Journal of Coastal Research	28	1A	99–104	West Palm Beach, Florida	January 2012

Restoration of Seagrass Habitat in New Jersey, United States

Paul A.X. Bologna[†] **and Michael S. Sinnema**[‡]

[†]Department of Biology and Molecular Biology
Montclair State University
Montclair, NJ 07043, U.S.A.
bolognap@mail.montclair.edu

[‡]LGA Engineering, Inc.
750 Vassar Avenue
Lakewood, NJ 08701, U.S.A.

www.cerf-jcr.org

www.JCRonline.org

ABSTRACT

BOLOGNA, P.A.X. and SINNEMA, M.S., 2012. Restoration of seagrass habitat in New Jersey, United States. *Journal of Coastal Research,* 28(1A), 99–104. West Palm Beach (Florida), ISSN 0749-0208.

Seagrass restoration has often not been successful due to poor site planning, physical disturbance, transplant timing incompatibility, and physical and biological disturbances. As such, these factors are important for successfully restoring seagrasses, and global success has greatly increased. We conducted restorations in the mid-Atlantic region of the United States to reestablish this valuable habitat. Our restoration efforts in New Jersey involved transplants of both *Zostera marina* (eelgrass) and *Ruppia maritima* (widgeon grass). We found that *Z. marina* site success and transplant survival increased over the scope of this 4-year investigation (66%–100% and 34%–43%, respectively). However, *R. maritima* success was heavily dependant upon the year planted; with limited success in 2002 (12%) and high success during 2003 (80%), most likely related to the brown-tide bloom and nonbloom associated with these planting years. For both species restored, ecosystem function was becoming established by the end of the study, demonstrated by their ability to trap and bind fine particulate matter. We provide evidence from this study that seagrass restoration is a viable option for coastal managers and that once established, seagrasses can recover and expand.

ADDITIONAL INDEX WORDS: *Zostera, Ruppia,* eelgrass, widgeon grass.

INTRODUCTION

Seagrasses are dominant coastal plant communities in shallow, soft sediment regions throughout the globe. They are considered among the most productive ecosystems because of their high rates of primary production, which leads to high standing stock biomass. This energy fuels secondary production rates in these systems, which provides a broad foundation for coastal food webs (Heck *et al.*, 2008). Additionally, the aboveground and belowground structures associated with seagrasses provide structural habitat for numerous organisms, especially larval and early juvenile stages of commercially and recreationally important species. As such, seagrass beds provide refugia and trophic resources for many associated species (Unsworth and Cullen, 2010). Beyond biological productivity, seagrasses are system engineers. The structures that they create help to reduce water velocity, attenuate wave action, and increase particle deposition. Their root and rhizome systems subsequently bind these particles and stabilize sediments (Bos *et al.*, 2007). Therefore, it is essential that we understand these crucial systems to ensure stability and productivity of our coastal communities.

Seagrasses are one of the most sensitive indicators of long-term water quality and can be used as a barometer of coastal ecosystem health (Dennison *et al.*, 1993; Krause-Jensen, Greve, and Nielsen, 2005). Changes in the vitality and distribution of these vascular plants generally signal a decline in water quality. Seagrass decline has become a common occurrence in many shallow, temperate, and tropical regions of the world (Orth *et al.*, 2006a, Waycott *et al.*, 2009) and may be a result of anthropogenic nutrient input (Hauxwell, Cebrian, and Valiela, 2003; van Katwijk *et al.*, 2010), disease (Short, Ibelings, and den Hartog, 1988; Short, Matheisson, and Nelson, 1986), and generalized coastal development (Short and Burdick, 1996; Valiela *et al.,* 2000). Frequently, humans are the direct or indirect source of seagrass loss, and as such, it is our responsibility to address the root causes of the declines as well as accelerate the recovery in regions where declines or anthropogenic disturbances are present.

Bologna *et al.* (2000) investigated *Zostera marina* and *Ruppia maritima* distributions in Barnegat Bay, New Jersey, to assess their health and distribution. Results from that study indicated that during the last 25 years total seagrass coverage in the southern portion of the bay had declined by 62%. A broader assessment of seagrasses in New Jersey indicated a 2000- to 3000-ha loss during this same time period (Lathrop *et al.,* 2001). In New Jersey, it has been shown that macroalgal accumulations can cause significant reductions in *Z. marina* biomass through direct smothering, thus eliminating critical recruitment habitat (Bologna, Wilbur, and Able, 2001). Additionally, there are numerous studies that have shown brown-tide blooms significantly reduce light availability to *Z. marina*, which may have caused reductions of eelgrass distribution in the mid-1980s in Long Island, New York, embayments (Dennison, Marshall, and Wigand, 1989). This shading effect may be responsible for changes in health and biomass of *Z. marina* (Bologna *et al.,* 2000), and reduced

DOI: 10.2112/JCOASTRES-D-10-00062.1 received 20 April 2010; accepted in revision 17 August 2010.
© Coastal Education & Research Foundation 2012

recovery of seagrasses revegetation after an algal smothering event caused a rapid decline in coverage in New Jersey (Bologna, Gibbons-Ohr, and Downes-Gastrich, 2007). Since *Z. marina* losses may pose a substantial threat to community structure (*i.e.*, loss of essential habitat), understanding the implications of seagrass loss, as well as how seagrass restoration may reverse this trend, is essential for wise coastal management.

During the last 20 years great advances have occurred in submerged aquatic vegetation (SAV) restoration/mitigation techniques (see Fonseca, Kenworthy, and Thayer, 1998). However, no successful examples exist for New Jersey (Bologna and Sinnema, 2006; Reid, MacKenzie, and Vitaliano, 1993). Some of the limitations to restoration in New Jersey may relate to timing of field activities and techniques. Previous recommendations for New Jersey restoration efforts suggested that activities should occur during the spring (April–June), in accordance with populations located in more northerly habitats (Fonseca, Kenworthy, and Thayer, 1998). However, in New Jersey's estuarine waters, these recommendations may result in the placement of plants in a highly stressed situation with warm water, poor water clarity, and the potential for overgrowth by algae occurring during the summer (Bologna, Wilbur, and Able, 2001). Additionally, prior to 1998, limited data existed regarding *Z. marina* biology and life history (Vaughan, 1982) and no data existed regarding *R. maritima*. Because New Jersey is on the cusp of northerly and southerly populations of *Z. marina*, their restoration survival may be better aligned with mid-Atlantic populations, which indicate higher survival for fall plantings in the Chesapeake Bay (Orth and Moore, 1982). Bologna and Sinnema (2006) showed high initial success and flowering for fall transplanted *Z. marina,* but low survival and no growth for transplanted *R. maritima*. As such, the recommendations of Bird, Jewett-Smith, and Fonseca (1994) for early spring plantings of *R. maritima* in the mid-Atlantic may be more germane for New Jersey. These findings guided our restoration efforts in this study.

METHODS

Study Sites

Restoration efforts were conducted in Barnegat Bay, New Jersey (40°0′N 74°5′W to 39°30′N 74°18′W) from 2001 to 2004 (Figure 1). Barnegat Bay is a wind driven, barrier island protected, lagoonal system. It has two oceanic inlets in the south and middle of the bay and a northerly connection through the Intercoastal Waterway canal. Lathrop *et al.* (2001) estimated total seagrass coverage to be 6083 ha. During the summer of 2001 we investigated sites for restoration. Preplanting site selection included the assessment of site-specific water quality (*e.g.,* salinity, oxygen, water clarity), sediment characteristics (*i.e.,* sand *vs.* mud), and relative sheltering from physical activity (*e.g.,* boat traffic, open fetch). Sediment type and site protection were used collaboratively to minimize the potential that a site would receive too much physical disturbance or could accumulate substantial quantities of drift algae. This information, along with historical knowledge of SAV presence in the region (Macomber and Allen, 1979), was used to identify sites for restoration activities. Five donor beds were also selected for restoration efforts and included Shelter Island, Barnegat Inlet, and Herring Island (*Z. marina* restorations) and Seaside and Mordecai (*R. maritima* restorations). These sites were chosen for their accessibility and the presence of dense, healthy stands of seagrass (Figure 1).

Seagrass Restorations

During the course of restoration activities, we principally used the peat pot planting unit method, but also included a bundled-staple unit methodology for sites that exhibited higher physical characteristics (see Fonseca, Kenworthy, and Thayer, 1998, for technique descriptions). We transplanted *Z. marina* during the fall (September–November) to eight sites in 2001, six sites in 2002, and four sites in 2004. All sites were planted either as four 7×7 m plots with planting units at 1-m spacings (196 m^2) or four 6.75 m $\times$ 6.75 m plots with planting units at 0.75-m spacings (175 m^2). Spacing differences were used to assess the efficacy of closer spacing among restoration efforts. Plots were located using a global positioning system (GPS), and corners of restoration plots were anchored with 1-inch PVC pipe for monitoring location identification. Restoration efforts were conducted in the following manner: restoration planting units were collected from the identified donor sites, and planting units were transferred to restoration sites and planted during the same day. *Ruppia maritima* was transplanted during May and June to eight sites in 2002 and five sites in 2003. Sites were planted using peat pots at 1-m spacing in 7 m $\times$ 7 m grids in the same manner described above.

Restoration Monitoring

Restoration sites were monitored approximately 8–12 months after restoration efforts. The variability of monitoring occurred due to logistical constraints and weather-related constraints. Sites were monitored in the following manner: the original plot locations were identified using GPS, corner stakes were relocated, and the overlying grid was laid upon the planting grid. A survey of each grid was completed with an assessment of the survival of planting units and the abundance of shoots.

To test whether restoration activities could facilitate trapping and binding of fine particulate material in the region, we collected 2.54 cm diameter core samples to a depth of 5 cm. Sediment samples were collected prior to restoration activities and during the monitoring events. Samples were returned to the laboratory, dried to constant weight at 80°C, and then sieved through a graded sieve series (4, 2, 1, 0.5, 0.25, 0.125, 0.063 mm, pan). Weights were recorded and mean sediment size calculated. For each sediment sample, a mean phi (Φ) was calculated via the Folk and Ward method (1957) produced in the Gradistat 4.0 program developed by Blott and Pye (2001). Comparisons in mean phi were made between prerestoration and postrestoration activities using paired t-tests, because initial sediment sizes were not identical for all sites. Comparisons were made by assessing each seagrass species and whether the site showed successful establishment of planting units or not.

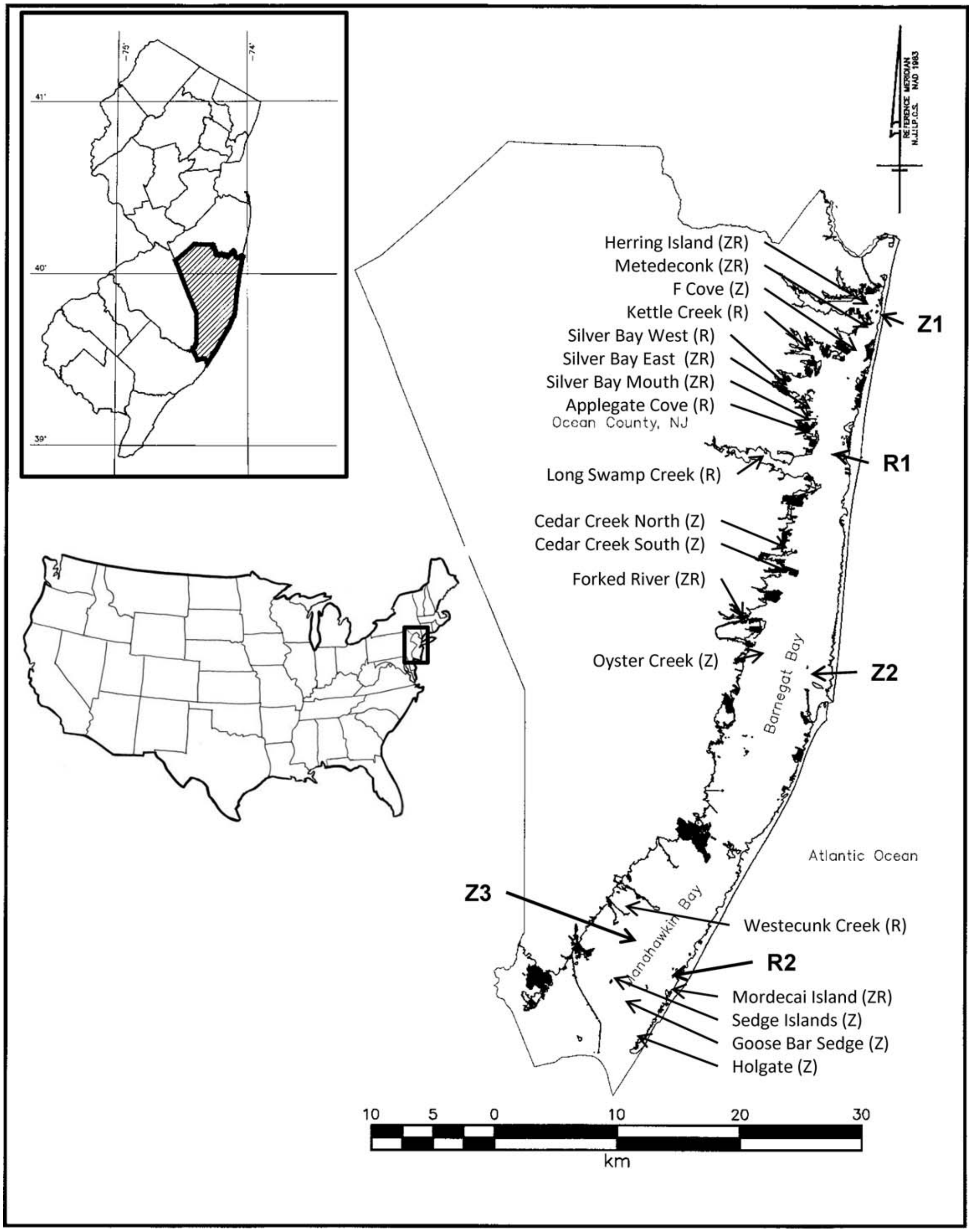

Figure 1. Restoration site locations in Barnegat Bay, New Jersey. Sites denoted with "Z" and/or "R" received *Zostera marina* and/or *Ruppia maritima* restorations, respectively. Donor sites depicted with Z or R indicate species present: Z1 Herring Island, Z2 Barnegat Inlet, Z3 Shelter Island, R1 Seaside, R2 Mordecai.

RESULTS

Zostera marina

During the 3 years of planting efforts, *Z. marina* planting unit survival varied widely among sites (0–94%, Figure 2a). Four sites showed no transplant survival, while seven showed >50% survival. During the course of this investigation, our ability to hone the restoration location selection criteria led to an increase in site success with survival rates of 75% in 2001, 66% in 2002, and 100% in 2004. Transplant survival rates

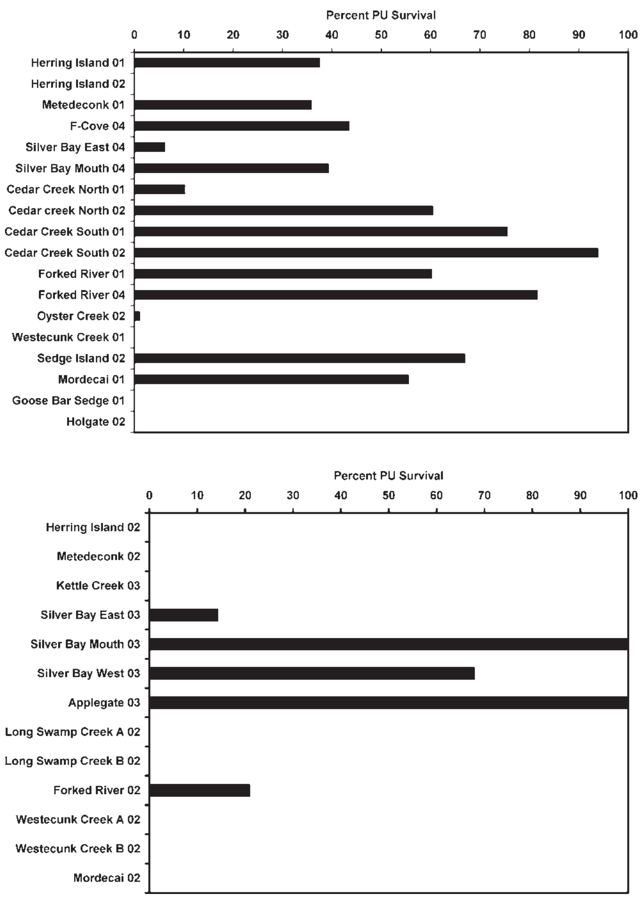

Figure 2. Survival rate of transplanted SAV planting units (PU) for individual sites. Sites designated by name and year transplanted from north to south in Barnegat Bay. (a, upper) *Zostera marina,* (b, bottom) *Ruppia maritima*.

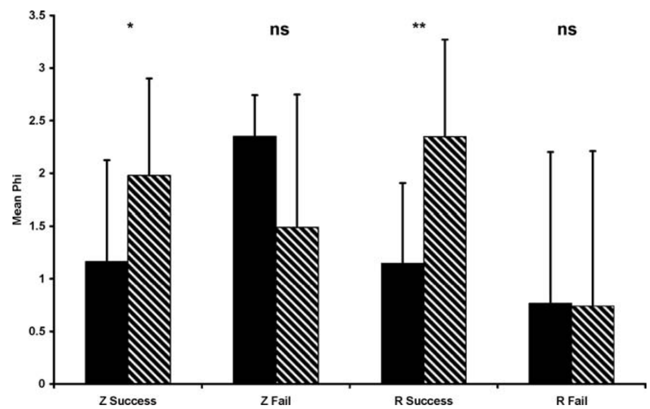

Figure 3. Mean phi (Φ) value (±standard deviation) comparisons for sediment samples collected during restoration events, assessed by species and restoration success (Z = *Zostera marina*, R = *Ruppia maritima*).

within successful sites also increased, with 34.3% survival in 2001, 37.04% in 2002, and 42.6% in 2004. Overall, shoot density varied among sites (0.025–188 m^{-2}), but averaged 17.7 m^{-2}. Reproductive shoots were present in all but one of the sites; however they were not quantified. Additionally, no difference was seen in success between sites planted at 1-m spacing and 0.75-m spacing, nor between peat pot *vs.* bundled, stapled planting units.

For most sites that showed no survival, evidence of physical disturbance by storms and/or ice scour were present. Westecunk Creek (2001), Holgate (2002), and Herring Island (2002) had adjacent salt marshes that had been sheared with numerous large chunks of peat redistributed into the restoration plots. Additionally, peat pots and planting staples were found deposited on the salt marsh surface.

Ruppia maritima

Survival of *R. maritima* was very different between years of planting. In fact, only one site in 2002 showed any survival, while plantings in 2003 showed 80% site survival (Figure 2b). Additionally, several sites planted in 2003 showed not only 100% planting unit survival, but 100% spatial coverage of the monitored region planted and beyond the borders of the 7 m × 7 m grids used. This demonstrated substantial lateral growth and expansion for these sites. Since quantitative shoot counts were not possible in the field, 10.54 cm cores were collected to assess shoot density. On average, shoot density was 5862 m^{-2}, with one site exceeding 10,000 m^{-2}.

Sediments

Results from our sediment collections showed that when sites were successful, sediment size structure was shifted to smaller size fractions and mean phi increased significantly (Figure 3). Specifically, for *Z. marina,* successful sites showed a mean increase of 0.814 phi units (t_8 = 2.35, $p < 0.05$), while sites showing no survival showed a −0.86 decrease (t_5 = 1.84, $p > 0.1$). A more impressive change was observed for *R. maritima* sites, which showed a mean phi increase of 1.2 (t_4 = 6.66, $p < 0.003$), while unsuccessful sites showed a minor decrease of 0.03 phi (t_1 = 1.06, $p > 0.4$).

DISCUSSION

The global importance of seagrasses as essential habitat for fish and invertebrates has been established for decades (Heck, Nadeau, and Thomas, 1997). Their ecosystem contributions include nutrient cycling, reductions in flow regimes and particulate removal, sediment stabilization and reduced erosion, and dissipation of storm energy to coastal communities. Along the Atlantic Coast of the United States, *Z. marina* declines have been linked to various factors including disease (Short, Ibelings, and den Hartog, 1988), macroalgal smothering (Bologna, Gibbons-Ohr, and Downes-Gastrich, 2007; Hauxwell *et al.,* 2001), and changes in water quality (Kemp *et al.,* 2004). The problem facing many coastal managers relates to minimizing losses and increasing coverage through restoration efforts. In New Jersey, the wasting disease outbreak in the 1930s is thought to be responsible for the elimination of *Z. marina* in the southern part of the state. Since limited natural transport of seeds can occur across long distances, active restoration may lead to significant increases in spatial coverage if successfully reestablished in these regions (*sensu* Orth *et al.,* 2006b). Essentially, once small populations are established, they can expand through vegetative growth and local seed dispersal.

Our results show mixed success between species and among sites and years (Figure 2). Several major factors associated with lack of success include water clarity changes due to brown-tide development, winter ice scour, and fall storm events. Each of these factors may have resulted in the loss of planting units, but in different ways. For *Z. marina*, the water clarity issue was limited in scope as a result of the planting scheme of fall restorations. This afforded these plants several months of fall growth and then rapid spring growth and flowering before the onset of brown-tides and warm summer temperatures. Perhaps we were fortunate that the brown-tide events of 2001 and 2002 (Gastrich *et al.*, 2004) did not appear to substantially affect our *Z. marina* transplants, since brown-tide is known to have significant impact on *Z. marina* (Dennison, Marshall, and Wigand, 1989). *Zostera marina* appeared to be most affected by the physical disturbance of storms and ice as a result of the planting time and the prevalence of these physical disturbances in the fall and winter. These results are similar to those of Davis and Short (1997), who identified ice scour as a significant loss of planting units, and to those of Reusch and Chapman (1995), who demonstrated losses due to storms. For *R. maritima*, the late spring restoration efforts of 2002 led to early success but substantial loss of planting units within 1 year (Figure 2b). Remarkably, sites restored in 2003 showed high planting unit survival and substantial growth and expansion. For two sites, we achieved not only 100% survival, but 100% spatial coverage on the site and substantial growth beyond the 196-m^2 transplanted region. This explosive growth was also seen by Bell, Robbins, and Jensen (1999), who witnessed a 15-m expansion in bed dynamics for *Halodule wrightii*. The primary difference between years was the large-scale brown-tide events in the bay in 2002 (Gastrich *et al.,* 2004), which substantially decreased light availability, and the lack of brown-tide development in 2003 (Lathrop and Haag, 2005). In fact, Lathrop and Haag (2005) showed significantly lower Secchi disk depths (0.8 m *vs.* 1.2 m) between brown-tide years (2001/2002) and non–brown-tide years (2003/2004) indicating greater light penetration during this later period. This difference in brown-tide bloom development in 2003 may potentially be due to the extremely high spring recruitment of blue mussels (*Mytilus edulis*) into Barnegat Bay with filtration potentials exceeding 15 m^3 of water m^{-2} day^{-1} (Bologna *et al.*, 2005), limiting the development of brown-tides (*sensu* Cerrato *et al.,* 2004). Moore, Neckles, and Orth (1996) and Moore, Wetzel, and Orth (1997) found that significant light reduction was extremely detrimental to seagrass transplants in the Chesapeake Bay, and this line of reasoning would lead to the limited restoration success in 2002 for *R. maritima,* but few limitations in 2003.

One of the important ecology benefits of seagrasses is their potential to trap fine particulate material and then bind these particles into the sediment through the development of the rhizome mat. Our results showed that when restoration efforts were successful, mean phi size increased (Figure 3), while grain size did not change for sites that were unsuccessful. This demonstrated that our efforts were able to remove fine particulate material and sediment them into the benthos. This has been shown by Fonseca and Fisher (1986) and Bos *et al.* (2007), who demonstrated that sediment size structure decreases in the presence of functioning seagrass habitat. Our results hold true for both *Z. marina* and *R. maritima,* despite the lower planting unit survival rates for the successful *Z. marina* planting. The very large change in phi for *R. maritima* was directly related to the substantial growth and expansion within the restoration time frame. With shoot densities exceeding 5000 m^{-2}, these beds were providing predicted ecosystem functions.

CONCLUSION

Through our research, we have demonstrated that seagrasses in New Jersey can be restored and the optimal timing of restoration efforts has been identified. This is a critical step in the management of these communities, since continued coastal development puts environmental pressure on the system. If we are to effectively manage the resource, there needs to be clear evidence that restoration and mitigation of habitats can succeed. In this way, coastal decision makers can address the potential habitat loss and develop clear objectives for restoration, mitigation, and remediation of impacted sites. Long-term goals of regional habitat requirements are needed to ensure that seagrass habitats are present and providing food, refuge, and ecosystem functions to coastal bays. Only through research, wise development, and enforceable mandates can we plan for a sustainable coastal future.

ACKNOWLEDGMENTS

The authors would like to thank D. Maio, S. You, S. Margiotti, E. Wolek, P. Sokoloff, J. Misiuk, K. Luckhardt, S. Regetz, R. Papagian, R. Fiorile, R. Watson, A. Raichle, J. Patterson, and A. Lettieri, for their assistance in the field and laboratory. We also would like to thank L. Hazard, M. Egan, D. Vanderklein, and three anonymous reviewers for critical assessment of this manuscript. This research was supported by the New Jersey Department of Environmental Protection through the EPA 319H program (P.A.X.B. and M.S.S.) and laboratory facilities support through Montclair State University. The views expressed herein do not necessarily reflect the views of any of those organizations.

LITERATURE CITED

Bell, S.; Robbins, B., and Jensen, S., 1999. Gap dynamics in a seagrass landscape. *Ecosystems*, 2, 493–504.

Bird, K.T.; Jewett-Smith, J., and Fonseca, M.S., 1994. Use of in vitro propagated *Ruppia maritima* for seagrass meadow restoration. *Journal of Coastal Research*, 10, 7342–737.

Blott, S.J. and Pye, K., 2001. Gradistat: a grain size distribution and statistics package for the analysis of unconsolidated sediments. *Earth Surface Processes and Landforms,* 26, 1237–1248.

Bologna, P.; Fetzer, M.; Moody, E., and McDonnell, S., 2005. Assessing the potential benthic-pelagic coupling in episodic blue mussel (*Mytilus edulis*) settlement events within eelgrass (*Zostera* *marina*) communities. *Journal of Experimental Marine Biology and Ecology,* 316, 117–131.

Bologna, P.; Gibbons-Ohr, S., and Downes-Gastrich, M., 2007. Recovery of eelgrass (*Zostera marina*) after a major disturbance event in Barnegat Bay, New Jersey, USA. *Bulletin of the New Jersey Academy of Sciences,* 52, 1–7.

Bologna, P.; Lathrop, R.; Bowers, P., and Able, K., 2000. Assessment of Submerged Aquatic Vegetation in Little Egg Harbor, New Jersey. Technical Report 2000-11. New Brunswick, New Jersey: Institute of Marine and Coastal Sciences, Rutgers, the State University of New Jersey. 30p.

Bologna, P. and Sinnema, M., 2006. Assessing the Success of Eelgrass Restoration in New Jersey. *In:* Treat, S. and Lewis, R. (eds.), *Seagrass Restoration: Success, Failure and the Costs of Both*. Valrico, Florida: Lewis Environmental Services, Inc., pp. 79–90.

Bologna, P.; Wilbur, A., and Able, K., 2001. Reproduction, population structure, and recruitment limitation in a bay scallop (*Argopecten irradian*s Lamarck) population from New Jersey, USA. *Journal of Shellfish Research,* 20, 89–96.

Bos, A.; Bouma, T.; de Kort, G.; and van Katwijk, M., 2007. Ecosystem engineering by annual intertidal seagrass beds: sediment accretion and modification. *Estuarine, Coastal and Shelf Science,* 74, 344–348.

Cerrato, R.M.; Caron, D.A.; Lonsdale, D.J.; Rose, J.M., and. Schaffner, R.A., 2004. Effect of the northern quahog *Mercenaria mercenaria* on the development of blooms of the brown-tide alga *Aureococcus anophagefferens*. *Marine Ecology Progress Series,* 281, 93–108

Davis, R.C. and Short, F.T., 1997. Restoring eelgrass, *Zostera marina* L., habitat using a new transplanting technique: the horizontal rhizome method. *Aquatic Botany,* 59, 1–15.

Dennison, W.; Marshall, G., and Wigand, C., 1989. Effect of "brown tide" shading on eelgrass (*Zostera marina* L.) distributions. *In:* Cosper, E.; Bricelj, M., and Carpenter, E. (eds.), *Novel Phytoplankton Blooms*. New York: Springer-Verlag, pp. 675–692.

Dennison, W.; Orth, R.; Moore, K.; Stevenson, J.; Carter, V.; Kollar, S.; Bergrsrom, P., and Batiuk, R., 1993. Assessing water quality with submersed aquatic vegetation: habitat requirements as barometers of Chesapeake Bay health. *BioScience,* 43, 86–94.

Folk, R.L. and Ward, W. C., 1957. Brazos River bar: a study in the significance of grain size parameters. *Journal of Sediment Petrology,* 27, 3–26.

Fonseca, M. and Fisher, J., 1986. A comparison of canopy friction and sediment movement between four species of seagrass with reference to their ecology and restoration. *Marine Ecology Progress Series,* 29, 15–22.

Fonseca, M.S.; Kenworthy, J.W., and Thayer, G.W., 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*. Silver Spring, Maryland: NOAA Coastal Ocean Office, 222p.

Gastrich, M.; Lathrop, R.; Haag, S.; Weinstein, M.; Danko, M.; Carone, D., and Schaffner, R., 2004. Assessment of brown tide blooms, caused by *Aureococcus anophagefferens*, and contributing factors in New Jersey coastal bays: 2000–2002. *Harmful Algae,* 3, 305–320.

Hauxwell, J.; Cebrian, J.; Furlong, C., and Valiela, I., 2001. Macroalgal canopies contribute to eelgrass (*Zostera marina*) decline in temperate estuarine ecosystems. *Ecology,* 82, 1007–1022.

Hauxwell, J.; Cebrian, J., and Valiela, I., 2003. Eelgrass *Zostera marina* loss in temperate estuaries: relationships to land-derived nitrogen loads and effect of light limitation imposed by algae. *Marine Ecology Progress Series,* 247, 59–73.

Heck, K.; Carruthers, T.; Cuarte, C.; Hughes, A.R.; Kendrick, G.; Orth, R.; and Williams, S., 2008. Trophic transfer from seagrass meadows subsidize diverse marine and terrestrial consumers. *Ecosystems,* 11, 1198–1210.

Heck, K.; Nadeau, D., and Thomas, R., 1997. The nursery role of seagrass beds. *Gulf of Mexico Science,* 15: 50–54.

Kemp, W.M.; Batuik, R.; Bartleson, R.; Bergstrom, P.; Carter, V.; Gallegos, G.; Hunley, W.; Karrh, L.; Koch, E.; Landwehr, J.; Moore, K.; Murray, L.; Naylor, M.; Rybicki, N.; Stevenson, J.C., and Wilcox, D., 2004. Habitat requirements for submerged aquatic vegetation in Chesapeake Bay: water quality, light regime, and physical-chemical factors. *Estuaries,* 27, 363–377.

Krause-Jensen, D.; Greve, T.; and Nielsen, K., 2005. Eelgrass as a bioindicator under the European water framework directive. *Water Resources Management,* 19, 63–75.

Lathrop, R.G. and Haag, S., 2005. Brown Tide Assessment Project Years 2000–2004: Developing Indicators of Brown Tide Blooms in NJ Coastal Waters. Final Report, NJDEP/DSRT Contract No. SR04-047. Trenton, New Jersey: New Jersey Department of Environmental Protection, Division of Science, Research and Technology, 126p.

Lathrop, R.; Styles, R.; Seitzinger, S., and Bognar, J., 2001. Use of GIS mapping and modeling approaches to examine the spatial distribution of seagrasses in Barnegat Bay, New Jersey. *Estuaries,* 24, 904–916.

Macomber, R. and Allen, D., 1979. The New Jersey Submerged Aquatic Vegetation Distribution Atlas Final Report. Technical Report. Washington, DC: Earth Satellite Corporation.

Moore, K.A.; Neckles, H.A., and Orth, R.J., 1996. *Zostera marina* (eelgrass) growth and survival along a gradient of nutrients and turbidity in the lower Chesapeake Bay. *Marine Ecology Progress Series,* 142, 247–259.

Moore, K.; Wetzel, R., and Orth, R., 1997. Seasonal pulses of turbidity and their relations to eelgrass (*Zostera marina* L.) survival in an estuary. *Journal of Experimental Marine Biology and Ecology,* 215, 115–134.

Orth, R.J.; Carruthers, T.; Dennison, W.C.; Duarte, C.M.; Fourqurean, J.W.; Heck, K.L.; Hughes, A.R.; Kendrick, G.A.; Kenworthy, W.J.; Olyarnik, S.; Short, F.T.; Waycott, M., and Williams, S., 2006a. A Global Crisis for Seagrass Ecosystems. *BioScience,* 56, 987–996.

Orth, R.J.; Luckenbach, M.L.; Marion, S.R.; Moore, K.A., and Wilcox D.J., 2006b. Seagrass recovery in the Delmarva Coastal Bays, USA. *Aquatic Botany,* 84, 26–36.

Orth, R. and Moore, K., 1982. The Biology and Propagation of *Zostera marina*, Eelgrass, in the Chesapeake Bay, Virginia. Applied Science and Ocean Engineering, Special Report Number 265. Gloucester Point, Virginia: Virginia Institute of Marine Science, 187p.

Reid, R.; MacKenzie, C., and Vitaliano, J., 1993. A Failed Attempt to Re-establish Eelgrass in Raritan Bay (New York/New Jersey). NOAA/NMFS/NEFSC, Ref. Doc. 93, 27p.

Reusch, T.B. and Chapman, A.R.O., 1995. Storm effects on eelgrass (*Zostera marina* L.) and blue mussels (*Mytilus edulis* L.) beds. *Journal of Experimental Marine Biology and Ecology* 192, 257–271.

Short, F. and Burdick, D., 1996. Quantifying eelgrass habitat loss in relation to housing development and nitrogen loading in Waquoit Bay, Massachusetts. *Estuaries,* 19, 730–739.

Short, F.; Ibelings, B., and den Hartog, C., 1988. Comparison of a current eelgrass disease to the wasting disease of the 1930s. *Aquatic Botany,* 30, 295–304.

Short, F.; Matheisson, A., and Nelson, J., 1986. Recurrence of the eelgrass wasting disease at the border of New Hampshire and Maine, USA. *Marine Ecology Progress Series,* 29, 89–92.

Unsworth, R. and Cullen, L., 2010. Recognising the necessity of Indo-Pacific seagrass conservation. *Conservation Letters,* 3, 63–73.

Valiela, I.; Geist, M.; McClelland, J., and Tomasky, G., 2000. Nitrogen loading from watersheds to estuaries: verification of the Waquoit Bay nitrogen loading model. *Biogeochemistry,* 49, 277–293.

van Katwijk, M.; Bos, A.; Kennis, P.; and de Vries, R., 2010. Vulnerability to eutrophication of a semi-annual life history: a lesson learnt from and extinct eelgrass (*Zostera marina*) population. *Biological Conservation,* 143, 248–254.

Vaughan, D., 1982. Production ecology of eelgrass (*Zostera marina* L.) and its epiphytes in Little Egg Harbor, New Jersey. New Brunswick, New Jersey: Rutgers, the State University of New Jersey, Ph.D. thesis, 137p.

Waycott, M.; Duarte, C.; Carruthers, T.; Orth, R.; Dennison, W.; Olyarnik, S.; Calladine, A.; Fourqurean, J.; Heck, K.; Hughes, A.R.; Kendrick, G.; Kenworthy, W.; Short, F., and Williams, S., 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences,* 106, 12377–12381.