolved inorganic forms), macroalgae abundance, and light attenuation correction due to tides have also been used as proxy for water column light attenuation factors (Dennison *et al.* 1993; Koch and Beer 1996; Batiuk e*t al.* 1992; Batiuk *et al.* 2001).

Light limitation as the main water quality factor impacting seagrass abundance and distribution seems to be well established. However, the struggle to develop a mechanistic understanding of light requirement thresholds on seagrass persistence and establishment continue to stimulate much debate among seagrass researchers today. Research on the minimum water clarity conditions for *Z. marina* at the canopy depth frequently suggest a threshold minimum of 22% of surface irradiance (Batiuk et al. 1992; Fonseca 1998; Batiuk 2001) This threshold has been challenged by seagrass researchers over the past five years. Seagrass restoration models in Chesapeake Bay (Batiuk et al. 1992), Narragansett Bay (Kopp et al. 1995), and Buzzards Bay (Short et al. 2000) have used bathymetry and the critical depth that corresponds to 20% surface irradiance to characterize potential *Z. marina* habitat. Batiuk et al. (2001) have refined this approach to account for epiphytic light attenuation and have adjusted minimum light requirement to 15% of surface irradiance. Other researchers have determined that requirements for *Z. marina* seedlings should be at least 47% of surface irradiance (Bintz and Nixon, 2001). Short (personal communication, 2000) recommends threshold irradiance of 50- 60% of surface irradiance.

Batuik *et al.* (2001) provide a novel approach for establishing a new light requirement criteria; percent light at leaf surface (PLL). PLL is calculated by means of an algorithm derived from empirical studies and mechanistic modelling to predict the total attenuation of light at the leaf surface. PLL includes the amount of light attenuated by epiphytes attached to the leaf surface. This information has been used to establish the minimum light requirement of different seagrass species. The minimum light requirement takes into consideration water column attenuation, attenuation due to tide level, and epiphytic light attenuation. However, Batiuk et al. (2001) assume that epiphytic biomass on seagrass leaves can be predicted by DIN (Dissolved inorganic nitrogen, DIP (Dissolved inorganic phosphorus), TSS, and K_d (the diffuse light attenuation coefficient). Tidal current speed and exposure terms are not included as independent variables. They acknowledge this as a potential source of error and future research need, because current speed and exposure disturbance are known to play a role on the assemblage of epiphyte communities (Horner *et al*. 1990). Therefore, use of wave energetics as a spatial model input in a habitat requirement model that includes epiphytic light attenuation should proceed with caution, given the chances of unquantified co-variance of parameters.

Limitations for any of these threshold light limit estimates need to be considered. For instance, the epiphytic light attenuation model developed for Chesapeake Bay should not be applied to other estuaries ‘off the shelf’ without new empirical studies, model calibration, and correspondence analysis (Neckles H., personal communication). Epiphytic communities are likely to vary regionally between estuaries, and spatially and temporally within estuaries. Estuaries with higher concentrations of epiphytic grazers (not accounted for in the Chesapeake model) could seriously weaken the statistical relationships established by Batiuk et al. (2001).

What is important for the decision support model user, is to understand the extent of uncertainty associated with spatial modelling of light penetration. Where high resolution spatially explicit light data is available, it is possible to model potential habitat restoration sites with many of the suggested minimum light attenuation scenarios. However, common constraints include: interpolating light attenuation data points from a few stations to representative regions of the larger estuary and potential model interactions.

Where light data is insufficient, threshold limits for nutrient concentrations, chlorophyll-*a*, and TSS that correspond to light attenuation have been proposed in Fonseca et al. (1998) and Batiuk et al. (2001). Dennison et *al.* (1993) established median threshold growing season concentrations of DIN and DIP, <.10 μM and <.67 μM, respectively, as seagrass habitat requirements for Chesapeake Bay. However, measures of Inorganic forms of N and P in an estuary can be problematic due to the potential rapid turnover of these constituents, especially during the growing season (Tomasko and Lapointe 1991). An alternative proxy proposed by many researchers for nutrient enrichment (and thus light attenuation) is concentrations of phytoplankton chlorophyll. Researchers have converged on sustained levels of Chlorophyll-*a* in the range of10-15 mg/liter, as indicative of nutrient enrichment and water quality degradation (Batuik et al. 1992; Valiela et al. 1990). Correspondence between chlorophyll-*a* concentrations and light attenuation vary by research effort. Research conducted in Narragansett Bay by Kopp et al. (1995) reported very high concentrations of Chlorophyll-*a* and low values of light attenuation. In another region of the same estuary chlorophyll concentrations corresponded reasonably well (r^2=60) with light attenuation of PAR (Granger et al. 2000). However, regression equations of light attenuation vs. chlorophyll-a concentrations have large intercept values, suggesting that the relationship is restricted to large algal blooms (S.Nixon, personal communication).

Again, parameter interactions have the potential to subsume parameter complexity and lead to inaccurate model output (violating criterion two). If light attenuation is caused by suspended solid constituents other than algal biomass there is potential that phytoplankton may be light limited and nutrient saturated (Batiuk et al. 2001; Mann 2000). These internal dynamics need to be considered by the model user when sifting through available data sets and their applicability to representing actual seagrass light requirements.

Abundance of macroalgae is considered an important factor influencing seagrass communities (Fonseca et al. 1998). However, time intensive surveys are necessary to quantify cover and abundance, and it is difficult to account for macroalgal species that drift with tidal currents. Nutrient enrichment can stimulate growth of macroalgae which compete with seagrass species for light and space and can overgrow seagrass beds (Short et al, 1993**).** Fonseca recommends that potential restoration sites with over 50% macroalgal cover be rejected. However, Fonseca et al. (1998) do not explicitly state when to make these measurements. Should they be made during the maximum duration of biomass present or a mean monthly maxima during the growing season? Researchers do not provide site selection model users with clear guidelines for incorporating macroalgae abundance or proxies into the decision support spatial model. This has significant implications if macroalgae are the leading disturbance to seagrass beds in a particular estuary. Macroalgal shading and smothering have been implicated in seagrass restoration transplant failures through out the Northeastern, U.S (Short et al. 2000). Therefore, criterion one (inadequate or missing process level understanding) needs to be considered a factor to evaluate the validity of site selection model that is applied to an estuary that contain ephemeral drift macroalgal communities, such as *Ulva lactuca*.

A problem experienced in Chesapeake Bay's seagrass restoration work that represents possible criterion three violation are issues with spatial variability of water quality data. Batiuk et al. (2001) use water quality data collected from many mid-channel stations to represent seagrass conditions near shore. Their findings report that station pairs less than 2 km from each other had habitat conditions that were indistinguishable 90% of the time. They did not report statistics for stations separated by greater distances.

The GIS spatial modeler, who in many cases may not be a seagrass scientist, should be expected to understand not only the quality of data used in model inputs (criterion three) but the statistical properties of the model inputs themselves. This point is especially important when considering light requirements. Batiuk et al. (2001), Short et al. (2000) and other researchers describe light attenuation as a model 'parameter.' Although it may appear to be an issue of semantics, light attenuation does not fit the operational definition of 'parameter' presented in this paper. Light attenuation is influenced by a complex suite of physical, chemical, and biological factors that vary in time and space. Light attenuation measures should be considered a variable, not a constant property of an estuarine water unit unless locations were classified categorically by light limitations during prime growing periods in which case light could be considered a "functional" parameter. Making this distinction between 'variable' and 'parameter' allows the spatial modeler to acknowledge up front issues that may arise with this model input.

D. Bathymetry/Critical Depth

Bathymetric data are used to predict potential habitat restoration locations by examining the relationships of depth with existing seagrass distribution. Depth is a necessary data source in any restoration planning initiative (Fonseca et al. 1998; Short et al. 2000). Critical depth is a metric that links current seagrass distribution with bathymetric data and corresponds to the depth thresholds of existing seagrass. The benefit of this metric is that it considers total plant and environment interactions and is measured by real plant responses without experimental error. One of the issues of using a critical depth threshold to model potential habitat with other factors such as light attenuation level is the fact that the deepest shoots of seagrass (critical depth) may depend on the resources extended to them via stolons from clonal plants located in the interior of the seagrass bed (Fonseca 1998). This is typical for plant communities dominated by clonal growth forms with an interconnected root system. Plants growing in more desirable locations are able to translocate photosynthetic energy via root system to plants growing in deeper or more stressful environments. Furthermore, critical depth is measured for existing and established seagrass beds and may not be representative of establishing seagrass beds (seedlings, or young transplants).

In most estuaries bathymetric data have been collected by the National Oceanic and Atmospheric Administration (NOAA). However, near shore areas where seagrasses are more likely to grow are often absent in these spatial datasets. Spatial model users need to be aware of scale issues when analyzing spatial bathymetric data and other coverages, such as seagrass distribution and light. Criterion three: data resolution and accuracy, needs to be considered in the use of this factor. If bathymetric data are produced at a different scale from mapped seagrass beds, it is very possible when overlaying these coverages to find seagrass occurrence in upland locations. Establishing frequency distribution graphs of seagrass distribution with depth of distribution is recommended by Short et al. (2000) to establish critical depth thresholds. It is very important that the resolution of both bathymetry and seagrass distribution maps are adequate to perform this analysis.

D. Sediment Characteristics

1. Sedimentation and Burial

No empirical data exist for *Z. marina* to indicate critical burial rates in the Northeast. Merkel (1992) suggests .3 mm/day for *Z. marina* on the West Coast of the U.S. High apparent sedimentation rates may be related to exposure and tidal currents but ephemeral and episodic events can cause acute sedimentation, affecting seagrass. In contrast to “new” material accumulating in the long term, much may be due to relatively short-term deposition and resuspenstion (S. Nixon, personal communication). This factor has not been used in seagrass restoration spatial models, reported in the literature due to a number of reasons. Measuring sediment burial rates is costly due to the time and logistics necessary to conduct surveys. Annual burial rates may be very different than maximum burial rates that may occur on shorter time periods and which are more likely to be important in influencing seagrass growth. Research is lacking in establishing maximum burial thresholds for seagrass species. This would pose a difficulty in selecting a meaningful statistic for sediment burial to be meaningful. Furthermore, spatially modelling burial rates would require a significant amount of empirical data collection to accurately model areas at the estuary scale. Thus, process level information is necessary to fill gaps in monitoring and assessment. Critical burial rates are important for seagrass restoration planning on a site by site basis, once locations have been identified using other criteria; however, given the cost of data acquisition, this factor may be a source of “noise” or uncertainty in most decision support models.

2. Sediment Surface Depth:

No empirical data exist relating seagrass distribution/cover with sediment surface depth. Z marina has shallow roots and is probably not be inhibited by shallow sediments (Fonseca, 1998). Sediment surface depths are measured by sediment cores and subaquaous soil mapping. Again, it would be impractical to attempt to model estuarine areas without accurate and fine resolution sediment boundaries. Spatial models would be limited by their ability to interpolate data collected by point locations into accurate spatial coverages due to the amount of sampling that would be required to map sediment surface depths. This factor may be more useful as ancillary information collected from the field to further refine a site selection model output. Future research and site selection model testing could potentially identify this factor as an important model input parameter.

3. Sediment Grain Size

Grain size characteristics of seagrass habitat requirement thresholds are discussed in the literature. Batiuk et al. (2001) do not consider grain size characteristics in their model to identify potential seagrass habitats. Short (2000) recommends threshold values as general guidelines due to the uncertainty of empirically derived relationships. Short recommends against sites with silt/clay content greater than 70%, while sites with rock/cobble substrates (no percent reported) should be rejected as a candidate for restoration. If spatial sediment data are available to the spatial modeler, issues of scale and data resolution need to be considered.

4. Sediment Biogeochemistry

Adequate mechanistic understanding of sediment porewater characteristics that may limit seagrass restoration requirements are not well understood, especially seagrass growth and sediment organic carbon content relationships. High levels of organic carbon in fine grain sediments have the potential to produce toxic anaerobic conditions that will limit the establishment of seagrass plants. High sulfide levels in sediments are known to be toxic to higher salinity seagrass species, i.e. *Z. marina* (Batiuk et al. 2001). Goodman et al. (1995) found that porewater sulfide levels exceeding 400μM reduce seagrass leaf photosynthesis. If sediment characteristic data on sulfide and organic carbon content do exist for a given estuary, the spatial modeler should be cautious about including these factors in the model if only point location data are all that is available. Interpolating points of biogeochemical characteristics to a larger surface area has the potential to severely affect model accuracy, as biogeochemical soil properties are especially difficult to model spatially (Addiscott and Wagenet 1985). Statistical based interpolation techniques such as kriging could be used to assess the accuracy of extrapolating point data. Sediment phytotoxins also have the potential to be an important factor that should be considered in a restoration site selection approach. However, future research is needed to quantify their importance as well as the interaction with other model-input criteria, such as nutrient enrichment.

E. Seagrass Distribution: Current/Historical

A review of seagrass restoration literature highlights the importance of having high-resolution distribution maps of current seagrass populations. It is intuitive that restoration practitioners should know current distribution prior to restoring. Historical evidence of seagrass distribution has been collected in a number of ways: historic aerial photographs, seed and plant burial evidence, oral interviews, herbaria specimens, and historic maps (Kopp et al 1995; Fonseca 1998). Issues of data resolution, and quality need to be accounted for by the modeler.

Seagrass beds are dynamic environments as evident from the factors presented. The NOAA Coastal Change Analysis Program offers guidelines for mapping seagrass species using remote sensing techniques. Site surveys conducted at one point in time may not reflect true current distribution. Fonseca (1998) recommends 10 years of aerial photography to determine an accurate distribution of seagrass in a particular location, but it is unlikely that most states have this level of spatial coverage. Furthermore, remote sensing techniques often miss the deep-water distribution of seagrass due to the limitation of photographic penetration into the water column (Cottrell et al. 2000).

Fonseca et al. (1998) and Short et al. (2001) suggest incorporating the distribution of existing seagrass populations as a model factor in order to select restoration locations that are distinct from sites that already support seagrass and which may ultimately colonize adjacent unvegetated locations if habitat conditions are suitable. Care should be taken when modelers develop frequency distribution relationships with depth or light, because deep-water seagrass beds may be missed during remote sensing surveys. Spatial modelers need to keep in mind the resolution and scale of seagrass distribution data when combining other spatial coverages.

E. Bioturbation

Bioturbation of seagrass plants is caused by animals that either directly or indirectly disturb seagrass shoots and seagrass root systems, causing plants to be damaged, dislodged, ingested, or exposed to erosive forces. Bioturbation has been documented as a factor that can negatively impact seagrass restoration projects (Orth 1975; Wigand and Churchill 1983; Davis and Short 1997).

One of the first rigorous experimental studies was conducted by Wigand and Churchill (1988). These researches examined ten species for possible predator activity on eelgrass seeds and seedlings. They determined that under experimental conditions *Ilyanassa obsoleta, Littorina littorea*, and *P. longicarpus* preyed on seedlings when alternative food sources were not available. Crustaceans damaged up to 93% of the seeds while P. longicarpus damaged 93% of seedlings. The size of the crustacean P. longicarpus was determined to be a factor in seedling damage. Individuals of P. longicarpus with 9-mm carapace lengths were more predaceous than the 7-mm size class. I. obsoleta was the most predaceous snail on seedlings, inflicting rasp-like wounds along seedling leaves and cotyledonary sheath. Fish species preyed on less than 5% seeds and seedlings. P.longicarpus seed consumption was 2% when exposed to eelgrass with an alternate food source present but increased to 19% with eelgrass alone. The researchers acknowledged that captive behavior effects and seasonal differences in feeding could have been a factor in the experiments.

Davis et al. (1998) present mesocosm and in situ results that establish a link with green crab density and decreased restoration transplant survival. Green crabs forage in the top few cm of the sediment surface and mechanically damage shoots. Having a well-established root system is especially crucial during the initial establishment of transplants or recently established seedlings. A well-developed root system can prevent sediment penetration by foraging and burrowing crab species (Valentine et al. 1994). Other organisms considered as potential *Z marina* bioturbators are listed in Short *et al.* (2001). No researchers have attempted to spatially model bioturbation due to the difficulty in quantifying bioturbators in time and space in a spatial context. Short et al.

(2000) use bioturbation as a model parameter as a last phase in their site selection model where site specific field data have been collected. Bioturbator abundance was measured at a subset of locations and thresholds were established to further reject restoration sites after an initial screening process with other factors.

Davis et al. (1998) present mesocosm and in situ results that establish a link with green crab density and decreased restoration transplant survival. In their research conducted under laboratory settings, greater green crab density (g> 7 crabs/ m^2) resulted in higher shoot damage than moderate crab density (4 crabs/m^2). Crab density thresholds were established by observing crab densities in the field. Field densities were obtained by placing two 1.25 m^2 quadrats at a transplant site and observed for 1 hour. The number of observations made was not reported. Laboratory findings showed that 39% of transplanted shoots were lost within 1 week with crab densities of 4 /m^2, though there was no evidence of shoot consumption.

Further studies need to be conducted to determine applicability of threshold limits by bioturbator species for use of this variable in estuaries conducting this site selection model. Batiuk et al. (2001) establish no bioturbation habitat requirements. Research conducted in Nova Scotia, Canada has implicated green crab bioturbation as a leading cause of natural eelgrass decline. This work remains unpublished but should be tested in other regions where eelgrass and green crab distribution overlap. It is apparent that bioturbation is an important variable to include in measuring potential restorability but it may require intensive field sampling to accurately characterized predator activity at a location. It seems appropriate that this factor be used once a regional model has been developed to initially select sites that can then be field checked.

II. Overview of a current site selection technique: The Transplant Suitability Index Model (Short et al. 2002)

The Transplant Suitability Index Model (TSI) is a recent modeling approach advanced by Short et al. (2001) that uses a GIS based decision support model to select the most suitable eelgrass restoration transplant locations. The TSI appears to be an excellent cost effective tool to model potential restoration sites. TSI model is presented in this section to review one of the only GIS decision support tools found in the published literature.

The TSI is a multiplicative index that rates estuarine environments for transplant suitability by modeling existing environmental data and incorporating field derived site specific data to calculate a final suitability index rating per spatial unit of estuary under consideration. Site selection criteria considered in this model include: historic eelgrass distribution, current eelgrass distribution, bathymetry, water quality (eutrophication index), **light**, sediment grain size, wave exposure, proximity to existing eelgrass beds,

bioturbation, and **eelgrass survivability (% survival, growth, and Leaf N)**. {Bold refers to field data collected during test transplants by Short et al. (2002)}.

GIS analyses are performed to provide a first order screening level analysis of preliminary transplant suitability (PTSI) by using existing data sets and recommended model inputs. Index values are then prescribed to model parameters with a rating system of 0-least favorable to 3-greatest restoration potential. Index values of all parameters are assigned for a representative cell size; and all selection criteria values for each cell area are multiplied together to provide a favorability rating for eelgrass transplant suitability. A problem with multiplying factors together is that results are somewhat artificially spread out (S.Nixon, personal communication). For instance, a site with a “1” value for each of 4 factors will only be 1/16 as good as a site with a “2” value for each variable. Once the PTSI is completed, results are output as maps and the highest ranked sites are selected for the active restoration component of the model. Test transplants are conducted at the highest rated locations and measurements are made to evaluate conditions of test transplants for one growing season. Test transplant results are then included as additional model parameters on a site by site basis with threshold values established and criteria rated on a favorability scale from 0 to 2.

Threshold Values for Model Inputs used in Short et al. (2002)

Model Input	PTSI Score	Factor Threshold
Historical Eelgrass Distribution	1	previously unvegetated
	2	peviously vegetated
Current Eelgrass Distribution	0	currently vegetated
	1	currently unvegetated
Proximity to Natural Eelgrass Beds	0	< 100m
	1	≥ 100m
Sediment	0	rock,cobble
	1	> 70% silt/clay
	2	cobble free <70% silt/clay
Wave Exposure	0	>mean + 2 SD
	1	< mean + 2 SD
Water Depth	0	too shallow, too deep
	1	shallow edge of reference bed
	2	average of reference bed
Water Quality phytoplankton DIN, TON, eutrophication index	0	poor
	1	fair
	2	good

The performance of these test sites are combined with field monitoring of light, bioturbators, and leaf N. Calculations of PTSI and the additional criteria are made for the subset of sites chosen in the initial screening to produce a final transplant suitability index (FTSI) that rates the most favorable test transplant sites that can be used for future full scale restoration.

This TSI model has the potential to increase the success rate of eelgrass transplanting in the Northeast, where previous restoration efforts have been challenged by the high costs and difficulties associated with the intensive field monitoring required to determine site suitability. In this way, more time is spent using test eelgrass transplants that integrate habitat suitability conditions of a particular site, rather than expending significant resources collecting site information prior to eelgrass transplanting. However, there are significant constraints to adequately implementing the PTSI in a spatial GIS model. Part II of this paper will present an adaptation of this approach to Narragansett Bay will compare the results of the TSI model with actual restoration results.

PART II. DEVELOPMENT AND TESTING OF A NARRAGANSETT BAY ELGRASS RESTORATION SITE SELECTION MODEL (NBERSSM)

A. PURPOSE & OBJECTIVES

Given the analysis presented in the first part of this paper, a modified approach to identify eelgrass restoration sites in Narragansett Bay is presented, following Short et al (2001) as a guide. Here, I provide the details on the development of this modified approach which is known as the “Narragansett Bay Eelgrass Restoration Site Selection Model.” The model builds on a decade of work by the University of Rhode Island, Save The Bay (STB), state and federal agencies, local schools, and community volunteers involved in planning, mapping, cultivating, and transplanting eelgrass in Narragansett Bay. The model is intended to overcome the inadequacies of current site selection procedures and promote successful eelgrass restoration. The need for this model is underscored by results from the Rhode Island Department of Environmental Management – Narragansett Bay Estuary Program (RIDEM-NBEP) and STB's 1996 Critical Habitat Mapping Study—that identify less than 100 acres of eelgrass remaining in Narragansett Bay. Work by the University of Rhode Island Graduate School of Oceanography has demonstrated the potential for eelgrass restoration throughout Bay waters.

Eelgrass restoration success rates utilizing a spatial model developed by Kopp et al. (1995) for Narragansett Bay resulted in 17% success rate (1 out of 6 site survival) while the TSI model resulted in 50% success rate (2 out of 4 site survival) in Buzzards Bay (B. Kopp, personal communication). Like the TSI model, the decision support tool I present differs from previous eelgrass site selection procedures by incorporating an active restoration component into the modelling process. The Narragansett Bay Eelgrass Restoration Site Selection Model (NBERSM) incorporates a Preliminary Transplant Suitability Index (PTSI) and results from actual eelgrass transplants to generate a final model output of full-scale eelgrass restoration potential among test locations.

The most relevant criteria for identifying potential eelgrass restoration locations is whether eelgrass can in fact survive at a particular location. Eelgrass persistence is the ultimate test of suitability as plants integrate all aspects of the physical, biological, and chemical attributes of a site. By using a rapid cost-effective transplant technique many eelgrass plants can be deployed at the previously screened sites in a short period of time. In this way, more time is spent using test eelgrass transplants that integrate habitat suitability conditions of a particular site; rather than expending significant resources collecting site information prior to full scale eelgrass transplanting.

By documenting the NBERSSM procedures and testing the model performance, I hope to improve the efficiency and efficacy of eelgrass restoration programs in Narragansett Bay. The results of these analyses will be organized as follows:

I. **Development of the NBERSSM**, which includes the methods and results of the PTSI model, the performance of eelgrass test sites (the active restoration component), and the results of the Final Transplant Suitability Index.

II. NBERSSM Model Evaluation, which presents the methods, results, conclusions, and discussion of how well the PTSI performed and an evaluation of select model input factors. Future directions of work are all presented .

I. Development of the Narragansett Bay Eelgrass Restoration Site Selection Model

A. Methodology:

Step 1. Seagrass Science Technical Team Input:

The first task in developing the NBSSERM model was to convene a science advisory team in order to identify and evaluate the most relevant environmental data to be used as habitat requirement factors in the initial PTSI screening model. After these model variables were prioritized for importance to Narragansett Bay eelgrass populations, this technical team identified and discussed the existing data resources that could be used to measure the most important habitat factors. The accuracy and availability of the data and the ability to spatially model the highest priority datasets were evaluated by the team.

Step 2. Contracting the necessary GIS expertise

In order to develop the GIS-based PTSI model, two GIS consultants were contracted to perform the necessary GIS analyses. Jeff Hollister and Michael Traber from URI-Environmental Data Center were contracted to obtain, interpolate, and grid the datasets recommended by the technical team and myself.

Step 3. Computing the PTSI model for Narragansett Bay

The Preliminary Transplant Suitability Index, the initial site screening model, was calculated by multiplying the selected eelgrass habitat factors together. The model (see schematic in Figure 1) uses these datasets and raster-based GIS functions to form a composite picture of potential eelgrass restoration sites within Narragansett Bay. Based upon the resolution of the model input data layers, micro-processor speed and performance of the anticipated model user's computers, a pixel size of 30m by 30m was determined to be an adequate resolution to use in the site selection models. The PTSI model was run for the geographic area that represented a total of 96,000 acres of Narragansett Bay in RI and MA. Based upon the prioritization, evaluation, and selection of model input factors by the technical advisory team and URI Environmental Data Center GIS expertise the following five habitat factors were included in the PTSI model:

1. **Current Eelgrass Distribution**
2. **Historic Eelgrass Distribution**
3. **Water Column Light**
4. **Critical Depth**
5. **Local Knowledge & Expert Testimony**

The PTSI model uses the aggregate of the model input factors that were selected for reasons previously stated and reported in detail below. Each component or factor ranks each pixel with a restoration suitability score consisting of three possible values:

0 – no potential for restoration
1 – moderate potential for restoration (neutral value)
2 – high potential for restoration

It should be noted that additional habitat factors were identified by the technical team and selected as model input variables to be included in the site specific Final Transplant Suitability Index. These additional factors included: light, bioturbation, temperature, and transplant survival; and were measured during the test site phase of the project at locations selected from the PTSI model output. The Final PTSI score is calculated by multiplying each PTSI factor score. These factors are presented in Step 9. I now present in detail the data sources used to represent the PTSI model input factors and the rationale for establishing eelgrass restoration suitability scores.

1. Model Input Factor: Current Eelgrass Distribution

Based upon recommendations presented in the first section of the paper as well as the availability of recently completed eelgrass mapping data, current eelgrass distribution was selected as an important model input factor. This model input factor was based upon a compilation of raw point and polygon distribution coverages that were later converted into a raster environment. Several datasets were used to estimate extent and location of current eelgrass beds in Narragansett Bay and presented in Table II.1. As suggested by Short *et al* (2002), each data-set was buffered by 100 m, rasterized (30 m pixels) and combined into a single raster data-set. Pixels that were within the 100 m buffer of existing eelgrass were given a value of 0 (no potential for eelgrass restoration). All other pixels were given a value of 1 (neutral potential for eelgrass restoration). The rationale for buffering existing eelgrass vegetation is to prevent transplant efforts from being conducted in areas that may be naturally recolonized. Thus areas immediately adjacent to existing eelgrass vegetation were precluded.

Table II.1. Current Eelgrass Datasets (reproduced here with permission. See http://www.edc.uri/edu/eelgrass for more information on available Narragansett Bay Eelgrass Datasets)

Data Layer	Description
Current CRMC Eelgrass Lines	**1999-2000 CRMC & RIDEM** shoreline eelgrass survey.
Current CRMC Eelgrass Points	**1999-2000 CRMC & RIDEM** shoreline eelgrass survey.
Current RIDEM/F&W Eelgrass Points	**1999-2000 RI DEM Department of Fish & Wildlife** eelgrass points taken from field observations in Narragansett Bay. Not geographically referenced; extents are only approximations.
Current NBEP & Save The Bay Eelgrass Polygons	**1996 Narragansett Bay Estuary Program** polygon data from 1996 true color aerial photography delineation. Synonymous with RIGIS data.
Current NBEP Eelgrass Lines	**1996 Narragansett Bay Estuary Program** line data from 1996 true color aerial photography delineation. Synonymous with RIGIS data.

2. Model Input Factor: Historic Eelgrass Distribution

Historic eelgrass distribution and mapping data was also selected as a value added habitat factor and included in the PTSI model. Again, this model factor was included for reasons previously discussed as well as the availability of existing historic eelgrass spatial coverages, as reported by Kopp et al. (1998). Two datasets were used to estimate the historic distribution of eelgrass (Table 2). Since the point data set had no extent information, it was assumed that a 100m buffer would provide some extent information without overestimating the area. The buffered point data set was merged with the polygon data set and rasterized. Areas that had a historic record of eelgrass were given a value of 2 (high potential for eelgrass restoration). All other areas were given a value of 1 (neutral potential for eelgrass restoration). The rationale for keeping this habitat factor as a value-added score was two fold. Kopp et al. (1998) acknowledge their reconstruction of historic eelgrass distribution represents an approximation of historic extent and some areas of the Bay may have been under-represented due to lack of historic records. Secondly, subsequent changes in subtidal benthic conditions of a site that historically supported eelgrass, such as changes due to sediment loss and erosion, may make that a historic location inappropriate to restoration. Changes over time may have allowed other subtidal locations to be appropriate for eelgrass colonization even if no historic evidence exists that eelgrass was supported in the location. Therefore, It was decided that sites without historic eelgrass distribution evidence should not be rejected; thus a neutral value is assigned.

Table II.2. Historic Eelgrass Distribution (reproduced here with permission. See http://www.edc.uri/edu/eelgrass for more information on available Narragansett Bay Eelgrass Datasets)

Data Layer	Description
Historical RIAF Eelgrass Points	**1848-1994 Rhode Island Aqua Fund** historical point data. Collected from personal accounts (P), herbarium specimens (H), literature review (L), or NOAA charts (C).
Historical RIAF Eelgrass Polygons	**1848-1994 Rhode Island Aqua Fund** historical polygon data. Collected from personal accounts (P), herbarium specimens (H), literature review (L), or

	NOAA charts (C).

3. Model Input Factor: Light

The technical team meeting results, the site selection recommendations reported in the literature, and the availability of current data sources, were reasons I included light as an important model input factor. Three sources of light data were used in the Preliminary Transplant Suitability Index (PTSI), see table 3. The primary source of light data was collected by Oviatt *et al.* between 1997 and 1998 (in press). Data on light extinction was collected bi- weekly at 16 stations located in Narragansett Bay and the lower Providence River. A second set of light extinction data was collected in 1994 by Kopp *et al*. This data set includes one comprehensive survey of 50 stations through out Narragansett Bay, the Providence River, Greenwich Bay, and The Sakonnet River (1995). The third set of light data was a time series taken within Greenwhich Bay and it's associated coves between 1996-1997 (Granger et al. 1998). Only data from May to September, the peak growth months for eelgrass in Narragansett Bay, was used from each time series. Bathymetry data was from NOAA.

The light extinction coefficient (k) at each station and date was calculated and then a grid for each date was interpolated using the Inverse Distance Weighting (IDW) algorithm. The maximum extent of each interpolation was dictated by the station locations. Using Formula 1. we calculated area of the bottom that received 20% and

Formula 1: $I_z/I_o=e^{-kz}$

I_z = Incident Light at Depth

I_o = Incident Light at Surface

e = log

k = extinction coefficient

z = depth

50% light for each date. Areas that received less than 20% light were given a restoration suitability score of 0, areas that received at least 20% light were given a value of 1 and areas that received at least 50% light were given a value of 2. Finally, all the dates were multiplied together, using the map calculator function in ArcView, yielding a map depicting the number of surveys at which each area received a certain percentage of light. A high value indicated that the area received 50% light for a majority of the survey dates while a value of zero indicated that the site received less than 20% light during at least one survey. A final grid for each light data set was produced where a value of 0 was given to all areas that received less than 20% light, a value of 1 was given to areas that received at least 20% light or 50% light for less than ½ the surveys, and a value of 2 was given to all areas that received 50% light for more than ½ the surveys.

Table II.3: Light Data

Data Set	Location	Number of surveys/year	Number of Survey Stations
Oviatt et al, 2001	Narragansett Bay and Lower Providence River	9 1997-1998	16
Granger et al, 1998	Greenwich Bay and associated coves	5 1996	5
Kopp et al, 1995	Narragansett Bay, Providence River, Sakonnet River	1 1994	50

The three final light grids were combined to create a single estimate of light availability. The datasets were combined in the following manner: If a cell had a value of 2 (high potential for eelgrass restoration) in any of the three datasets, then the combined data set cell was given a value of two. Cells that were classified as a one in any of the three datasets and were not already classified as a 2 were given a value of 1 (moderate potential for eelgrass restoration). Remaining cells were classified as 0 (no potential for eelgrass restoration).

4. Model Input Factor: Critical Depth

Bay-wide statistics for eelgrass depths were calculated using current eelgrass polygons and bathymetry, derived from NOS 500,000 soundings, of Rhode Island waters, collected over several decades. Arcview was used to sample the bathymetry grid with current distribution of eelgrass polygons and to calculate descriptive statistics for polygon depth, depicted in the frequency distribution histogram shown in Figure II.1.

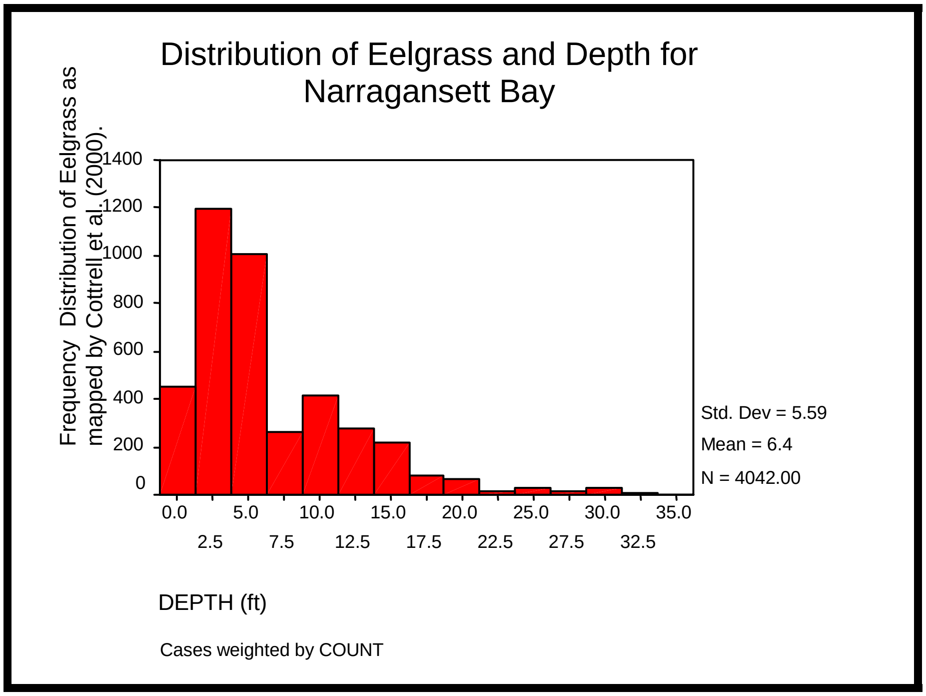


Figure II.1 Frequency distribution histogram, calculated in Arcview and exported to SPSS 8.0

Utilizing the frequency distribution histogram as a guide, depths were categorized into three classes with values of 0 (no potential for eelgrass restoration), 1 (suitable for eelgrass restoration), or 2 (high potential for eelgrass restoration). The 0 class was all depths greater than 12 feet (> Mean + 1 S.D.). Values from 0 to 1 (< Mean - 1 S.D.) and from 6.4 to 12 (Mean + 1 S.D.) were re-classed into the 1 class. Values from 1m up to 6.4m (Mean – 1 S.D to mean) were deemed to be most important for eelgrass (i.e. shallow enough to allow for proper light conditions and deep enough to mitigate for ice shear and/or desiccation) and classified to a value of 2.

5. Model Input Factor: Local Knowledge, Site Visits, and Expert Testimony

The final model input factor used in the PTSI model was the Expert Testimony factor. A binary grid was created that represented additional information obtained from knowledge of the area and from site visits. If an area was known to have poor conditions (i.e., macroalgae, persistent water quality problems, inappropriate sediment types, use conflicts, etc.) it was given a value of 0. All other areas were given a value of 1. This particular data set, expert_input, is a reclassed grid of areas which are known, either through local knowledge or specific site visits, to have conditions that are unfavorable for eelgrass. This data was obtained during a technical team meeting held in April 2001 with 10 technical experts from RIDEM, RI Ocean State Fisherman Assoc., URI, Save The Bay, and Brown University. Polygons of unsuitable areas were hand drawn during the meeting on paper maps and later transferred digitally into ARCVIEW.. These are areas that would not have been identified as having poor restoration qualities by the other data sets used in the development of the PTSI model. Because of the constraints of depicting poor water quality conditions as discussed in Part I, the only way of incorporating high macroalgal abundance was to digitize areas (coves and other poorly flushed water bodies) based upon first hand local and expert knowledge. In the final PTSI model, this polygon data set is used to eliminate these well known poor quality areas from consideration for restoration. Cells were assigned a value of 1 to represent no known problems. Cells with a value of 0 had one or more of the following problems: abundant bioturbators, poor water quality, macro-algae conflicts, prior failed restoration attempts. The leading factor for site elimination was known drift algae locations. Currently, no surveys have ever been conducted in Narragansett Bay to map macro-algal communities. The difficulty in mapping ephemeral drift algae populations was discussed in earlier sections of this paper, although it remains a very important factor contributing to restoration site failure. Although the expert testimony model input factor is not based upon published data, it was the only information source available based upon observations of field seasoned professionals who knew the Bay extremely well via SCUBA diving, commercial fishing, and conducting estuarine research. All sites excluded by this habitat factor were field verified to confirm the presence of water quality impacts.

Step 4. Calculating the PTSI Model

The PTSI was calculated by multiplying each factor within a cell, in the same way reported by Short et al. (2001). A zero in any of the selected habitat factor datasets would remove that pixel from future consideration. The PTSI ranged from 0 to 8 with 0, 1, 2, 4, and 8 as possible scores. Only pixels with scores of 8, 4, and 2 were considered for test transplants.

PTSI INDEX

0 – unsuitable for eelgrass restoration
1 - very low suitability for eelgrass restoration
2 - low suitability for eelgrass restoration
4 – moderate suitability for eelgrass restoration
8 - high suitability for eelgrass restoration

Total Area by PTSI Class

PTSI-Score	# of Test Sites	Area (acres)
Unsuitable PTSI-0	0	100,032
Low suitability PTSI-2	4	40,252
Moderate suitability-PTSI-4	6	3,507
High suitability-PTSI-8	9	907

Step 5. Selection of Eelgrass Test Transplant Sites

The PTSI model effectively narrowed down a 96,000 acre estuary to a subset of locations with high restoration potential. Of these PTSI scored sites, representing 44,600 acres, 30 sites were chosen for possible test transplants. Test transplant locations were chosen to equally represent the three major geographic regions of Narragansett Bay: East Passage, West Passage, and Sakonnet Passage (and Mount Hope Bay). A subset of twenty locations were evaluated by a team of technical advisors during May 2001. The amount of test sites chosen was based solely upon the logistics and resources available to carry out restoration within a 1 week time period. This was done in order to remove the possible effects of differential growth and problems staggering transplants over a long time period. There are negative impacts that are known to affect eelgrass plants when installing transplants later in the growing season, especially when algal production and water temperature might be more detrimental to recently transplanted plots vs. plots that had already gone through initial transplant shock. These sites were further screened for test transplanting based on the following factors: potential use conflicts, possible eelgrass habitat area (determined by the PTSI model), sediment type, depth, and macroalgal abundance.

Step 6. Obtaining Necessary Regulatory Requirements

The Coastal Resources Management Council (CRMC) requires that all proposed human activities that occur that have the potential to change or modify existing RI coastal resources are subject to Coastal Zone Management Act regulations. For this reason, a Category A assent was required by the CRMC in order for the test transplants to be implemented. Letters of support from eleven of the cities and towns that encompassed the locations of proposed transplant locations were necessary in order for CRMC to issue a permit. After this was accomplished a permit was issued by CRMC, allowing the test phase to be conducted.

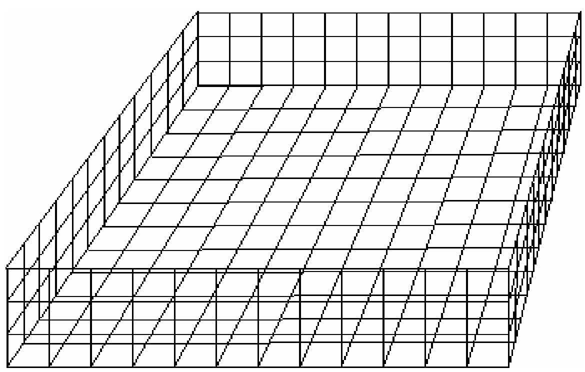
Step 7. Conducting Eelgrass Test Transplants

Test transplants were deployed using the TERFS^{tm} (Transplanting Eelgrass Remotely with Frames) method, based on techniques described by Short and Coles (2002) and presented in Figure II.2. TERFS^{tm} consist of a 0.25m^2 frame containing 100 plants per frame. Test transplants conducted for 2001 were planted with 5 TERFS^{tm} that occupied a total vegetated area of 1.25 m^2 . The test transplant shoot density of 100 shoots/0.25 m^2 or 400 shoots/m2 is equivalent to natural locations in Narragansett Bay. During the Spring 2002 test transplants, the number of frames were reduced from five to four, reducing the number replicates per site and the amount plants per test site to 400 plants.

To create a TERF, vegetative shoots were harvested from existing beds in Newport and Jamestown, RI. All reproductive shoots were removed and plants were sorted by volunteers into bundles of fifty plants. Volunteer divers were used to collect eelgrass plants by harvesting small palm sized clods of eelgrass shoot and intact rhizome material. Each collection was spaced apart at least 1 meter from each other in order to minimize disturbance. Shoots were bundled, 50 at a time, and placed in iced coolers for transport. Shoots were stored in flowing seawater tanks or submerged in situ in lobster cars until their deployment at restoration locales, up to 72 hours.

The TERF^{tm} restoration technique relies on weighted coated wire frames, modified lobster pots, which are used to anchor attached eelgrass plants to the bottom of the seabed. Each frame (outside diameter) is 0.36 m^2, shaped like a box, having two bricks attached outside of two of the parallel sides. One hundred harvested vegetative shoots are attached in pairs to the lower side of the wire intersections using a dissolvable thread. This process is accomplished while the frame is suspended over seawater so as to minimize desiccation. After lowering the frame into the water at the transplant site, the weight of the frame forces the eelgrass rhizomes into the sediment, allowing the plants to secure a foothold over a three to five week period. After the securing period, the threads will have dissolved away and the frames are recovered for later use. 19 out of the proposed 20 test sites were established in June 2001. The 20th site was not established due to a delay in receiving permission at the proposed test site in Jamestown, RI.

Figure II.2. TERFS^{tm} frame



Outside Frame Diameter: W-61 by L-61 by H-15 cm

Step 8. Performance monitoring of test sites

Habitat requirement factors that were previously identified by the science advisory team were monitored at each of the test transplant locations. All test transplant sites were monitored for the following conditions: light data (% surface irradiance by LICOR 4Pi Sensor), temperature, relative abundance of bioturbators, macroalgal abundance, epiphytic cover, wasting disease, and salinity. Test sites were monitored throughout the first season's growth. Survivability of eelgrass shoots (eelgrass shoots observed at end of one growing season and at the end of one year/total planted) were determined for all TERF^{tm} units deployed at the end of the first the growing season, and the end of 2 growing seasons for the 2001 test transplants. Bioturbator surveys were conducted visually through out the growing season by counting total bioturbating organisms observed immediately adjacent to or in eelgrass test plots. Relative abundance of macroalgae and epiphyte growth was quantified at least twice per growing season using Braun-Blanquette percent cover scale. Relative abundance of macroalgae and epiphyte growth are not presented in this analysis due because sampling did not occur at all the test locations. Furthermore, lack of time to organize, evaluate, and interpret the existing data prevented inclusion in this paper. All in situ surveys, as described above, were conducted underwater via SCUBA.

Step 9. Incorporation of Test Site Performance into the final model output

The survivorship of eelgrass plants and the monitoring data obtained over one season's growth were used to determine the Final TSI model output. Once monitoring data was entered into spreadsheets, the following variables were added to the TSI model as model factors to calculate the final TSI scores of the test site locations: Temperature, Bioturbator Abundance, Survival Percentage, Light, and PTSI Score. Each of these variables was reclassified to a 0,2, or 1, based upon threshold limits determined by science advisors and best professional judgement and then multiplied together to calculate the final TSI. Performance monitoring data is summarized in the results section of the paper and restoration suitability score threshold values for each metric are presented with the rationale used to separate suitability values.

Light, Bioturbation, Temperature, and Test Site Survival data were re-categorized into four additional model input factors for inclusion in the final screening to rank the 19 sites in the Final Transplant Suitability Index. The following section presents how the data collected for each performance factor were reclassified categorically.

Threshold Values used in the NBERSSM FTSI Model

Model Input	PTSI Score	Factor Threshold
Historical Eelgrass Distribution	1	no historic record of occurance (100m)
	2	historic record of occurance (100m)
Current Eelgrass Distribution	0	currently vegetated w/in 100m
	1	currently unvegetated w/in 100m
Critical Depth	0	> mean + 1 SD (>12 ft)
	1	< mean - 1 SD & mean +SD (0-1 ft & 6.4-12 ft)
	2	mean - 1 SD to mean (1-6.4 ft)
Water Quality: Light	0	< 20% Light
	1	at least 20% Light or 50% Light for less than 1/2 of surveys
	2	50% light for more than 1/2 of surveys
Expert Testimony	0	known poor habitat quality (macroalgae, water qulity impacts)

a. Light TSI

Light measurements were taken at most stations at least 1/month throughout the summer using a LI-COR 2 Pi quantum light sensor. A value kd was calculated for each survey of light, using methods described in <ins>*Section 1.B Water Quality Light*</ins>. Kd was used to estimate the depth at which 20% and 50% light would reach the bottom (see methods for formula 1.). These depth values were compared to the station depths to calculate the number of sampling times each light level would reach the bottom. It is important that eelgrass (*Zostera marina*) receive enough light for photosynthesis. Duarte (1991) and Dennison (1993) have shown that eelgrass requires a minimum of 20% of the light at the surface of the water to reach its blades for photosynthesis and growth. Short et. al.(1995) have found that > 50% surface irradiance will increase its growth rate, while 47% surface irradiance was recommended by Bintz and Nixon (2001). We took a conservative approach and used 50% surface irradiance as the threshold for the highest TSI score.

Light TSI Index:

2 = The station received 50% light or greater at the bottom for more than 50% of the surveys

1 = The station received at least 20% light at the bottom

0 = The station received less than 20% light at the bottom during at least one survey.

b. Bioturbation TSI

Bioturbators were measured by summing the total abundance of crabs for all observations per site as displayed in Chart II. Based upon the limited literature references and personal communication with Dr. Stan Cobb, we determined that swimming crab bioturbators had the potential to be more destructive to eelgrass plants than non-swimming crabs. Therefore, the threshold limits established for rating the impact of bioturbator activity was based upon the type of bioturbator species and their abundance. If No Swimming Crabs Present- Blue Crab *Callinectes sapidus* or Lady *Crab Ovalipes ocellatus* the following thresholds were used to rank sites and assign a TSI score

Non Swimming Crab Bioturbator Index
> 25 crabs =TSI score 0
1-25 crabs= TSI score 1
0 crabs = TSI score 2

If Swimming Crabs were Present- Blue Crab *Callinectes sapidus* or Lady Crab *Ovalipes ocellatus,* The following thresholds were used to rank sites and assign a TSI scored:

Swimming Crab Bioturbator Index
>10 crabs = TSI score 0
1-9 crabs = TSI score 1
0 crabs = TSI score 2

c. Test Site Survival TSI

Test site survival was determined after one season's growth, as presented in Table II.7. All shoots were counted for each TERF frame. The total number of shoots from all replicate TERFS (five or four) were divided by the total plants transplanted to calculate total site survival after one growing season. Based upon Fonseca et al. (1998) it was determined that greater than 50% survival of initial plantings should be considered very successful given the difficulties of establishing new plants. Sand-Jensen (1994) identified a minimum patch size of 32 shoots (surviving over two years) in his study to evaluate the effect of patch size on eelgrass survival. For this reason, we used 50% survival per site which would represent on average at least 50 eelgrass shoots surviving per TERF plot. It is common that even under favorable conditions initial plant losses will occur due to transplant shock. Furthermore, there is high probability of bed expansion if plants are sited in areas with suitable environmental conditions.

TSI Shoot Survival Index was classified as follows:

TSI 0 = survival from 0 up to 25%
TSI 1= survival from 25% to 50%
TSI 2= survival greater than 50%

d. Temperature TSI

The threshold temperature of when eelgrass is known to go into heat stress and senesce is reported in the literature to be 25 C (Evans et al. 1986). TSI Score is based upon the amount of temperature observations (sampling days) where the 25 degree Celsius exceedence threshold is violated. Temperature was measured using a YSI, Incorporated Model 85 Handheld Oxygen, Conductivity, Salinity, and Temperature system.

Temperature TSI Index was classified as follows using Table II.9:

TSI 0= when a site exceeds 25 C at least 2 sampling dates
TSI 1= site does not exceed 25 C for more than one sampling date

a. PTSI TSI

Based upon Short et al. (2001), the PTSI value originally calculated for the test site location was used as a model input factor to further rank the nineteen locations.

PTSI Scores were reclassified:
PTSI 0 or 1= TSI 0
PTSI 2 = TSI 1
PTSI 4 or 8= TSI 2

b. Final Transplant Suitability Index (FTSI)

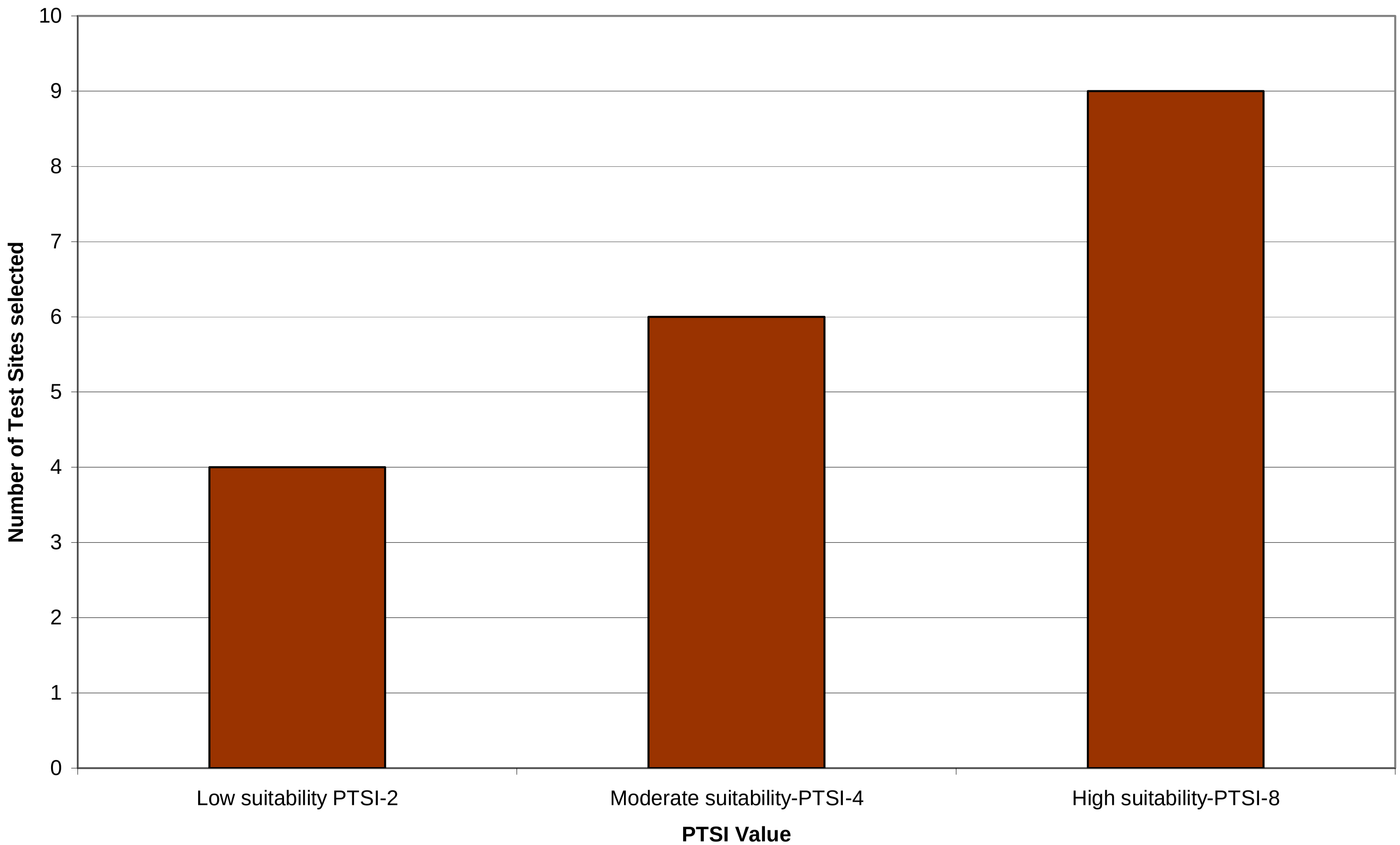
The final TSI was calculated for the nineteen locations to provide the model user with a further ranking of the nineteen test site locations.

Final TSI Score/test site location =
light_TSI * temp_TSI * bioturbator_TSI * survival_TSI * PTSI_TSI

C.RESULTS

Results of the Narragansett Bay Eelgrass Restoration Site Selection Model will be presented in multiple sections in order to organize the variety of different data sets, GIS output maps, tables, and other results in a discernable way. Results are first presented on the initial site screening PTSI model output, which include summary statistics derived from GIS calculations as well as GIS map outputs separated by geographic areas of Narragansett Bay. Restoration test site performance results will be presented from the monitoring of all 19 locations during the first growing season. Furthermore, the results of reclassifying the site performance factors into additional model input factors to be used in the final site selection model will also be presented. The final Transplant Suitability Index, calculated for the 19 test locations, is presented in the form of summary statistics and GIS map outputs.

II. Chart II.1 PTSI rank of the Eelgrass Test Sites Selected



III. Test Site Performance Results

a. Light Conditions

Table II.4 Summary of Light Conditions for 18 Test Sites

Station	PTSI Score	Mean Irriadiance*	Irradiance Std	n	Mean Kd**	Kd Std	Depth, meters***
ep1	4	0.51	0.10	2	-0.67	0.19	-1.0
ep2	8	0.72	0.02	4	-0.67	0.05	-2.0
ep3	4	0.83	0.01	2	-0.62	0.05	-1.0
sk1	2	0.36		1	-1.03	0.00	-1.0
sk2	8	0.63	0.06	2	-0.76	0.15	-2.0
sk3	4	0.88	0.04	4	-0.45	0.15	-1.0
sk4	8	0.63	0.09	5	-0.47	0.15	-2.0
sk5	4	0.68	0.11	2	-0.39	0.16	-1.0
wp1	8	0.36	0.08	5	-1.04	0.25	-1.0
wp10	2	0.36	0.08	2	-0.69	0.14	-1.5
wp2	2	0.22	0.05	4	-1.01	0.15	-1.5
wp3	4	0.58	0.11	2	-0.56	0.19	-2.0
wp4	8	0.47	0.05	3	-0.76	0.11	-1.0
wp5	8	0.54	0.01	2	-0.61	0.02	-1.8
wp6	4	0.28	0.12	2	-0.89	0.29	-1.5
wp7	2	0.55	0.10	4	-0.60	0.19	-1.5
wp8	8	0.66	0.07	7	-0.70	0.18	-0.8
wp9	8	0.48	0.04	2	-0.53	0.06	-2.0

* calculated for each sample date from $Io/Iz=e^{-(kd*depth)}$

** calculated as the mean Kd determined for each survey

*** calculated in ARCVIEW by sampling NOS soundings

Table II.6 Bioturbators Found at all Sites 2001*

Station #	PTSI Score	Date(s) 2001	Total visits n	Survey 1	Survey 2	Survey 3	Bioturbator total
ep1	4	7/18/, 8/18	2	9 green crabs	0	NA	9
ep2	8	7/6, 7/10. 7/16	3	0	0	0	0
ep3	4	7/18/	3	50 green crabs	0	0	50
ep4	8	7/20/, 8/19	2	1 unkown	0	NA	1
sk1	2	7/20, 8/19	2	0	2 Blue	NA	2
sk2	8	7/6. 7/20, 8/19	3	0	17 green	0	17
sk3	4	7/6. 8/19	2	3 ladies	3 blue, spider. Lady	NA	6
sk4	8	7/6, 7/20, 8/19	3	3	0	4	7
sk5	4	7/20, 8/19	2	4 spider, green, hermit	1horseshoe	NA	5
wp1	8	7/18, 8/22	2	6 (3) blue, hermit. Lady, spider	0	NA	6
wp2	2	7/19, 8/1, 8/18	3	3 lady, (2)spider	8 spider	5 dead spider	16
wp3	4	7/9, 8/18	2	5 (2) blue, unk	0	NA	5
wp4	8	7/24, 8/22	2	0	4 (3) spider, green	NA	4
wp5	8	7/12, 8/18	2	3 (2) spider, green	5 spiders	NA	8
wp6	4	7/12, 7/24, 2/27, 8/18	4	0	0	0	0
wp7	2	7/12, 8/18	2	6 Blue	1 Blue	NA	7
wp8	8	7/9, 7/18, 8/18	3	3 unk	1 spider	NA	4
wp9	8	7/18, 8/18	2	8 (7) green, spider	2 hermit	NA	10
wp10	2	8/1, 8/18	2	1blue	0	NA	1

*** Bioturbators sampled by counting all crabs occuring in or adjacent to test sites. Surveys performed using SCUBA**

Mean abundance of crabs from all sites = 8.3 crabs
Mean Number of Visits/Site= 2.4 site observations

c. Site Survival

Table II.6 Site Survival Summary Statistics

Station ID	PTSI Score	n	Mean Survival 1 season(SD)	Survival-1 season Std	Mean Survival-1 year (SD)
EP1*	4	5	7.4	3.4	0
EP2	8	5	0	0.0	0
EP3	4	5	0	0.0	0
EP4	8	5	0.2	0.4	0
SK1	2	5	1.2	1.8	0
SK2	8	5	2	3.1	0
SK3	4	5	74.4	17.2	115
SK4	8	5	54.6	25.9	17
SK5	4	4	5.5	7.1	0
WP1	8	5	15.4	14.5	0
WP10	2	5	1.8	2.7	0
WP2	2	5	3	6.2	0
WP3	4	5	6.4	7.1	0
WP4	8	5	82.4	9.1	110
WP5	8	5	83.8	45.6	114
WP7	2	5	29.6	15.4	13
WP8	8	5	49.8	12.7	1.2
WP9	8	5	61.4	15.1	81
Total Survival (all sites/1 season)*					**27.7**
Total Survival (all sites/1 year)*					**26.5**
* site failure was due to human error-transplanting in intertidal zone EP1 is excluded from summary survival statistics					

Chart II.2 Test Site Survival 2001 & 2002

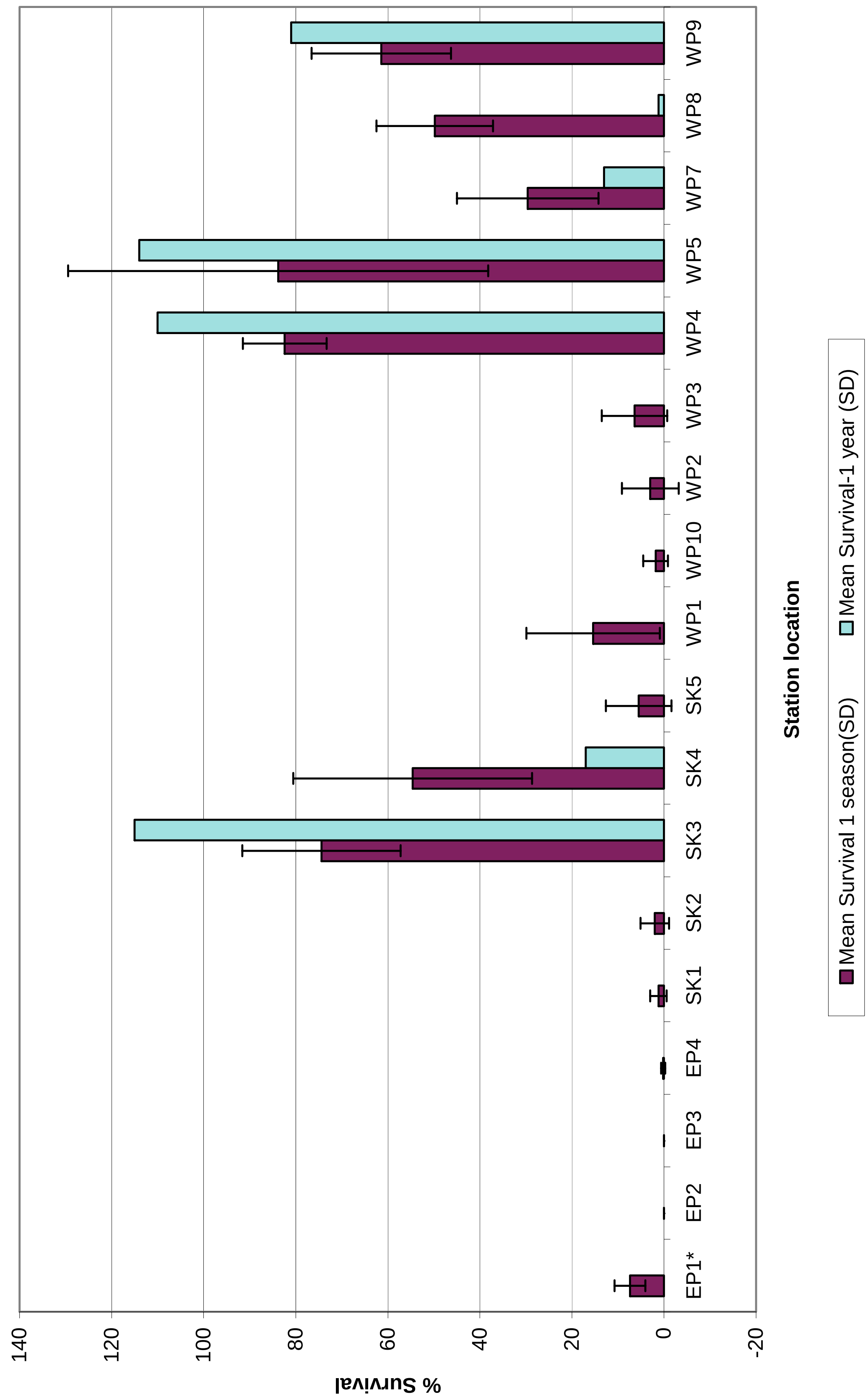


Table II.7 Test Site Temperature Summary

Station #	PTSI Score	Station description	date	temp(^oC)	depth (m)
ep1	4	Potters Cove	7/18/2001	21.7	0.2
ep1	4	Potters Cove	8/7/2001	24.2	2.4
ep1	4	Potters Cove	9/19/2001	21.0	0.2
ep2	8	Bristol Harbour	8/1/2001	27.4	3.3
ep2	8	Bristol Harbour	8/29/2001	24.5	5.5
ep2	8	Bristol Harbour	9/19/2001	20.5	2.5
ep2	8	Bristol Harbour	9/19/2001	21.2	0.2
ep2	8	Bristol Harbour	10/3/2001	19.5	0.2
ep2	8	Bristol Harbour	10/3/2001	19.4	1.2
ep2	8	Bristol Harbour	10/10/2001	15.0	0.2
ep2	8	Bristol Harbour	10/19/2001	15.0	2.1
ep3	4	Hog Island	7/10/2001	23.8	0.5
ep3	4	Hog Island	7/10/2001	23.8	1.0
ep3	4	Hog Island	7/17/2001	22.4	?
ep3	4	Hog Island	8/7/2001	24.7	1.5
ep4	8	Kickimuit	7/20/2001	24.4	?
ref1		Cornelius Island	7/27/2001	21.1	1.5
ref1		Cornelius Island	8/15/2001	23.9	1.2
ref1		Cornelius Island	9/9/2001	23.1	0.1
ref1		Cornelius Island	9/9/2001	23.0	1.5
ref1		Cornelius Island	9/26/2001	20.9	0.2
ref1		Cornelius Island	9/26/2001	21.3	1.2
ref1		Cornelius Island	11/1/2001	12.2	1.7
ref1		Cornelius Island	11/1/2001	12.3	0.2
sk1	2	Nannaquaket Pond	7/20/2001	22.9	
sk1	2	Nannaquaket Pond	8/29/2001	24.7	1.9
sk2	8	The Cove	7/6/2001	24.3	?
sk2	8	The Cove	7/20/2001	21.0	?
sk2	8	The Cove	8/29/2001	26.0	1.2
sk3	4	Fogland North	7/6/2001	23.5	?
sk3	4	Fogland North	8/29/2001	24.8	0.9
sk3	4	Fogland North	9/19/2001	21.6	0.2
sk3	4	Fogland North	9/19/2001	21.5	0.8
sk3	4	Fogland North	10/3/2001	18.1	0.8
sk3	4	Fogland North	10/3/2001	17.9	0.2
sk3	4	Fogland North	10/19/2001	14.8	0.0
sk3	4	Fogland North	10/19/2001	14.8	0.8
sk3	4	Fogland North	10/19/2001	15.5	0.2
sk3	4	Fogland North	10/19/2001	15.5	1.5
sk3	4	Fogland North	11/1/2001	14.1	0.2
sk4	8	Fogland South	7/6/2001	22.6	0.5
sk4	8	Fogland South	7/6/2001	22.6	1.0
sk4	8	Fogland South	7/6/2001	22.6	1.5

sk4	8	Fogland South	7/6/2001	22.6	2.0
sk4	8	Fogland South	7/6/2001	22.6	2.5
sk4	8	Fogland South	7/6/2001	22.6	3.0
sk4	8	Fogland South	8/10/2001	26.2	2.4
sk4	8	Fogland South	8/29/2001	24.3	1.3
sk4	8	Fogland South	9/19/2001	21.0	0.2
sk4	8	Fogland South	9/19/2001	20.7	1.9
sk4	8	Fogland South	10/3/2001	17.6	0.2
sk4	8	Fogland South	10/3/2001	17.6	1.9
sk5	4	Sapowet Point	7/20/2001	21.1	?
sk5	4	Sapowet Point	8/29/2001	25.6	1.1
sk5	4	Sapowet Point	9/19/2001	20.7	1.5
sk5	4	Sapowet Point	9/19/2001	21.1	0.2
sk5	4	Sapowet Point	10/3/2001	18.0	0.2
sk5	4	Sapowet Point	10/3/2001	17.7	1.6
wp1	8	Buttonwood	7/17/2001	24.7	?
wp1	8	Buttonwood	8/1/2001	22.9	4.1
wp1	8	Buttonwood	8/15/2001	25.6	1.8
wp1	8	Buttonwood	9/12/2001	23.7	?
wp1	8	Buttonwood	9/12/2001	23.7	1.9
wp2	2	Sandy Point	6/27/2001	25.1	0.0
wp2	2	Sandy Point	6/27/2001	24.1	B
wp2	2	Sandy Point	7/19/2001	22.9	0.0
wp2	2	Sandy Point	7/24/2001	23.4	0.1
wp2	2	Sandy Point	8/1/2001	24.5	1.3
wp2	2	Sandy Point	8/15/2001	24.0	2.0
wp2	2	Sandy Point	9/12/2001	23.8	0.2
wp2	2	Sandy Point	9/12/2001	23.5	2.3
wp2	2	Sandy Point	9/26/2001	21.2	0.2
wp2	2	Sandy Point	9/26/2001	21.2	1.8
wp2	2	Sandy Point	10/19/2001	13.5	2.8
wp2	2	Sandy Point	10/19/2001	14.2	0.2
wp2	2	Sandy Point	11/1/2001	13.3	0.0
wp2	2	Sandy Point	11/1/2001	13.2	1.0
wp3	4	Patience	8/1/2001	23.6	1.5
wp4	8	Sauga Point	8/15/2001	28.5	1.1
wp4	8	Sauga Point	9/12/2001	22.4	0.2
wp4	8	Sauga Point	9/29/2001	22.3	1.2
wp4	8	Sauga Point	9/26/2001	0.2	0.2
wp4	8	Sauga Point	11/1/2001	20.6	1.3
wp4	8	Sauga Point	11/1/2001	13.0	0.2
wp5	8	Poplar Point	8/15/2001	24.1	0.9
wp5	8	Poplar Point	9/7/2001	22.4	0.2
wp5	8	Poplar Point	9/7/2001	22.2	2.0
wp5	8	Poplar Point	9/26/2001	20.9	0.2
wp5	8	Poplar Point	9/26/2001	21.0	1.5
wp5	8	Poplar Point	11/1/2001	13.9	0.0

wp6	4	Rome Pt. North	8/15/2001	23.3	1.7
wp7	2	Rome Pt. South	8/15/2001	23.3	1.5
wp7	2	Rome Pt. South	9/7/2001	22.3	0.2
wp7	2	Rome Pt. South	9/7/2001	21.7	2.0
wp7	2	Rome Pt. South	9/26/2001	20.6	2.5
wp7	2	Rome Pt. South	9/26/2001	20.7	0.2
wp7	2	Rome Pt. South	11/1/2001	13.7	0.0
wp7	2	Rome Pt. South	11/1/2001	13.6	1.7
wp8	8	Providence Pt.	8/1/2001	22.8	1.7
wp8	8	Providence Pt.	8/15/2001	22.0	2.0
wp8	8	Providence Pt.	9/12/2001	23.2	0.1
wp8	8	Providence Pt.	9/12/2001	22.6	2.5
wp8	8	Providence Pt.	9/19/2001	20.6	0.2
wp8	8	Providence Pt.	9/19/2001	20.3	2.5
wp8	8	Providence Pt.	9/26/2001	20.6	0.2
wp8	8	Providence Pt.	9/26/2001	20.7	105.0
wp8	8	Providence Pt.	10/19/2001	14.9	0.2
wp8	8	Providence Pt.	10/19/2001	14.9	2.7
wp8	8	Providence Pt.	11/1/2001	13.1	0.2
wp8	8	Providence Pt.	11/1/2001	13.0	2.0
wp9	8	Prudence West	7/17/2001	23.1	?
wp9	8	Prudence West	8/15/2001	21.2	1.5
wp9	8	Prudence West	8/15/2001	24.0	0.5
wp9	8	Prudence West	9/12/2001	21.30	0.2
wp9	8	Prudence West	9/12/2001	21.0	1.8
wp10	2	Rocky Point	8/1/2001	23.9	0.7
wp10	2	Rocky Point	9/19/2001	20.3	0.2
wp10	2	Rocky Point	9/19/2001	20.2	2.0

Table II.8 Final Transplant Suitability Index Results

Site_id	Site_name	PTSI	Area with High Restoration Potential (acres)*	Area with High & Moderate Restoration Potential (acres)**	Light_TSI	Bioturbator_TSI	Survival_TSI	PTSI_TSI	Temperature_TSI	Final TSI	Suitability
EP1	Potter's Cove	4	3.79	45.24	0	1	0	2	1	0	unsuitable
EP2	Bristol Harbor	8	13.59	21.17	2	2	0	2	0	0	unsuitable
EP3	Hog Island Cove	4	0	23.17	2	0	0	2	1	0	unsuitable
EP4	Kickimuit River	8	9.14	187.63	0	1	0	2	1	0	unsuitable
SK1	Nannaquaket Pond	2	106.51	156.65	0	1	0	1	1	0	unsuitable
SK2	The Cove	8	133.92	234.2	0	1	0	2	1	0	unsuitable
SK3	Fogland North	4	0	32.76	2	1	2	2	1	8	high
SK4	Fogland South	8	8.02	66.63	1	1	2	2	1	4	moderate
SK5	Sapowet Point	4	0	145.96	2	1	0	2	1	0	unsuitable
WP1	Buttonwoods	8	168.91	399.99	0	1	0	2	1	0	unsuitable
WP10	Rocky Point Cove	2	0	0.22	0	1	0	1	1	0	unsuitable
WP2	Sandy Point	2	37.21	285.01	0	0	0	1	1	0	unsuitable
WP3	Patience Island	4	0	0.89	1	1	0	2	1	0	unsuitable
WP4	Sauga Point	8	24.96	71.08	1	1	2	2	1	4	moderate
WP5	Poplar Point	8	10.92	38.33	1	1	2	2	1	4	moderate
WP7	Rome Point-South	2	0	0	0	1	1	1	1	0	unsuitable
WP8	Providence Point	8	8.02	27.63	1	1	1	2	1	2	low
WP9	Prudence- West	8	9.36	23.84	1	1	2	2	1	4	moderate
	TOTAL AREA		**534.35**	**1760.4**							

* Sum of the area of contiguous cells with a PTSI value of “8” or high restoration suitability

** Sum of the area of contiguous cells with a PTSI value of "8" and "4" -high and moderate restoration suitability

F. Discussion of The FTSI and Test Site Performance

Table II.6 and Chart II.2 show the results of test transplant plots at 18 sites in Narragansett Bay. Of the original 19 stations, station WP6, located at the north end of Rome Point in North Kingstown, was the only transplant that was destroyed by unknown circumstances. For this reason, only 18 test sites are included in the statistical analyses reported in this study. Furthermore, one test site, Ep1, located in Potter's Cove on Prudence Island was established in intertidal habitat due to human error. It is presumed that desiccation and exposure led to the quick destruction of this test site. It is clear that this problem reflects map scale and resolution constraints of the bathymetry data used in the PTSI model input. The PTSI model inaccurately identified Ep1 as a subtidal location. For this reason, Ep1 is excluded in the statistical analyses reported in Section III of this study and excluded from summary survival statistics.

Table II.8 reports the final TSI calculation for eelgrass restoration sites selected from the PTSI model. Ratings for each variable are multiplied to generate the final restoration suitability score. 13 sites were rejected by the FTSI model output, given values of zero, and 6 locations were ranked from low suitability to high suitability. Only one location was ranked as highly suitable for eelgrass restoration, 4 sites were ranked as moderately suitable for restoration, and one location was ranked as low suitability. Based upon the results of the FTSI model output, three of the highest scoring locations were selected for full-scale eelgrass restoration and implemented in Spring 2002. The results of the full-scale restoration are presented in Part III. Model Evaluation and Testing of the NBERSSM.

Table II.9 Model Area Summary Table

	Area (acres)
Total Modelling Extent-PTSI*	144,660
PTSI-High and Moderate Suitability	47,000
Sub-Area selected for FTSI screening	1,700
High and Moderate Restoration Potential based upon FTSI model	281

* Although Narragansett Bay is 96,000 acres
our model extent also included areas offshore of the Bay this was due to the GIS grids used in the
analysis which extended due south of the Bay into Block Island Sound

Table II.10 Potential Restoration Based on FTSI Ranking of Test Transplants

FTSI Score	Area (acres)
2-Low Suitability	28
4-Moderate Suitability	200
8-Hight Suitability	53
Total Area	281

Table II.9 demonstrates the potential restoration area by FTSI class.

It should be emphasized that while the PTSI model screened the entire area of Narragansett Bay (96,000 plus 48,600 acres of RI sound), the FTSI evaluates only sites where test transplants occurred. The FTSI screened a total area of 1781 acres, calculated in ARCVIEW by summing the total area of contiguous cells, identified in the PTSI model, for each of the 19 restoration sites. Considering that only 100 acres of eelgrass remain in Narragansett Bay, the FTSI model output suggests that approximately 250 acres are potentially restorable of a the subset area represented by the 19 test sites (1781 acres)..

Chart II.2 demonstrates the variable level of test site success achieved through the overall test transplant project. Even though sites were chosen with known habitat constraints, a test transplant success rate of 28% is twice the success achieved by previous restoration attempts in Narragansett Bay. Percent survival of test transplants, surveyed after one full year, remain 27%. It should be noted that transplant sites were located over the full range of PTSI values, though no test sites were performed in sites rejected by the PTSI model. Chart II.1 represent the number of test sites conducted within each suitability class, ranging from low to high suitability. When examining The FTSI GIS Map Output there is an apparent pattern in the relationship between location in Narragansett Bay and site performance. Research has demonstrated the existence of a north to south pollution gradient in Narragansett Bay with the bulk of pollutants, nutrient loadings, entering from the head of the Bay in the Providence River and decreasing with increased distance south (Granger et al. 2000). All test sites failed north of the northern tip of Prudence Island with increasing test site success in more southern locations of the Bay. It should also be noted that no naturally occurring populations of eelgrass are known to persist north of Prudence of Island. If we assume that the results of the FTSI model obtained from the 19 test sites are representative of the 47,000 acres that were identified as suitable for restoration by the PTSI model, than we can make a coarse estimate of the total restoration potential of Narragansett Bay as a whole. 49% (PTSI suitability) X 15% (FTSI % of PTSI) = 7% or 7,050 acres of the total bay area has eelgrass restoration potential

PARTR III. NBERSSM Model Testing and Evaluation

A. Purpose:

The PTSI model discussed in this paper identified thousands of acres of suitable eelgrass habitat. Further screening of these sites was accomplished with the Final Transplant Suitability Index that ultimately identified approximately 253 acres of highly and moderately suitable eelgrass restoration habitat of the 1781 acres represented by the test sites. The goal of this analysis is to examine the relationship between in situ restoration success, measured as % survival, and model output predictions. I will evaluate the performance of the PTSI and FTSI model output by testing whether sites inititially predicted as highly suitable for restoration in fact succeeded. This will provide insight for future and current restoration practitioners to understand the strengths and weakness of the model and I hope it will inspire future model users to continue to use, test, and refine the site selection model. This will ultimately contribute to increasing restoration success in the field.

In order to evaluate the performance of the NBERSSM, both the PTSI model and the FTSI model output are compared to in situ restoration success. Additionally, two of the individual model input factors that were measured for the FTSI model output, light and bioturbation, are individually evaluated to examine their contribution to model output by examining whether they were good indicators of transplant success. Finally, the FTSI scores are tested against the actual results of transplant success from the three full-scale restoration projects that were conducted in the Spring of 2002 at locations receiving the highest FTSI scores. The performance of these restoration sites will be used to test whether the Narragansett Bay site selection model is a useful tool in identifying the best sites to conduct restoration. Guidance to improve future restoration site selection and potential management applications of the Narragansett Bay Eelgrass Restoration Site Selection Model will also be presented.

This analysis is a particularly important validation of the NBERSSM model. If site survival was not significantly different among low scoring locations and high scoring locations then the PTSI model has not increased the capability of GIS models to identify potential eelgrass restoration sites that have the highest chances for successful plant establishment. Any random placement of transplants would provide the restoration practitioner with an equal chance of obtaining the same results. It should be emphasized that no test sites were conducted at PTSI scores of 0 or “unsuitable.” If there are significant differences in transplant survival among PTSI classes then the PTSI model has the potential to benefit restoration practitioners and begins to validate the assumption that the model user has chosen habitat factors and threshold limits that best reflect the most important considerations for successful eelgrass establishment.

B. Methodology:

1. PTSI testing

In order to examine the ability of the PTSI model to predict restoration success, mean site survival data are reported by PTSI class categories: Low Suitability for eelgrass restoration (2), Moderate suitability for eelgrass restoration (4), and High suitability for eelgrass restoration. Analysis of Variance (ANOVA) was conducted on the test transplant '% survival after one growing season' data to test whether there were significant differences in transplant survival among PTSI classes. In other words, did survivorship differ among sites that were classified as low restoration potential, medium restoration potential, and high restoration potential. It is beyond the ability of the data in this study to test whether the PTSI model can predict eelgrass transplant success.

Multiple comparisons were made with the Tukey test for % survival to test whether there were significant differences between sites that were rated as high restoration potential, low restoration potential and medium restoration potential. Sites that were ranked with the highest restoration potential PTSI score should be the sites that led to the most successful restoration results in the field, as measured by test transplant % survival. Site failures should be sites that were ranked the lowest by the PTSI model. The Tukey test provides a useful statistic to confirm or reject this. The following null hypotheses are tested for the analysis:

H_o: Transplant survival does not differ among sites ranked by the PTSI as low, medium, and high suitability for eelgrass restoration

H_o: Transplant survival does not differ between sites ranked by the PTSI as low, medium, and high suitability for eelgrass restoration

2. FTSI Testing

Following the above method for evaluating the PTSI, Analysis of Variance (ANOVA) was also conducted to compare FTSI model output with % survival data from the 2002 full scale restoration sites. These sites were selected from the three locations that had the highest FTSI scores representing classes. Survival after one growing season' data was used to test whether there were significant differences in restoration success among the FTSI classes. In other words, did survivorship differ among sites that were classified as low restoration potential, medium restoration potential, and high restoration potential. Multiple comparisons were made with the Tukey test for % survival to test whether there were significant differences between sites that were rated as high restoration suitability, low restoration suitability and medium restoration suitability. Sites that were ranked with the highest restoration potential FTSI score should be the sites that led to the most successful restoration results in the field, as measured by test transplant % survival. Reduced site success should be sites that were ranked the lowest by the FTSI model. The Tukey test provides a useful statistic to confirm or reject this. The following null hypotheses are tested for the analysis:

H_o: Transplant survival does not differ among sites ranked by the FTSI as medium, and high suitability for eelgrass restoration

H_o: Transplant survival does not differ between sites ranked by the FTSI as medium, and high suitability for eelgrass restoration

a. 2002 Full Scale Restoration Methods:

Restoration at three locations in Narragansett Bay (wp5 Poplar Point, wp9 Prudence West, and sk3 Fogland Point) occurred in May 2002. Donor bed collection, transplant techniques, and performance monitoring were identical to methods used in 2001 test site transplants. For these details, refer to the methods outlined in Step 7. in the previous section. The only difference between the test site transplant methods and the full scale restoration was the full scale restoration sites were significantly larger, involving the placement of 64 TERF frames over an .5 acre area per restoration location. Restoration success, measured as % survival after one season's growth, was calculated by randomly sampling half of all TERF frames per site. Total shoots per TERF were counted using SCUBA and performed by Wendy Norden-restoration ecologist for Save The Bay, and two volunteers.

3. Evaluation of Individual Factors used in the FTSI Model: Light

Univariate Statistics were used to analyze the relationship between light conditions measured during the test transplant's first growing season and the success of the transplant. Analysis of Variance (ANOVA) was conducted on the test transplant '% survival after one growing season' data to test whether there were significant differences in transplant survival among sites with different light levels. Light conditions were sampled and calculated using methods described previously. ANOVA was conducted on the categorized LIGHT-TSI value ranking and % survival. Scatter plots and regression equations are presented for Site survival and continuous mean Kd per restoration site and site survival and the mean Irradiance per restoration site in this analysis. Univariate statistics were used to analyze the relationship of light measured during the test site's first growing season and the success of the transplant. Analysis of Variance (ANOVA) was conducted on the test transplant % survival to test whether there were significant differences in transplant survival among sites with different light level classes.

4. Evaluation of Individual Factors used in the FTSI Model: Bioturbation

Transplant survival was compared to bioturbator monitoring data as reflected in the calculation of the Bioturbator TSI score. Scatter plots and regression equations are presented for Site survival and the actual abundance of bioturbators per test site location.

C. Results:

1. ***PTSI Model***

Table III.1 Test Site Mean Survival among PTSI ranked Sites*

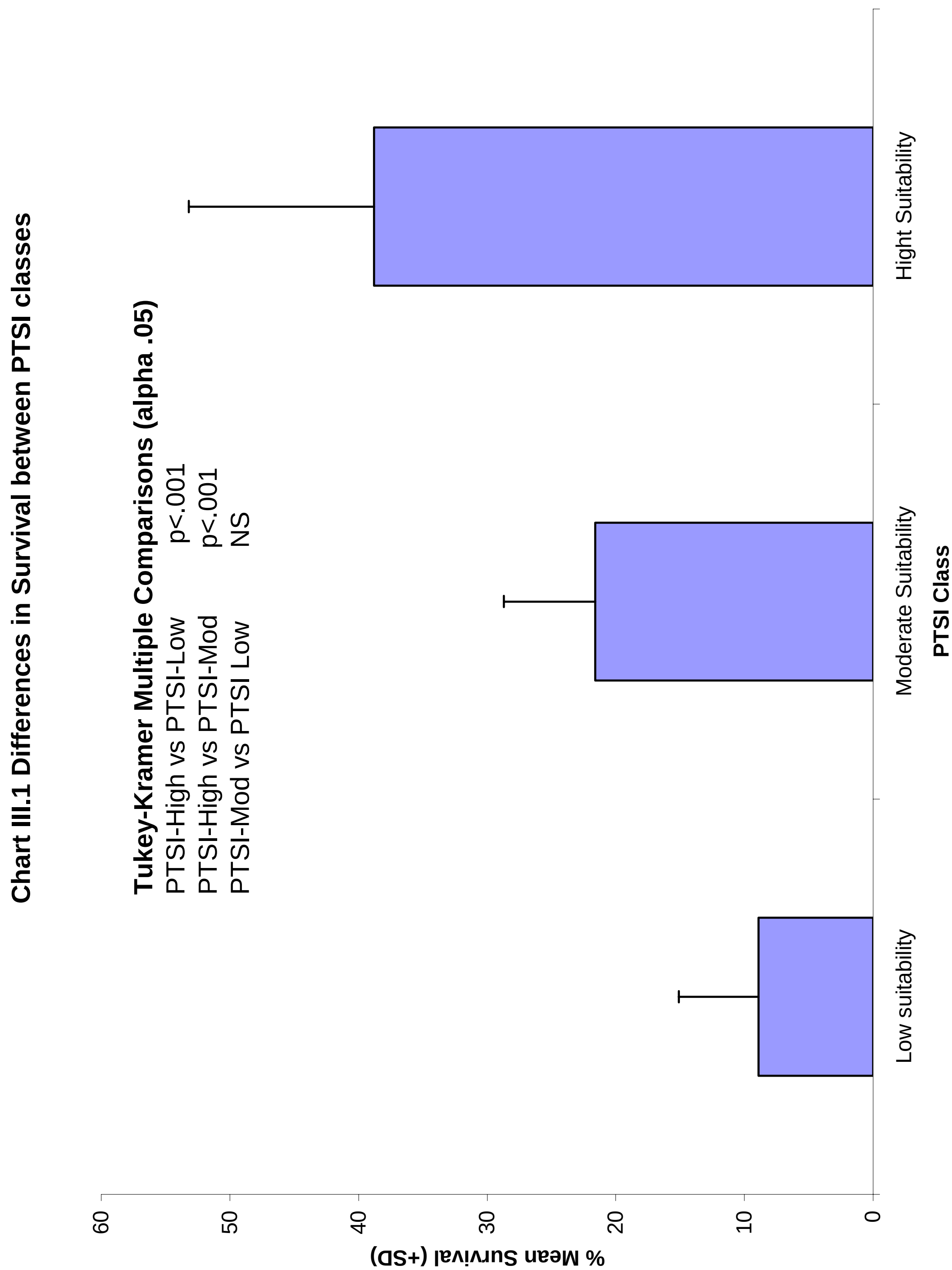
PTSI Score	n=# sites with that rank	Mean % Survival (1season)	STD	# sites <25% SURVIVAL	# SITES >-25 <50% SURVIVAL	# SITES >-50% SURVIVAL
2-Low Suitability	4.0	8.9	6.2	4.0	1.0	0.0
4-Moderate Suitability	4.0	21.6	7.1	3.0	0.0	1.0
8-High Suitability	9.0	38.8	14.4	4.0	1.0	4.0

***Excluding sites EP1 and WP6**

Statistical Results:

H_o: Transplant survival does not differ among sites ranked by the PTSI as low, medium, and high suitability for eelgrass restoration------**Rejected**
H_o: Transplant survival does not differ between sites ranked by the PTSI as low, medium, and high suitability for eelgrass restoration**------Rejected**

Transplant survival is different among PTSI classes (F ratio = 7.04, d.f. 2 and 86, p <.0015) when running the ANOVA at the .05 probability level. Tukey test explains where the greatest significant differences exist between the potential restoration classes, 2, 4, and 8. Transplant survival was most different between sites with low restoration potential (PTSI-2) and sites with high restoration potential (PTSI-8).



2. FTSI Model Results

Table III.2 Results from 2002 Full Scale Restoration

Restoration Site *	n-samples	Mean %Survival	Std
Poplar Point - WP5 (4)	34	58	17
Prudence West - WP9 (4)	34	61	18
Fogland Point - SK3 (8)	64	81	13

***( ) FTSI SCORE**

Means for Oneway Anova

FTSI score	N	Survival Mean	Std Error	Lower 95%	Upper 95%
4	66	59.5606	1.8575	55.885	63.236
8	64	80.8594	1.8863	77.127	84.592

Std Error uses a pooled estimate of error variance

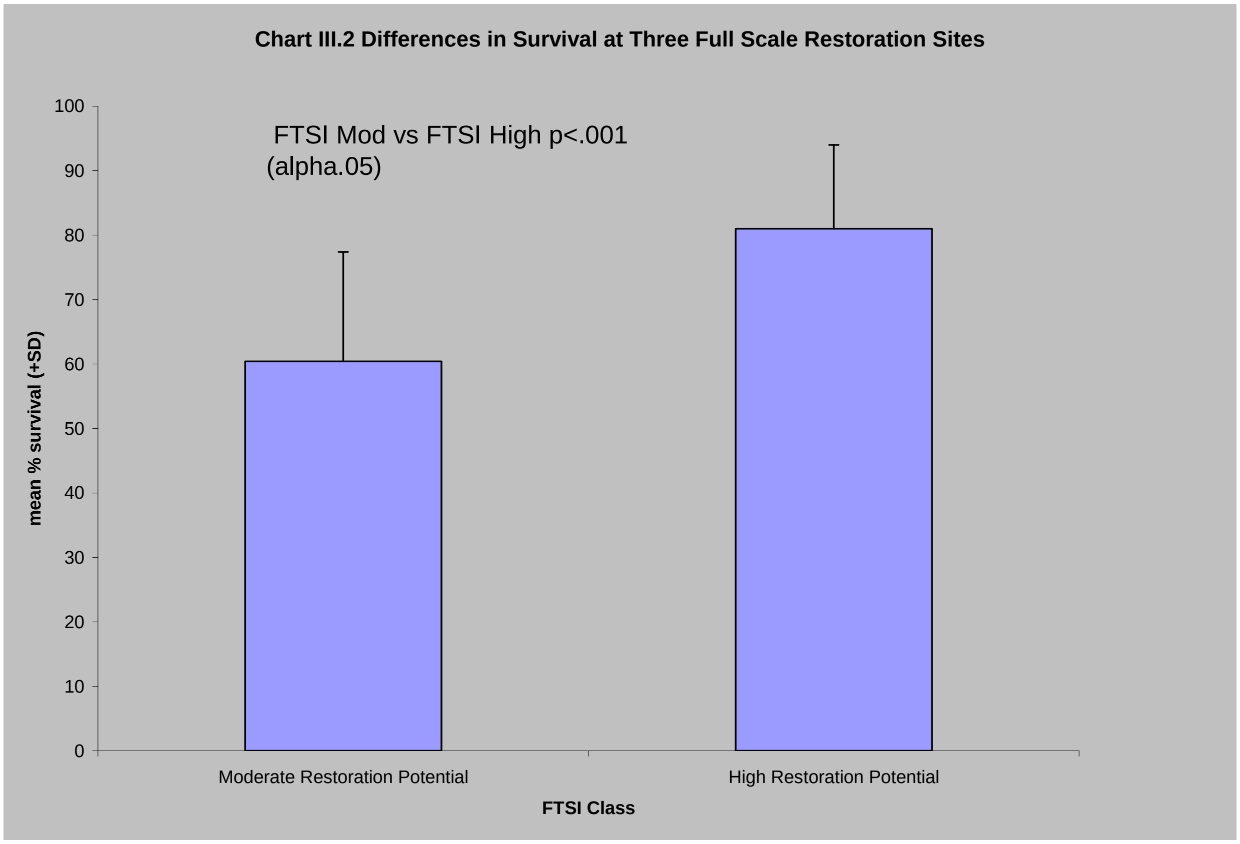
N= number of samples

Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
FTSI	1	14739.731	14739.7	64.7278	<.0001
Error	128	29147.992	227.7		
C. Total	129	43887.723			

–

Transplant survival is different between the two FTSI classes Moderate Suitability and High Suitability (F ratio = 64.7, d.f. 1 and 129, p <.0001) when running the ANOVA at the .05 probability level.



3. Individual Factors and Test Site Survival: Light Results

Table III.3 % Survival According to Light TSI Class

Light TSI Score	N	Mean Survival %	Std Dev	Std Err Mean
0	30	8.8333	13.5598	2.4757
1	30	56.4000	33.7808	6.1675
2	19	20.7368	34.1253	7.8289

N= the number of replicates (5 replicates/transplant site) used to compute the mean % survival

2 = The station received 50% light

or greater at the bottom for more than 50% of the surveys

1 = The station received at least 20% light

0 = The station received less than 20% light during at least one survey

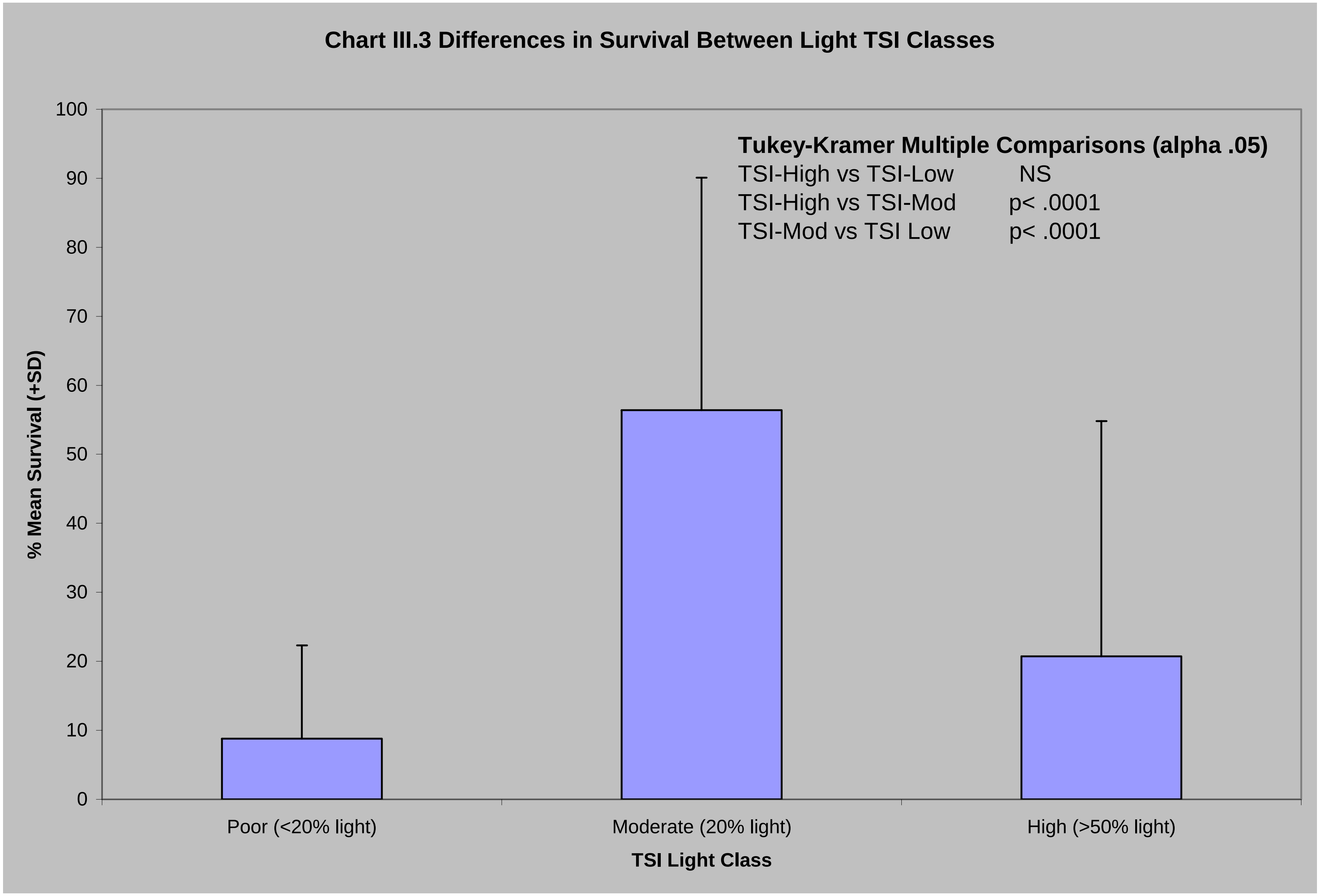


Chart III.3 Differences in Survival Between Light TSI Classes
Tukey-Kramer Multiple Comparisons (alpha .05)
TSI-High vs TSI-Low NS
TSI-High vs TSI-Mod p< .0001
TSI-Mod vs TSI Low p< .0001
% Mean Survival (+SD)
100
90
80
70
60
50
40
30
20
10
0
Poor (<20% light)
Moderate (20% light)
High (>50% light)
TSI Light Class

Transplant survival is different among the three Light-TSI classes (F ratio = 26.1, d.f. 2 and 81, $p < .0001$) when running the analysis at the .05 probability level. Tukey test explains where the greatest significant differences exist between sites with different light conditions. Tukey test results show that survivorship at sites with a TSI-1-intermediate light score are significantly different than sites with a TSI-2-highest light score. However, test site survival was not a significant between the highest light TSI score-2 and the lowest light TSI score-0. The regressions reported in Charts III.4-III.7 display the lack of any significant relationship between light conditions and site survival.

Chart III.4 Light Attenuation Coefficient (-Kd) and % Survival

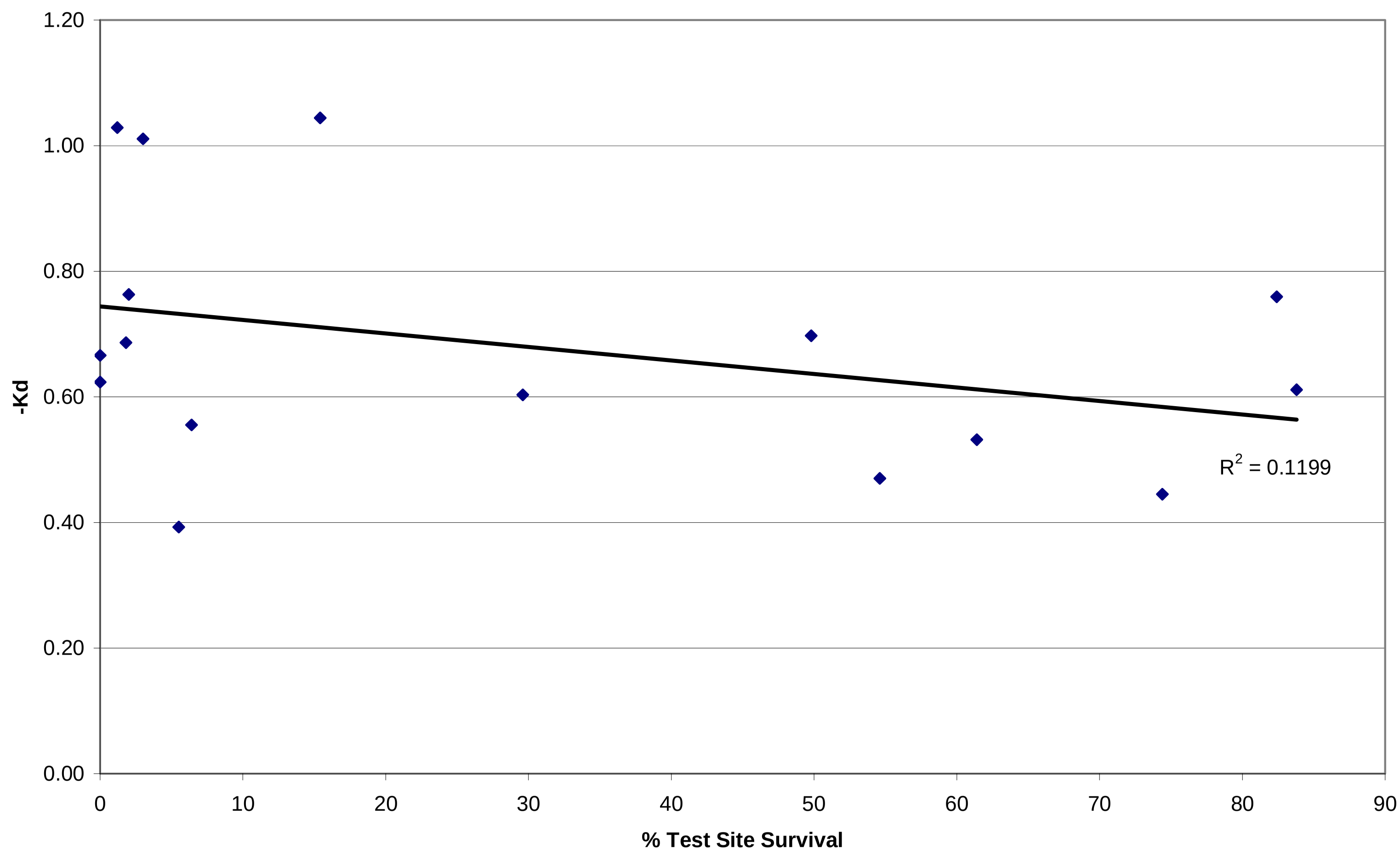


Chart III.5 Mean Irradiance and % Survival

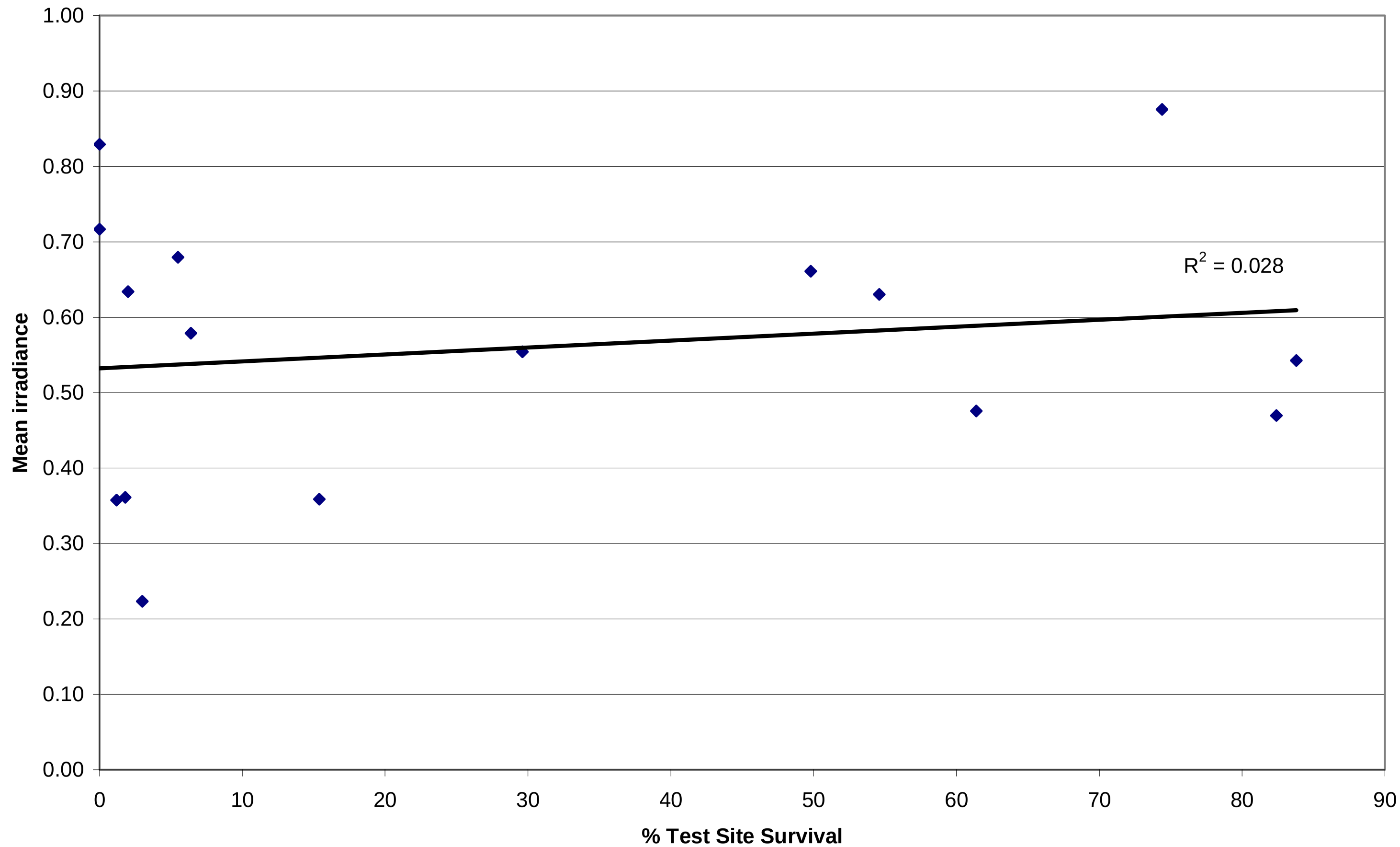
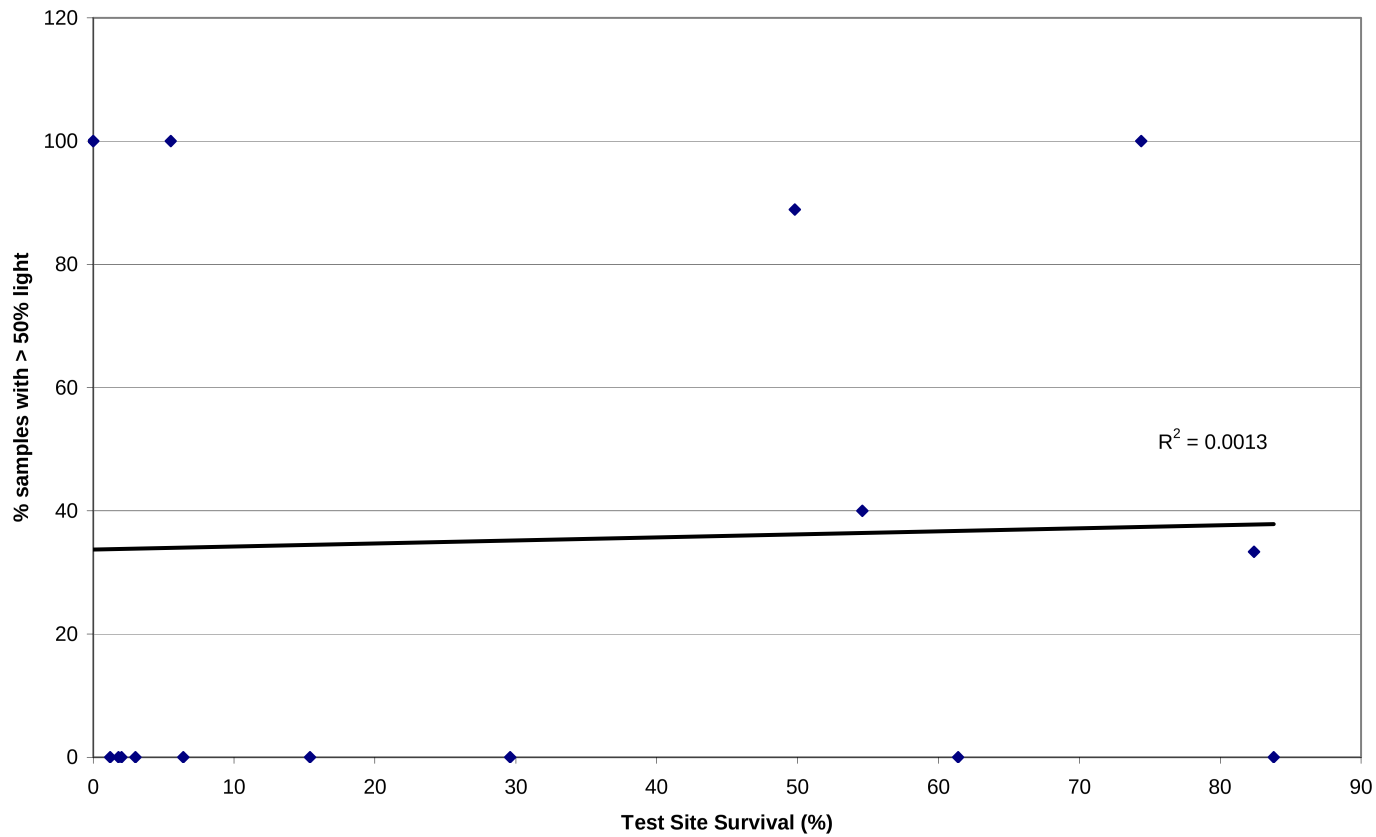
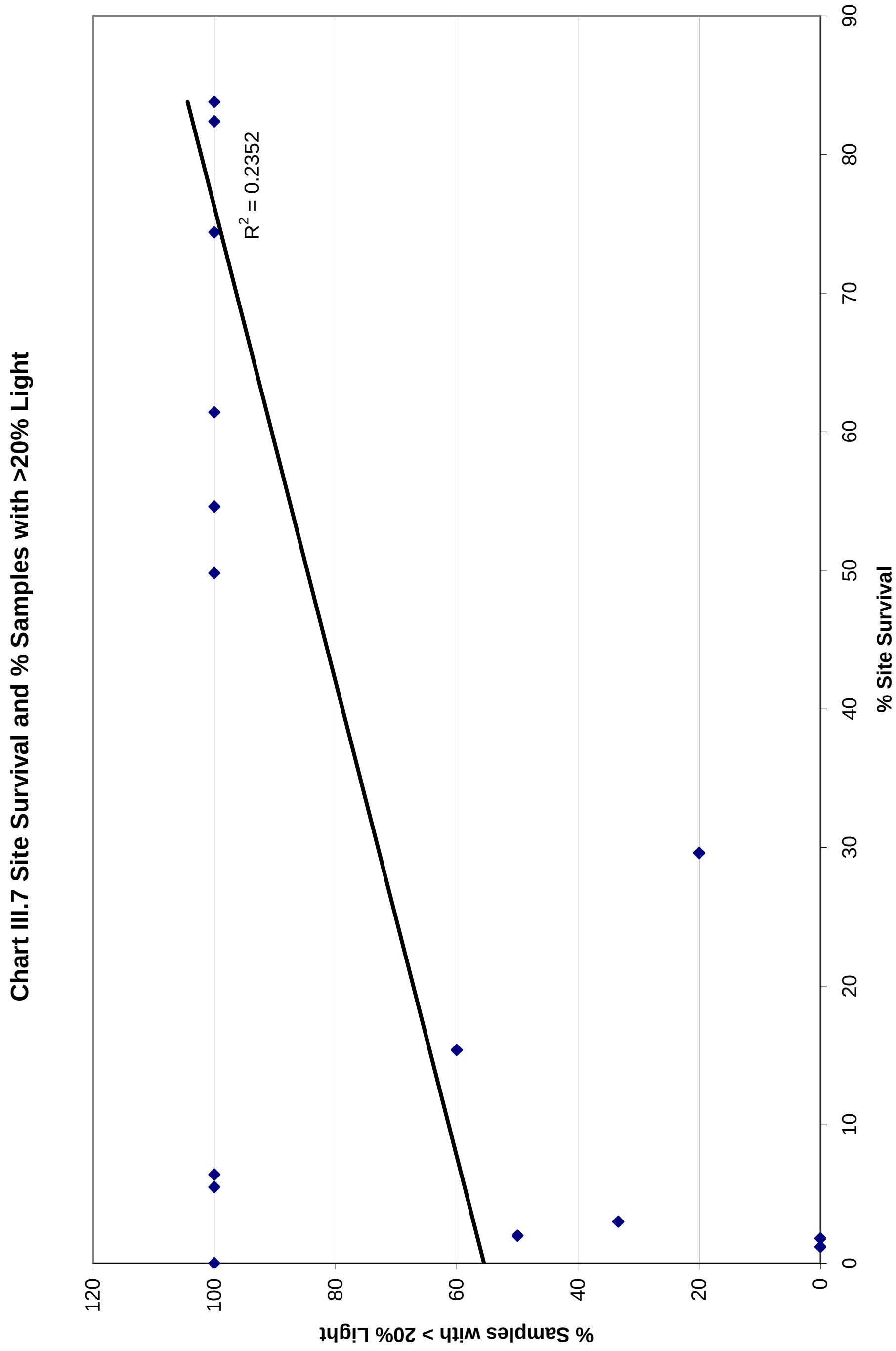


Chart III. 6 Survival and % Samples with > 50% Light





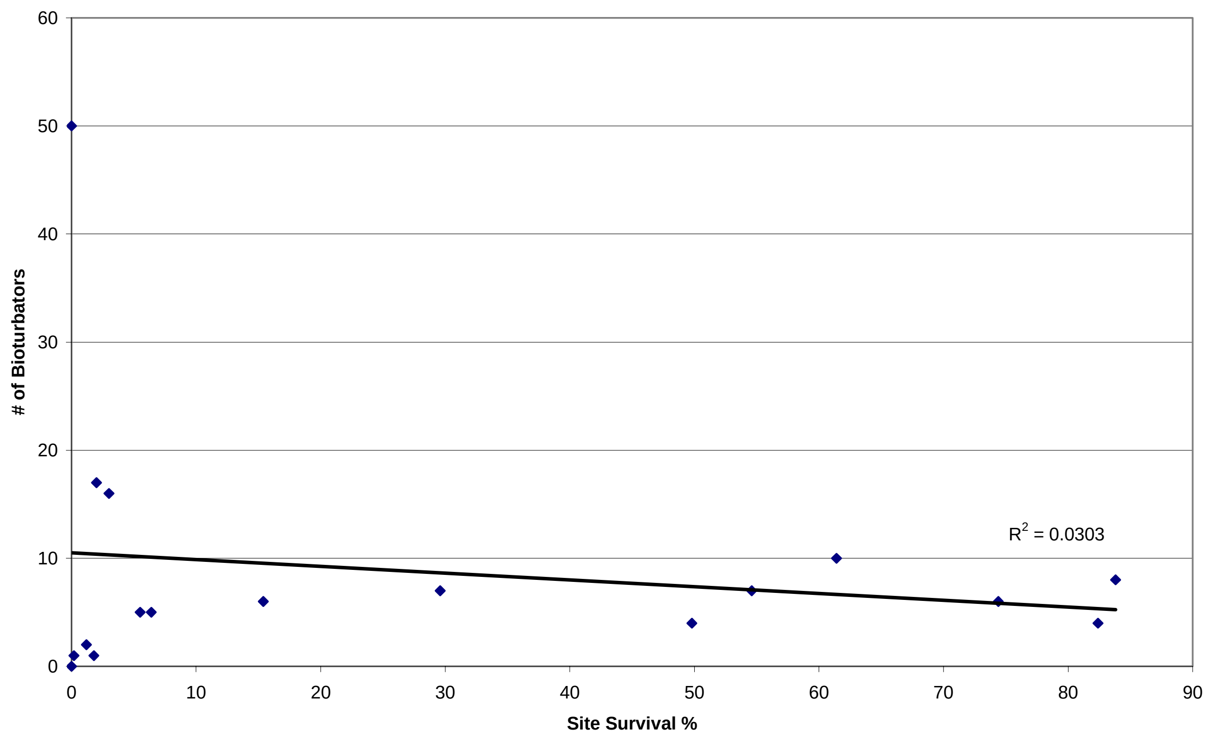
4. Individual Factors and Test Site Survival: Bioturbation Results

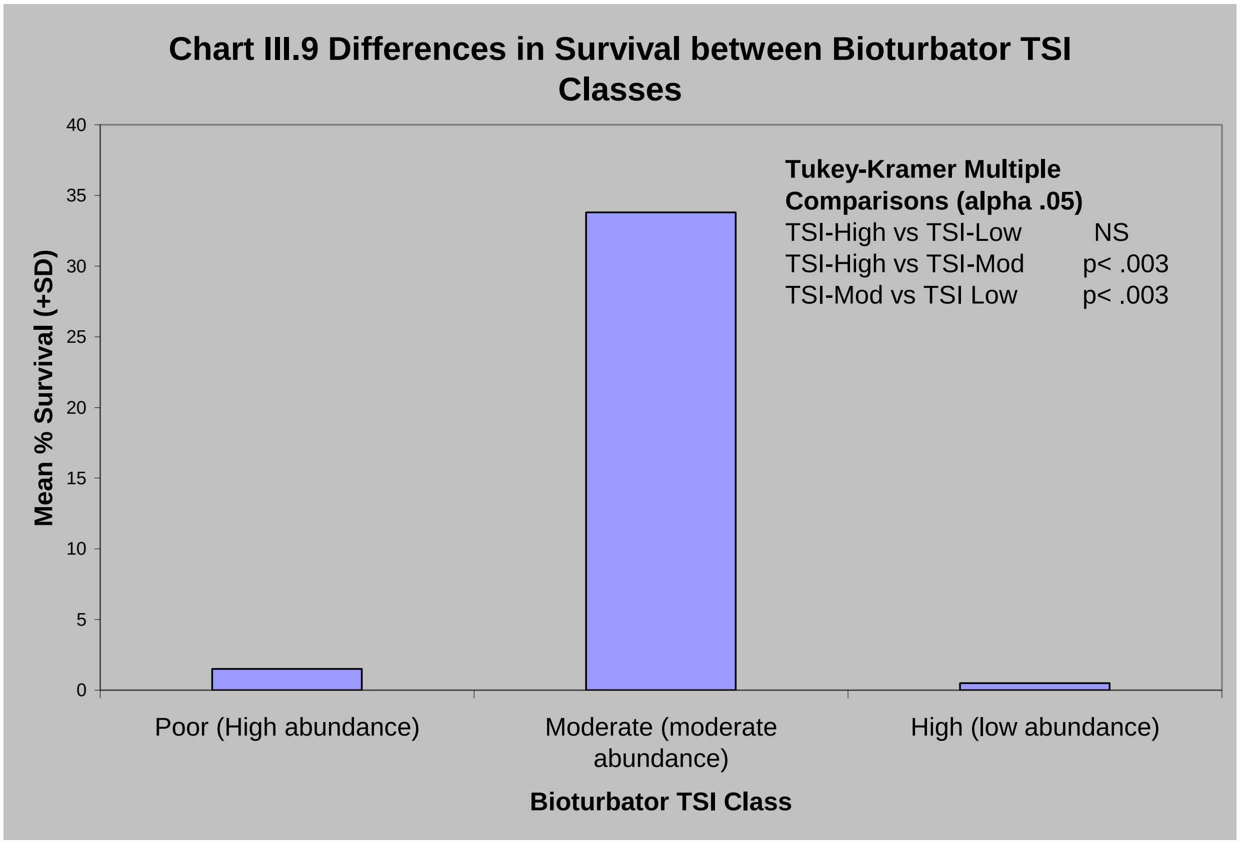
Table III.4 Bioturbator TSI and % Survival Means and Std Deviations

Bioturbator-TSI Score	N=replicates	%Survival Mean	Std Dev	**# sites <25% SURVIVAL**	**# SITES >-25 <50% SURVIVAL**	**# SITES >-50% SURVIVAL**
0	10	1.5000	4.4033	2	0	0
1	69	33.8696	35.5581	7	2	5
2	5	0.0000	0.0000	1	0	0

Transplant survival is different among the three Bioturbator TSI classes (F ratio = 6.25, d.f. 2 and 83, p <.0030) when running the analysis at the .05 probability level. Tukey test explains where the greatest significant differences exist between sites with different bioturbator scores. Tukey test results show that survivorship at sites with a Bioturbator TSI-1 are significantly different than sites with a BioturbatorTSI-0. Test site survival was not significantly different between the highest Bioturbator TSI score-2 and the lowest Bioturbator TSI score-0.

Chart III.8 Bioturbator Abundance and Site Survival





D. Discussion:

a. PTSI Model

Results from the Anova (Tukey) tests confirm that the best performing test transplants, as measured by the % survival after one growing season, were also rated the highest suitability for restoration by the PTSI model. It should be emphasized that normality assumptions were violated when performing this and other tests even after multiple attempts to transform the negatively skewed distribution of '%survival' by adding 1 and square root transformation (Zar, 1999). Chart III.1 demonstrates that sites with high suitability (PTSI-8) had a mean survival rate of 39%, sites with moderate suitability (PTSI-4) had a mean survival rate of 22% and sites with the lowest suitability (PTSI-2) had a mean survival rate of 9%. The results of these tests suggest that the PTSI model is a useful <u>pre-screening</u> tool for eelgrass site selection. The combination of habitat factors and threshold settings that were used in the PTSI model apparently approximate the necessary conditions that are important for eelgrass persistence and restoration success. Examination of restoration success of the 2002 large scale eelgrass transplants is an ultimate test of whether the PTSI model and the further screening of the FTSI model output can increase the success of eelgrass restoration in Narragansett Bay.

b. FTSI Model Evaluation

The FTSI Model screened locations selected by the PTSI model output by incorporating monitoring results and the performance of eelgrass test transplants. By examining the end result to both models-the performance of the large scale restoration site- I can evaluate whether the combination of a GIS spatial model and test plots can increase eelgrass restoration success. Initial results after one growing season show positive results. A mean restoration success of 67% for all three large scale restoration sites has been achieved after one growing season. Table III.2 report the performance of the large scale restoration projects. Anova and Tukey tests were performed on the survival data of the three large scale restoration sites resulting in a significant difference in restoration success between FTSI classes. The three large scale restorations were ranked as follows: Poplar Point (FTSI-4 moderately suitable), Fogland Point (FTSI-8 highly suitable, and Prudence West (FTSI-4 moderately suitable). A full evaluation of the FTSI would entail conducting a large scale restoration in all the representative classes of FTSI ranked sites-including a FTSI class of 0 or “no restoration potential”. However, the over-riding goal of this project was to establish persistent and successful eelgrass transplants and thus a more rigorous study design was not possible. The logistics and resources necessary to conduct these large scale plots in areas ranked as unsuitable by the FTSI made this type of full analysis unfeasible.

c. Discussion of Individual habitat factors: *Light*

Two habitat factors were examined to test whether any relationships between individual habitat factor conditions and site survival existed. Indirectly, this analysis allows us to examine how important these habitat factors are to modelling habitat suitability and whether they are good indicators for restoration success.

As discussed in earlier sections of this paper, light penetration reaching the seagrass canopy has been considered to be one of the most important factors that contribute to eelgrass survival.   Overall, when testing the light factor alone and its relationship to eelgrass survival, there is a significant difference in site survival between locations receiving at least 20% light and locations receiving less than 20% light.
However, close examination of the statistical results in Table II.14 show that light levels sustained over 50% did not show a very significant relationship with greater eelgrass survival. Stations receiving at least 20% surface light had the highest survival rates. Stations receiving the most light did not have as high transplant success (only 20% mean survival) as stations receiving at least 20% light (58% mean survival).

In order to examine the light and survival relationship further I plotted scatter plots of three different ways to measure the light variable: 1.

Plotting the mean diffuse light attenuation coefficient per station 2. Plotting the mean % irradiance reaching the bottom per station and 3. Plotting light using the percent of surveys where light levels exceeded 20% and 50%. All the scatter plot charts III.4-III.7 demonstrate very weak correlations between light and survival, when reviewing R^2 values. Chart III.4 representing Kd and survival does depict a nesting of low survival sites in regions of the plot with the greatest Kd (higher Kd corresponds with less light). The regions of the plot with the highest survival also depict the least Kd values. An important consideration in comparing Kd values between sites is the fact that not all of the test transplant locations were located at the same depths. Therefore, a site with low Kd values but deeper eelgrass, will ultimately mean less light reaching plants than a shallow location with a higher mean Kd. However, the relative difference in depth between locations was not significant and variation due to depth measurements conducted at different tidal levels would seem to contribute more to the depth variation between sites.

Scatter plot Chart II.6 is the only other plot to yield interpretable results and confirms the relationship between light levels above 20% and site survival. In this plot the majority of points with the highest site survival correspond with the greatest number of samples achieving at least 20% light. When examining the descriptive statistics reported in Table II.14, it is important to note that sites with excellent light conditions had a wide range of success (0-90%) but also had much fewer records (n=19); while sites rated with a TSI-0 and TSO-1 had many more records (n=30).

Light conditions in Narragansett Bay can change rapidly over time and space and it is important to recognize that the light data may reflect this type of variability. Ultimately, light data should be measured from remotely deployed photometers and set out to capture continuous data over the growing season. However, the logistics and equipment necessary to carry this out continues to challenge estuarine and marine researchers. Light data used in this study were collected at one point in time and give us only a snapshot of an integrated daily condition of a site. This is how most researchers depend upon this method (Short and Coles 2001). Changes in cloud cover can significantly influence values taken at the same station within the same time period so variability in light data is common. However, light sampling occurred at least twice per station with a maxima of seven sampling periods per station and represents a very reasonable level of effort. Therefore, I can neither rule out nor conclude that variability and lack of sample power in the light monitoring dataset has contributed to erroneous results and misinterpretation.

It would appear that sites with the highest light might have had other factors, not incorporated as model inputs in the TSI or PTSI models that contributed to transplant performance. This might also suggest that increasing light threshold values above the 20% irradiance threshold may not be necessary. It may be more appropriate to lump the TSI categories into two distinct groups rather than three. More appropriate thresholds for a new TSI Light index might consist of two classes: sites with light conditions below 20% surface irradiance and sites with light conditions above 20% surface irradiance.

c. Bioturbation

The statistical tests show that there is a significant relationship between bioturbator abundance and transplant survival. Although the scatter plot in III.8 does not show a very good correlation between survival and bioturbator abundance, when comparing Bioturbator TSI index score given to a site with site survival, a significant relationship does exist between survival and bioturbators. This is depicted in Chart III.4. It is clear that the threshold values chosen for the bioturbator TSI are problematic for the final TSI model output. A total of 9 test sites received a bioturbator score of 0 or 1 and had the worst survival, while only 1 site, scoring bioturbator TSI 2, had the greatest survival. One of the problems, as reviewed in the first section of this paper, is quantifying bioturbators. For the purposes of this study, bioturbators were limited to crab observations, though it was clear that other species have the potential to disturb establishing eelgrass shoots. Adult winter flounder and summer flounder were observed using restoration sites with observed deleterious impacts on the recently transplanted shoots. Another problem with this data set is the lack of observations conducted over the course of the growing season. Bioturbator surveys were carried out at a minimum of two surveys/ site and a maximum of three surveys per site. It is unclear whether the frequency of observations is great enough to capture the natural variation that exists in crustacean abundance and distribution. We made the assumption that swimming crabs were more disruptive to eelgrass plants than non swimmers. It is likely that certain non swimming crabs, such as green crabs and spider crabs, could impact eelgrass shoots just as severely.

E. Future Directions for Eelgrass Restoration Site Selection in RI

The Narragansett Bay Eelgrass Restoration Site Selection model (NBERSSM) as reported in this paper shows promising results. Sites identified as high restoration suitability were the sites that actually grew eelgrass most successfully. The sites selected from the FTSI model for full scale restoration are demonstrating the highest level of restoration success that has ever been achieved in Southern New England with a mean survival of 67% for one season's growth. Ultimately this is the acid test for the restoration practitioner with limited resources. To continue this study to its completion, the final TSI index should be tested against the success of large-scale restoration transplants at the same locations after one full year. The performance of these sites will be a measure of the ability of the NBERSSM to lead to successful restoration at the full restoration scale.

Due to a lack of time and resources a number of tasks which would improve the evaluation of the NBERSSM and the refinement of habitat factors were not conducted. Due to reasons discussed earlier the sampling of macroalgae and epiphyte cover conducted at the test plant sites were excluded from the FTSI model. These data should be re-examined and applied to the FTSI. Indirectly, macroalgae are incorporated in the PTSI model as the expert testimony variable.

To conduct a more rigorous evaluation of the NBERSSM that includes both the PTSI and FTSI models; Principal Component Analysis (PCA) statistics might be considered. Principal component analysis followed by multiple regressions would allow me to truly test which factors used in the models were most important in accounting for the variation in site survival. Unfortunately, the lack of time and replicates prevented this type of analysis. More data is now emerging. Along with the full scale eelgrass restoration sites, five additional test eelgrass transplants were conducted in new sites in Narragansett Bay. Incorporation of the first season's growth data into the analyses presented in this paper would benefit this study by increasing sample power and TSI class representation.

It is my hope that future eelgrass restoration practitioners will continue to update the FTSI model with the results of continued test sites, test the performance of the PTSI and FTSI models, and refine the habitat factors and threshold limits that are at the heart of the screening process. Future users might conduct more rigorous sensitivity analysis of the model input factors and threshold limits. How sensitive are model outputs, i.e. favorability of one site over another, to changes in model inputs?

The model input data used in the PTSI model should be accurate, current, and relevant to real seagrass life requirements in order for the model to be continue to be an effective tool for future eelgrass restoration practitioners in Rhode Island. In order for this to happen a number of studies will need to be initiated to update existing data sources and to create new monitoring programs. Based upon the technical team meetings that were held to support this project and the results from this report, the following tasks are recommended to improve the application of the PTSI model and support future eelgrass restoration planning:

Priority for Immediate Applications

1. Remote sensing studies to re-map eelgrass beds in Rhode Island are necessary. Current distribution is now based upon seven year old data. This will be necessary in order to quantify eelgrass abundance trends in the state.

2. Bathymetric mapping and re-sampling could be conducted on a statewide level

using new technology, side scan sonar, especially focusing on near shore areas not covered by NOS surveys. This will improve the accuracy of the PTSI model and minimize chances of modelling intertidal or exposed areas.

3. An eelgrass reference site monitoring program should be established to properly determine restoration site performance success in relation to naturally occurring eelgrass populations. This will allow us to answer the question of whether restoration success is due to a good year or bad year for eelgrass? Reference monitoring can also be used to replace the literature derived habitat factor thresholds in both the PTSI and FTSI models. For instance, the light suitability index can be based upon light thresholds that are based upon the actual light conditions that exist at

reference eelgrass beds. All the FTSI indices can be related to how close a test site performs in relationship to what is measured at a reference site. If a reference eelgrass bed is known to persist with a bioturbator abundance of x crabs/square meter, then a test site can be paired with the reference bed bioturbator measurement. If the test site greatly exceeds the amount of bioturbators than the reference location, then it will receive a lower Bioturbator rating.

Long Term Research Questions

1. Light monitoring should be considered throughout RI state waters at fine spatial and temporal resolution – with a focus on measurements of light during the growing season—March through November

2. Updated PTSI models should be run and tested on existing eelgrass populations to test monitoring data thresholds and sites should be stratified by location. Eelgrass beds growing in different regions of the state are known to express phenotypic differences, such as the timing of flower initiation and seed release dates.

3. Subaquaous soil mapping should be conducted and an evaluation of the value of these data to restoration be undertaken. Soils are critical to the establishment of eelgrass plants and seeds, as stated in the first portion of this paper. Soil mapping units may provide a better understanding of water quality conditions as well, as high organic/anoxic sediment map units will invariably covary with areas of nutrient enrichment and poor potential seagrass habitat which may be difficult to quantitatively capture in the current habitat factors.

4. Macroalgae Population Monitoring: Drift algae and sessile forms should be assessed statewide to quantify the potential for macroalgal production. This can be done either by remote sensing or population sampling techniques. Again, the lack of any database to incorporate into the PTSI model required that field surveys be conducted and incorporated into the “expert testimony” data layer. Many of the test sites that failed were caused by macroalgal smothering. Many locations that were identified in the PTSI model met the light requirements for eelgrass but vast areas of these locations are severely impacted by macroalgae production, especially the drift algae *Ulva lactuca*. This data has the potential to vastly improve the ability of the PTSI model to select appropriate restoration sites.

5. Temperature monitoring data should be interpolated using remote sensing techniques similar to work performed by Dr. Jack Mustard, Brown University, to map seasonally high water temperature regions. This data will be very useful for the site selection process. Further testing of temperature and plant growth threshold limits should be conducted to

validate the whether the 25 C temperature threshold is applicable to RI eelgrass populations.

6. Test sites need to be continually executed on a seasonal basis, ideally spring and fall test transplants should be conducted at sites over multiple years. This will assist restoration practitioners in a number of ways: to determine appropriate sites to invest in full scale restoration, to support further calibration and validation of the PTSI model, and to test whether certain locations are more suited for Fall or Spring transplanting

7. Water Current Speed Data layer, interpolated for RI coastal waters; will provide an important data set for site selection models. Again, current speed threshold values will need to be determined by empirical studies to develop a reasonable basis for site rejection and approval.

The site selection tools in the NBERSSM have resulted in an average full-scale restoration site success level of 67%. FTSI model output identified over 250 acres of Narragansett Bay to be moderately and highly suitable for eelgrass restoration. If the FTSI sites are representative of the additional acreage of untested locations identified by the PTSI model (47,000 acres), then the model suggests that approximately 7000 acres of the Bay may be suitable for eelgrass restoration. This assumption will need to be tested by conducting future test sites and FTSI model updating. Future test transplanting and updating of the FTSI and PTSI models will be necessary to identify more areas for restoration. It is clear to me that the Narragansett Bay Eelgrass Restoration Site Selection Model is a useful screening tool that has resulted in increasing the level of restoration success in Narragansett Bay.

Improving the success of eelgrass transplant techniques in Narragansett Bay requires a cost effective and time saving methodology to evaluate potential restoration locations with the most easily attainable and useful site selection criteria. The PTSI and FTSI screening tools that constitute the Narragansett Bay Eelgrass Restoration Site Selection Model meet these requirements. By relying on existing and updateable GIS datasets and in situ test site performance, the NBERSSM will be able to be continually used, tested, and refined by future model users.

REFERENCES:

Addiscott, T .M. 1998. Modelling concepts and their relation to the scale of the problem. Nutrient Cycling in Agroecosystems. 50:239-245.

Addiscott, T.M., A.P. Whitmore, and D.S. Powlson. 1991. Farming, Fertilizers, and the Nitrate Problem, CAB International.

Addiscott, T.M , and R.J. Wagenet,. 1985. Concepts of solute leaching in soils: a review of modelling approaches. Journal of Soil Science 36:411-424.

Batiuk, R., R. Orth, K. Moore, J. C. Stevenson, W. Dennison,L. Staver, v. Carter, N Rybicki, R. Hickman, S. Kollar and S. Bieber. 1992. Submerged Aquatic Vegetation Habitat Requirements and Restoration Targets: A Technical Synthesis. Chesapeake Bay Program, Annapolis, Maryland. 217 pp.

Batuik et al. 2001 Chesapeake Bay Submerged Aquatic Vegetation Water Quality and Habitat-Based Requirements and Restoration Targets: A Second Technical Synthesis. Chesapeake Bay Program, Annapolis Maryland.

Bell, J.D. and Pollard,D.A. 1989. Ecology of fish assemblages and fisheries associated with seagrasses p. 565-609. In Biology of Seagrasses. Elsevier, NY

Bintz, J. and S.W. Nixon. 2001 Response of eelgrass (Zostera marina) (I.) to reduced light. In press Marine Ecological Progress Series.

Cottrell, H., I.Huber, L.Carlson, and A.Lipsky. 1999. Coastal Wetland Habitat Mapping in Narragansett Bay, Rhode Island/Massachusetts. In Proceedings of Estuarine Research Federation 1st Biennial International Conference, New Orleans, Louisiana.

Davis, R.C, F.T. Short, and D.M. Burdick. 1998. Quantifying the effects of green crab damage to eelgrass transplants. Restoration Ecology, 6:3:297-302.

Dennison, W. C., R. J. Orth, K. A. Moore, J. C. Stevenson, V. Carter, S. Kollar, P. W. Bergstrom and R. A. Batiuk. 1993. Assessing water quality with submersed aquatic vegetation. Habitat requirements as barometers of Chesapeake Bay health. Bioscience 43:86-94.

ESRI, Inc. 1996. Arcview Spatial Analyst: Using the Arcview Spatial Analyst. ESRI, Inc., Redlands, CA, USA. pp 143.

Evan, A.S., K.L. Webb, and P.A. Penhale. 1986. Photosynthetic temperature acclimation in two coexisting seagrasses, *Zostera marina* L. and *Ruppia martima* L. Aquatic Botany, 24:185-197.

Fonseca, M.S., et al. 1998. Guidelines for the Conservation and Restoration of Seagrasses in the United States and Adjacent Waters. NOAA Coastal Ocean

Program Decision Analysis Series No.12. NOM Coastal Ocean Office, Silver Spring, MD. 222pp.

Fonseca, M. S. and J. A. Cahalan. 1992. A preliminary evaluation of wave attenuation by four species of seagrass. Estuarine, Coastal and Shelf Science 35:565-576.

Goodman, J. L., K. A. Moore and W. C. Dennison. 1995. Photosynthetic responses of eelgrass (Zostera marina) to light and sediment sulfide in a shallow barrier lagoon. Aquatic Botany 50:37-48.

Groffman, P., and R.J. Wagenet. 1994. Advances and pitfalls in expressing land qualities at different scales. Vol. 6a. Commission V Symposia. P. 662-681. In Trans. Int. Congr. Of Soil Sci., 15 , Acapulco, Mexico. July 1994. Int. Soc. Of Soil Science, Institute fur Bodenforshchung, Universitat fur Bodenhultur, Gregor-Mendel-Strasse 33, Vienna, Austria.

Granger, S., M. Brush, B. Buckley, M. Traber, M. Richardson, and S.W. Nixon. 2000. An assessment of eutrophication In Greenwich Bay. Paper No.1 in: M. Schwartz (ed.) Restoring Water Quality In Greenwich Bay: A White paper Series. Rhode Island Sea Grant, Narragansett, RI. 20pp.

Horner, R. R., E. B. Welch, M. R. Seeley and J. M. Jacoby.1990. Responses of periphyton to changes in current velocity, suspended sediment and phosphorus concentration. Freshwater Biology 24:215-232.

Koch, E. W. and S. Beer. 1996. Tides, light and the distribution of *Zostera marina* in Long Island Sound, U.S.A. Aquatic Botany 53:97-107.

Kopp, B.S., A.M. Doherty, and S.W. Nixon. 1995. A Guide to Site Selection for Eelgrass Restoration Projects in Narragansett Bay, Rhode Island. Providence RI: Technical Report to RIDEM Narragansett Bay Project. 128 pp.

Kopp, B.S. and F .T .Short. 1999. Year 1 Progress Report of the New Bedford Harbor Eelgrass Habitat Restoration Project. Submitted to the New Bedford Harbor Trustee Council, Gloucester MA. 23 pp.

Merkel, K. W .1992. A field manual of transplantation techniques for the restoration of pacific coast eelgrass meadows. Pacific Southwest Biological Services, Inc. National City, CA. 42pp.

Orth,R.J. 1975. Destruction of eelgrass, *Zostera marina*, by the cownose ray, *Rhinoptera bonasus*, in the Chesapeake Bay, Virginia. Chesapeake Science 16:205-208

Orth, R.J., Heck, K.L Jr. Van Montfrans J. 1984. Faunal communities in seagrass beds: A review of the influence of plant structure and prey characteristics on predatory-prey relationships. Estuaries 7: 339-350

Oviatt C., A. Keller and L. Reed. 2001. Annual primary production in Narragansett Bay with no bay-wide winter spring phytoplankton bloom. In review.

Save The Bay, Rhode Island DEM, Coastal Resources Management Council, National Oceanic and Atmospheric Administration. 2000. Unpublished meeting minutes from: Rhode Island Coastal Habitat Restoration Planning and Information System, Seagrass Habitat Restoration Technical Team Meeting held 5 December 2000, University of Rhode Island, Narragansett Campus.

Short F.T., Davis, R.C., Kopp, B.S., Short, C.A., Burdick, D.M. 2002.
Site-selection model for optimal transplantation of eelgrass *Zostera marina* in the northeastern US. Mar Ecol Prog Ser 227: 253-267.

Short, F.r., D. Burdick, J. Wolf, and G.E. Jones. 1993. Eelgrass in Estuarine Research Reserves along the East Coast, U.S.A., Part I: Declines from pollution and disease and Part II: Management of eelgrass meadows. NOAA, Coastal Ocean Program Pub. 107 pp.
Short, F.T. and R.G. Coles (editors). 2001. Global Seagrass Research Methods. Elsevier Science B.B. Amsterdam. 473pp

Tomasko, D.A. and B.E. Lapointe. 1991. Effects of the number of short shoots and presence of the rhizome apical meristem on the survival and growth of transplanted seagrass *Thalasia testudinum*. Contr. Mar. Sci. 52:41-48.

Valiela, I., J. Costa, K. Foreman, J.M. Teal, B. Howes and D. Aubrey. 1990. Transport of groundwater-borne nutrients from watersheds and their effects on coastal waters. Biogeochemistry 10: 177 -197.

Wagenet, R.J. and J.L. Hutson. 1996. Scale dependency of solute transport modeling/GIS Applications. Journal of Environmental Quality 25:499-510.

Wigand, C. and A.C. Churchill. 1988. Laboratory studies on eelgrass seed and seedling predation. Estuaries, 11:180-183.

Zar, J.H. 1999. Biostatistical Analysis-Fourth Edition. Prentice Hall, Upper Saddle River, New Jersey. 663 pp.

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Appendix E. RIDEM Anadromous Fish Restoration Plan

Appendix F. USFWS Coastal Wetland Restoration Analysis for Narragansett Bay

Appendix G: Literature Review of Eelgrass (*Zostera marina*) Restoration Seeding Techniques

Introduction:

The use of eelgrass seeds has been recognized by seagrass restoration scientists as an idealized restoration technique to restore eelgrass beds through out the coastal waters of North America and Europe (Fonseca et al.1999; Orth et al. 2000). The labor involved with seed collection, the quantity of potential propagules, and delivery of seed to sediments minimizes damage to extant donor sites, increases genetic diversity, and reduces project costs and labor. Although a wide body of eelgrass restoration guidance documents exists for restoration practitioners and resource managers, the literature has focused on whole plant restoration and water quality restoration planning, for example: Fonseca et al. (1999) and Short and Coles (2001). With the potential for significant resources becoming available from federal restoration programs (Restore Americas Estuaries & NOAA Restoration Partnership, US Department of Agriculture-2002 Farm Bill, U.S. Coastal Restoration Act of 2000), the development of eelgrass restoration programs is anticipated to increase in coastal regions through out the United States. Ongoing and future restoration programs will incorporate eelgrass seeding methods into overall restoration project design. The need for a current review of eelgrass seeding techniques is apparent. Consequently, the purpose of this paper is to synthesize the results of eelgrass seed ecology and seed restoration literature from North America and Europe and discuss the implications for establishing successful seed restoration. Keeping the restoration practitioner in mind, this paper will review and discuss the following aspects of eelgrass seed ecology and current seed restoration methods:

1. **Reproductive strategy of natural populations**
    - germination studies conducted in situ and under laboratory conditions,
    - seedling development and survival,
2. **The role of predators on seed abundance and successful seedling establishment, including predator control techniques;**
3. **Flowering shoot collection/seed storage methods, and a**
4. **Current review of North American seed restoration programs**.

A Rationale for Seed Restoration Techniques:

It has been well-established that seagrass species-such as eelgrass, support diverse communities of finfish, shellfish and invertebrate species (Orth et. al., 1984; Bell and Pollard, 1989; Heck et. al., 1989 and Howard et. al., 1989). Recognizing these values, resource managers and restoration practitioners have utilized a variety of whole plant restoration techniques to watershed-based water quality improvement projects to restore eelgrass populations for at least the past fifty years (Lamson, 1947). Given the continuing decline of seagrass populations, a phenomenon that has been described in the published literature, I contend that restoration efforts will need to scale up projects to ecologically meaningful levels. The labor, costs, and scale of eelgrass projects using whole plant transplanting techniques, such as sods and anchored shoots have not yet been successful in returning populations to an approximation of historic. Furthermore, efforts to install pollution controls to improve water quality conditions and allow for natural eelgrass will still need to incorporate the ability of species to recolonize new habitats. Some researchers contend that eelgrass recovery will proceed slowly in these instances, given the species reproductive strategy. The lack of eelgrass re-colonization following the wasting disease epidemic in the 1930's through out coasts of the mid-Atlantic may be evidence of this (Orth et al. 1994).

Due to the great amount of potential reproductive material derived from eelgrass seeds and the ability to disperse seed material over large areas, eelgrass seeding techniques will be critically important to restoration efforts whose goals are to significantly increase restoration project areas. Whole plant restorations conducted in North America have been carried out at the sub-acre level with few transplants exceeding 10 acres in size. Current estimates indicate that seed restoration projects have the potential to increase restoration areas from the sub acre level to the 10-30 acre level or greater (Traber, personal communication). Large scale eelgrass restoration projects that are currently being planned in the U.S include: a 70 acre restoration project in Laguna Madre, Texas, and in RI, USA the US Army Corps of Engineers are considering eelgrass restoration projects that will total 180 acres. I believe that the amount of whole plant shoots that would need to be collected from donor locations to support this level of restoration would cause significant detriment to donor sites. Consequently, incorporating seed restoration technologies into large scale restoration programs will be paramount to achieve large restoration goals. Comprehension of eelgrass reproductive biology and the results of seed restoration experiments conducted to date will be necessary to effectively harness this potential.

I. Reproductive Strategy of natural Populations of Z. marina in North America

Studies conducted along the Pacific and Atlantic coasts of North America tested whether differences in reproductive strategy existed among eelgrass populations along its full latitudinal range of distribution (Phillips, 1983; Keddy, 1987). Reproductive strategies of eelgrass and other submerged aquatic vegetation are generally characterized by asexual or clonal vegetative growth and sexual reproduction via fertilized seeds. The results of this research demonstrate the variability in reproductive maintenance techniques eelgrass populations utilize. Understanding eelgrass reproductive strategy in naturally occurring populations is crucial to planning successful restoration projects.

Phillips (1983) demonstrated that the three distinct life history strategies of eelgrass in North America are dependent upon habitat and biogeographical conditions by studying populations along the Pacific Ocean of North America. At it's southern latitudinal extreme, eelgrass plants are unable to survive stresses induced by high summer water temperatures (Gulf of California populations), causing eelgrass plants to exhibit an annual life history strategy. In these populations 100% of eelgrass plants flower, develop seeds, and perish. Subsequent seed germination in fall and winter allows eelgrass populations to persist from year to year. At the northern edge of its range, eelgrass populations exhibit a perennial life history strategy; however, physical disturbances (ice scour and freezing temperatures) cause increased incidence of flowering plants and increased rate of seed germination. Population maintenance is also dependent upon sexual reproduction. Along the central portion of eelgrass distribution-populations may exhibit both perennial life history strategy and annual growth form-which is dependent upon habitat type. Intertidal habitats with greater fluctuation of physical and chemical extremes (temperature, salinity, and relative humidity, exposure, faunal predation) cause eelgrass plants to persist on annual basis (Keddy, 1987). Similar to extreme northern and southern latitude populations-these plants are characterized by 100% flowering shoots and increased seed germination rates. Where as, eelgrass growing in subtidal habitats exhibit a perennial growth form dominated by asexual reproduction, as subtidal habitats are protected from extreme stressors and disturbance.

In summary, according to Phillips (1983), there is a greater incidence of flowering shoots in Pacific coast populations growing at the latitudinal extremes, distributed in intertidal habitats, and that follow an annual growth form. In temperate climates of the Pacific Coast, subtidal eelgrass habitats have fewer flowering shoots and annual growth forms are rare-populations are maintained by clonal vegetative shoot expansion. Eelgrass populations of Atlantic coasts display similar life history strategies as Phillips (1983) reported. Phillips (1983) reports that eelgrass populations in the Atlantic exhibited greater proportion of flowering in intertidal habitats than in subtidal habitats.

The amount of seeds produced by extant eelgrass beds and the timing of seed release, are paramount to planning seed restoration efforts. If restoration efforts are to be sustainable, seeds collected from donor sites should not negatively impact the donor site’s ability to maintain its population. Seed collected for restoration activities needs to be timed with natural reproductive timing of eelgrass beds in a given region. Knowledge of the factors that influence seed production is presented.

Research conducted by Keddy (1987) reported the reproductive potential of annual vs. perennial eelgrass populations in Nova Scotia. Keddy (1987) defines reproductive potential as the quantity of flowers and the number of spathes per shoot. Annual populations of eelgrass in Nova Scotia represent the highest reproductive potential of any other populations, with over 78,000 seeds/m2. Seed production of annual eelgrass was seven times that of perennial populations in this study. Perennial populations in Chesapeake Bay have significant less percent of total shoots that flower and produce seeds with flowering shoots accounting for 11-19% and an average of 8127 seeds/ m^2 (Fishman and Orth 1996). **See Table 1.** for a synopsis of studies that report reproductive output of eelgrass populations across its range of distribution.

The timing of seed production along the Atlantic Coast varies at local and regional scales but is largely controlled by rising water temperatures. Seed production follows a South to North latitudinal gradient, beginning in April and

Location	**Seeds/m-2**	**Researcher**
S. Oyster Point, NY	2776	Gates (1983)_
Smith Point, NY	5818	Gates (1983)_
Great South Bay, NY	1802	Churchill, 1978
Chesapeake Bay, NY	8127	Silberhorn et. al (1983)
El Infienello, Mexico	19883	Phillips et. al (1983)
North Adriatic Sea, Italy	1700	Curiel et al. (1997)
Nova Scotia, Canada	78000	Keddy (1987)

Table 1. Seed Production reported by researchers in North America and Europe

	Seed Maturity	**Anthesis**	
North Carolina	April	February	Dillion (1971)
Virginia	May	April	
New York	Nune	May	Churchill and Riner (1978)
Nova Scotia	July	June	Keddy and Patriquin (1978)

Table 2. Seed Maturity and Anthesis along Latitudinal Gradient of E. North America

May in North Carolina and July and August in Nova Scotia, Canada (Fishman and Orth 1996). Granger et al. (2000) reported a one month lag period in seed release between an eelgrass population in Narragansett Bay and a site in Rhode Island Sound (an estimated distance of 15 miles). See **Table 2.** For a representation of seed maturity and anthesis dates across its range of distribution.

Studies carried out in the North Adriatic Sea, Italy observed the longest flowering period of any other previously studied location. Curiel et al. (1987) reported a 6 month flowering period and concluded that seed production (1600-1700 seeds/m^2) and flowering shoot abundance (2-3% of total shoots) was very limited for that region.

Factors affecting reproduction

Intraspecific competition, disturbance, and regional adaptive history are factors that have been quantified in assessing the reproductive potential of eelgrass. Keddy (1987) concluded that intraspecific competition existed between perennial and annual populations where they co-occurred. Competition for available sunlight limited reproductive potential of annual eelgrass by perennial eelgrass shoots through shading effects. Keddy (1987) suggested that as water depth increases, plants put more energy into growing towards the water surface and less energy flow into flowering shoot production. In the comparative study of eelgrass seed germination and growth by Lent et al. (1995) seeds were grown in tanks with sediment pots. This study concluded that eelgrass flowering was related to a higher demand for Nitrogen and that genetic variation did not differ between perennial and annual populations. Transplant studies have indicated that phenotypic (environmental factors) control reproductive potential while genetic influences showed less control (Keddy 1987, Lent et al. 1995). For instance, seeds harvested from annual eelgrass populations will give rise to both annual and perennial growth forms depending upon the environment they are planted in.

Both Phillips (1983) and Silberhorn et al. (1983) discuss the affect of physical stress as a potential mechanism for increasing flowering shoot production. Disturbance caused by a migrating shoal that over ran a study site was believed to have increased reproductive potential at a location in Chesapeake Bay where the disturbed site had 25% flowering shoots compared to 2-3% flowering at nearby undisturbed sites (Silberhorn et al. 1983). Curiel et al. (1997) predicted that the low reproductive potential of eelgrass in the North Adriatic Sea was negatively impacted by poor adaptation to the climate of the region, presumably due to high water temperatures. Ewanchuck (1995) speculated that sexual reproduction was controlled by environmental stress (temperature, salinity, and desiccation), based upon interpretation of results from Phillips et al. (1983). Keser et al. (2002) observed a trend over a 15 year period, where seed bearing shoot abundance disappeared during months of August and September from 1991 to 2002. During this study a significant increase in mean annual and daily seawater temperatures were also observed. However, rising water temperature is an environmental cue for senescence that occurs once plants have completed their reproductive cycle. Further studies of disturbance should be carried out to support the disturbance- reproduction hypothesis.

Mapping surveys conducted in Chesapeake Bay has led some researchers to believe that natural recolonization of eelgrass does not occur over great distances from existing eelgrass beds. New eelgrass colonies have been observed up to 7.3 km from existing populations, suggesting that natural recovery of eelgrass populations to suitable habitats may be hampered by the ability of natural beds to disperse reproductive material (Orth et al. 1994). The same study concluded that the limited reproductive ability of eelgrass might explain the lack of recovery of populations impacted by the wasting disease epidemic of the 1930’s. Given these constraints to natural recolonization, physical restoration via seeding or whole plant transplanting will remain a vital tool for restoration practitioners. Comprehension of seed/shoot dispersal and the ability of natural recolonization are therefore paramount to effectively plan future restoration projects. Natural dispersal characteristics and recolonization potential of extant beds should be understood to better target areas for restoration that meet habitat requirements but will not be recolonized due to metapopulation “bottlenecks”, such as that caused by the lack of seed/shoot sources.

The dispersal of flowering shoots and propagules have been studied by a variety of researchers over the past two decades. Eelgrass seeds exhibit atelochory, defined as a lack of physical dispersal characteristics-which may in fact be eelgrass’s physical dispersal characteristic. Seeds are barrel shaped and negatively buoyant, demonstrating that their morphology is engineered for short distance dispersal and quick sinking. Orth et al. (1994) give a good account of seed dispersal potential in natural populations of Chesapeake Bay. They concluded that eelgrass seeds settle rapidly from the water column and move only a few meters by tidal currents, supporting the dogma that eelgrass seeds are not exported long distances. Over the course of this 3 year study, 15 m was the maximum distance that seeds were moved. In these field experiments seeds were hand broadcasted into 5 m^2 plots. Sampling conducted after germination, revealed that 80% of all seedlings were found within the 5m^2 plots. Additionally, laboratory flume experiments were set up to determine threshold velocities for seed movement. Using the laboratory derived values, geophysical effects (tidal current velocities) that occurred at the study plots should have been strong enough to move seeds greater distances than observed. Quick seed burial and drag forces influenced by benthic microtopography are probable mechanisms that lead to seed burial and consequent retention in these energetic sites. Furthermore, results from genetic flow studies conducted by Ruckelshaus (1996) concluded that pollen dispersal from naturally flowering eelgrass populations was also very limited. In this research pollen grain abundance decreased with distances greater than 15 m. De Cock (1980) documented that the viability of pollen grains after release was on the order of several hours.

Gates (1983) provides an excellent review of the fate of reproductive material in eelgrass populations. By quantifying total seed yield at natural sites and comparing subsequent seed counts in sediment cores through the course of a year, Gates demonstrated that seed yield did not equal seed recovery and thus seeds were being exported from the site. Gates concluded that seeds were exported by a number of potential mechanisms: bioperturbation by fauna, flowering shoot rafting, and released at the site of flowering beds. A discussion of these seed dispersal mechanisms is presented.

Flowering Shoot drifting

Since the late 1960's, researchers, such as McRoy (1968), have observed the rafting of flowering shoots in tidal currents. Flowering shoots, filled with unreleased maturing seeds, become dislodged from the rhizome and are transported by tidal currents to new locations. Researchers have suggested that this phenomenon has the potential of being a significant dispersal mechanism in delivering seeds to distant locations (McRoy 1968, Gates 1983). Gates (1983) found that seed depth and incorporation into the seed bank were different between low and high wave energy sites. Over 2000 seeds or 36% of the potential seed yield were exported from a study site with high wave exposure. Fewer seeds were recovered in higher energy areas and seeds were not found deeper than 15 cm in the sediment. In a more protected site seeds were found as deep as 25 cm in finer sediments and flowering shoots did not become attached and export seeds from the site, during the same sampling period. Other than Gates' research, few studies have quantified the significance of this. Seed dispersal studies conducted by Orth et al. (1994) conclude that flowering shoot rafting remains a gap in the science and needs to be quantified further.

Seed Dispersal Gas bubble floating

The phenomenon of gas bubble floatation and transport has been identified as a possibly important seed dispersal mechanism and quantified by a number of researchers (Churchill 1985). Gas bubbles adhering to the surface of the seed coat allow seeds to overcome their negative buoyancy and float on the water surface until the bond of attraction is broken between the gas bubble and seed coat. Gates (1983) reported that eelgrass seeds have been observed floating as far as 300-400 meters. Churchill (1985) demonstrate that 5-13% of seeds produced in a population can be released with gas bubbles, and transported for as long as 40 minutes and as far as 200 meters. Other researchers who have observed this phenomena attributed gas bubble seed release to a very specific set of physical conditions. A combination of warm water temperatures, high solar radiance, and reduced turbulence have been associated with gas bubble seed release in Chesapeake Bay eelgrass populations (Orth personal communication).

Dispersal of seeds via Faunal Ingestion

Eelgrass seeds have been known to be eaten directly or indirectly by a wide variety of vertebrates and invertebrates, including waterfowl, benthic infauna, mobile epifauna, and demersal fishes (Wigand and Churchill, 1983; Orth et al. 1994). Waterfowl have not been identified as important seed eaters in Chesapeake Bay due to the timing of their abundance and seed availability, though waterfowl have the potential of being more important in northern areas, according to (Short and Churchill 1983). The viability of ingested seeds has the potential to be impacted by digestive processes due to the thinness of the eelgrass seed's seed coat.

Whether dispersal mechanisms will allow sufficient reproductive material to reach a suitable habitat type for recolonization is an outstanding gap in the science of eelgrass biology. If the portion of exported reproductive material does not turn into new beds or incorporated into the local bed of origin, the genetic material does not serve an important role in population maintenance. Few studies, other than Ewanchuck (1995), have attempted to quantify this question. Ewanchuck reported that following a large die off of eelgrass in San Diego, California, seedling recruitment accounted for 20% of the total recovering population.

II. Germination studies conducted in situ and under laboratory conditions

Understanding the factors that influence seed germination are key to developing feasible seed restoration techniques. The viability of seeds collected for restoration, the techniques used to preserve seeds for planting, and the germination capacity of collected seeds under laboratory conditions and in situ are critical aspects of implementing seed restoration programs. In the following section a review of landmark eelgrass germination studies is presented. Seed germination is defined as the cracking of the seed coat and initiation of the axial hypocotyl.

Seed germination research has been carried out since 1920's (Setchell 1929, Taylor 1957). Lamounette (1977) was the first to carry out statistically rigorous laboratory experiments to describe factors that influenced eelgrass seed germination. Laboratory experiments were performed on seeds collected from Great South Bay, NY. Seeds were tested for their viability using 1% TTC (2,3,5-

triphenyl tetrazolium chloride) dye. Seed viability was used to establish the germination capacity of the population.

Lamounette stored collected seeds in 10 ppt filtered seawater that was changed monthly. Seed were sterilized HgCL2. Many researchers have utilized disinfection and sterilization techniques to minimize seed decomposition and tank fouling to reduce deleterious experimental effects. Lamounette performed a number of experiments to test temperature, salinity, seed coat scarification, and storage effects on seed germination rates. Under the various experimental regimes, Lamounette found that the greatest germination rates occurred at 15 C and at the lowest salinity levels. Disturbances to the seed coat (multiple scarification types) resulted in 4-10 times the germination rate than non scarified seed coats. Lamounette concluded that under laboratory conditions eelgrass seeds are strongly influenced by temperature, seeds do not require dormancy (cold period) to germinate, and that seeds are less viable after eight months of storage.

Previous to Lamounette's studies, Churchill and Riner (1975) determined that a mature plant produced a mean of 48 ovaries, and a maximum of 2 seeds would germinate under laboratory conditions. Lamounette concluded that 2-4% of the total ovaries produced by a plant were capable of germinating. However, viability testing indicated that 41% of the total ovaries produced viable seed. Lamounette also found that seeds could germinate after exposure to desiccation for up to 24 h.

Phillips (1983) not only presented an impressive landmark study on eelgrass reproductive strategies for the full range of pacific coast populations but also conducted a variety of laboratory seed germination experiments. Using sea water culture chambers, biweekly water exchange, and a photoperiod of 16 hours of light and 8 hours of dark, Phillips tested the effect of temperature and salinity on germination rates from five sites in 10 C and 28-30 ppt salinity.

Germination Rates (Phillips 1983)

Gulf of California, Mexico	***94%***
Carlsbad, California	**14%**
Puget Sound & Maine	**2-10%**
Alaska	**1%**

Phillips also tested seed germination rates at lower salinity levels and found that seeds from northern latitudes (Puget Sound and Maine) displayed significantly greater germination, reporting 41-65% germination. Under all of these treatments the maximum rate of seed germination was within the first 4 months. The overall results in this study indicated that seed germination rates are inhibited by high salinity in both East and West Coasts populations with the exception for Gulf of California population; and that temperature effects were shown to be insignificant. The results of these experiments have been challenged in light of more recent studies.

Gates (1983) provides an excellent analysis of seed germination capacity in the wild. Gates sampled eelgrass populations at two locations in New York. By comparing the amount of seedlings and the number of seeds present in the sediment, Gates was able to determine that the germination capacity at the two sites was 44-50%. Eelgrass populations in Peconic Bay exhibited 76%-93% germination capacity, according to Churchill (1983), and Orth and Moore (1983) determined that eelgrass populations in Chesapeake Bay exhibited 38% germination capacity. Harrison (1993) determined that seed germination was 10% under field conditions. However, Ewanchuck (1995) showed that in California eelgrass populations the potential reproductive output was very different from actual reproductive output, as measured by seedling survival. He estimated that the potential contribution of sexual production to population maintenance was less than .5% of the potential reproductive output.

In the early 1990s, researchers began to test seed germination in the laboratory using conditions more representative of wild eelgrass populations by incorporating sediments and creating the anaerobic conditions that were known to exist in seagrass bed benthos (Moore et al. 1993, Churchill 1992, Lent et al. 1995). Churchill (1992) demonstrated that eelgrass seeds could successfully germinate under anaerobic seawater conditions, resulting in germination rates ranging from 75-90%. Churchill also concluded that eelgrass seeds germinate in sediments below the redox discontinuity layer. However, Orth et al. (1994) reported that seeds planted below redox discontinuity layer had reduced germination success. Under experimental conditions, Churchill demonstrated that an eelgrass seed has the ability to extend its axial hypocotyl to a maximum of 32mm towards the sediment surface, though lengths as great as 50mm were observed by Churchill in the field. In these laboratory experiments seeds were planted in beach sediments at varying depths. 30% of planted seeds rotted at 10mm where as 62% of planted seeds rotted at 31 and 37mm depths.

Orth et al. (2000) reviewed much of the historic seagrass germination literature and concluded that the validity of earlier germination studies should be questioned in light of recent studies that demonstrate the influence of sediment O2 levels on seed germination rates. The critique appropriately points out that experimental conditions, reported in the published literature, were conducted under aerobic and soil-less conditions. Studies such as Gates (1983) and Phillips (1983) examined the effects of temperature and salinity while more recent studies now suggest that sediment O2 levels or other sediment properties may be an important determinant to seed germination (Moore et al. 1990).

Orth et al. (2000) also criticize previous laboratory experiments for not maintaining as close to field conditions as necessary to obtain ecologically meaningful experiments. They specifically cite issues related to seed scarification, seed disinfection, seed storage time, and application of acid to seed coats. Orth et al (2000) concluded that variation in seed germination under field conditions is caused by an interaction of temperature and sediment O2.

Seed Dormancy

Seed dormancy is another property of seed germination potential that has been studied by researchers in Europe and North America. Comprehension of seed dormancy characteristics in eelgrass populations is especially critical due to its implications for seed storage and seed restoration planting windows.

Harrison (1991) conducted research to test whether seed dormancy existed in an annual population of eelgrass in the Netherlands. Studies performed by Orth and Moore (1983) and Churchill (1983) reported that when warmer water temperatures in summer drop below 15 C in fall, seedlings are immediately observed in wild populations, though the majority of seed germination occurs in late winter. Harrison interprets this phasing of seedling establishment as a basis for dormancy characteristics in some seeds. Harrison (1991) determined that eelgrass seed coat color represented degrees of seed germination onset. Green seeds had weaker seed coats than brown seeds, and thus germinated earlier in the season. Greater lignin content of brown seeds makes seed coat tougher and less susceptible to early seed germination onset. In other words eelgrass seed coat color is proxy for dormancy characteristics.

Harrison's studies have been further elucidated by studies of viability, germination, color, and dormancy conducted by Linden (1992). In these laboratory experiments the germination of colored seed coats was tested with treatments of scarification- disinfection, salinity and temperature effects. Linden concluded that seed coat color indicated the different dormancy periods of eelgrass seed and the different duration that colored seeds remained viable. Linden's experiments also determined that lower salinity induced greater germination by enhancing the rupture of the seed coat. Linden postulated that most subtidal eelgrass populations do not undergo rapid or dynamic changes in salinity, suggesting that alternation of temperature may cause germination in habitats with high salinity. Linden also concluded that disinfection of seeds increased germination in green, white, tan seed coat colors but not in black.

Other Seed Germination Factors

An applicable study for the restoration practitioner was conducted by Granger et al. (2000) to determine the effects of seeding density and planting depth on germination rates. In these experiments seeds were planted during the fall in flow through seawater tanks under ambient conditions with natural eelgrass sediments. Dark gray to green colored seeds were selected for experiment and placed in .5, 1, 2, and 3 cm depths in sediment trays. Seeds planted in 2 cm of sediment exhibited 50-60% germination while seeds planted greater than 3 cm or surface planting had the least germination. This study also concluded that seeding density influenced seed germination. Seed densities of 4000 seeds/ m2 had significantly lower germination and fewer successful seedlings than 500, 1000, 2000 seeds/ m2. Seed germination was defined as the emergence and development of true leaves.

III. **Seedling Development and Survival**

Surprisingly, research findings quantifying the successful establishment of seedlings in the field is scant. While a wealth of literature exists documenting the effects of genotype and phenotypic factors on seed germination, flowering, vegetative reproduction, and population maintenance, very few studies have been able to track the development of seedlings. Knowing that under a certain treatment regime-seeds are able to germinate at a given rate is little benefit to the restoration ecologist interested in knowing what percent of successfully germinated seeds will develop into seedlings. A limited number of studies do report the fate of laboratory or field planted eelgrass seeds. The results and/or observations of these studies are presented.

In germination studies conducted by Orth et al. (1994) seeds buried too deep (greater than 15 cm) did not develop into seedlings. In the same study 3-39% of viable seeds hand-broadcasted in Chesapeake Bay sites germinated into seedlings but seedlings did not survive. The focus of this research was to quantify seed transport and dispersal, as discussed previously, and not to evaluate seed to seedling development. In the research conducted by Gates (1983) at two study sites in NY, eelgrass plants were observed from seed germination to seedling stages from fall to the following Spring. Seedling populations reached maximum numbers by November, remained steady until March, but by May all seedlings had disappeared. Gates (1983) admitted that it was difficult to distinguish seedlings from vegetative clonal shoots; and this was could be a possible source of error. Ewanchuck (1995) and Gates (1983) have also raised this problem. Ewanchuck suggested that the seasonality of recruitment of new seedlings and the sampling problems that have effected many studies might under estimate sexual reproduction in natural beds.

Research conducted by Granger et al. (2000) in laboratory settings (see above discussion) determined that seedling development did not differ among seeds collected from different locations. This seems to be substantiated by the research of Keddy and Patriquin (1978) who demonstrated that seeds harvested from perennial and annual populations and replanted gave rise to both growth forms. Morphological variation was related to environmental differences (phenotypic) and not genotypic. However, an ongoing comparative eelgrass seed germination/growth experiment being carried out by the University of Rhode Island Graduate School of Oceanography is demonstrating that seeds collected from different locations along a latitudinal gradient of the Western Atlantic will germinate and develop into seedlings at different times. Southern eelgrass populations and populations known to flower earlier due to warmer climatic conditions are quicker to germinate and the first to emerge from the sediment surface and develop into seedlings than seeds derived from northern latitude populations.

Granger et al. (2000) also reported that seedling development and lateral shoot growth was effected by seeding density. A seeding density of 4000 seeds/m2 resulted in fewer successful seedlings than densities of 500, 1000, or 2000 seeds/m2. Seedlings grown from these seed density experiments displayed the greatest number of lateral shoots per plant at lower seed densities. Regardless of the initial seed densities planted, by the end of the second growing season all of the experimental plots reached a common shoot density of 400-600 shoots/m2, a value equivalent to naturally occurring local populations. They have also observed seeds planted directly in field locations in Narragansett Bay to become seedlings and persist for several years.

Harwell and Orth (1999) conducted experimental exclosure tests on seeds planted in Chesapeake Bay (full discussion in predator section). Interestingly, they claim that 15% of seeds broadcasted on unvegetated sediments survived to seedling stage. However, this figure comes from unpublished data and the experimental design and details concerning seedling survival are not apparent. Sand-Jensen (1994) found that patches of eelgrass with more than 32 shoots survived over a 2 year study. Does this represent a minimum patch size? It remains unclear whether seedling survival in these and other studies reached a minimum patch size or if seedlings derived from seeding methods failed to develop into adult plants due to site conditions where eelgrass life requirements were not met.

One of the only rigorous studies of seedling recruitment was conducted by Ewanchuck (1995) in San Diego, California. This research is important because it supplies important clues into the role of seed/seedling recruitment in maintaining eelgrass beds in mid-latitudinal perennial plant populations. This study examined four locations in San Diego Bay, California. Total seedling recruitment at these study sites was less than 5% of the total shoot density. Seedlings in this study persisted through the duration of the study period of sixteen months. In another location, reported in this study, the decrease in leaf shoot density due to epiphytic fouling was followed by a fall and winter increase in seedling recruitment. Ewanchuck identified increased light availability and changes in edaphic factors as possible mechanisms for enhancing seedling recruitment. The number of seedlings between unvegetated areas and vegetated areas was highly significant in all study sites. This study also showed that new biomass added by seedling production was no different from total biomass added by vegetative recruits. Because of the small sampling size and limited geographic scope of this study, this research should be conducted throughout the mid-Atlantic distribution of eelgrass populations where this data is lacking. Ewanchuck concluded that vegetative reproduction is more important in additions of biomass to the populations he studied.

Role of predators on seed abundance and seedling establishment

Predation of eelgrass seeds and seedlings by a diverse range of fauna have been implicated by many researchers as a source of seed loss (Gates 1983, Orth et al. 1994). The major research on seed/seedling predation is discussed.

One of the first rigorous experimental studies was conducted by Wigand and Churchill (1988) who examined ten species for possible predacious activity on eelgrass seeds and seedlings. They determined that under experimental conditions *Ovalipes ocellatus*, *Pagurus longicarpus*, and *Panopeus herbstii* preyed on seeds when alternative food sources were not available. *Ilyanassa obsoleta, Littorina littorea*, and *P. longicarpus* also preyed on seedlings when alternative food sources were not available. Researchers used seeds and seedlings that measured between 3.6- 20mm in length in all experiments.

Crustaceans damaged up to 93% of the seeds while *P. longicarpus* damaged 93% of seedlings. The relative thinness of the eelgrass seed coat offered little protection from crabs. The size of the crustacean *P. longicarpus* was determined to be a factor in seedling damage. Individuals of *P. longicarpus* with 9 mm carapace length's were more predaceous than the 7 mm size class*. I. obsoleta* was the most predaceous snail on seedlings, inflicting rasp-like wounds along seedling leaves and cotyledonary sheath. Fish species preyed on less than 5% seeds and seedlings. P.longicarpus seed consumption was 2% when exposed to eelgrass with an alternate food source present, but increased to 19% with eelgrass alone. The researchers acknowledged that captive behavior effects and seasonal differences in feeding could have been a factor in the experiments.

Fishman and Orth (1996) conducted experiments in the York River, Virginia to quantify predator effects on seed abundance using a number of exclosure and enclosure experiments. Enclosures were stocked with likely eelgrass predators, croaker Micropogonia undulates and blue crab *C. sapides*. Exclosures were tested using full top-bottom-and side cages and cages with just sides. All experiments were conducted with 6 mm mesh size enclosures and exclosures. *C. sapides* enclosures had significantly less seed abundance than full exclusion cages. Split seeds were observed in C. sapides treatments. Within one week 96% of seeds in the uncaged treatment were lost. The least seed abundance was found in uncaged and partial cage exclosures. Seed abundance was unaffected in Micropogonia undulates enclosures. This experiment concluded that seed predation can cause up to 65% of seed losses (caused by C. sapides) and the researchers believed that predation may be dependent on predator and primary food abundance. At the time of the experiments infaunal abundance ranged from 409-602 individuals / m2 that included *Spiochaetopterus ocelots, Clymenella torquata, Nereis spp.,* capitellids, oligochaetes*, Tagelus spp.,* phoronids, and nemerteans. The abundance of infauna was thought to be very low when compared to 2000-8000 individuals/ m2 that had been recorded at unvegetated shoals in other years. The researchers hypothesized that a lack of prey could have played a role in influencing predator feeding behavior. Seeds located in existing eelgrass beds may resist predation due to the wide availability of alternative food sources, less abundant seed predators, and the refugia offered by shoot bases and rhizomes.

Davis et al. (1998) present mesocosm and in situ results that establish a link with green crab density and decreased restoration transplant survival. Although this study presents data on transplanted (whole plant) shoot damage- it is transferable to seedling predation-as developing seedlings like a recently transplanted eelgrass site lack well developed below ground rhizome system, making them susceptible to bioturbating organism such as green crabs.
In their research conducted in laboratory settings, greater green crab density in this study (g> 7 crabs/ m^2) resulted in higher shoot damage than moderate crab density (4 crabs/m^2). Crab density thresholds were established by observing crab densities in the field. Field densities were obtained by placing two 1.25 m^2 quadrats at a transplant site and observed for 1 hour. The number of observations made was not reported. Laboratory findings showed that 39% of transplanted shoots were lost within 1 week with crab densities of 4 /m^2, though

there was no evidence of shoot consumption. In a separate experiment conducted in situ, cage exclosures were erected to exclude potential bioturbators using just side exclosure caging. This type of exclosure did not eliminate green crabs from entering sites.

Green crabs forage in the top few cm of the sediment surface and mechanically damage shoots. Having a well established root system is especially crucial during the initial establishment of transplants or recently established seedlings. A well developed root system can prevent sediment penetration, as suggested by Valentine et al. (1994). Research conducted in Nova Scotia, Canada has implicated green crab bioturbation as a leading cause of natural eelgrass decline. This work remains unpublished but should be tested in other regions where eelgrass and green crab distribution overlap.

(Harwell and Orth, 1999) attempted to develop seed restoration exclosure devices to protect seeds from potential predators. Using mesh burlap bags enclosed with seeds in small packets and anchored to sediments, experiments were carried out in the field and laboratory. Seedling survival in the field was reported to be 41-56% by using the burlap bag exclosure, 5-15% survival for unprotected seeds, and 50% survival for laboratory conditions. However, after eight months, seedlings were killed when seed bags were buried by 50 cm of sediment. Interestingly, Harwell and Orth report that a 3-12 fold increase in seedling success can be achieved by using these burlap exclosure even though all seedlings eventually perished. Unpublished data, reported by these researchers, indicate that small .25 meter plots of eelgrass remained from seed bag exclosures after 14 months. Churchill et al. (1978) reported that a seed tape enclosure material did not enhance seed germination.

IV. North American seeding techniques and current seed restoration programs

Few studies have been published that report seed collection, storage, and winnowing methods to support seed restoration activities. Granger et al. (2000) document methods for collection, storage, and winnowing to support laboratory experiments on seeding density. In Narragansett Bay flowering shoots were collected once seeds began releasing from spathes. 50 person hours were required to harvest enough flowering shoots to equal 500,000 seeds processed. Shoots were stored in flow through seawater tanks under ambient conditions. 80% of seeds were released from spathes in the storage tanks during first four weeks after collection. Tanks were drained weekly and filtered through a 500 micron mesh filter bag. Seeds were then screened and winnowed to separate out detritus. Once seed material was cleaned and separated out, they were transferred to oxygenated flow through tanks. Seeds were held under ambient conditions for several months before germination began to occur.

This is similar to methods performed by Harwell and Orth (1999) who also stored seeds in circular ambient oxygenated seawater tanks but used shade cloth to limit sunlight into seed chamber. Seed enclosure experiments conducted by Harwell and Orth (1999) achieved similar results with 100% of seeds releasing within 6 weeks of collection. Harwell and Orth (1999) selected seeds for their experimental work using seed coat rigidity guidelines based on Moore et al (1993). These researchers also report that their use of seed bag enclosure devices, consisting of 15 seeds per bag, employed half the labor of transplanting whole shoot 15 plant bundles of eelgrass. Churchill et al. (1978) reported that a seed tape enclosure material did not enhance seed germination.

Besides research and restoration being conducted in Narragansett Bay (University of Rhode Island-Graduate School of Oceanography) and Chesapeake Bay (Virginia Institute of Marine Science) no other published literature exists on current seed restoration programs and techniques. However, a number of eelgrass seed restoration projects have either been completed, are underway, or are planned for in the near future. The following projects appear in gray literature sources. The Peconic Estuary Program Comprehensive Conservation and Management Plan reports in its Federal FY 1994 Action Plan that eelgrass seed collection activities were carried out in East Hampton Harbors to establish an eelgrass seed bank. This project was conducted by the Marine Program of Cornell Cooperative Extension and the East Hampton Natural Resources Department. The success of this seed collection and seeding project was not reported.

A seed restoration project is currently being carried out by the Jacques Cousteau National Estuarine Research Reserve, the Alliance for a Living Ocean, Rutgers University, Chesapeake Bay Foundation, and NOAA-Restoration Center. In this multi faceted eelgrass restoration project in Little Egg Harbor, NJ, eelgrass seeds are collected from wrack deposited flowering shoots. Flowering shoots are sorted and allowed to mature until seeds are released. The project does not report how seeds are stored or for how long. Seeds are placed in biodegradable containers and inserted into sediments at planting sites. Again, details are not given for this project, nor when this will be carried out. Using a seawater green house in Sequim, Washington, the Sequim Marine Sciences lab is working on methods to increase seed production of flowering shoots by manipulating light and temperature conditions (Seattlepi.com September 1, 2000).

Large scale eelgrass seeding projects have been conducted in Chesapeake Bay over the past five years utilizing hand cast techniques. Results from these projects have not yet appeared in the published literature. However, restoration success is being achieved in these projects (Orth personal communication).

Innovative techniques incorporating mechanized seed planting have been developed by researchers from the University of Rhode Island Graduate School of Oceanography. Using an underwater towed sled, eelgrass seeds are pumped into a modified seed drill and planted within one centimeter of the sediment surface. This planting system provides a number of benefits:

- large quantities of seed can be planted over a large surface area in a small amount of time
- seeds are delivered below the sediment surface, reducing exposure to potential bioturbators
- and seed planting density can be controlled by altering pumping rates

Initial results of seed germination rates from mechanized planting with gel media and hand planted seeds in field plots and aquarium trials did not show significant advantages of the mechanized technique. Reduced germination rates were attributed to the organic enrichment effects caused by the Knox gel media type (CICEET, 2002). Since these initial trials, experiments have been conducted on alternative gel types including: Sodium bentonite, fumed silica, and Potassium propenoate propenamide copolymers. Germination conducted in these laboratory experiments were achieving over 50% germination (defined as percent of seeds producing an emergent seedling (CICEET, 2002).

Seeding experiments were conducted with the seeding sled in the summer and fall of 2001 using the Knox gel but at a reduced gel to seed ratio. Researchers were able to collect over 1.75 million seeds for planting into two locations with a total restoration area of 800 m^2. Seeds were collected and stored using methods reported in Granger et al. (2000). Investigators were able to plant seeds at a rate of 100 m^2 area/hour and planting at approximately 2000 seeds/m2. Results from these two sites have shown mixed results though a full report of these trials is currently in press. While both sites ultimately failed, germination and seedling development at one of the locations did achieve initial success with seedlings persisting for eight months. This site ultimately failed in the spring of 2002, though site failure was attributed to disturbance from an unusually high abundance of transient spider crabs that had moved into the restoration area. It should be noted that immediately adjacent to the seeding plots, whole plant transplants carried out by this author also experienced significant bioturbator damage during the same time period as the uprooting of seedlings from the seed trial. Experiments to incorporate alternative gel media continue to be investigated by these researchers.

Currently, eelgrass seeding using the mechanized planting device is being incorporated with whole plant transplanting in a large scale restoration project in Narragansett Bay RI. This effort is being carried out by URI GSO, Save The Bay, and the US Department of Agriculture. Experiments are now underway to quantify the effects of seeding areas within transplanted plots and to test the effects of different types of pumping gel media. Results from this partnership will elucidate the effectiveness of seed techniques in the Northeastern United States. URI researchers are in the process of publishing a eelgrass seeding guidebook based upon the results of their experiments that will be directed at restoration practitioners.

Conclusion and Future Directions for Eelgrass Seed Restoration Programs:

The use of eelgrass seeds as a sustainable alternative to whole plant transplanting is a new and potentially important method of eelgrass restoration. This technique can minimize the cost per acre of restoration while reducing the impacts that current techniques have on donor sites. Three major themes emerge from this synthesis of eelgrass seed ecology literature and current seed restoration programs that are relevant to improving eelgrass restoration success:

1. Filling the major gaps in the science of seed ecology and mechanistic understanding of eelgrass reproductive strategies as they relate to seeding techniques and restoration planning
2. Better reporting and coordinated monitoring of efforts and experiments that conduct seed restoration projects.
3. Continuing to conduct seed restoration projects throughout the full range of eelgrass distribution

In the following section I summarize ways of addressing these three themes and pose key questions that remain unanswered.

1. Mechanistic Understanding and Gaps in the Science:

Gaps in the mechanistic understanding of characteristics of eelgrass reproductive biology and population dynamics have been identified in this paper. In this section some of the major gaps in the science are discussed.

A. Seedling Development

Better monitoring of seedling development at restoration locations and in naturally occurring eelgrass populations is necessary. Additional studies will be necessary to further quantify the role of seedlings in bed recovery. Because seed viability decreases after a short period of time, less than a year, reproductive material removed from a location to support restoration efforts could have an impact on the donor location's ability to recover from disturbance events. It will be very important for restoration practitioners to plan how much reproductive material can be extracted from eelgrass seed donor sites without causing substantive impacts on existing populations. Further research is thus required to quantify the importance of seed/seedlings in bed maintenance. The research reviewed in this paper suggests that collection of reproductive shoots, especially in mid latitudes, should pose limited risk to extant populations. However, In the interest of conservative restoration approaches, collection of flowering material from a variety of donor locations will minimize the possible impacts of removing reproductive material that could be important in local eelgrass bed maintenance.

B. Eelgrass Seed Genetics and Implications for restoration

Better comprehension of genetic differences and consequent plant characteristics can benefit restoration efforts. Comparative studies of seeds used from different locations should continue to be studied, considering the gap in the science of understanding regional differences in phenotypic and genotypic characteristics of eelgrass seeds. For instance, can seed material collected from southern regions of the Mid-Atlantic be utilized in restoration efforts in more northern latitudes. Since flowering shoot initiation occurs earlier in these reaches, planting southern seeds in northern latitudes can occur earlier with expected germination and seedling development also occurring earlier. This may give eelgrass seedlings a competitive survival advantage over late developing genetic material, and allow seedlings to establish and withstand known disturbance effects such as bioturbators that have been observed in removing restoring seedling patches.

Studies conducted along the Pacific coast have shown reduced genetic diversity of transplanted eelgrass populations compared to naturally occurring locations. However, Ruckelshaus (1996) showed that random genetic drift was not a cause for population differentiation, though high rates of clonal propagation resulted in effective densities low enough for drift to play an important role in population structure. Ruckelshaus cautioned that the genetic uniformity of eelgrass populations, presumably due to the lack of genetic isolation, make the species especially vulnerable to large scale disturbances, such as the wasting disease epidemic in the 1930s.

Increased understanding of the influence of genetic diversity on transplant and seeding success has the potential to increase restoration success. Keddy and Patriquin (1978) showed that seeds collected from perennial and annual populations gave rise to both growth forms when planted in suitable habitats. Results of transplant experiments conducted by Keddy (1987) also re-affirmed this dogma that suggests that morphological variation is related to environmental differences (phenotypic) and not genotype. However, Keddy and Patriquin (1978) also reported that shoots grown from seeds collected from three distinct populations showed some morphological variation, especially in leaf characteristics. Shoots from one population had narrower leaves than plants grown from seeds from the other populations. Therefore, there may be some rationale for a genotypic response. For Instance, if certain populations of eelgrass are better adapted to warmer water temperatures and reduced water quality conditions, which has been suggested in populations in Chesapeake Bay; then this seed material may offer advantages when incorporated into restoration programs in northern locations with less tolerant genotypes.

C. Natural Eelgrass Recolonization and Dispersal

Research reviewed in this paper support the fact that eelgrass sexual reproduction, especially in populations located in the mid-latitudes of its distribution, exerts a limited influence in recolonization to distant locales. It is important to restoration ecologists to understand the reproductive biology of eelgrass populations for restoration planning initiatives that incorporate natural recruitment and recolonization as a restoration tool. For instance, what are the chances of extant beds in supplying the necessary genetic material to begin new beds, how much of that material i.e. seedlings are necessary to generate a new bed? Understanding the fate of reproductive material is also important to evaluate the impacts of flowering shoot collection for seed restoration projects. Reproductive materials generated by these natural eelgrass populations (excluding rafting flowering shoots and gas bubble transported seeds) would appear to be used locally in maintaining the seeds' bed of origin. This is suggested in Ewanchuck (1995). Propagules that are not consumed by potential predators (benthic infauna etc.) and are exported from a flowering population enter a game of chance. Will long distance transport allow viable seeds to fall out of the water column, incorporate into sediments, germinate, and become seedlings in an appropriate habitat? Another important question related to the success of natural re-colonization is whether there exists the necessary genetic variability of new eelgrass colonies to persist in new locations.

D. Bioturbator Impacts on seeds and seedlings

Evaluation of seed/seedling predation has been limited to few experiments as outlined in previous sections of this paper. Predation pressure on seeds and seedlings will need to be understood by restoration practitioners in order to account for seed losses, learn to avoid problem sites, or to implement seed protective measures.

2. Better Reporting and Expanded Monitoring Programs:

Every restoration project is an experiment. Well designed restoration monitoring programs will result in good adaptive management and advance the science of eelgrass seed restoration ecology. Eelgrass restoration monitoring programs will benefit from adapting approaches taken from salt marsh restoration monitoring programs where standardized protocols and cooperative monitoring programs are now being implemented over vast geographic regions (Roman et al. 2001). Standardizing these approaches will allow for the adequate information and technological transfer to restoration practitioners outside of the academic sector.

Restoration practitioners, such as Non governmental organizations, are able to mobilize resources in ways to cost effectively implement seed and whole plant restoration projects. They can increase the level of success of seed restoration projects by increasing the level of replication in carrying out projects over multiple time frames and locations. However, seagrass ecologists will need to work with these groups to develop logistically possible monitoring programs and supply the necessary expertise to analyze and interpret monitoring data.

Seed restoration projects have been observed to be very successful in Chesapeake Bay though no published literature have reported the level of success or failure projects are attaining. Success criteria also need to be better articulated and possibly standardized by researchers. It is critical that researchers work cooperatively together to publish restoration findings in peer reviewed literature and at least in gray literature sources. It can be acknowledged that some restoration practitioners and academic researchers may be hesitant to publish and report project failures due to the possible negative effect this research may have in obtaining potential future funding. However, learning from these failures is crucial to increasing our ability to apply adaptive management techniques and increase the body of knowledge of seed restoration ecology.

3. Continued Restoration Projects and Experiments:

The best way to increase the success of eelgrass restoration projects is to continue to conduct them using the best available technologies and to continue to experiment with new techniques. Learning from the success and failures will ultimately result in better results. For instance, some researchers have suggested that adult plants and seeds should be restored together to create new beds with multiple age classes (Harwell and Orth, 1999).

Positive feedback loops exist within eelgrass communities that reinforce suitable habitat conditions for further recolonization and growth. Irlandi (1986) describe the effects of seagrass shoot density and cover on sediment stabilization in areas adjacent to vegetated areas. The positive feedback of eelgrass colonization in high wave energy areas is characterized by the increasing wave energy dampening effect that new plants can have on adjacent unvegetated areas. The establishment of eelgrass cover in a suitable unvegetated location will produce environmental conditions that are beneficial for both whole plant transplants and seedlings. The restoration projects being conducted by URI, Save The Bay, and USDA to combine seeding and transplanting will provide critical data for future restoration efforts. These type of projects should also be extended to other regions of the species distribution for comparative purposes.

Eelgrass restoration programs have relatively low success rates compared to other coastal habitat restoration projects (Fonseca et al. 1999). Site selection procedures have been identified as one of the most important factors in whole plant transplant site failures. Seeding projects should take advantage of recent advances to improve eelgrass restoration success via cost effective site selection models by implementing small scale test sites into the restoration planning phases. This will save time and reduce overall costs of implementing the full scale restoration. Knowing whether a location can actually support eelgrass, preferably for more than once growing season, is crucial to understanding whether site conditions caused the failure of a restoration or the restoration technique being used. This allows researchers and practitioners to quantify the effects of the restoration techniques being used on the success of the project.

Seed restoration projects will benefit by increasing the number of sites tested in order to incorporate the high level of variability and possibility of site failure due to external effects such as bioturbators. Seeding projects should be conducted at locations that are known to meet minimum habitat life requirement criteria. Test plots over multiple years and at multiple sites will assist in determining appropriate locations to conduct larger scale restoration projects.

References:

Ackerman, J.D. 1993. Pollen germination and pollen tube growth in the marine angiosperm, Zostera marina L. Aquatic Botany, 46:189-202.

Bell, J.D. and Pollard,D.A. 1989. Ecology of fish assemblages and fisheries associated with seagrasses p. 565-609. In Biology of Seagrasses. Elsevier, NY

Curiel, D., A. Rismondo, F. Scarton, and M. Marzocchi. 1997. Flowering of Zostera marina in the Lagoon of Venice (North Adriatic, Italy). Botanic Marina, 40: 101-105.

Churchill, A.C. 1992. Growth characteristics of Zostera marina seedlings under anaerobic conditions. Aquatic Botany, 43:379-392.

Churchill, A.C, G. Nieves, and A.H. Brenowitz. 1985. Flotation and dispersal of eelgrass seeds by gas bubbles. Estuaries, 8(4):352-354.

Cooperative Institute for Coastal and Estuarine Environmental Technology (CICEET). 2002. " Density-dependent effects on grazing and success for seed generated seagrass (Zostera marina L.) plants." CICEET Progress Reports for 2/2002 and 7/2001.

Davis, R.C, F.T. Short, and D.M. Burdick. 1998. Quantifying the effects of green crab damage to eelgrass transplants. Restoration Ecology, 6:3:297-302.

De Cock, A.W.A.M. 1981. Development of the flowering shoot of Zostera marina L. under controlled conditions in comparison of the development in two different natural habitats in the Netherlands. Aquatic Botany, 10:99-113.

Ewanchuck, P.J. 1995. Population growth of eelgrass (Zostera marina l.): The relative importance of sexual versus asexual reproduction. M.S. thesis, San Diego State University, San Diego, California, 144 pp.

Fisher, D. 2000. Whidbey eelgrass project scores unique success. Seattle Post, Sept. 1.

Fishman, J.R, and R.J Orth. 1996. Effects of predation on Zostera marina L. seed abundance. J. Exp. Mar. Biol. Ecol., 198:11-26.

Gates, K.L. 1984. A Comparison of two Zostera marina L. seed banks in Great South Bay, New York. M.S. thesis, Adelphi University, 62 pp.

Granger, S.L., M.S. Traber, and S.W. Nixon. 2000. The influence of planting depth and density on germination and development of Zostera marina L. seeds. Biol. Mar. Medit. 7(2): 55-58.

Harrison, P.W. 1991. Mechanisms of seed dormancy in an annual population of

Zostera marina (eelgrass) from the Netherlands. Can. J. Bot., 69:1972-1976.

Harwell, M.C and R.J. Orth. 1999. Eelgrass (Zostera marina L.) seed protection of field experiments and implications for large-scale restoration. Aquatic Botany, 64:51-61.

Heck, K.L Jr., Able, K.W., Roman, C.T., Fahay, M.P. 1995. Composition, abundance, biomass, and production of macrofauna in a new england estuary: comparisons among eelgrass meadows and other nursery habitats. Estuaries 18(2) 379-389.

Irlandi, E.A. 1996. The effects of seagrass patch size and energy regime on growth of a suspension-feeding bivalve. Journal of Marine Research, 54, 161-185.

Keddy, C.J. 1987. Reproduction of annual eelgrass: variation among habitats and comparison with perennial eelgrass (Zostera marina L.). Aquatic Botany, 27:243-256.

Keser M., Senarton, J.T., Vozarik, J.M. and J.F. Foertch. 2002. Changes in Eelgrass (Zostera marina L.) Density, biomass, and Reproductive characteristics in Long Island Sound near Millstone Point, CT (USA): 1985-2000. In press.
Lamounette, R.G. 1977. A study of the germination and viability of Zostera marina L. seeds. M.S. thesis, Adelphi University, 42pp.

Lamson, A.L. 1947. Annual report January 1, 1947 – December 31, 1947. Connecticut State Board of Fisheries and Game Eelgrass Restoration Program, 16 pp.

Linden, R.E. 1992. Factors influencing the germination of Zostera marina L. (Eelgrass) seeds. M.S. thesis, University of Connecticut, 70pp.

Moore, K.A, R.J. Orth, and J.F. Nowak. 1993. Environmental regulation of seed germination in Zostera marina L. (eelgrass) in Chesapeake Bay: effects of light, oxygen and sediment burial. Aquatic Botany, 45:79-91.

Orth, R.J., M.W. Luckenbach, and K.A. Morre. 1994. Seed dispersal in a marine macrophyte: Implications for colonization and restoration. Ecology, 75: 1927-1939.

Orth, R.J., M.C. Harwell, E.M. Bailey, A. Bartholomew, J.T. Jawas, A.V. Lombana, K.A. Moore, J.M Rhode, and H.E. Woods. 2000. A review of issues in seagrass seed dormancy and germination: Implications for conservation and restoration. Mar. Ecol. Prog. Ser., 200:277-288.

Orth, R.J., Heck, K.L Jr. Van Montfrans J. 1984. Faunal communities in seagrass beds: A review of the influence of plant structure and prey characteristics on predatory-prey relationships. Estuaries 7: 339-350

Phillips, R.C., Grant, W.S., and McRoy, C.P. 1983. Reproductive strategies of eelgrass (Zostera marina L.). Aquatic Botany, 16: 1-20.

Roman, C.T., M.J. James-Pirri, and J.F. Heltshe. 2001. Monitoring Salt Marsh Vegetation: A Protocol for the long-term ecosystem monitoring program at Cape Cod National Seashore. CACO files.

Ruckelshaus, M.H. 1996. Estimation of genetic neighborhood parameters from pollen and seed dispersal in the marine angiosperm Zostera marina L. Evolution, 50(2):856-864.

Short F.T., Davis, R.C., Kopp, B.S., Short, C.A., Burdick, D.M. 2002. Site-selection model for optimal transplantation of eelgrass *Zostera marina* in the northeastern US. Mar Ecol Prog Ser 227: 253-267.

Silberhorn, G.M, Orth, R.J. and Moore, K.A. 1983. Anthesis and seed production in Zostera marina L. (eelgrass) from the Chesapeake Bay. Aquat. Bot., 15: 133-144.

Van Lent, F. and J.M. Verschuure. 1995. Comparative study on populations of Zostera marina L. (eelgrass): experimental germination and growth. J. Exp. Mar. Biol. Ecol., 185:77-91.

Wigand, C. and A.C. Churchill. 1988. Laboratory studies on eelgrass seed and seedling predation. Estuaries, 11:180-183.

Appendix H. Rare Animals and Plants in Rhode Island

RARE NATIVE ANIMALS OF RHODE ISLAND
Revised: March, 2006

ABOUT THIS LIST

The list is divided by vertebrates and invertebrates and is arranged taxonomically according to the recognized authority cited before each group. Appropriate synonomy is included where names have changed since publication of the cited authority.

The Natural Heritage Program's *Rare Native Plants of Rhode Island* includes an estimate of the number of "extant populations" for each listed plant species, a figure which has been helpful in assessing the health of each species. Because animals are mobile, some exhibiting annual long-distance migrations, it is not possible to derive a population index that can be applied to all animal groups. The status assigned to each species (see definitions below) provides some indication of its range, relative abundance, and vulnerability to decline. More specific and pertinent data is available from the Natural Heritage Program, the Rhode Island Endangered Species Program, and the Rhode Island Natural History Survey.

STATUS. The status of each species is designated by letter codes as defined:

(FE) Federally Endangered (7 species currently listed)

(FT) Federally Threatened (2 species currently listed)

(SE) State Endangered Native species in imminent danger of extirpation from Rhode Island. These taxa may meet one or more of the following criteria:

1. Formerly considered by the U.S. Fish and Wildlife Service for Federal listing as endangered or threatened.
2. Known from an estimated 1-2 total populations in the state.
3. Apparently globally rare or threatened; estimated at 100 or fewer populations range-wide.

Animals listed as State Endangered are protected under the provisions of the Rhode Island State Endangered Species Act, Title 20 of the General Laws of the State of Rhode Island. This law states, in part (20-37-3):

"No person shall buy, sell, offer for sale, store, transport, export, or otherwise traffic in any animal or plant or any part of any animal or plant whether living or dead, processed, manufactured, preserved or raw if such animal or plant has been declared to be an endangered species by either the United States secretaries of the Interior or Commerce or the Director of the R. I. Department of Environmental Management."

(ST) State Threatened Native species that are likely to become State Endangered in the future if current trends in habitat loss or other detrimental factors remain unchanged. In general, these taxa have 3-5 known or estimated populations and are especially vulnerable to habitat loss.

(C) Concern Native species not considered to be State Endangered or State Threatened at the present time, but are listed due to various factors of rarity and/or vulnerability. Species listed in this category may warrant endangered or threatened designation, but status information is presently not well known.

(SH) State Historical Native species which have been documented for the state during the last 100 years, but which are currently unknown to occur. When known, the year of the last documented occurrence in Rhode Island is included.

FUTURE REVISIONS

The listing of rare species is an ongoing process requiring annual revisions to reflect the best scientific information available concerning the circumstances of rarity, as well as our increased knowledge of the native fauna. Submission of additional data on species currently listed, or on other species which may warrant listing, is encouraged. Information may be sent to:

Rhode Island Natural Heritage Program
Rhode Island Department of Environmental Management
Division of Planning & Development
235 Promenade Street
Providence, Rhode Island 02908
Telephone: (401) 222-2776 ext.4308

Rhode Island Endangered Species Program
Rhode Island Dept. of Environmental Management
Division of Fish and Wildlife
Great Swamp Management Area
West Kingston, Rhode Island 02892
Telephone: (401) 789-0281

INVERTEBRATES

The task of evaluating the status of invertebrates in Rhode Island has been initiated for several selected groups. At this time the list primarily includes freshwater bivalves (clams and mussels) and the following insect groups: lepidopterans (moths and butterflies), odonates (dragonflies and damselflies), silphids (burying beetles), and cicindelids (tiger beetles). Additional taxa will be added in the future upon the completion of further research and inventory. The following publications are a partial listing of taxonomic references:

Boyd, H.P. and Associates. 1982. *Checklist of Cicindelidae: The Tiger Beetles*. Plexus Publishing, Marlton, New Jersey. 1-31.

Hodges, R.W., et. al. 1983. *Check list of the Lepidoptera of America north of Mexico*. E.W. Classey Ltd. and Wedge Entomological Research Foundation. 1-284.

Johnson, R.I. 1980. *Zoogeography of North American Unionacea (Mollusca: Bivalvia) north of the maximum Pleistocene glaciation*. Bull. Museum Comparative Zoology. 149:77-189.

Paulson, D.R. and S.W. Dunkle. 1999. *A checklist of North American Odonata, including English name, etymology, type locality, and distribution.* Slat. Mus. Nat. Hist. Occ. Pap. 56.

BIVALVE MOLLUSKS

Unionoida (freshwater mussels)

Margaritiferidae (pearlshells)

Margaritifera margaritifera	Eastern Pearlshell	SE

Unionidae (unionid mussels)

Alismidonta varicosa	Brook Floater	SH (1897)
Lampsilis radiata	Lampmussel	C
Ligumia nasuta	Eastern Pond Mussel	C
Strophitus undulatus	Squawfoot	C

CRUSTACEANS

Amphipoda (amphipods)

Crangonyctidae (freshwater amphipods)

Synurella chamberlaini	Coastal Swamp Amphipod	C

INSECTS

Coleoptera (beetles)

Cicindelidae (tiger beetles)

Cicindela dorsalis dorsalis	Northeastern Beach Tiger Beetle	FT/SH (1978)
Cicindela formosa generosa	Pine Barrens Tiger Beetle	ST
Cicindela hirticollis	Seabeach Tiger Beetle	ST
Cicindela limbalis	Claybanks Tiger Beetle	C
Cicindela marginata	Salt Marsh Tiger Beetle	ST
Cicindela patruela	Barrens Tiger Beetle	SH (1921)
Cicindela purpurea	Purple Tiger Beetle	C
Cicindela rufiventris	Red-bellied Tiger Beetle	C
Cicindela tranquebarica	Dark-bellied Tiger Beetle	ST

Silphidae (burying beetles)

Nicrophorus americanus	American Burying Beetle	FE

Staphylinidae (rove beetles)

Lordithon niger	Black Lordithon Rove Beetle	C

Lepidoptera (butterflies and moths)

Lycaenidae (coppers, hairstreaks, elfins, & blues)

Lycaena epixanthe	Bog Copper	C
Satyrium acadica	Acadian Hairstreak	C
Satyrium caryaevorum	Hickory Hairstreak	C
Mitoura hesseli	Hessel's Hairstreak	C
Incisalia henrici	Henry's Elfin	C
Incisalia irus	Frosted Elfin	ST
Incisalia polia	Hoary Elfin	C
Fixsenia favonius ontario	Northern Hairstreak	C
Parrhasius m-album	White M Hairstreak	C

Nymphalidae (brush-footed butterflies)

Speyeria idalia	Regal Fritillary	SH (1990)
Boloria bellona	Meadow Fritillary	C
Enodia anthedon	Northern Pearly Eye	C

Hesperiidae (skippers)

Erynnis brizo	Sleepy Duskywing	C
Erynnis persius	Persius Duskywing	SH (1950)
Poanes massasoit	Mulberry Wing	C
Poanes viator zizaniae	Broad Winged Skipper	C
Atrytonopsis hianna	Dusted Skipper	C

Noctuidae (noctuid moths)

Abagrotis crumbi benjamini	Benjamin's Abagrotis	C
Acronicta lanceolaria	A Noctuid Moth	C
Apharetra purpurea	Blueberry Sallow	C
Aplectoides condita	A Noctuid Moth	C
Grammia speciosa	An Arctiid Moth	C
Lithophane viridipallens	Pale Green Pinion Moth	C
Metarranthis pilosaria	Coastal Swamp Metarranthis	C
Papaipema appassionata	Pitcher Plant Borer	C
Papaipema leucostigma	Columbine Borer	SH
Spartiniphaga inops	Spartina Borer	C
Zale sp. (*)	Pine Barrens Zale	C
Zale submediana	A Noctuid Moth	C

(*) a full scientific name for this species has not been published.

Saturniidae (saturnid moths)

Citheronia regalis	Royal Walnut Moth	SH (1939)
Citheronia sepulcralis	Pine Devil	SH
Hemileuca maia maia	Barrens Buckmoth	C

Odonata (dragonflies and damselflies)

Coenagrionidae (pond damselflies)

Enallagma pictum	Scarlet Bluet	C
Enallagma recurvatum	Pine Barrens Bluet	C
Lestes unguiculatus	Lyre-tipped Spreadwing	C
Nehalennia integricollis	Southern Sprite	ST

Gomphidae (clubtails)

Ophiogomphus aspersus	Brook Snaketail	ST
Progomphus obscurus	Common Sanddragon	C
Stylurus scudderi	Zebra Clubtail	ST
Stylurus spiniceris	Arrow Clubtail	C

Aeshnidae (darners)

Aeshna mutata	Spatterdock Darner	C
Anax longipes	Comet Darner	C

Corduliidae (emeralds)

Cordulegaster obliqua	Arrowhead Spiketail	C
Neurocordulia obsoleta	Umber Shadowdragon	C
Somatochlora georgiana	Coppery Emerald	C
Williamsonia lintneri	Ringed Boghaunter	SE

Libellulidae (common skimmers)

Leucorrhinia glacialis	Crimson-ringed Whiteface	ST
Libellula auripennis	Golden-winged Skimmer	C

VERTEBRATES

The following reference is used:

August, P.V., Enser, R.W. and L.L. Gould. 2001. *Vertebrates of Rhode Island*. Vol. 2. Biota of Rhode Island. Rhode Island Natural History Survey, Kingston, RI.

FISH

Petromyzontidae (lampreys)

Lampetra appendix	American Brook Lamprey	ST

Acipenseridae (sturgeons)

Acipenser oxyrhynchus	Atlantic Sturgeon	SH
Acipenser brevirostrum	Shortnose Sturgeon	FE (SH)

AMPHIBIANS

Plethodontidae (lungless salamanders)

Gyrinophilus porphyriticus	Northern Spring Salamander	C

Pelobatidae (spadefoot toads)

Scaphiopus holbrookii	Eastern Spadefoot	SE

Ranidae (true frogs)

Rana pipiens	Northern Leopard Frog	C

REPTILES

Note: Several reptiles are covered under regulations of the Rhode Island Division of Fish and Wildlife, which identifies several species as "protected", i.e., that possession without a permit is prohibited at all times. Species designated under these regulations are indicated by "P" in the status column.

Cheloniidae (sea turtles) - offshore waters only.

Caretta caretta	Loggerhead Sea Turtle	FT
Eretmochelys imbricata	Hawksbill Sea Turtle	FE
Lepidochelys kempii	Kemp's Ridley Sea Turtle	FE

Dermochelyidae (leatherback turtles) - offshore waters only.

Dermochelys c. coriacea	Atlantic Leatherback	FE

Emydidae (turtles)

Clemmys guttata	Spotted Turtle	P
Clemmys insculpta	Wood Turtle	C/P
Malaclemys t. terrapin	Northern Diamondback Terrapin	SE/P
Terrapene carolina	Eastern Box Turtle	P

Colubridae (colubrid snakes)

Carphophis amoenus	Eastern Worm Snake	C
Elaphe obsoleta	Black Rat Snake	C
Heterodon platirhinos	Eastern Hognose Snake	C
Thamnophis sauritus	Eastern Ribbon Snake	C

Viperidae (vipers)

Crotalus horridus	Timber Rattlesnake	SH (1972)/P

BIRDS

Note: Birds are listed based on the status of *breeding* populations in Rhode Island.

Podicipedidae (grebes)

Podilymbus podiceps	Pied-billed Grebe	SE

Ardeidae (herons)

Botaurus lentiginosus	American Bittern	SE
Ixobrychus exilis	Least Bittern	ST
Ardea herodias	Great Blue Heron	C
Ardea albus	Great Egret	C
Egretta caerulea	Little Blue Heron .	C
Egretta thula	Snowy Egret	C
Bubulcus ibis	Cattle Egret	C
Nycticorax nycticorax	Black-crowned Night Heron	C

Nyctanassa violacea	Yellow-crowned Night Heron	C

Threskiornithidae (ibises)

Plegadis falcinellus	Glossy Ibis	C

Anatidae (swans, geese, ducks)

Anas crecca	Green-winged Teal	C
Anas discors	Blue-winged Teal	C
Anas strepera	Gadwall	C
Lophodytes cucullatus	Hooded Merganser	C

Accipitridae (eagles, hawks)

Haliaeetus leucocephalus	Bald Eagle	FT
Pandion haliaetus	Osprey	C
Circus cyaneus	Northern Harrier	SE
Accipiter striatus	Sharp-shinned Hawk	SH (1939)
Accipiter cooperii	Cooper's Hawk	C
Accipiter gentilis	Northern Goshawk	C
Falco peregrinus	Peregrine Falcon	SE

Rallidae (rails, gallinules)

Rallus elegans	King Rail	C
Rallus longirostris	Clapper Rail	C
Porzana carolina	Sora	C
Gallinula chloropus	Common Moorhen	SH (1970)

Charadriidae (plovers)

Charadrius melodus	Piping Plover	FT

Haematopodidae (oystercatchers)

Haematopus palliatus	American Oystercatcher	C

Scolopacidae (sandpipers)

Catoptrophorus semipalmatus	Willet	C
Bartramia longicauda	Upland Sandpiper	SE

Laridae (gulls, terns)

Sterna dougallii	Roseate Tern	FE/SH (1979)
Sterna antillarum	Least Tern	ST

Tytonidae (barn owls)

Tyto alba	Barn Owl	SE

Strigidae (owls)

Asio otus	Long-eared Owl	C
Aegolius acadicus	Northern Saw-whet Owl	C
Caprimulgidae (goatsuckers)		
Chordeiles minor	Common Nighthawk	C
Picidae (woodpeckers)		
Dryocopus pileatus	Pileated Woodpecker	C
Tyrannidae (flycatchers)		
Empidonax virescens	Acadian Flycatcher	C
Alaudidae (larks)		
Eremophila alpestris	Horned Lark	C
Hirundinidae (swallows)		
Hirundo pyrrhonota	Cliff Swallow	SH (1991)
Troglodytidae (wrens)		
Troglodytes troglodytes	Winter Wren	C
Cistothorus palustris	Marsh Wren	C
Parulidae (warblers)		
Vermivora chrysoptera	Golden-winged Warbler	SH (1960)
Parula americana	Northern Parula	ST
Dendroica caerulescens	Black-throated Blue Warbler	ST
Dendroica cerulea	Cerulean Warbler	SE
Dendroica fusca	Blackburnian Warbler	ST
Protonotaria citrea	Prothonotary Warbler	C
Helmitheros vermivorus	Worm-eating Warbler	C
Icteria virens	Yellow-breasted Chat	SE
Emberizidae (sparrows)		
Pooecetes gramineus	Vesper Sparrow	SH (1984)
Ammodramus henslowii	Henslow's Sparrow	SH (1940)
Ammodramus savannarum	Grasshopper Sparrow	ST
Ammodramus maritimus	Seaside Sparrow	C
Zonotrichia albicollis	White-throated Sparrow	C
Junco hyemalis	Dark-eyed Junco	C

MAMMALS

Soricidae (shrews)

Sorex fumeus	Smoky Shrew		C
Sorex palustris	Water Shrew		C

Leporidae (rabbits, hares)

Sylvilagus transitionalis	New England Cottontail		C

Muridae (mice)

Synaptomys cooperi	Southern Bog Lemming		C

Felidae (cats)

Lynx rufus	Bobcat	ST	

Balaenopteridae (rorquals)

Balaenoptera physalus	Fin Whale		FE
Megaptera novaeangliae	Humpback Whale		FE

Balaenidae (right whales)

Eubalaena glacialis	North Atlantic Right Whale		FE

RARE NATIVE PLANTS OF RHODE ISLAND - January, 2002

Prepared by
Richard W. Enser
Rhode Island Natural Heritage Program
Rhode Island Department of Environmental Management
Providence, Rhode Island 02908

The flora of Rhode Island includes roughly 1700 plant taxa of which approximately 1300 (77%) are considered to be native. The following list identifies those members of the native flora which are the rarest in Rhode Island and most in need of conservation. All plant taxa listed herein are currently being tracked by the Rhode Island Natural Heritage Program through comprehensive mapping and computerized databases. Information regarding the location and status of rare elements, including plants, animals and natural communities, is used to establish priorities for land preservation and to provide guidance within the environmental review process.

The Rhode Island Natural Heritage Program was established in 1978. During the first year of operation an initial listing of rare plants was derived from two previously published lists: *Endangered Plants of Rhode Island*, by Dr. Irene Stuckey; and *Rare and Endangered Vascular Plant Species in Rhode Island*, by Dr. George L. Church and Richard L. Champlin. The latter publication was the Rhode Island contribution to a regional assessment of rare plants prepared by the New England Botanical Club in cooperation with the U.S. Fish and Wildlife Service. More recently, the New England Plant Conservation Program (NEPCoP - established in 1991) has conducted an exhaustive reassessment of the region's flora in preparation of *Flora Conservanda*: New England - the NEPCoP list of plants in need of conservation. This list, published in 1996, provides a regional perspective to the Rhode Island list, and a clear picture of regional conservation priorities.

Since 1978, the Natural Heritage Program has gathered information from many sources, particularly herbaria, published reports, and botanical field notes to refine the Rhode Island state list. The author, along with several other professional and amateur botanists, has also spent considerable time verifying the locations and identities of rare plants throughout the state. This combined effort has made the Natural Heritage Program's database the largest repository of rare plant information in Rhode Island.

The rare plant list is amended annually to reflect the most up-to-date knowledge of plant distribution, status, and taxonomy. Although the number of plants on the rare list has remained relatively constant, certain species have been deleted when found to be more common or less vulnerable to extirpation than originally thought, while others have been added following similar status assessment, or when newly discovered in Rhode Island. (These may be new colonizers or may have been overlooked in the past.) The January 2002 edition of the *Rare Native Plants of Rhode Island* includes 309 plants, or approximately 24% of the state's native flora.

ABOUT THIS LIST

The list is arranged alphabetically by botanical family, genus and species. Trinomials are used to describe certain subspecies and varieties.

Nomenclature

The taxonomic authority for scientific names is:

Gould, L.L., R.W. Enser, R.L. Champlin, and I.S. Stuckey. 1998. *Vascular Flora of Rhode Island: A list of Native and Naturalized Plants.* Volume 1 of The Biota of Rhode Island project. Rhode Island Natural History Survey, Kingston, RI.

Copies of Gould, *et.al.* are available from:

Rhode Island Natural History Survey
Room 101, the Coastal Institute in Kingston
1 Greenhouse Road
University of Rhode Island
Kingston, RI 02881-0804

Extant Populations

The number cited refers to extant populations known since 1985. (There are a few instances of populations being destroyed since this date - these are not included in the count.) The number of distinct populations of some species, especially aquatics, is often difficult to determine. Population numbers for these species are based on the assumption that occurrences within the same reach of a river, or separate portions of a pond, lake, or other contiguous wetland system are considered one population.

Status

The status of each species is designated by letter codes as defined below:

(FE) Federally Endangered	(1 RI species currently listed)
(FT) Federally Threatened	(2 RI species currently listed)

(SE) State Endangered — Native taxa in imminent danger of extirpation from Rhode Island. These taxa may meet one or more of the following criteria:

1. A taxon formerly considered by the U.S. Fish & Wildlife Service for listing as Federally endangered or threatened. These species were identified as **C2** (Category 2) taxa for which information indicated that proposing to list under the Federal Endangered Species Act was potentially appropriate, but for which sufficient data on biological vulnerability and threat were not currently available to support proposed rules. The US Fish & Wildlife Service is currently not designating Category 2 species.
2. A taxon with 1 or 2 known or estimated total populations in the state.
3. A taxon apparently globally rare or threatened, estimated to occur at approximately 100 or fewer sites range-wide.

Plants listed as State Endangered are protected under the provisions of the Rhode Island State Endangered Species Act, Title 20 of the General Laws of the State of Rhode Island. This law states, in part (20-37-3):

"No person shall buy, sell, offer for sale, store, transport, import, export, or otherwise traffic in any animal or plant or any part of any animal or plant whether living or dead, processed, manufactured, preserved or raw (if) such animal or plant has been declared to be an endangered species by either the United States secretaries of the Interior or Commerce or the Director of the Rhode Island Department of Environmental Management."

(ST) State Threatened — Native taxa which are likely to become State Endangered in the future if current trends in habitat loss or other detrimental factors remain unchanged. In general, these taxa have 3-5 known or estimated populations and are especially vulnerable to habitat loss.

(C) Concern — Native taxa not considered to be State Endangered or Threatened at the present time, but are listed due to various factors of rarity and/or vulnerability.

(SH) State Historical — Native taxa which have been documented for Rhode Island during the last 150 years but for which there are no extant populations. When known, the year of last documented occurrence is included.

Note on Status Designation:

For most listed plants the definitions outlined above have been adhered to when assigning status. In some cases, especially for those species which have not received intensive field inventory, the "Concern" category is assigned even if only 1-2 populations are known to be extant. These species are targeted for additional inventory and may be assigned to other categories when their actual status in Rhode Island has been clarified. Taxa so-defined are designated with an asterisk (*).

Global Rank

Each taxon has been assigned a “global rank” that reflects its rarity and vulnerability to extinction throughout the world. Global ranks were originally derived by The Nature Conservancy and are used by all Natural Heritage Programs as a standardized method of determining the status of each taxon throughout its range. These ranks are defined as follows:

G1 = Critically imperiled throughout its range due to extreme rarity (5 or fewer sites or very few remaining individuals) or extremely vulnerable to extinction due to biological factors.

G2 = Imperiled throughout its range due to rarity (6-20 sites or few remaining individuals) or highly vulnerable to extinction due to biological factors.

G3 = Either very rare and local throughout its range (21-100 sites), with a restricted range (but possibly locally abundant), or vulnerable to extinction due to biological factors.

G4 = Apparently secure throughout its range (but possibly rare in parts).

G5 = Demonstrably secure throughout its range (but possibly rare in parts).

GH = No extant sites known, but may be rediscovered.

GX = Believed to be extinct.

T-ranks: Sometimes amended to a G rank to indicate the designation is for a subspecies or variety, not the rarity of the species as a whole.

Double Ranks (i.e., G2G3): The first rank indicates rarity based on current documentation. The second rank indicates the probable rarity after all historical records and potential habitats have been surveyed. Denotes taxa needing additional survey.

NEPCoP Status

Designation indicates the inclusion of the taxon on *Flora Conservanda*: New England. The New England Plant Conservation Program (NEPCoP) list of plants in need of conservation. Taxa on this list are assigned to one of 5 Divisions based on the following definitions:

Div 1: Globally rare taxa occurring in New England. Taxa in this division have been ranked as globally rare (G1 through G3, or T1 through T3) under criteria described above.

Div 2: Regionally rare taxa with fewer than 20 current occurrences within New England.

Div 3: Locally rare taxa that may be common in part of New England, but have one or more occurrences of biological, ecological, or possible genetic significance.

Div 4: Historic taxa that once existed in New England, but have not been seen since 1970.

IND: Indeterminate taxa which are under review for inclusion in one of the above divisions, but issues of taxonomy, nomenclature, or status in the wild are not clearly understood.

County of Occurrence

Each county is designated by the first three letters of its name as follows:

PRO = Providence; BRI = Bristol; KEN = Kent; NEW = Newport; WAS = Washington. In addition, Block Island (actually part of Washington County) is designated by the letters BLO. Counties are identified for extant populations only, except in the case of Historic species in which the last county of occurrence is given.

FUTURE REVISIONS

The listing of rare species is an ongoing process requiring annual revisions to reflect the best scientific information available concerning the circumstances of rarity, as well as our increased knowledge of the native flora. Submission of additional data on species currently listed, or on other species which may warrant listing, is encouraged. Information should be sent to:

Rhode Island Natural Heritage Program
Rhode Island Department of Environmental Management
235 Promenade Street
Providence, Rhode Island 02908
Telephone: (401) 222-2776 extension 4308
Fax: (401) 222-2069
Email: renser@dem.state.ri.us

ACKNOWLEDGMENTS

Many people have shared their knowledge of the Rhode Island flora, helping me to provide the most complete assessment of rare plants in the state. In particular I would like to thank the following individuals: William E. Brumback, Caren A. Caljouw, Richard L. Champlin, Gilbert George, Lisa L. Gould, Julie Lundgren, Les Mehrhoff, Joanne Michaud, William Nichols, Christopher Raithel, Thomas Rawinski, Bruce Sorrie, Irene Stuckey, Gordon Tucker and Frances Underwood.

Rhode Island Rare Plant Scorecard - 2002

	SE*	ST	C	SH**	Total
Pteridophytes	6	2	12	1	21
Gynosperms	-	1	2	-	3
Angiosperms	45	54	125	61	285
Total	51	57	139	62	309

*includes one Federally Threatened, and one Federally Endangered species

** includes one Federally Threatened species

Species	Common Name	Extant Pop	Counties of Occurrence	State Status	Global Rank	NEPCoP List
PTERIDOPHYTES						
Aspleniaceae (Spleenwort Family)						
Asplenium montanum	Mountain Spleenwort	1	PRO	SE	G5	Div 2
Asplenium rhizophyllum	Walking Fern	1	PRO	SE	G5	
Asplenium trichomanes	Maidenhair Spleenwort	9	PRO;KEN;WAS	C	G5	
Equisetaceae (Horsetail Family)						
Equisetum fluviatile	Water Horsetail	3	PRO;WASC	G5		
Equisetum hyemale ssp. *affine*	Rough Horsetail	7	PRO;KEN;WAS	C	G5T5	
Equisetum sylvaticum	Woodland Horsetail	10	PRO;BRI;WAS	C	G5	
Dryopteridaceae (Wood Fern Family)						
Gymnocarpium dryopteris	Oak Fern	3	PRO;KEN	ST	G5	
Matteuccia struthiopteris var. *pensylvanica*	Ostrich Fern	5	PRO;KEN	C	G5T5	
Woodsia ilvensis	Rusty Woodsia	0 (1977)	PRO	SH	G5	
Isoetaceae (Quillwort Family)						
Isoetes echinospora	Spiny Quillwort	2	PRO;KEN	C	G5	
Isoetes engelmannii	Engelmann's Quillwort	2	PRO;KEN	C	G4	
Isoetes riparia	River Quillwort	4	PRO;KEN	C	G5	Div 2
Lycopodiaceae (Clubmoss Family)						
Lycopodiella alopecuroides	Foxtail Clubmoss	1	WAS	SE	G5	Div 2
Lycopodium annotinum	Stiff Clubmoss	1	PRO	SE	G5	
Ophioglossaceae (Adder's-tongue Family)						
Botrychium lanceolatum ssp. *angustisegmentum*	Triangle Grape-fern	2	PRO;KEN	C	G5T4	
Botrychium matricariifolium	Daisyleaf Grape-fern	4	PRO;KEN	C	G5	
Botrychium simplex	Dwarf Grape-fern	5	PRO;KEN;WAS	C	G5	
Ophioglossum pusillum	Northern Adder's-tongue	1	WAS	SE	G5	Div 3
Pteridaceae (Maidenhair Fern Family)						
Pellaea atropurpurea	Purple Cliff-brake	1	PRO	SE	G5	
Schizaeaceae (Curly-grass Fern Family)						
Lygodium palmatum	Climbing Fern	6	PRO;KEN;WAS	C	G4	
Thelypteridaceae (Marsh Fern Family)						
Phegopteris connectilis	Long Beech Fern	3	PRO;KEN	ST	G5	
GYMNOSPERMS						
Taxaceae (Yew Family)						
Taxus canadensis	Ground Hemlock	3	PRO	C	G5	
Pinaceae (Pine Family)						
Larix laricina	American Larch	3	PRO;KEN	ST	G5	
Picea mariana	Black Spruce	9	PRO;KEN	C	G5	
ANGIOSPERMS						
Aceraceae (Maple Family)						
Acer pensylvanicum	Striped Maple	4	PRO	C	G5	
Acer spicatum	Mountain Maple	1	PRO	ST	G5	
Alismataceae (Water-Plantain Family)						
Sagittaria graminea	Grass-leaved Arrowhead	4	KEN;WAS	C	G5	
Sagittaria subulata	River Arrowhead	0 (1895)	PRO	SH	G4	Div 2
Sagittaria teres	Slender Arrowhead	3	WAS	SE	G3	Div 1

Species	Common Name	Extant Pop	Counties of Occurrence	State Status	Global Rank	NEPCoP List
Amaranthaceae (Amaranth Family)						
Amaranthus pumilus	Seabeach Amaranth	0 (1897)	NEW	FT/SH	G2	Div 4
Apiaceae (Parsley Family)						
Angelica atropurpurea	Large Angelica	1	PRO	SE	G5	
Angelica lucida	Seaside Angelica	3	WAS;BLO	ST	G5	IND
Cryptotaenia canadensis	Honewort	5	PRO	C	G5	
Hydrocotyle verticillata	Saltpond Pennywort	0 (1895)	BLO	SH	G5	Div 2
Ligusticum scothicum	Scotch Lovage	10	NEW;WAS	C	G5	
Lilaeopsis chinensis	Lilaeopsis	0 (1900)	PRO	SH	G5	Div 3
Osmorhiza longistylis	Anise-root	3	PRO;KEN ST	G5		
Ptilimnium capillaceum	Mock Bishop's Weed	7	NEW;WAS;BLO	C	G5	
Taenidia integerrima	Yellow Pimpernel	0 (1886)	PRO	SH	G5	Div 2
Zizia aptera	Heart-leaved Golden Alexanders	0 (1920)	WAS	SH	G5	Div 2
Zizia aurea	Golden Alexanders	10	PRO	C	G5	
Araceae (Arum Family)						
Orontium aquaticum	Golden Club	1	WAS	SE	G5	
Araliaceae (Ginseng Family)						
Aralia racemosa	Spikenard	5	PRO;KEN	C	G5	
Panax quinquefolius	American Ginseng	1	PRO	SE	G4	
Asclepiadaceae (Milkweed Family)						
Asclepias amplexicaulis	Blunt-leaved Milkweed	7	PRO;KEN;WAS	C	G5	
Asclepias exaltata	Poke Milkweed	4	PRO;KEN;WAS	C	G5	
Asclepias purpurascens	Purple Milkweed	0 (1906)	WAS	SH	G4G5	Div 2
Asclepias quadrifolia	Four-leaved Milkweed	4	PRO	ST	G5	
Asclepias tuberosa	Butterfly Milkweed	8	WAS;BLO	C	G5	Div 3
Asclepias verticillata	Whorled Milkweed	4	PRO;BRI;NEW	C	G5	
Asteraceae (Aster Family)						
Artemisia campestris ssp. *caudata*	Tall Wormwood	3	NEW;WAS	C	G5T5	Div 3
Aster concolor	Eastern Silvery Aster	0 (1925)	WAS	SH	G4	Div 2
Aster infirmus	Cornel-leaved Aster	0 (1965)	PRO	SH	G5	Div 2
Aster laevis	Smooth Blue Aster	4	KEN;WAS	C	G5	
Aster macrophyllus	Large-leaved Aster	5	PRO;WASC	G5		
Bidens connata	Swamp Beggar's-ticks	2	WAS;BLO	C	G5	
Bidens coronata	Tickseed Sunflower	3	WAS	C	G5	
Cacalia suaveolens	Indian-plantain	0 (1930)	PRO	SH	G3G4	Div 4
Chrysopsis falcata	Sickle-leaved Golden Aster	8	KEN;WAS	C	G3G4	
Chrysopsis mariana	Maryland Golden Aster	1	BLO	ST	G5	Div 2
Cirsium horridulum	Yellow Thistle	1	BLO	ST	G5	IND
Coreopsis rosea	Pink Tickseed	7	KEN;WAS	C	G3	Div 1
Eupatorium aromaticum	Snakeroot	0 (1979)	WAS	SH	G4G5	Div 2
Eupatorium leucolepis var. *novae-angliae*	New England Boneset	5	NEW;WAS	SE	G5T1	Div 1
Gnaphalium purpureum	Purple Cudweed	0 (1913)	BLO	SH	G5	Div 2
Helianthus divaricatus	Woodland Sunflower	3	PRO;BRI;WAS	C	G5	
Liatris scariosa var. *novae-angliae*	Northern Blazing Star	4	WAS;BLO	SE	G5T3	Div 1
Prenanthes serpentaria	Lion's-foot	1*	PRO	C	G5	Div 2
Rudbeckia laciniata	Green-headed Coneflower	1	PRO	ST	G5	
Sclerolepis uniflora	Sclerolepis	1	PRO	SE	G4	Div 2
Solidago elliottii	Elliott's Goldenrod	2	BLO;WAS	C	G5	
Solidago flexicaulis	Zigzag Goldenrod	2	PRO	ST	G5	
Solidago rigida	Stiff-leaf Goldenrod	0 (1921)	WAS	SH	G5	Div 2
Berberidaceae (Barberry Family)						
Caulophyllum thalictroides	Blue Cohosh	2	PRO	ST	G5	
Boraginaceae (Borage Family)						
Onosmodium virginianum	False Gromwell	0 (1886)	PRO	SH	G4	Div 2

Species	Common Name	Extant Pop	Counties of Occurrence	State Status	Global Rank	NEPCoP List
Brassicaceae (Mustard Family)						
Arabis drummondii	Rock-cress	1*	PRO	C	G5	Div 3
Arabis missouriensis	Missouri Rock-cress	1*	PRO	C	G4	IND
Draba reptans	Carolina Whitlow-Grass	0 (1902)	PRO	SH	G5	Div 2
Caesalpiniaceae (Caesalpinia Family)						
Senna hebecarpa	Wild Senna	0 (1971)	KEN	SH	G5	
Campanulaceae (Bluebell Family)						
Lobelia dortmanna	Water Lobelia	9	PRO;WASC	G4		
Caprifoliaceae (Honeysuckle Family)						
Linnaea borealis	Twinflower	0 (1930)	WAS	SH	G5	
Lonicera caerulea	Mountain Fly-honeysuckle	2	KEN;WAS	C	G5	
Lonicera dioica	Mountain Honeysuckle	4	PRO;KEN;WAS	C	G5	
Sambucus racemosa var. *pubens*	Red-berried Elderberry	0 (1878)	PRO	SH	G5T4	
Triosteum aurantiacum	Wild Coffee	5	PRO;WASC	G5	Div 3	
Triosteum perfoliatum	Feverwort	4	PRO;WASC	G5	Div 2	
Viburnum alnifolium	Hobblebush	5	PRO	C	G5	
Viburnum nudum var. *nudum*	Swamp Haw	1	WAS	ST	G5	Div 2
Caryophyllaceae (Pink Family)						
Arenaria caroliniana	Pine Barren Sandwort	0 (1918)	WAS	SH	G5	Div 4
Arenaria groenlandica var. *glabra*	Smooth Sandwort	2	WAS	ST	G4	Div 2
Arenaria stricta	Rock Sandwort	1	NEW	SE	G5	
Honckenya peploides var. *robusta*	Seabeach Sandwort	6	NEW;WAS;BLO	C	G5T4	
Silene stellata	Starry Campion	0 (1935)	WAS	SH	G5	Div 2
Spergularia canadensis	Northern Sand-spurrey	1*	WAS	C	G5	
Chenopodiaceae (Goosefoot Family)						
Atriplex glabriuscula	Smooth Orache	2	WAS	C	G4	
Chenopodium leptophyllum	Goosefoot	2	NEW;WAS	C	G5	IND
Cistaceae (Rock-rose Family)						
Helianthemum dumosum	Bushy Rockrose	6	WAS;BLO	SE	G3	Div 1
Helianthemum propinquum	Low Rockrose	4	BRI;WAS	C	G4	
Hudsonia ericoides	Golden Heather	4	PRO;KEN;WAS	C	G4	
Clusiaceae (St. John's-wort Family)						
Hypericum adpressum	Creeping St. John's-wort	4	WAS	ST	G2G3	Div 1
Cuscutaceae (Dodder Family)						
Cuscuta coryli	Hazel-dodder	0	PRO	SH	G5	Div 2
Cornaceae (Dogwood Family)						
Cornus rugosa	Round-leaved Dogwood	4	PRO	C	G5	
Cyperaceae (Sedge Family)						
Carex alata	Winged Sedge	1*	BLO	C	G5	
Carex albicans	Covered Sedge	2*	PRO	C	G5	
Carex buxbaumii	Buxbaum's Sedge	1*	PRO	C	G5	Div 3
Carex collinsii	Collins' Sedge	0 (1979)	WAS	SH	G4	Div 2
Carex cumulata	Piled Sedge	3	PRO	C	G4	
Carex exilis	Bog Sedge	5	PRO;WASC	G5		
Carex limosa	Mud Sedge	0 (1892)	PRO	SH	G5	
Carex mitchelliana	Mitchell's Sedge	1*	PRO	C	G3G4	Div 1
Carex pedunculata	Long-stalked Sedge	2	PRO	ST	G5	
Carex polymorpha	Variable Sedge	1	PRO	SE	G3	Div 1
Carex schweinitzii	Schweinitz's Sedge	0 (1895)	PRO	SH	G3	Div 1
Carex sparganioides	Burreed-like Sedge	0 (1948)	PRO	SH	G5	Div 3
Carex sterilis	Sterile Sedge	0 (1878)	PRO	SH	G4	Div 2
Carex striata var. *brevis*	Walter's Sedge	1	WAS	SE	G4T4	Div 2
Cyperus odoratus	Fragrant Umbrella-sedge	1*	BLO	C	G5	IND
Cyperus squarrosus	Awned Umbrella-sedge	1	KEN	SE	G5	

Species	Common Name	Extant Pop	Counties of Occurrence	State Status	Global Rank	NEPCoP List
Cyperaceae (continued)						
Eleocharis equisetoides	Horsetail Spike-rush	8	PRO;KEN;WAS	C	G4	Div 2
Eleocharis fallax	Deceitful Spike-rush	0	WAS	SH	G4G5	Div 2
Eleocharis melanocarpa	Black-fruited Spike-rush	1	WAS	SE	G4	
Eleocharis rostellata	Small-beaked Spike-rush	4	BRI;WAS	C	G5	IND
Eleocharis tricostata	Three-angled Spike-rush	1	WAS	SE	G4	Div 2
Eriophorum gracile	Slender Cotton-grass	2	PRO;WASST	G5		
Eriophorum vaginatum var. *spissum*	Hare's Tail	0 (1904)	PRO	SH	G5T5	
Eriophorum viridicarinatum	Bog Cotton-grass	2	KEN;WAS	C	G5	
Fuirena pumila	Umbrella Grass	2	WAS	SE	G4	Div 3
Lipocarpha micrantha	Tiny-flowered Sedge	2	KEN;WAS	ST	G4	
Rhynchospora inundata	Inundated Horned Rush	4	PRO;WASSE	G3G4	Div 2	
Rhynchospora macrostachya	Tall Beaked Rush	5	PRO;KEN;WAS	ST	G4	
Rhynchospora scirpoides	Long-beaked Bald Rush	2	WAS	SE	G4	
Rhynchospora torreyana	Torrey's Beaked Rush	1	WAS	SE	G4	Div 2
Scirpus etuberculatus	Swamp Bulrush	1	WAS	SE	G3G4	Div 1
Scirpus hudsonianus	Northern Cotton-grass	0 (1907)	PRO	SH	G5	
Scirpus longii	Long's Bulrush	1	WAS	SE	G2	Div 1
Scirpus maritimus	Saltmarsh Bulrush	4	BRI;NEW	C	G5	Div 2
Scirpus smithii	Smith's Bulrush	3	WAS	ST	G5	
Scirpus subterminalis	Water Bulrush	3	NEW;WAS	C	G4G5	
Scirpus torreyi	Torrey's Bulrush	3	WAS	C	G5	
Scleria pauciflora	Carolina-whipgrass	3	WAS	ST	G5	Div 2
Scleria reticularis	Reticulated Nut-rush	3	WAS	ST	G3G4	Div 1
Scleria triglomerata	Whipgrass	2	WAS	ST	G5	Div 2
Droseraceae (Sundew Family)						
Drosera filiformis	Thread-leaved Sundew	0 (1988)	WAS	SH	G5	
Elatinaceae (Waterwort Family)						
Elatine triandra var. *americana*	American Waterwort	2	WAS;BLO	C	G4	IND
Ericaceae (Heath Family)						
Andromeda glaucophylla	Bog Rosemary	1	PRO	SE	G5	
Gaultheria hispidula	Creeping Snowberry	2	PRO	ST	G5	
Gaylussacia dumosa var. *bigeloviana*	Dwarf Huckleberry	4	PRO;WASC	G5T4		
Kalmia polifolia	Pale Laurel	2	PRO	ST	G5	
Lyonia mariana	Staggerbush	0	WAS	SH	G5	Div 4
Rhododendron periclymenoides	Pinxter-flower	1*	WAS	C	G5	
Fabaceae (Bean Family)						
Crotalaria sagittalis	Rattlebox	1	WAS	ST	G5	
Desmodium ciliare	Small-leaved Tick-trefoil	2	PRO;WASST	G5		
Desmodium sessilifolium	Sessile-leaved Tick-trefoil	2	WAS	ST	G5	Div 2
Lupinus perennis	Wild Lupine	8	PRO;KEN;WAS	C	G5	Div 3
Strophostyles umbellata	Pink Wild Bean	1	KEN	ST	G5	
Tephrosia virginiana	Goat's-rue	6	PRO;KEN;WAS	C	G5	
Fagaceae (Oak Family)						
Quercus prinoides	Dwarf Chestnut Oak	7	KEN;WAS	C	G5	
Quercus stellata	Post Oak	2	WAS	C	G5	
Fumariaceae (Fumitory Family)						
Adlumia fungosa	Climbing Fumitory	2	PRO	SE	G4	
Corydalis sempervirens	Pale Corydalis	10	PRO;KEN;WAS	C	G4G5	
Gentianaceae (Gentian Family)						
Gentiana andrewsii	Closed Gentian	0 (1915)	PRO	SH	G4	Div 2
Gentianopsis crinita	Fringed Gentian	4	PRO;KEN;WAS	ST	G4	
Sabatia kennedyana	Plymouth Gentian	4	NEW;WAS	SE	G3	Div 1
Sabatia stellaris	Sea-pink	4	WAS	ST	G5	Div 2
Geraniaceae (Geranium Family)						
Geranium bicknellii	Bicknell's Geranium	1	PRO	ST	G5	
Geranium robertianum	Herb-Robert	3	NEW	C	G5	

Species	Common Name	Extant Pop	Counties of Occurrence	State Status	Global Rank	NEPCoP List
Grossulariaceae (Gooseberry Family)						
Ribes hirtellum	Smooth Gooseberry	4	KEN;WAS	C	G5	
Haemodoraceae (Bloodwort Family)						
Lachnanthes caroliniana	Carolina Redroot	4	KEN;WAS	ST	G4	
Haloragaceae (Water-milfoil Family)						
Myriophyllum alterniflorum	Alternate-flowered Water-milfoil	0 (1864)	PRO	SH	G5	
Myriophyllum pinnatum	Pinnate Water-milfoil	2	WAS;BLO	ST	G5	IND
Proserpinaca pectinata	Comb-like Mermaid-weed	1*	WAS	C	G5	
Iridaceae (Iris Family)						
Sisyrinchium fuscatum	Sandplain Blue-eyed Grass	0 (1900)	WAS	SH	G5	
Juncaceae (Rush Family)						
Juncus debilis	Weak Rush	1	WAS	ST	G5	Div 2
Juncaginaceae (Arrow-grass Family)						
Triglochin palustre	Arrow-grass	0 (1878)	NEW	SH	G5	
Lamiaceae (Mint Family)						
Hedeoma pulegioides	American Pennyroyal	4	PRO	C	G5	
Lycopus rubellus	Stalked Water-horehound	2*	PRO;WASC	G5	Div 2	
Monarda fistulosa	Wild Bergamot	0 (1965)	WAS	SH	G5	
Physostegia virginiana	False Dragonhead	2	KEN	C	G5	
Stachys hyssopifolia	Hyssop-leaved Hedge-nettle	3	WAS	ST	G5	Div 3
Lentibulariaceae (Bladderwort Family)						
Utricularia biflora	Two-flowered Bladderwort	2	WAS	ST	G5	Div 2
Utricularia geminiscapa	Paired Bladderwort	5	WAS	C	G4G5	
Utricularia gibba	Humped Bladderwort	4	PRO;KEN;WAS	C	G5	
Utricularia intermedia	Flat-leaved Bladderwort	2	PRO;KEN	C	G5	
Utricularia minor	Small Bladderwort	0 (1920)	PRO	SH	G5	
Utricularia resupinata	Reversed Bladderwort	4	WAS	C	G4	Div 2
Utricularia subulata	Zigzag Bladderwort	4	PRO;WASC	G5	Div 2	
Liliaceae (Lily Family)						
Aletris farinosa	Colicroot	8	PRO;KEN;WAS	C	G5	
Allium tricoccum	Wild Leek	3	PRO;KEN;NEW	C	G5	
Lilium canadense	Canada Lily	3	PRO	ST	G5	
Lilium philadelphicum	Wood Lily	9	PRO;KEN;WAS	C	G5	
Smilacina trifolia	3-leaved False Solomon's Seal	1	PRO	ST	G5	
Streptopus roseus	Rose Twisted-stalk	2	PRO	ST	G5	
Trillium erectum	Purple Trillium	2	PRO;KEN ST	G5		
Linaceae (Flax Family)						
Linum intercursum	Sandplain Flax	1	WAS	SE	G4	
Linum medium var. *texanum*	Common Yellow Flax	1*	WAS	C	G5T5	Div 2
Linum sulcatum	Grooved Flax	0 (1844)	PRO	SH	G5	Div 2
Lythraceae (Loosestrifes)						
Rotala ramosior	Toothcup	1	KEN	SE	G5	Div 2
Oleaceae (Ash Family)						
Fraxinus nigra	Black Ash	6	PRO;KEN C	G5		
Onagraceae (Evening-primrose Family)						
Circaea alpina	Small Enchanter's Nightshade	4	PRO;KEN C	G5		
Epilobium palustre	Marsh Willow-herb	1	WAS	ST	G5	
Ludwigia sphaerocarpa	Round-fruited False Loosestrife	1	WAS	SE	G5	Div 2
Orchidaceae (Orchids)						
Arethusa bulbosa	Swamp Pink	4	PRO;KEN;WAS	SE	G4	Div 3
Calopogon tuberosus	Grass Pink	7	PRO;KEN;WAS	C	G5	
Coeloglossum viride var. *virescens*	Long-bracted Green Orchis	2	PRO	ST	G5T5	
Corallorhiza maculata	Large Coralroot	7	PRO;KEN	C	G5	
Corallorhiza odontorhiza	Autumn Coralroot	1	PRO	SE	G5	Div 3

Species	Common Name	Extant Pop	Counties of Occurrence	State Status	Global Rank	NEPCoP List
Orchidaceae (continued)						
Corallorhiza trifida	Early Coralroot	4	PRO;KEN	C	G5	
Cypripedium calceolus var. *parviflorum*	Small Yellow Lady-slipper	2	PRO	SE	G5	IND
Cypripedium calceolus var. *pubescens*	Large Yellow Lady-slipper	3	PRO	SE	G5	IND
Isotria medeoloides	Small Whorled Pogonia	1	PRO	FT	G2G3	Div 1
Liparis lilifolia	Lily-leaved Twayblade	2	PRO;KEN	SE	G5	Div 2
Liparis loeselii	Yellow Twayblade	2	PRO;NEW	ST	G5	
Listera cordata	Heartleaf Twayblade	0 (1897)	WAS	SH	G5	Div 3
Malaxis unifolia	Green Adder's-mouth	1	PRO	SE	G5	
Galearis spectabilis	Showy Orchis	1	PRO	SE	G5	
Platanthera blephariglottis	White Fringed Orchid	4	BRI;KEN;WAS	ST	G4G5	
Platanthera ciliaris	Yellow Fringed Orchid	2	WAS	SE	G5	Div 2
Platanthera flava var. *herbiola*	Pale Green Orchid	4	NEW;KEN;WAS; BLO	SE	G4T4	
Platanthera hookeri	Hooker's Orchid	0 (1983)	PRO	SH	G5	
Platanthera hyperborea	Northern Green Orchid	2	PRO	ST	G5	
Platanthera orbiculata	Round-leaved Orchid	1	PRO	ST	G5	
Platanthera orbiculata var. *macrophylla*	Large Round-leaved Orchid	1	PRO	ST	G5T4	
Platanthera psycodes	Small Purple Fringed Orchid	9	PRO;KEN;WAS	C	G5	
Spiranthes lucida	Shining Ladies'-tresses	0 (1960)	PRO	SH	G5	
Spiranthes tuberosa	Little Ladies'-tresses	1	WAS	SE	G5	
Spiranthes vernalis	Spring Ladies'-tresses	3	NEW;WAS	C	G5	
Orobanchaceae (Broom-rape Family)						
Conopholis americana	Squaw-root	8	PRO;NEW;WAS	C	G5	
Oxalidaceae (Wood sorrel Family)						
Oxalis violacea	Violet Wood-sorrel	3	NEW;WAS	SE	G5	Div 2
Papaveraceae (Poppy Family)						
Sanguinaria canadensis	Bloodroot	6	PRO;KEN;WAS	C	G5	
Poaceae (Grass Family)						
Aristida longespica	Slim-spike Three-awn	6	PRO;BRI;NEW; KEN;WAS	C	G5	
Aristida purpurascens	Purple Needlegrass	2	BLO	ST	G5	Div 2
Elymus villosus	Downy Wild Rye	1*	PRO	C	G5	Div 2
Leptochloa fascicularis var. *maritima*	Saltpond Grass	0 (1913)	BLO	SH	G5T3	Div 1
Oryzopsis pungens	Northern Ricegrass	1*	WAS	C	G5	
Panicum amarum	Panic-grass	1*	WAS	C	G5	
Panicum philadelphicum	Philadelphia Panic-grass	1*	WAS	C	G5	
Panicum rigidulum	Long-leaved Panic-grass	1*	WAS	C	G5	
Panicum wrightianum	Wright's Panic-grass	2	WAS	C	G4	
Paspalum setaceum var. *psammophilum*	Tufted Beard-grass	1*	NEW	C	G5T4	Div 2
Poa languida	Weak Bluegrass	1*	WAS	C	G3G4	
Puccinellia pumila	Goosegrass	0 (1917)	WAS	SH	G4	
Setaria geniculata	Bristly Foxtail	1*	BRI	C	G5	
Sorghastrum nutans	Indian Grass	5	BRI;KEN;WAS;BLO	C	G5	Div 3
Spartina cynosuroides	Salt Reed Cordgrass	1*	PRO;KEN	C	G5	Div 2
Sphenopholis nitida	Shining Sphenopholis	1*	PRO	C	G5	Div 2
Sphenopholis obtusata	Prairie Wedgegrass	1*	PRO	C	G5	IND
Sphenopholis pensylvanica	Swamp Oats	0	PRO	SH	G4	Div 2
Sporobolus asper	Tall Dropseed	1*	WAS	C	G5	
Tripsacum dactyloides	Northern Gama-grass	8	BRI;NEW;WAS	C	G5	Div 2
Zizania aquatica	Wild Rice	6	KEN;WAS	C	G5	
Podostemaceae (River-weed Family)						
Podostemum ceratophyllum	Riverweed	0 (1890)	PRO	SH	G5	
Polygalaceae (Milkwort Family)						
Polygala cruciata	Cross-leaved Milkwort	3	NEW;WAS	C	G5	
Polygala verticillata	Whorled Milkwort	3	NEW;WAS	C	G5	IND
Polygonaceae (Buckwheat Family)						
Polygonum glaucum	Seabeach Knotweed	3	NEW;BLO	ST	G3	Div 1
Polygonum hydropiperoides var. *setaceum*	Strigose Knotweed	0 (1924)	WAS	SH	G5	IND

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Portulaceae (Purslane Family)						
Claytonia virginica	Meadow Beauty	0 (1838)	PRO	SH	G5	
Potamogetonaceae (Pondweed Family)						
Potamogeton confervoides	Alga-pondweed	0 (1929)	WAS	SH	G3G4	Div 1
Potamogeton pusillus var. *gemmiparus*	Slender Pondweed	1*	PRO	C	G5T3	IND
Primulaceae (Primrose Family)						
Glaux maritima	Sea Milkwort	0 (1917)	WAS	SH	G5	
Hottonia inflata	Featherfoil	6	PRO;NEW;WAS	C	G4	
Pyrolaceae (Shinleaf Family)						
Moneses uniflora	One-flowered Wintergreen	2	PRO	ST	G5	
Pyrola chlorantha	Green Pyrola	4	PRO;KEN	C	G5	
Pyrola secunda	One-sided Pyrola	2	PRO	ST	G5	
Ranunculaceae (Buttercup Family)						
Actaea rubra	Red Baneberry	5	PRO;KEN	C	G5	
Anemone cylindrica	Long-fruited Anemone	3	PRO	C	G5	
Anemone virginiana	Large Anemone	0 (1950)	PRO	SH	G5	
Anemonella thalictroides	Rue Anemone	5	PRO;WASC	G5		
Clematis occidentalis	Purple Clematis	1	PRO	SE	G5	
Hepatica americana	Hepatica	5	PRO	C	G5	
Ranunculus allegheniensis	Allegheny Crowfoot	1*	PRO	C	G4G5	Div 2
Ranunculus ambigens	Water-plantain Spearwort	1*	NEW	C	G4	Div 2
Ranunculus cymbalaria	Seaside Buttercup	0 (1948)	WAS	SH	G5	
Ranunculus flabellaris	Yellow Water-crowfoot	4	PRO	C	G5	
Ranunculus hispidus var. *hispidus*	Hispid Buttercup	1*	PRO	C	G5	IND
Ranunculus micranthus	Small-flowered Crowfoot	1	PRO	ST	G5	Div 2
Ranunculus sceleratus	Cursed Crowfoot	3	PRO;BRI	C	G5	
Ranunculus trichophyllus var. *calvescens*	White Water-crowfoot	1	WAS	ST	G5	
Thalictrum revolutum	Purple Meadow-rue	0 (1900)	PRO	SH	G5	
Rosaceae (Rose Family)						
Agrimonia pubescens	Hairy Agrimony	0 (1912)	PRO	SH	G5	
Dalibarda repens	Dewdrop	1	PRO	SE	G5	
Geum laciniatum	Hairy Herb-Bennet	0 (1920)	PRO	SH	G5	
Potentilla tridentata	Three-toothed Cinquefoil	0 (1979)	WAS	SH	G5	
Prunus pumila var. *cuneata*	Sand Cherry	3	KEN;WAS	C	G5T4	
Sanguisorba canadensis	Canadian Burnet	1	WAS	SE	G5	
Rubiaceae (Madder Family)						
Hedyotis longifolia	Long-leaved Bluets	0 (1966)	WAS	SH	G4G5	
Salicaceae (Willow Family)						
Populus heterophylla	Swamp Cottonwood	1	WAS	ST	G5	Div 2
Salix pedicellaris	Bog Willow	0 (1970)	PRO	SH	G5	
Saururaceae (Lizard's-tail Family)						
Saururus cernuus	Lizard's-tail	1	NEW	SE	G5	Div 2
Saxifragaceae (Saxifrage Family)						
Parnassia glauca	Grass-of-Parnassus	0 (1980)	PRO	SH	G5	
Penthorum sedoides	Ditch Stonecrop	7	PRO;KEN;WAS	C	G5	
Saxifraga pensylvanica	Swamp Saxifrage	5	PRO	C	G5	
Saxifraga virginiensis	Early Saxifrage	10	PRO;BRI;NEW	C	G5	
Scheuchzeriaceae (Pod-grass Family)						
Scheuchzeria palustris	Pod-grass	1	WAS	SE	G5	
Scrophulariaceae (Figwort Family)						
Agalinis acuta	Sandplain Gerardia	1	WAS	FE	G1	Div 1
Agalinis tenuifolia	Slender Gerardia	9	PRO;NEW	C	G5	
Castilleja coccinea	Painted Cup	0 (1908)	PRO	SH	G5	Div 2
Gratiola virginiana	Virginia Hedge-hyssop	1*	BLO	C	G4G5	Div 2
Limosella subulata	Mudwort	5	NEW;WAS;BLO	C	G4G5	
Penstemon digitalis	Tall White Beard-tongue	2*	PRO	C	G5	

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Scrophulariaceae (continued**)**						
Penstemon hirsutus	Northeastern Beard-tongue	1*	PRO	C	G4	
Scrophularia lanceolata	Hare Figwort	4	PRO;NEW;WAS	C	G5	
Scrophularia marilandica	Maryland Figwort	2	WAS	ST	G5	
Ulmaceae (Elm Family)						
Ulmus rubra	Slippery Elm	3	PRO;KEN C	G5		
Violaceae (Violet Family)						
Viola canadensis	Canada Violet	0 (1920)	PRO	SH	G5	
Viola palmata	Palmate-leaved Violet	2	PRO	ST	G5	IND
Viola pubescens	Smooth Yellow Violet	3	PRO;KEN	C	G5	
Viola rotundifolia	Round-leaved Yellow Violet	3	PRO	C	G5	
Viscaceae (Christmas-mistletoe Family)						
Arceuthobium pusillum	Dwarf Mistletoe	1	PRO	SE	G5	
Xyridaceae (Yellow-eyed Grass Family)						
Xyris montana	Northern Yellow-eyed Grass	3	PRO;WASST	G4		

Appendix I. North American Bird Conservation Plan for New England