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Technical Report TR-37

Eelgrass Restoration Used As Construction Impact Mitigation in Boston Harbor, Massachusetts

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Eelgrass Restoration Used As Construction Impact Mitigation in Boston Harbor, Massachusetts

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Abstract: The Massachusetts Division of Marine Fisheries (*MarineFisheries*) re-established eelgrass (*Zostera marina*) in Boston Harbor, Massachusetts as partial mitigation for assumed impacts from the construction of the 29-mile long "HubLine" natural gas pipeline in Massachusetts Bay. Restoration of eelgrass habitat provides shelter, food, and has the potential to positively affect abundance of a number of finfish and invertebrate species that were judged to be potentially impacted by the construction. Improved water quality from advanced wastewater treatment in this previously degraded estuary and deployment of a comprehensive restoration site-selection process were necessary precursors to a successful effort. However, despite municipal improvements to water quality, the number of possible restoration sites was severely limited by poor sediment quality. Hydrodynamic modelling efficiently focused our restoration efforts by indicating that natural spreading of eelgrass via seed shoots was likely within and near most of our selected transplant locations. Planting was conducted using a combination of hand- and frame-planting, and seed dispersal followed by survival and biological monitoring. Both vegetative and non-vegetative shoot density expansion was significant, and after 2 years, total areal coverage was over 2 hectares (~ 5 acres). Planted beds approached or exceeded healthy local natural beds both in habitat structure and faunal abundance and diversity. Outreach was an important part of this restoration project. We provided a "hands-on" educational experience for members of the community and promoted stewardship of this valuable resource.

Introduction

From 2004-2007, the Massachusetts Division of Marine Fisheries (*MarineFisheries*) restored eelgrass in Boston Harbor, MA as partial mitigation for assumed impacts to the environment and biota from the construction of the "HubLine" natural gas pipeline across Massachusetts Bay. Pipeline construction activities during 2002-2003 exceeded recommended time-of-year work windows and were determined to potentially impact a number of finfish and invertebrate species including crustaceans, flounder, gadids, and anadromous fish. Restoration of eelgrass habitat was chosen as a mitigation option because it had the potential to provide shelter and food and positively impact populations of these species.

The ecosystem value of eelgrass (*Zostera marina*) is well documented. Eelgrass acts to stabilize sediments, buffer wave energy, and provide habitat for juvenile fish and shellfish (Stauffer 1937; Orth et al. 1984; Heck et al. 1989; Hughes et al. 2002; Lazarri and Tupper 2002). Decline of this important marine plant has been tracked throughout its range (Jacobs 1979; Short et al. 1986; Valiela et al. 1992; Short and Burdick 1996). It has been estimated that 90% of eelgrass died off in the 1930s due to an outbreak of wasting disease (Tutin 1942). While wasting disease continues to occur sporadically (Short et al. 1986, 1987), natural re-population has been thwarted by degraded water quality from coastal development, which limits light essential for eelgrass growth (Batuik et al. 2000). This problem is compounded by the limited ability of eelgrass to disperse to suitable areas over long distances.

The clear relationship between eutrophication and eelgrass loss (Kemp et al. 1983; Valiela et al. 1992; Short et al. 1996; Hauxwell et al. 2001, 2003; Cardoso et al. 2004) underscores the futility of attempting restoration where water quality remains poor. In addition, physical and biological changes that can occur in an area when eelgrass is lost may inhibit natural re-vegetation (Rasmussen 1977; Duarte 1995; Short et al. 2002b). In fact, attempts to actively restore eelgrass have met with varied success, and many failures (Homziak et al. 1982; Thom 1990; Fonseca et al. 1998). Just 31% of restoration sites reviewed in Short et al. (2002a) succeeded in establishing eelgrass and many of these were only on a test transplant scale (<0.01 hectares). Careful site-selection is now recognized as an essential precursor to any restoration project (Fonseca et al. 1998; Short et al. 2002a; Kopp and Short 2003) and should improve the poor record of past attempts. Short et al. (2002a) developed a site-selection model with criteria based on some of their most successful transplant sites. Criteria included historical and current eelgrass distribution, proximity to natural eelgrass beds, sediment, wave exposure, water depth, and water quality. Further field testing is done at sites identified by the model, including light measurements and surveys of bioturbators. We used the Short et al. model (hereinafter the Short model) with some modifications as the basis for our site selection process in Boston Harbor.

Wastewater management upgrades in Boston Harbor presented an excellent opportunity to assess eelgrass restoration success in an area where most beds had disappeared due to severe eutrophication, but where major water quality improvements could enable its growth. The improvements in water and sediment qualities increased the potential for eelgrass growth, but it was unlikely that this growth would happen naturally on an acceptable time scale. *MarineFisheries* therefore undertook an eelgrass restoration project in the Harbor with the goal of "jump-starting" the re-colonization of eelgrass to this embayment. Rigorous attention to site selection was essential due to the massive physical and ecological changes which the Harbor had undergone.

Eelgrass spreads both vegetatively (rhizome expansion) and non-vegetatively (seeds). Vegetative spreading is limited to adjacent areas, so the natural spread of eelgrass to new areas must be accomplished by the dispersal of seeds. Eelgrass seeds are negatively buoyant and do not travel far within the water column once released from a vegetative shoot (Orth et al. 1994). Detached reproductive eelgrass shoots containing seeds can float long distances, and thus can start new meadows far from the bed of origin (Harwell and Orth 2002a and 2002b). However, such a scenario requires several assumptions: first, that seed shoots are available in the area from existing beds; second, that local currents carry the shoots to an area that is suitable for eelgrass propagation given requirements for water quality, depth (light), and sediment; third, that shoots will sink or the seeds drop out coincident with each shoot's travel over such an area; and, finally, that any seeds that do sink in suitable areas will germinate and survive burial, grazing, etc. to grow into a viable plant. We took these dynamics into account to determine the likelihood that eelgrass could have naturally re-colonized Boston Harbor, and to focus on areas in our site selection process where eelgrass would become self-spreading.

Methods

Study Area. Boston Harbor is a relatively shallow (4.9m average depth), tide-dominated estuary located on the western edge of Massachusetts Bay within the Gulf of Maine (Figure 1). The 125 km^2 area is broken up by numerous small islands. Tidal range averages 2.7 m (Signell and Butman 1992). The City of Boston (population 590,000) lies directly on the Harbor, and until the 1990's virtually the entire volume of sewage from the city and surrounding area (population 2 million) was discharged directly into the water body with minimal or no treatment. Prior to 1991, Boston Harbor received well over 100,000 metric tons of suspended solids annually (Knebel 1992; Rex et al. 2002). By the time *MarineFisheries*' efforts began in 2004, eelgrass in the Harbor had been reduced to 3 remnant beds Two of these beds had declined in area by over half between the 1995 and 2001 Massachusetts Department of Environmental Protection (DEP) Wetlands Conservancy Program surveys. A fourth bed shown on these surveys had virtually disappeared by the time we surveyed it in 2004. Cause of disappearance in the area was very likely poor water quality (http://www.mass.gov/dep/water/resources/maps/eelgrass/eelgrass.htm). A massive wastewater collection and treatment project for the Boston area, The Boston Harbor Project, curtailed sludge discharge in 1991. Over the next 9 years, effluent treatment was upgraded to secondary; in 2000, wastewater discharge was diverted from within Boston Harbor, via an outfall pipe extension, to an area 15 km offshore in Massachusetts Bay. Intensive studies conducted by the Massachusetts Water Resources Authority (MWRA), United States Geological Survey (USGS), and others have led to a good understanding of the impacts of the wastewater system upgrades on water and sediment quality in the Harbor.

Geometry of the estuary mouth, combined with regional bathymetry, define an ebb tide-dominated system, with net flushing of water from within the Harbor on each outgoing tide (Signell and Butman 1992). This flushing has accelerated water and sediment quality improvements since initiation of the wastewater treatment project. MWRA reported that, five years after the offshore transfer, almost all water-quality parameters had improved for the whole Harbor or for individual stations (Table 1; Taylor 2006).

Substrate within the Harbor is dominated by depositional sediment (silts, clayey silts, and sandy silts) in a patchy distribution. Gravel is also found within the Harbor (Knebel et al.1991; Knebel 1993; Knebel and Circé 1995). Tide- and wind-driven current patterns vary in different parts of the Harbor (Signell and Butman 1992; Knebel 1993); consequently improvements to both water and sediment have been more rapid in better-flushed areas (Taylor 2006; Tucker et al. 2006).

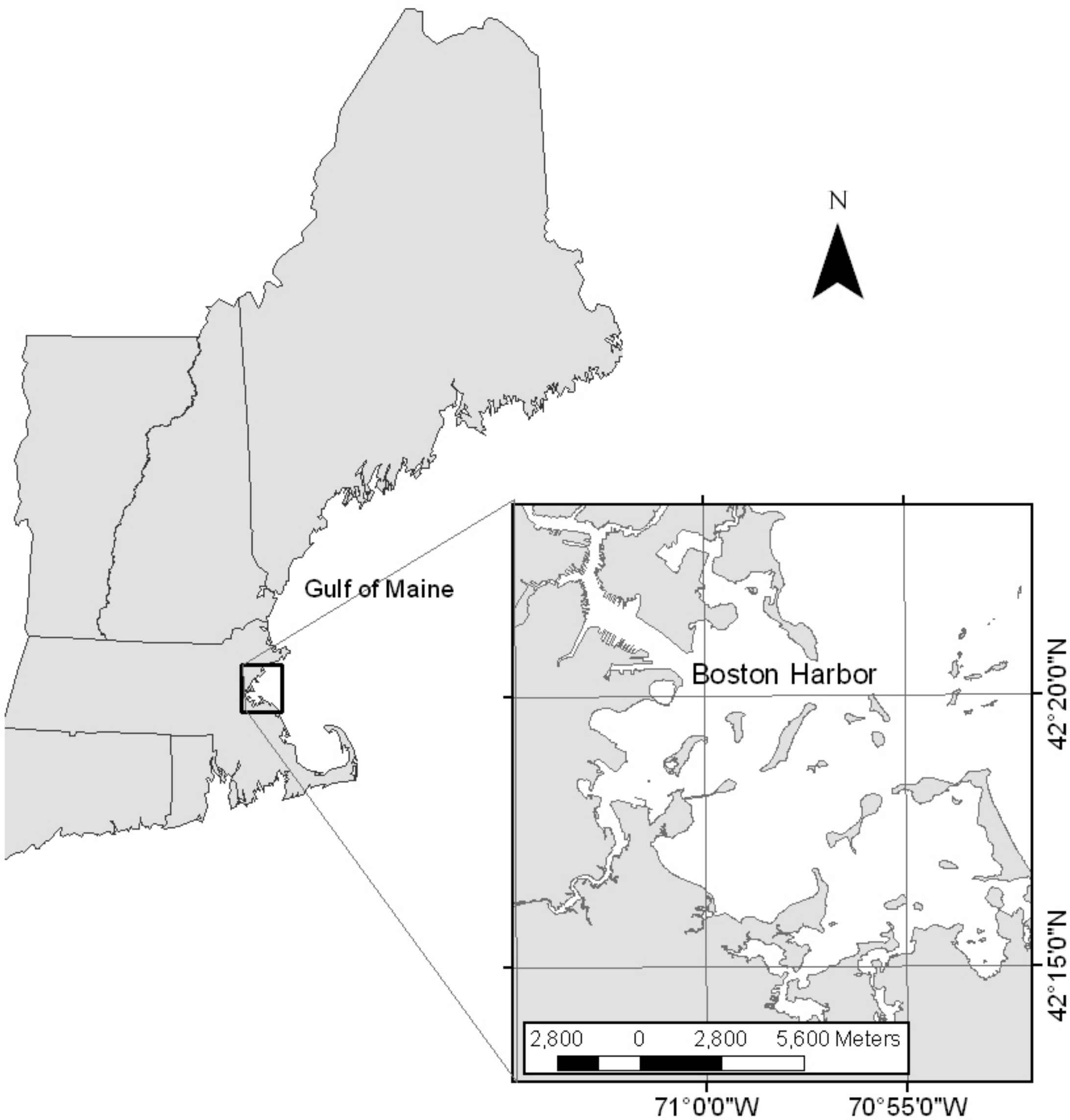


Figure 1. Boston Harbor, Massachusetts.

Table 1. Summary of differences in selected water-quality parameters in Boston Harbor at baseline and 5 years after outfall went online (adapted from Table 1 in Taylor (2006)). All current values are considered "improvements" for eelgrass except that shaded in gray. Recommended requirements for eelgrass (Batiuk et al. 2000) are provided for comparison where available (NA= Not available).

Variable	% increase (+) or decrease (-) at 5-y	Current value	Recommended requirements for eelgrass
Total nitrogen (TN) (μmol l-1)	-35	20.2 ± 2.9	NA
Dissolved inorganic nitrogen (DIN) (mg l-1)	-55	.074 ± .049	<0.15 (mg l-1)
Total Phosphorus (TP)	-28	1.48 ± 0.31	NA
Dissolved inorganic phospohorus (DIP)	-15	0.02 ± .0084	<0.02 (mg l-1)
Total chlorophyll-a (μg l-1)	-26	4.8 ± 2.4	<15
Total suspended solids (TSS) (mg l-1)	5	3.8 ± 1.1	<15
Percent organic carbon (POC) as % TSS	-33	12 ± 3	NA
Secchi depth (m)	4	2.7 ± 0.70	NA
Dissolved oxygen (DO) (mg l-1)	5	8.9 ± 1.3	NA

Existing water quality standards for eelgrass were met in Boston Harbor following the outfall diversion. However, the Harbor is dominated by fine-grained sediment. Sediment guidelines for eelgrass vary widely in the literature, ranging from a silt/clay fraction of <20% (Koch 2001) to < 70% (Short et al. 2002a). Pore water sulfide levels were recommended at <400 μM and Total Organic Carbon (TOC) < 5% (Koch 2001). Sediment in existing eelgrass beds is often very fine (Wanless 1981; Smith et al. 1988), largely due to the trapping and settling of suspended particles by leaves extending into the water column. Accumulation of organic matter and inability of oxygen to diffuse very far into fine sediments often creates anoxic sediment below a centimeter or so in existing eelgrass meadows (Klug 1980; Thayer et al. 1984; Huettel and Gust 1992). Nevertheless, very fine-grained sediment in unvegetated areas may be problematic for eelgrass transplants. It is easily re-suspended, and can worsen light attenuation; it is also subject to high porewater sulfide levels which can lead to H_2S toxicity in eelgrass (Barko and Smart 1983; Carlson et al. 1994; Goodman et al. 1995; Holmer et al. 1997; Koch 2001). Because of the variation in these parameters among stations noted by Tucker et al. (2006), and the wide range of recommended levels in the literature, it was important to conduct sediment quality work as a component of site selection.

Data Collection. *MarineFisheries*' Eelgrass Restoration Project efforts were initiated in spring 2004 and concluded in fall 2007. They encompassed site selection, permitting, test transplants, large-scale plantings, development and evaluation of planting methods, survival/expansion assessment, and comparison of habitat quality between planted and existing eelgrass beds and unvegetated control sites. In addition, an outreach component involved a number of hands-on volunteers and educated the public in multiple different forums. Estrella (2005) reported activities through June 2005, and Leschen et al. (2006 and 2007) summarized activities in 2005 and 2006 field seasons. This report combines information from the three previous reports, and also includes results from the 2007 field season.

Permitting. MassGIS eelgrass areal coverage was overlaid on Massachusetts Bay nautical charts with town water boundaries to determine municipal responsibility for each area. The Program Coordinator contacted shellfish constables and conservation commissions from towns surrounding Boston Harbor regarding our intent to harvest and transplant submerged aquatic vegetation (SAV). All constables were called and their respective town conservation commissions received a letter of introduction, a description of work to be accomplished, and a request for input on local permitting guidelines and requirements.

Considerable time was spent researching eelgrass restoration permit requirements with pertinent agencies and submitting appropriate documents. All necessary permit applications were filed including Notices of Intent with the seven affected towns (Boston, Hull, Hingham, Weymouth, Quincy, Revere, and Nahant – Winthrop had already been eliminated as a possibility before permitting began) and DEP. A PowerPoint presentation on our eelgrass restoration work was developed for communication to Town Conservation Commissions during our Notice of Intent hearings. Orders of Conditions were subsequently received from all towns. Approval was also obtained from the Army Corps of Engineers, the Massachusetts Historical Commission, and Board of Underwater Archeological Resources.

Site selection. Initial site selection began simultaneously with the permitting process. To narrow planting efforts to areas most likely to support eelgrass, environmental data specific to Boston Harbor was acquired (Estrella 2005). The site selection model produced by Short et al. (2002a) was modified and adapted to a GIS analysis (Estrella 2005; Leschen et al. 2006). This analysis was based on a grid of 100 m x 100 m cells covering the Boston Harbor area. Seven parameters were estimated at each cell: depth, exposure, historical eelgrass distribution, current eelgrass distribution, water quality, bioturbation, and sediment type. Parameters were assigned scores ranging from 0-2 (2 being the most suitable for eelgrass growth) based on literature values or from conditions at existing local reference eelgrass beds based on their suitability as eelgrass habitat (Table 2). All the parameter scores were then multiplied to get a Preliminary Transplant Suitability Index (PTSI) score (Short et al. 2002a) for each cell. Since the model employed a multiplicative index, a score of zero (0 - unsuitable) in any one parameter eliminated the site from further consideration, whereas high parameter scores made it more likely to support eelgrass. Each cell was color-coded to reflect the PTSI scores, which allowed mapping of areas with the most potential for eelgrass growth. PTSI results effectively focused the search for suitable sites, thus reducing the number of areas requiring further investigation.

The parameter estimates came from a variety of sources:

Depth: Average depth at mean low water was estimated by using point depth measurements taken from NOAA Electronic Navigational Chart (ENC) data. The depth points, along with MassGIS tidal flats and shoreline data (depth = 0), were spatially interpolated using the Inverse Distance Weighted (IDW) method to obtain depth estimates at each cell.

Exposure: The fetch in the NE direction was used as a surrogate for exposure, as it is the prevailing direction of winter storm winds. NE fetch was estimated using the MassGIS 1:25,000 shoreline data.

Historical SAV Distribution: Data produced by Mass DEP Wetlands Conservancy Program surveys in 1951, 1971, and 1995 were used for historical SAV distribution.

Current SAV Distribution: The most recent survey (2001) was used for current SAV distribution.

Water Quality: Water quality scores were made using point measurements of various water quality criteria taken by the Massachusetts Water Resources Authority (MWRA) and later supplemented by *MarineFisheries* Eelgrass Project staff. First, the median April to October value for each water quality criterion was estimated. These median values were then interpolated using the IDW method to obtain estimates at each cell.

Bioturbation: Values were based on density of bioturbating organisms such as green crabs and skates which were counted along 2 to 3, 50 m transects per site (2 m swath per transect).

Sediment Type: Using a polygon layer of sediment types for Massachusetts Bay developed by the U.S. Geological Survey (USGS), the percent composition of each sediment type was determined for each grid cell. The predominant sediment type for each cell was then used to derive a score for the sediment parameter. This score was revised as described below.

The potential transplant sites originally identified by the PTSI output were surveyed in the field. Sites were surveyed for actual depth (corrected for tide), presence of marinas, mooring fields, extensive shore armoring (rip-rap), and proximity to commuter ferry routes (wakes). Underwater surveys of sediment and bioturbators such as green crabs and skates were undertaken by SCUBA divers along two to three 50 m transects at each site (2 m swath transect-1). Sediment cores were collected every 5 m along the transect with a 15 cm long, 4.9 cm diameter flow-through cylindrical core. The sediment was dried and sieved by MDMF for preliminary grain size. A second set of samples was collected and sent for processing to a laboratory at Boston University (BU) where they were analyzed for grain size using standard methods of Poppe et al. (2000).

Our initial sediment groundtruth sampling indicated that the GIS sediment data layer from USGS was inaccurate in the shallow depth zone targeted for eelgrass restoration because data were extrapolated from deeper water. We therefore decided to remove the USGS sediment layer from the model and instead, conducted extensive

Table 2. PTSI scoring criteria for parameters used in evaluation of site suitability for eelgrass (*Zostera marina*) transplanting.

Parameter	PTSI score	GIS Data Source	Groundtruthing method
Depth	0 = <0.5m or > 4m 1 = 3 - 4m 2 = 0.5 - 3m	NOAA Navigational Chart, values based on reference beds	Depth soundings adjusted to low tide
Sediment type	0 = > gravel and >70% silt/clay 1 = coarse sand to very coarse sand 2 = <70% silt/clay to medium sand	USGS Open File 99-439	Underwater camera, Ponar grab samples, analysis of sediment cores
Exposure	0 = NE fetch > 2724 (max. fetch of existing bed) 1 = 1866 to 2274 m 2 = < 1866 m (average of existing beds	*MarineFisheries* calculations from existing beds	Visual: protection from NE
Historical SAV distribution	0 = previously unvegetated 1 = previously vegetated in 1 survey 2 = previously vegetated in 2 or more surveys	Mass DEP Wetlands Conservancy Program (WCP) Historical eelgrass distribution (1951, 1971, 1995) and current eelgrass distribution (2001)	Visual inspection with SCUBA
Current SAV distribution	0 = currently vegetated 2 = unvegetated	Mass DEP Wetlands Conservancy Program (WCP) Historical eelgrass distribution (1951, 1971, 1995) and current eelgrass distribution (2001)	Visual inspection with SCUBA
Water Quality	0 = >1 WQ value does not meet eelgrass requirements* 1 = meet all but one 2 = meet all requirements	MWRA BHWQM, CSORWM projects	Light attentuation measured with LICOR 1400 data logger
Bioturbation	0 = >1 crab/m2 1 = 1 crab/m2 2 = < 1 crab/m2	none figures based on Davis et al. 1998	50m sweep with 2m swath bar, counting crabs and skates/rays in each 10m segment

sediment groundtruthing. A Ponar grab sampler and an Atlantis underwater camera allowed us to quickly assess bottom type in an area. With the camera, rocky sites and areas of high macroalgal growth could quickly be eliminated. Gravelly areas were omitted by using the grab sampler, and black, anoxic areas noted for later consideration (only if such sediment proved suitable for transplantation). If the sediment did not show obvious problems a core sample was collected via SCUBA for grain size analysis. When high PTSI scores were upheld after groundtruthing, the area was selected for test transplanting.

Sediment grain size obtained at many sites was very fine (silt and clay) with a visible redox layer below ~2 cm. These observations of possible anoxic sediments in some areas raised concerns about bottom sediment quality, e.g., organic loading and H_2S toxicity (Barko and Smart 1983; Koch 2001; Carlson et al. 1994; Goodman et al. 1995). Therefore, we contracted analyses of TOC and pore water sulfide to help refine the transplant site selection process. An additional set of sediment cores was collected for analysis, stored intact and upright in coolers on the boat, and delivered to the Department of Environmental, Coastal and Ocean Sciences Laboratory at the University of Massachusetts, Boston. There they were analyzed immediately according to the methods of Cline (1969) and Hedges and Stern (1984), respectively. Briefly, sulfide in pore water was determined by sectioning the cores, isolating pore water by centrifugation (10,000g), 0.4 µm polycarbonate syringe filtration, and preservation by the addition of 2% zinc acetate (all performed in a nitrogen or argon atmosphere). Sediment for TOC analysis was dried at 60^{o}C, acidified with HCl to remove carbonates and analyzed by a Perkin-Elmer CHN Analyzer.

In spring 2007, we contracted Boston University (BU) to re-analyze sediment grain size from 12 original test transplant sites and pre-existing remnant beds with more accurate laboratory techniques than previously used.

Harvest and test transplants. Test transplanting began after potentially suitable sites had been identified and permits obtained. An existing bed north of Boston Harbor, in Lynn Harbor, Nahant, was adopted as the primary donor site after investigation confirmed it was extensive and dense (Figure 2). (A bed across the channel in Revere was examined, but eliminated as a donor site since it was not as dense as the Nahant bed).

Figure 2. Donor beds in Revere and Nahant.

Several steps were taken to minimize impact to the donor bed. A GPS-referenced 50 m transect line was used to avoid re-harvest of the same area. Divers then placed a 1 m^2 quadrat adjacent to the transect and harvested eelgrass shoots in one of two ways: the single-shoot method, described by Davis and Short (1997), or the clump method, adapted from Save the Bay, Rhode Island (Sue Tuxbury, personal communication). In the single shoot method, trained *MarineFisheries* divers fanned away sediment from the rhizomes and snapped off one shoot at a time with approximately 3-5 cm of rhizome. Shoots were grouped into bundles of 50. Alternatively, divers dug small clumps of eelgrass using a garden trowel, leaving sediment intact around the rhizomes of harvested shoots, and placed clumps in mesh dive bags. No more than 20% of standing stock was harvested from each quadrat using either method and quadrats with sparse eelgrass coverage were skipped. The quadrat was then flipped along the transect and harvest continued in this way until enough eelgrass was obtained.

Shoot counts were taken approximately every 2 months during the first and second harvest seasons to determine if harvest was having a long-term impact on the donor beds. Ten 0.25 m^2 quadrats were sampled along transect lines re-laid in the same location as harvest and also along control transects (where no harvest occurred). Data from harvest and control sites were compared using the Wilcoxon signed-rank test.

Sites which warranted primary phase test transplanting received four TERFs™ [weighted wire mesh frames to which paired eelgrass shoots were tied - Short et al. (2002b)] with 200 eelgrass shoots (50 per TERF™). These plantings were monitored for shoot survival and general health. Sites with best results were chosen for medium-scale transplanting in 2005 at which time the effectiveness of different planting methods and configurations was tested. TERFs™ were placed singly, together in a square pattern, and offset; 1 m^2 quadrats were hand planted; and, 1 m^2 PVC frames were deployed with string grid in the middle to which shoots were tied (a prototype designed to address problems experienced with TERFs™). A total of 1000 eelgrass shoots were planted in this pattern along a 50 m transect at each site (Figure 3).

Medium-scale transplant sites were monitored over the summer for survival and overall health of eelgrass shoots. Sites that did well received larger-scale plantings in fall 2005 or spring 2006. The PVC string frame design was modified into a PVC frame/jute mesh structure with an anchoring system (Figure 4). Volunteers tied pairs of eelgrass shoots at 25 junctions of the jute and 10" spikes were driven through pre-drilled holes in 2 of the frame corners to anchor the frame in the sediment; metal landscape and bamboo staples (bamboo barbeque skewers soaked in water and bent in half) were used to tighten up the jute. At the end of the season, the jute was cut away along the inside of the frame and left to biodegrade; frames and spikes were retrieved for reuse.

<u>Large-scale transplants.</u> Sites selected for large-scale transplants received either frame or hand-planting, or both, depending on local conditions. A slightly gravelly site was hand-planted because frames could not rest flat on

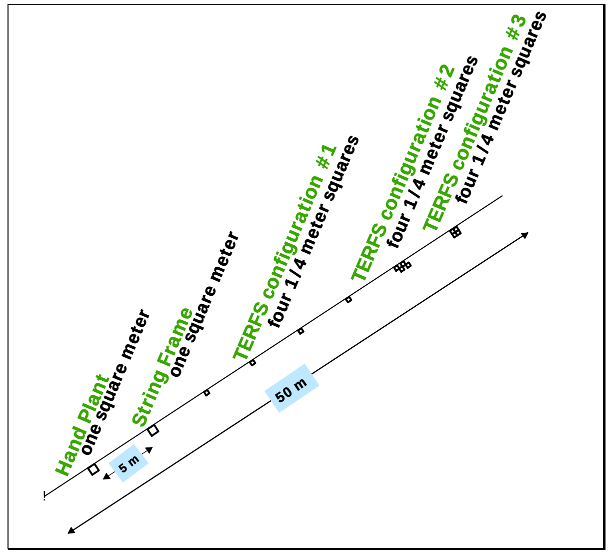


Figure 3. Secondary test transplants in Boston Harbor, 2005. Each site was planted with 1000 shoots in various patterns by different methods.

Figure 4. A PVC frame/jute mesh structure was constructed as a lighter-weight alternative to TERFs™.

the sediment. Muddier areas received frames to minimize agitation of sediment, which obscured visibility. At sandier sites we used both methods. Single shoots were planted using the horizontal rhizome method (HRM; Davis and Short 1997), where the rhizomes of 2 shoots are overlapped facing in opposite directions and held in place by a bamboo staple. Clumps were either tied into bundles of approximately 50 shoots prior to planting or planted "as is" with divers simply pulling them out of the mesh bag and estimating

50 shoots per square. Clumps were held in place using several bamboo staples. Hand-planted squares and string frames were arranged in a checkerboard pattern by alternating eighteen planted and unplanted ¼ m^2 quadrats (Figure 5). The planted squares contained approximately 50 shoots each. This pattern was adapted from a strategy used by Save the Bay in Rhode Island, the Maryland Department of Natural Resources, and others; it is designed to cover more ground than continuous planting of shoots, while providing voids for eelgrass to fill as it spreads. Initially, four to eight of these grid plots, spaced 30-50 m apart, were planted at each large-scale site. More were added later. Survival assessment of test- and medium-scale transplants was based on an assumed count of 50 shoots per square. However, at our large-scale planting sites we conducted baseline shoot counts within two weeks of planting to avoid compromising shoot survival estimates. This method accounted for: 1) bundler counting error, 2) more or less than 50 shoots actually being tied to the frames by volunteers, and 3) loss of shoots between the tying stage and transport/placement of the string frames on sediment.

Seeds. Eelgrass reproduces sexually by producing seeds and also spreads asexually by rhizome expansion. To determine if we could successfully grow eelgrass from seed, twelve fish totes of flowering shoots were harvested from Nahant in July 2005. Flowering shoots are generally longer and lighter-colored than vegetative shoots and can easily be spotted and plucked by divers. Shoots break off near the base so no digging or rhizome disturbance occurs. If left in place, these shoots would normally senesce and die after dropping their seeds (Orth et al. 1994; Granger et al. 2002).

Shoots were maintained in flow-through seawater tanks at the Marine Biological Laboratory in Woods Hole for approximately six weeks until seeds ripened and dropped from the leaves. Thereafter, vegetation was discarded and seeds were collected and sorted from detritus using a series of sieves (Granger et al. 2002; Figure 6). Sorted seeds were stored in lobster kreisels (narrow cylindrical tanks with circulating water) until late fall when they were planted. Approximately 300,000 seeds were collected and

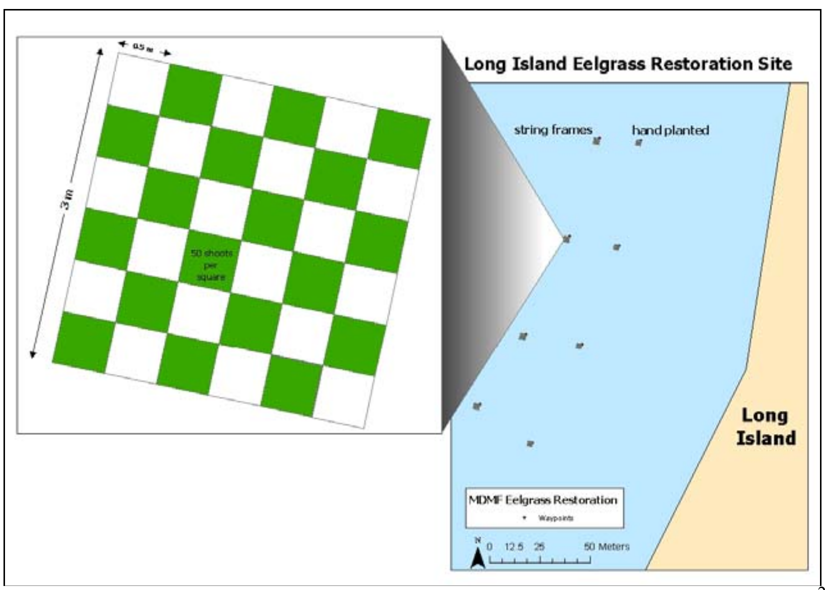


Figure 5. Planting pattern showing alternating planted and unplanted ¼ m^2 quadrats at a typical large-scale transplant site.

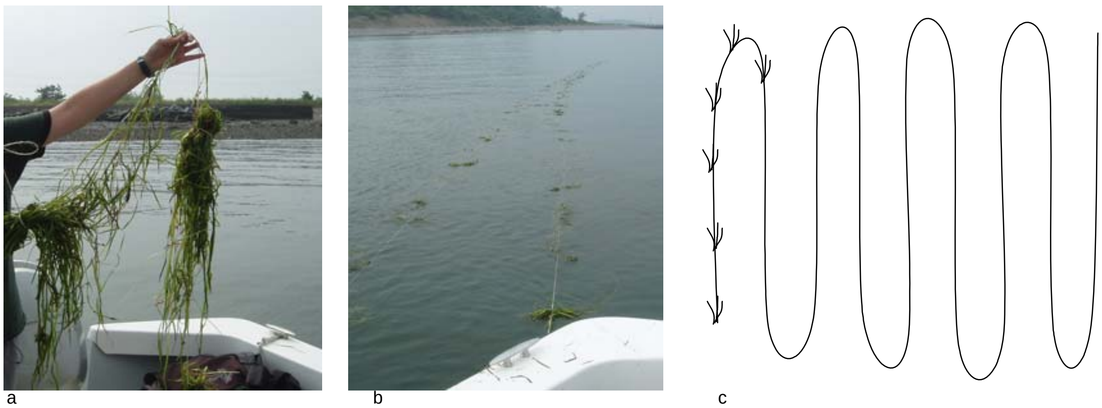
Figure 6. Clockwise from upper left: flowering shoots containing immature seeds; removing vegetation once seeds have dropped out; close-up of mature seeds; measuring seeds into bags for deployment.

distributed at three sites to complement the shoot planting. Divers scratched seeds into the sediment using a small garden claw at two of the sites and simply broadcast the seeds from the boat at the third site. We repeated these methods with approximately the same number of seeds at different sites in 2007.

In 2006 we tested an innovative seed planting technique in an attempt to reduce time and costs associated with the previous year's methods. Flowering shoots were harvested as before, but were transported directly to the planting site, where they were bundled in handfuls (average 22 shoots/bundle). Bundles were attached at intervals of 0.25-0.5 m along a continuous length of twine using a simple slip knot (Figure 7a). The lines trailed behind the boat (Figure 7b) and were staked to the seafloor in a zig-zag pattern by divers (Figure 7c).

Monitoring. Several measures of habitat structure and function were used to compare habitat function of our transplant sites to that of pre-existing natural beds and an unvegetated control site. Measures included survival and expansion, assessment of faunal communities, and habitat structure. Data were collected in July 2006 and 2007 from four locations: a pre-existing Boston Harbor natural bed, the Nahant donor site, our transplanted beds from 2005 and 2006, and an unvegetated control site near some of the planted sites. In 2007 we also conducted monitoring at one of the seeded areas.

Shoot density and size of plots were used to assess survival and expansion. Sites were evaluated for these parameters at the end of the summer for spring plantings and the following spring for fall plantings.



a b c

Figure 7. Handfuls of seed shoots tied in bundles along a length of twine (a). String of bundles trailing behind the boat (b). Planting pattern used to stake down bundles (c).

Thereafter shoot density and areal coverage monitoring continued at least once per year for the duration of the project.

To compare epibenthic/demersal and benthic faunal communities, we examined abundance, species richness, evenness, and diversity among sites and between years. Abundance was defined as the total number of organisms found at a site. Species richness (S) refers to the number of species found at each site. Evenness (relative abundance of the species present) was calculated using Shannon's Equitablility (EH) index. Species diversity indices take into account both richness and evenness. Shannon's (H) (also known as Shannon-Weaver or Shannon-Wiener) measures the chance of correctly predicting the species of the next individual collected. Simpson's Diversity (1-D) index is the probability that two individuals randomly selected from a sample will belong to different species (Krebs 1999). Though commonly used in ecology, the Shannon index assumes random sampling from an infinitely large population and that all species in the area sampled are present in the sample. These are assumptions that are rarely true in benthic monitoring efforts (Pielou 1975; Magurran 1988; Maciolek et al. 2004).

Since infaunal organisms could not be identified to species level in all cases, these analyses of diversity were performed with several caveats. Calculations of abundance were made for all taxa, including those identified only to higher taxonomic levels. Calculations based on species (i.e., species richness, evenness, diversity, and dominance) included only those taxa identified to species level or those treated as such (67 of 76). For example, Oligochaete spp.1 and Oligochaete spp. 2 were treated as species because they were known to be two different, though unidentified, species.

The top 3 contributors to the percentage of total species at each site were determined. An index of dominance (McNaughton 1967) was calculated as the sum of the percent contribution of the two most important species.

We selected several easily-measured proxies to evaluate habitat function (Evans and Short 2005). Provision of 3-dimensional structure was measured as shoot density, two-sided leaf area index (LAI), canopy height, and above-ground peak eelgrass biomass.

Data were tested using Shapiro-Wilk W test for non-normality. The non-parametric Kruskal-Wallis test was employed for each parameter, and then for all pair-wise comparisons (between sites in each year, and between years for each site) to determine significance at $P < 0.05$. StatsDirect statistical software version 2.6.5, available online, was used (http://www.statsdirect.com). Shannon and Simpson diversity indices and their SDs were also calculated using StatsDirect. Microsoft Excel Statistical Package Add-in was used for Shannon's Equitability.

Survival and expansion. Density was based on the mean shoot density count from nine 0.25 m^2 quadrats in each plot ± SD. To determine areal cover, we measured and multiplied distance between outermost shoots in perpendicular directions.

Epibenthic and demersal species monitoring. Ten 1 m^2 quadrats were distributed randomly within each site in one of two ways. At our 2005 and 2006 transplant sites, distribution of the ten quadrats among the plots was determined with a random numbers table. Quadrats were tossed from the boat into the planted areas at low tide when they were visible. At harvest, natural, control, and seeded sites, quadrats were attached at random intervals along a 50 m transect line (to facilitate finding them in poor visibility conditions) and pushed overboard. In all cases, sampling of quadrats was delayed for a minimum of ½ hour after placement to allow any disturbed fish and invertebrates to return to the area. A diver survey was chosen over seine or other net surveys for several reasons: depths at the sites made other methods extremely difficult and planted plots were too small for effective trawling which could also damage and uproot recent transplants. Since a visual SCUBA survey was the only feasible method in transplant plots, it was deployed throughout all sites for consistency. Pratt and Fox (2001) found that underwater visual transects sampled more species than gillnets in medium and heavy macrophyte cover.

Two divers slowly approached the quadrats, quickly assessed the species present in the first 30 seconds, and then more carefully counted and recorded numbers of each species. The divers observed each quadrat at the same time and the higher-recorded number of each organism was used in analyses. Species with large numbers (over 100) were estimated to the nearest 100 up to 1000, and then as 1000+. In vegetated plots, eelgrass was parted several times to gain visual access. One of each pair of divers "crawled" a gloved hand along the substrate at each quadrat to scare epibenthic fauna out of hiding.

Benthic infaunal species monitoring. We followed the University of New Hampshire, Jackson Estuarine Lab Standard Operating Procedures and the San Francisco Wetlands Regional Monitoring Plan protocols for sampling benthic infauna in eelgrass habitats, with modification of core size and number sampled. Twenty 4.9 cm diameter core samples were taken by divers from well-distributed, haphazard locations within each site. This method was chosen to minimize damage to transplanted beds that would have occurred with larger cores or grab samplers. At vegetated sites, all cores were taken where eelgrass was growing. Clear flow-through cores were inserted approximately 15 cm into the sediment and capped. Divers capped the bottom of the cores as they removed them from the sediment.

Cores were brought to the boat, where they were emptied and washed with seawater into a 0.5 mm mesh sieve (Eleftheriou and Holme 1984, Tetra Tech, Inc. 1987). Large pieces such as stones and shell debris were discarded after being rinsed and examined for organisms. Samples remaining after flushing were washed into labeled collection jars. Buffered formalin (4 oz. borax per gallon 40% formaldehyde) was added as fixative to samples in ambient seawater to equal approximately 10% of total volume, and several drops of Rose Bengal stain solution (4 g/L) were added to stain organisms and facilitate sorting (Raz-Guzman and Grizzle 2001; Holme and McIntyre 1984; Mudroch and MacKnight 1994). Samples were left in dilute formalin until they were processed in the laboratory. They were then poured through a 0.25 mm mesh sieve and rinsed several times into a waste collection container. After rinsing, samples were returned to jars with tap water in which they were stored in a refrigerator for a maximum 4 days before sorting (most samples were sorted on the same day as transfer). Samples were sorted in Petri dishes during examination with a dissecting microscope. Animals stood out with Rose Bengal stain and were removed with tweezers to small labeled collection vials containing 70% ethyl alcohol for later identification. Organisms were identified to species where possible by ENSR, Inc., Woods Hole, MA and data recorded in an Excel spreadsheet. Posterior fragments were discarded.

Habitat structure monitoring. Faunal presence and diversity in eelgrass beds have been correlated with the physical structure of the habitat (Evans and Short 2005, Fonseca et al. 1990). Trained divers estimated percent cover of eelgrass, macroalgae, and sessile invertebrates in the same 1 m^2 quadrats used to count fauna along transects. Algae and invertebrates were identified to species where possible or otherwise recorded as e.g., "un-id'd red algae, un-id'd sponge".

At each 1 m^2 quadrat, divers placed a 0.25 m^2 quadrat in a quadrant of the square, starting in the upper left corner and rotating clockwise with each successive 1 m^2 quadrat. This 0.25 m^2 quadrat was further divided with string into quarters, each of which was 0.0625 m^2 (Bosworth and Short 1993; Evans and Short 2005). We counted shoots within the 0.25 m^2 quadrat to obtain shoot density. To calculate LAI and aboveground biomass we cut and removed all above-ground vegetation from within two of the four 0.0625 m^2 subsections. In the lab, ten shoots from each sample were haphazardly chosen; length and width of these leaves were measured to the nearest mm, and leaf area calculated. LAI (m^2 leaf per m^2 area of seafloor) was calculated as 2-sided leaf area times density (Evans and Short 2005; Hauxwell et al. 2003). To determine epiphyte cover in the field, we estimated percent of the leaf area covered with epiphytes. In the lab, epiphytes were scraped from leaves using a glass slide or dull knife. All leaves and epiphytes from each site were then placed separately in a pre-weighed foil pouch and dried for 48 hr in a drying oven (60°C). Dry leaves and epiphytes were then weighed to obtain biomass in g per m^2 (Westlake 1965; Phillips 1990). Canopy height was measured in situ (80% of mean of maximum length shoots from each quadrat).

Efficiency of Harvest and Planting Methods. An efficiency analysis of hand- vs. frame-planting was conducted by recording the number of person-hours spent by divers, boat handlers, and shoreside volunteers, vs. the number of shoots planted in this effort. Results were averaged for 2 planting days.

The efficiency of the "clump harvest method" vs. the single shoot method was investigated. Number of shoots harvested and planted per dive hour (time spent in the water by divers) was calculated for each method. We used the same checkerboard pattern described above with 50 shoots planted in each square.

Modeling of seed shoot movement. We modeled the movement of seed shoots from pre-existing natural beds to areas which we found suitable for eelgrass in the Harbor in order to determine whether our restoration efforts were redundant, i.e., would eelgrass have colonized the Harbor without our efforts. Our previous field surveys had indicated that existing remnant beds, the source of reproductive shoots, may be scarce or non-existent in areas affected by water quality degradation, thereby severely limiting available seed stock. We also investigated whether seeds from our selected sites were likely to populate other suitable areas.

We used the model GNOME™ (General NOAA Operations Modeling Environment) to investigate the potential path of seed shoots that become detached from "parent" plants and float to the surface. GNOME™ is primarily used to simulate the movement of oil after a spill, but because it is tide and current driven, it was applicable in our research question. The model was first run to evaluate the distribution of seed shoots from historical (remnant) beds in Boston Harbor. The simulation was re-run using our successful transplant locations as start points to determine whether seeds from our transplants were likely to re-vegetate other suitable areas. We used Boston Harbor inputs of 1) wind typical for the time of year when seed shoots are maturing (early-mid-July) obtained from the NOAA National Data Buoy Center, 2) floating non-degradable objects (representing seed shoots), and 3) 2-week duration (the maximum time eelgrass shoots remain buoyant, and by which time they have dropped two-thirds of their seeds (Harwell and Orth 2002a)). (Massachusetts Bay current data were not included in the model inputs, which may affect distribution of shoots after they leave Boston Harbor, but they are not considered to significantly impact the results within Boston Harbor. The Nahant/Revere beds were also included in a model run, but did not affect results and were therefore left out of future model runs for simplicity).

Results and Discussion

Evaluation of Harvest Method. Shoot densities measured at harvest and control sites in the Nahant donor bed are presented in Figure 8. Differences were not significant ($p>0.05$) in all comparisons of control vs. harvest on any date, suggesting that our harvest methods had no detrimental impact on shoot density in donor beds.

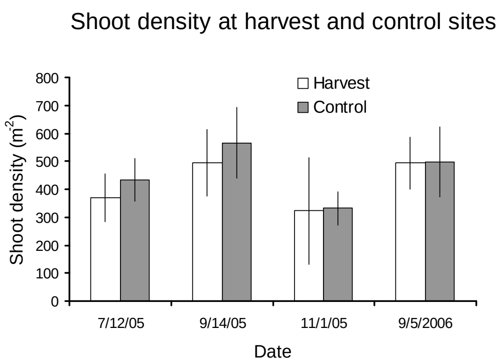
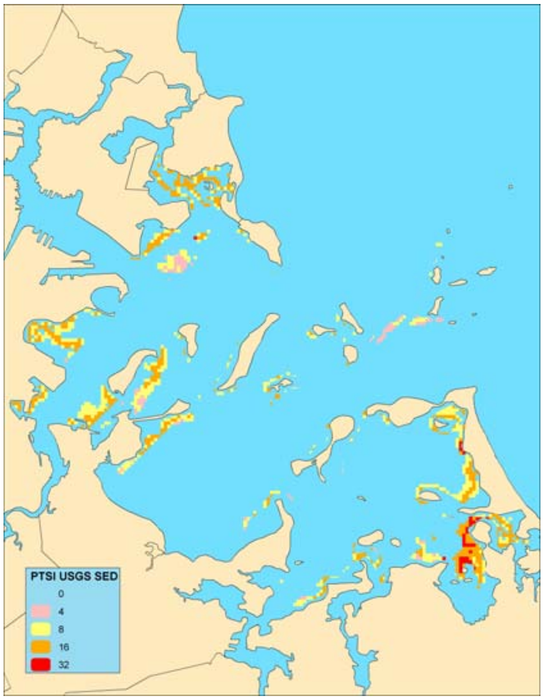


Figure 8. Eelgrass shoot densities at donor site in Nahant in 2005. Control and harvest data on each date were compared using the Wilcoxon signed-rank test. Error bars are ± SD.

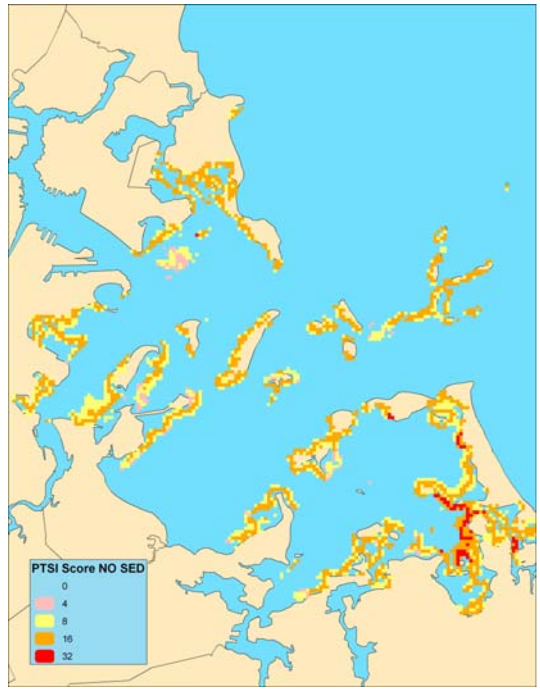
Site selection and test transplants. The GIS map generated from the original PTSI scoring (Figure 9A) was a starting point in our site selection process. The majority of the blue area (PTSI score of 0) was the result of unsuitable depth or exposure. This effectively focused our search along shallow segments of the shoreline that were protected from NE storm winds. Of the potential transplant areas originally identified with the PTSI output, six were eliminated due to presence of a marina, high energy environment, or incorrect depth, i.e., too shallow or too deep. The boat traffic associated with marinas makes transplanting impractical and potentially dangerous. Riprap reflects the wakes generated in shipping channels, creating energetic conditions unsuitable for eelgrass growth. Figure 9B shows the PTSI scores once the USGS sediment map layer was removed after groundtruthing revealed its inaccuracy in shallow water. Figure 9C depicts the scoring with the *MarineFisheries* sediment layer created from groundtruthing and the resulting limited area for restoration.

Twelve sites remained viable after sediment groundtruthing using Short model guidelines and were selected to receive test transplants (Figure 10). Shoot survival after primary test transplanting with four (4) TERFs™ ranged from 5% - 90% (Table 3).

However, several factors in addition to shoot survival influenced the decision to continue planting at a site after both primary and secondary test transplants were completed. Sediment at the Rainford E sites proved unsuitable; there were far more rocks and kelp than had been apparent on the initial visit, and despite initial high survival shoots later disappeared and the site was eliminated. Survival at the Thompson Island site was high; however, the grass looked very unhealthy, was covered with epiphytes and sediment, and up-rooted very easily when TERFs™ were removed. Because of these factors, and the prevalence of extremely soft, fine, anoxic sediment, the Thompson Island site was eliminated as was Lovell Island which was too shallow and gravelly to support eelgrass. Despite mediocre survival rates at some of the Long Island sites, remaining plants looked very healthy. The significant excavation by crabs (bioturbation) under TERFs™ at Long Island and Peddocks Island SE sites may have caused most of the eelgrass mortality, rather than poor growing conditions. Further planting by alternative methods was therefore pursued at these sites. Four sites, Long Island South (LIS), Peddocks SE (hence also referred to as “Peddocks”), and Weymouth were selected for secondary test transplants in the fall of 2005, with the intention of also planting Long Island North (LIN) in spring 2006.



A



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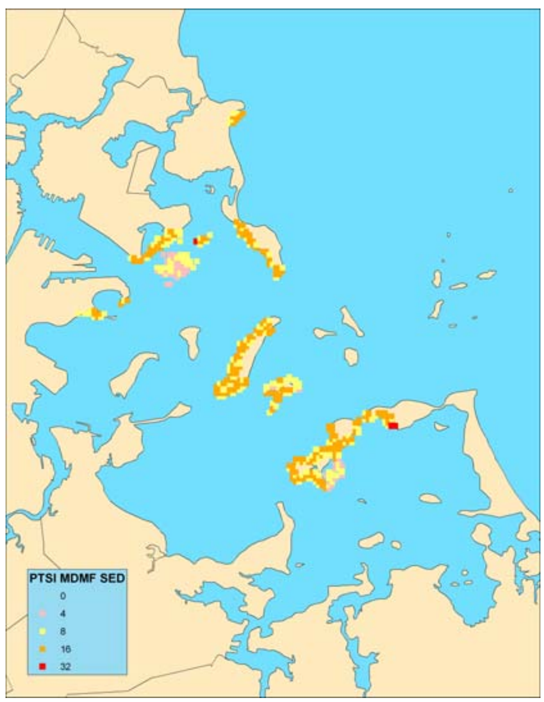


C

Figure 9. Results of PTSI scoring with USGS sediment layer (A). Higher scores indicate greater suitability for eelgrass growth based on the Short model. PTSI map with problematic USGS sediment layer removed (B) and PTSI scoring with *MarineFisheries* sediment layer (C).

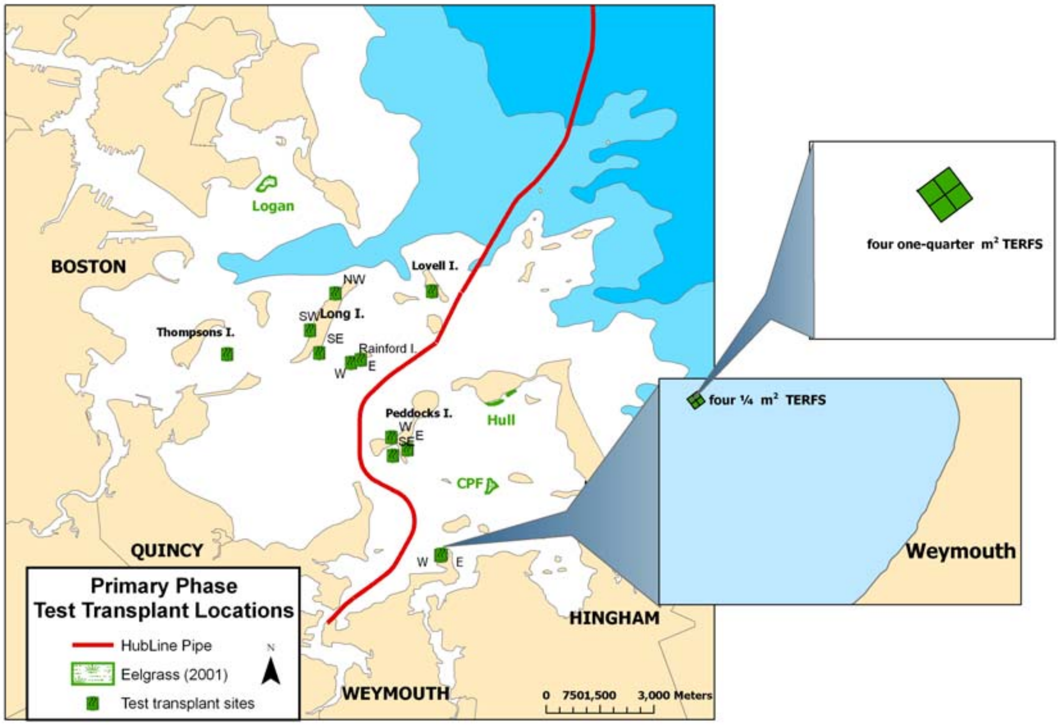


Figure 10. Primary test transplant locations in Boston Harbor, 2005. Each site was planted using four TERFs™ frames arranged in a square; each had approximately 50 eelgrass shoots attached. There are two sites at Rainford Island and Weymouth (they appear as one on the map due to their close proximity). CPF (Crow Point Flats), Hull, and Logan are pre-existing eelgrass beds).

SITE	% SHOOT SURVIVAL AFTER 6-8 WEEKS
Long I NW	50
Long I SW	45
Long I SE	75
Thompsons I	90
Rainford I	*
Rainford I E	87
Lovell I	5
Portuguese Cove (Peddocks I)	45
Peddocks I SE	85
Peddocks I E	70
Weymouth E	95
Weymouth W	82

Table 3. Percent shoot survival after 6-8 weeks at primary transplant sites. An Asterisk (*) indicates site buoy was gone and TERFs™ were not recovered, but sediment was deemed unsuitable anyway.

Combined shoot survival with TERFs™ at four secondary transplant sites ranged from 54-67%. However, these numbers may be artificially low for two reasons: 1) Percent survival was based on a planned baseline of 50 shoots per 0.25 m^2 TERF™, rather than a follow-up baseline count as we did later. The initial survival estimates from test transplants are therefore more useful in relative rather than absolute terms. Later survival estimates were more strongly correlated with follow-up baseline counts. 2) In general, we found that hand-planted shoots did much better than those in TERFs™ due to crab bioturbation under TERFs™ and uprooting of shoots during removal of the frames.

Prototype string frames showed potential; when they remained anchored, shoots did well and looked healthier than those in the TERFs™. Hand-planted quadrats remained free of excavation and did very well. Evaluation and selection of final sites was therefore subjectively based on health and vigor of remaining plants rather than strictly survival. It was felt that once equipment and techniques had been perfected, the secondary transplant locations, where remaining eelgrass was healthy, would be most conducive to eelgrass growth.

The pattern in which TERFs™ were planted (Figure 11) appeared to have less effect on survival than the planting technique (i.e., hand plant vs. TERFs™ vs. "string frames"). There was no statistical difference in survival among the single, offset, and square patterns of TERFs™ except at Peddocks (Figure 11). Here the offset arrangement did poorly, but crab excavation was again an important factor in these results. A single-factor ANOVA was used to determine whether differences in survival were evident between planting patterns at each site. Such differences were not significant ($P > 0.05$) at any site except Peddocks Island, where the offset pattern displayed significantly poorer survival than the other two patterns ($p = 0.01$). This result is also likely due more to crab excavation than TERFs™ arrangement.

Large-scale transplants. LIS, LIN, Weymouth, and Peddocks SE were selected for large-scale planting. Some sites were investigated further which led to additional plantings at Portuguese Cove (Figure 12).

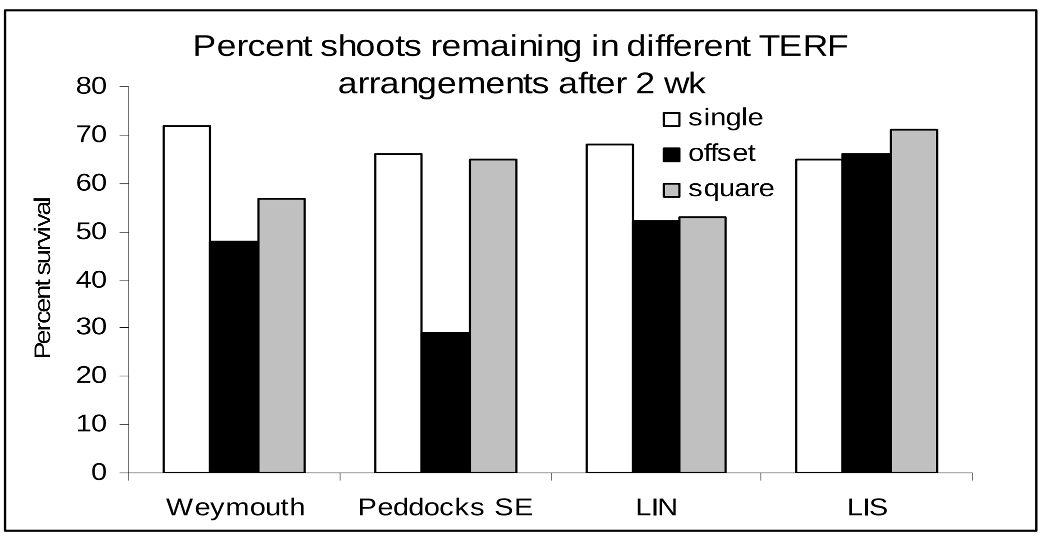


Figure 11. Percent survival of eelgrass shoots planted in various patterns at four sites in Boston Harbor, 2005. In the medium-scale test transplant, four TERFS™ were arranged in each of three patterns at four sites to assess the pattern's effect on survival. "Single" TERFs™ were placed linearly 5 m apart along a transect. "Offset" TERFS™ were laid in a checkerboard pattern. In the "square" pattern, 4 TERFS™ were laid adjacent to each other to form a square. N=12 at each site.

Eight grid plots were initially planted at LIS in fall 2005, 4 hand-planted and four with frames along two 150 m transects, respectively, bounding approximately one acre. Peddocks SE (frames) and Weymouth (hand-plant) were each planted with 4 plots in a square pattern, and encompassed ~ ½ acre per site. In spring 2006, Portuguese Cove and LIN were planted with 6 and 4 plots, respectively (plot size varied at these sites based on amount of eelgrass available from harvest), using a combination of hand- and frame-planting. More plots were added at each site through spring 2007. Figure 13 depicts plot configuration. Much of the area bounded by buoys at LIS filled in with eelgrass; this occurred to varying degrees at the other sites.

String frames. PVC string frames (Leschen et al. 2006) planted in fall 2005 were left in place during the following winter. Those planted in spring 2006 were retrieved at the end of the summer after eelgrass shoots had rooted. The string frames proved easy with which to work, deploy, and retrieve and their spiked anchoring system effectively prevented frame-shifting (Figure 14).

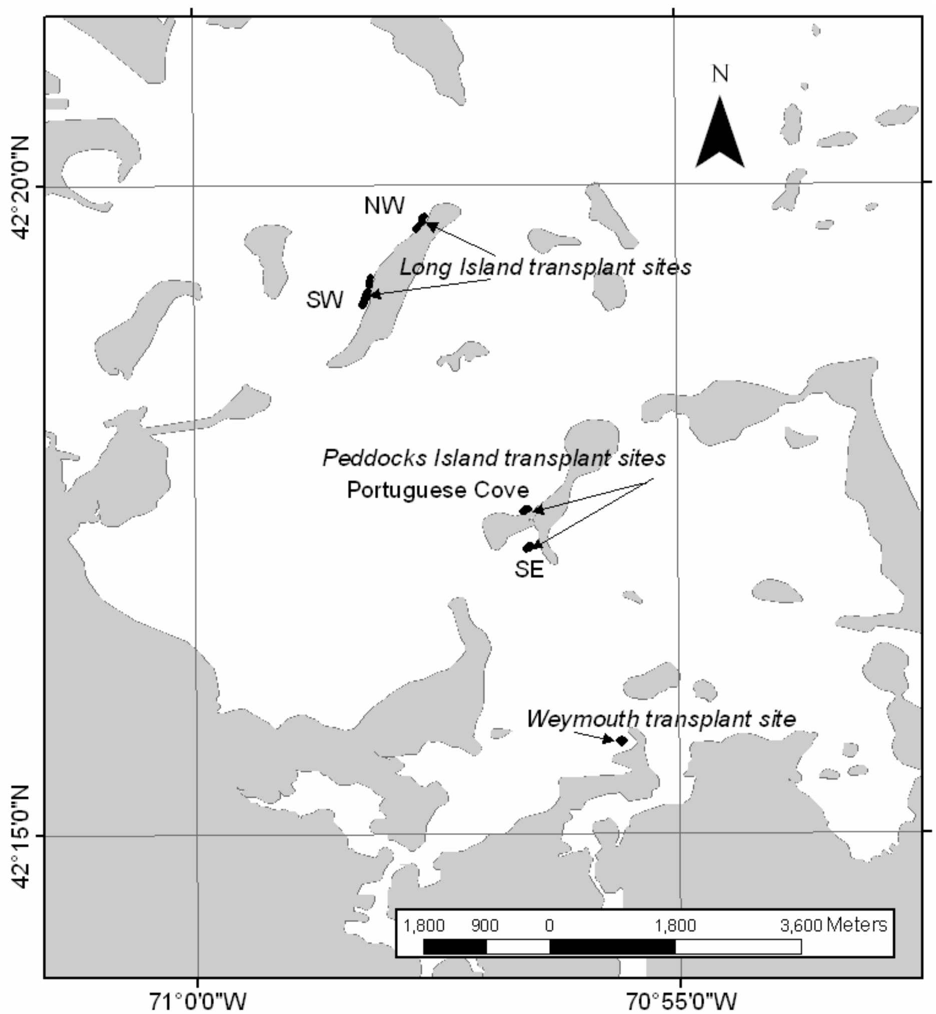


Figure 12. Large-scale transplant sites planted in 2005 and 2006.

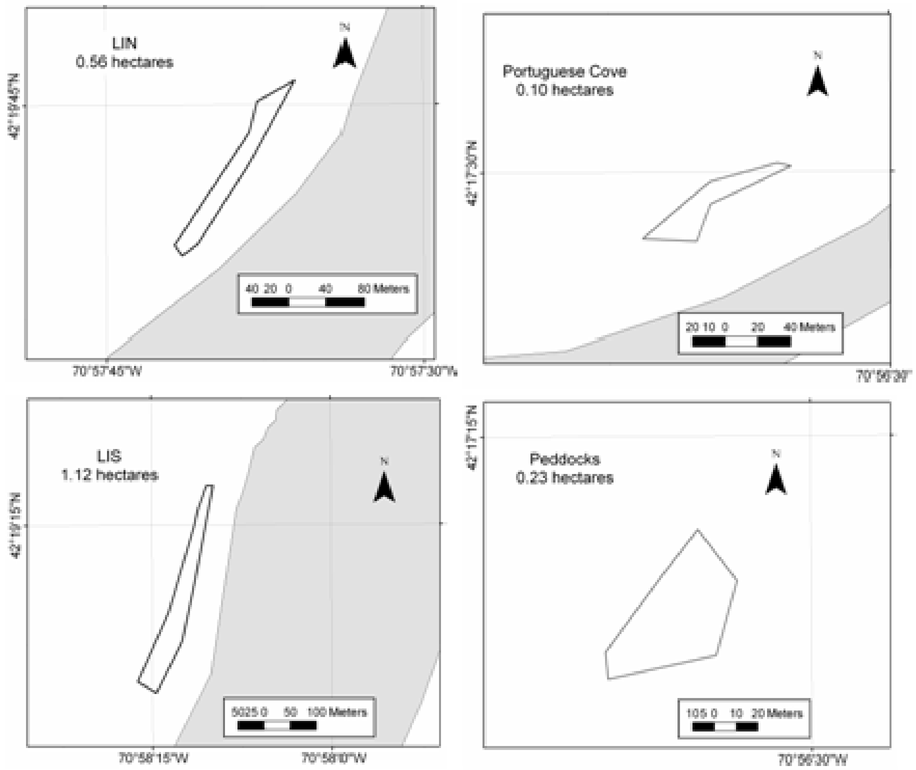


Figure 13. Enlargements of each large-scale transplant site and respective areal coverage.

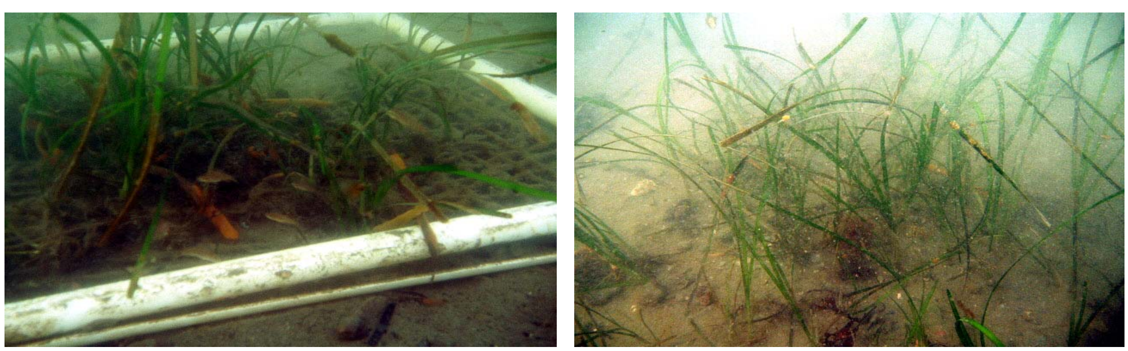


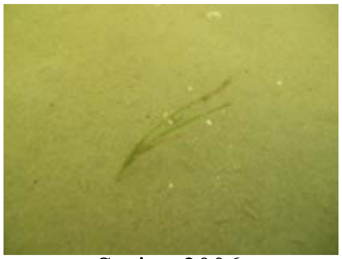
Figure 14. Photos of PVC frame in situ (left) and after frame has been removed (right). Note that jute is silted over.

Also, there were few crab excavations events with PVC frames (with the exception of Portuguese Cove), in contrast to our experience with TERFs at these same sites. While restoration efforts in other areas have experienced significant damage from green crabs, this species caused little or no destruction in our study area, despite its presence in low densities. Excavations at our sites were caused by *Cancer* spp. crabs and juvenile lobsters.

The jute mesh silted over fairly rapidly at all sites except Portuguese Cove, allowing eelgrass to root. Eelgrass within the frames generally increased greatly in density. However, expansion beyond the frames was limited since the PVC apparently provided a significant, though not insurmountable, barrier to vegetative spreading. This confinement was primarily a problem for frames planted in spring, which, in the future, could be resolved by removing the frames earlier in the summer.

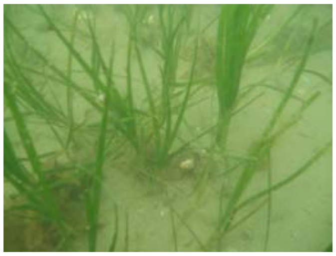
Seeds. Initial monitoring of seed germination in late April 2006 appeared to indicate a low germination rate (<1%) at both Peddocks SE and LIS from seeds planted in 2005. However, our site survey in July 2006 revealed a large, flourishing bed of eelgrass at the LIS seed-planting site. This bed continued to expand throughout the summer and by the end of August covered almost 180 m^2 (Figure 15).

Assessment in spring 2007 revealed an area of 3100 m^2 harboring at least some tufts or bunches of eelgrass which spread from the original 2005 seed planting (Figure 16); by fall 2007 most of the area exhibited fairly dense growth. Growth at Peddocks from the 2005 seed planting was less extensive and harder to measure due to poor visibility, nevertheless, this site showed promising growth and expansion. The LIN site, where seeds were simply broadcasted, covered approximately 100 m^2 by fall 2007. This cover was much less than at sites where divers scratched the seeds into the sediment. The additional, minimal effort by divers may have helped to conceal seeds from grazers and facilitated germination.



Spring 2006

Summer 2006



Fall 2006

Spring 2007

Summer 2007

Figure 15. Seed planting progression at LIS; initial sparse germination eventually spread into a large, dense bed.

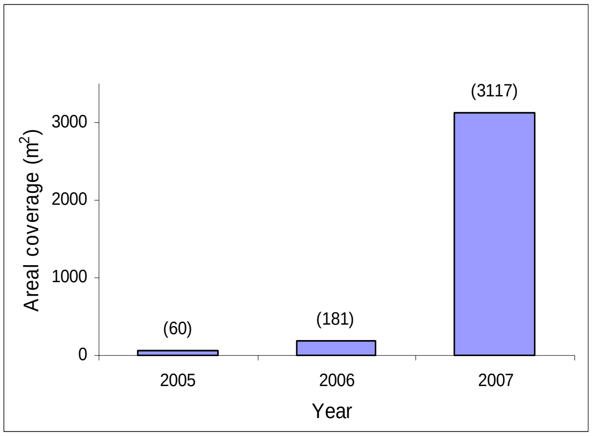


Figure 16. Areal coverage of seeded site at LIS, 2005-2007.

In May 2007, the area at LIS which was seeded in 2006 with staked reproductive shoot bundles, showed sparse shoot germination. At LIN only one or two shoots were observed. Further investigation of these sites over summer 2007 showed little if any growth. We speculate that grazers, much more active in July when these shoots were staked out, may have eaten most of the seeds; hermit crabs were observed carrying shoots away while we were staking bundles. Therefore, in 2007, we reverted to the highly successful 2005 seeding method.

The success of our seed planting efforts corroborates that of other projects (Orth et al. 2006; Pickerell et al. 2005; Maryland Department of Natural Resources). Seeding populated far more ground with eelgrass than our shoot transplant efforts, with a much smaller investment of time and resources. Large numbers of seed shoots can be harvested in 1-2 days, and seeds planted in another 1-2 days. Additional time and expense are involved in storing the seeds in a flow-through seawater tank and sieving the contents; but, overall, effort per area colonized is much less than transplanting shoots. Restoration efforts must still rely upon the site-selection process and test transplant stages to identify areas where seeds are likely to grow and spread. However, seed planting should be considered as an option to enhance the more labor- intensive shoot transplanting method.

Monitoring of Survival and expansion. Eelgrass plots planted in 2005 were evaluated in spring 2006. At Weymouth, few shoots survived the winter, and those remaining were in poor condition. We therefore decided to eliminate this site. Peddocks E and LIS were evaluated for density and expansion in spring and again in August/September, 2006. Spring 2006 plantings at LIN and Portuguese Cove were evaluated for survival/shoot density expansion in the summer/fall of that year (Table 4). More plots were added to all of the remaining sites over the 2006 field season, and into 2007. Hereafter, 2005 and 2006 plantings are distinguished, e.g. LIS 05 represents plots planted at Long Island South in 2005 and LIS 06 plots were planted in 2006.

Initial survival of 2005 plantings ranged from 41% (Weymouth) to 89% (Peddocks SE). Planting method did not appear to make a difference in survival; PVC string frame (hereinafter "frame") and hand planting at LIS 05 yielded similar survival rates. Expansion by the following spring (2006) also appeared to have little to do with planting method. For example, Weymouth (frames) declined continuously throughout the monitoring period until only a few shoots remained. At LIS 05, densities in hand-planted plots expanded only 71% vs. 127% for frame plantings, but at Peddocks SE, density in hand-planted quadrats expanded 116%. Initially high density increases in frames at LIS 05 and hand plantings at Peddocks SE slowed by the summer. In contrast, the initial slow density increase at LIS 05 hand-planted sites accelerated during this period. As a result,by September 2006, shoot density was fairly even across all 2005 sites (except Weymouth).

Sites planted in spring 2006 showed expansion by the following fall which ranged from 20% to 193% (Table 4). Planting method did not appear to be a consistent determinant of expansion; its effect varied by site. LIN frame plantings did extremely well (193% expansion) while hand plantings approximately doubled (93%). At Portuguese Cove, excavation by crabs and lobsters resulted in hand-planted areas faring considerably better (78%expansion) than frames (20%).

Table 4. Survival and shoot density expansion for 2005 and 2006 plantings. "Hand" and "frames" refers to planting method. "Overall" is the average of all plantings, regardless of method. "Survival" is the percent of originally planted shoots remaining alive. "Expansion" is the percent increase above the original planting density. Eelgrass planted in the spring was monitored for expansion in the fall. Eelgrass planted in late summer was monitored a month later for survival. NA = planting did not occur at that site and time; LIS=Long Island South; LIN=Long Island North.

2005 Plantings

Site	Plant method	Survival from Sept 05 planting to Oct 05	Shoot density expansion from fall 05 to spring 06	Shoot density expansion from spring 06 to summer 06	Total shoot density expansion since planting
Weymouth	frames	40.6%	-35.0%	-65.7%	-86.0%
LIS	overall	68.2%	95.3%	137.5%	210.4%
	frames	66.6%	126.7%	25.5%	205.2%
	hand	69.6%	71.1%	186.4%	241.3%
Peddocks E	hand	88.6%	116.2%	93.1%	283.0%

2006 Plantings

Site		Plant method	Shoot density expansion from spring 06 to fall 06	Survival from Aug 06 planting to Sept 06	
LIN	Area A	overall	144.1%	NA	
		frames	192.9%	NA	
		hand	93.2%	NA	
	Area B	hand	NA	78.9%	
LIS	Area A	hand	61.5%	NA	
	Area B	hand	NA	81.4%	
Portuguese Cove		overall	48.8%	NA	
		frames	19.8%	NA	
		hand	77.8%	NA	

Both LIN and LIS sites planted in August 2006, showed high survival one month later (~80%; Table 4). In summer 2007, all sites (LIS, Peddocks SE, LIN, and Portuguese Cove) looked healthy and most plots showed substantial shoot density increases and areal expansion (Figure 17), with two exceptions.

In one case, there were 2 plots at the southernmost end of the LIS site which had virtually disappeared. Prevailing currents run north along that area of shoreline, and we speculate that seeds produced in the plots were carried northward, filling in the northern segment of the LIS site, but leaving few seeds to re-populate the southernmost beds. It is also possible these plots had localized crab damage, because areas very close by were doing exceptionally well. The plot nearest the seeded area at LIS had merged with the seed bed by the time plots were measured in September 2007; it was no longer possible to distinguish the two from one another. The 2007 mean areal cover at LIS 05 excluded that plot and also the plots that had virtually disappeared at the southernmost end. The second exception was in 4 plots at LIN, where 2 plots had decreased in size, one had expanded slightly, but one had expanded significantly, accounting for the large SD seen in Figure 17 for that site.

Both 2005 sites increased significantly in areal covererage each year after planting ($P<0.05$ in all cases; Figure 17). The difference between Peddocks and LIS 05 sites did not differ significantly in any year. Density at Peddocks increased significantly between 2005 and 2006, and, although it continued to trend upward, 2007 data were not significantly different from 2006. LIS 05 showed the same pattern, except that 2006 and 2007 densities did not trend upward. There was no difference between the two sites in 2005 or 2006, but in 2007 density was significantly higher at Peddocks than LIS 05.

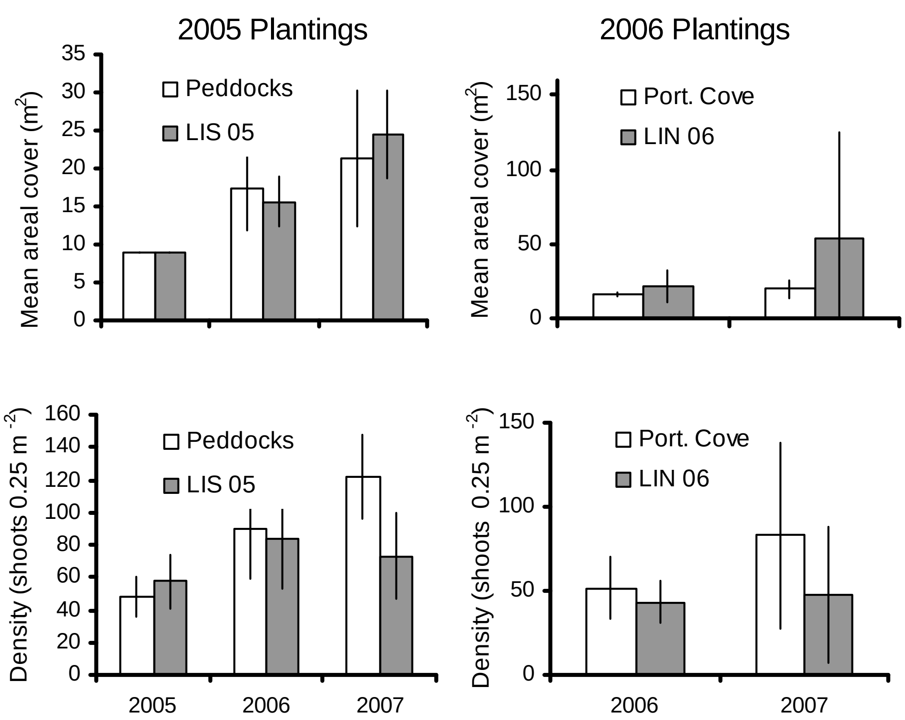


Figure 17. Mean density and areal cover (± SD) over the duration of the project (2005-2007) of plots planted in 2005 (LIS and Peddocks SE) and 2006 (Portuguese Cove and LIN).

At 2006 sites, there were no significant differences in areal cover between sites or years. There was more crab damage at Portuguese Cove than anywhere else, yet density increased significantly there between years, and was significantly higher than at LIN 06 in 2007.

All four large-scale planting sites exhibited healthy eelgrass, growth, and expansion, however, the patterns of growth differed among sites. LIS showed the most between-plot spreading, with all voids within the periphery of planted plots and seeded area filled or filling in (likely via seeds originating from planted plots and the seeded bed).

Individual planted plots were also spreading considerably, but with modest density increases. Peddocks SE plots expanded, but there was little between-plot spreading; the length and density of the eelgrass at this site exceeded all other transplant sites including the healthy donor bed at Nahant.

The sites planted in 2006, LIN and Portuguese Cove, showed evidence of between-plot spreading. Again, spreading seemed to be in either density increase (Portuguese Cove) or areal expansion (LIN 06), but not both.

Sediment Monitoring. Using results from BU's sediment analysis, we compared grain size composition among existing beds, successful transplant sites and 4 that failed: Thompson Island, two at Rainford (preliminary test transplants), and Weymouth (other sites that failed after preliminary transplants for reasons such as gravel, kelp, and boat traffic were excluded from the analysis). Sites with 35% or less silt/clay were successful. Those with >57% silt/clay failed (Figure 18).

All but one of the failed sites had less than Short et al.'s (2002a) recommended <70% silt/clay (Rainsford, 75%), and would not have been eliminated under that model. Though we had no data points between 35 and 57%, all of our successful sites, and all of the existing beds, had <35% silt/clay.

Surprisingly, sulfide and TOC levels did not exceed Koch's recommended levels at any sites except slightly at Thompson Island and Logan (pre-existing bed; Figures 19 and 20). However, levels at our Peddocks sites were higher than the LI sites. TOC levels there were close to those at Weymouth and Thompson, and sulfide levels at the Peddocks sites exceeded those at Weymouth and Thompson.

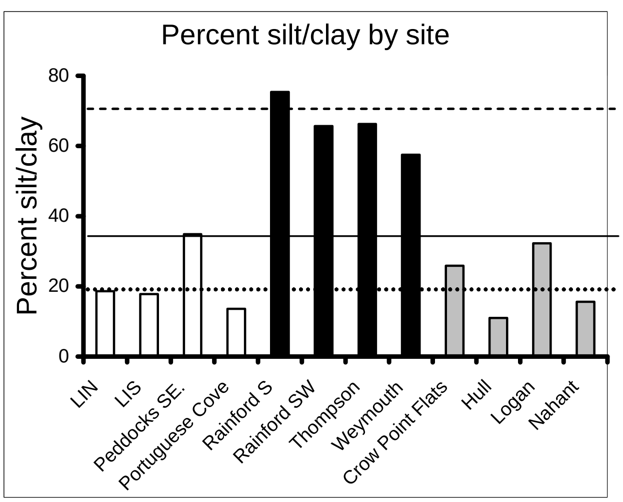
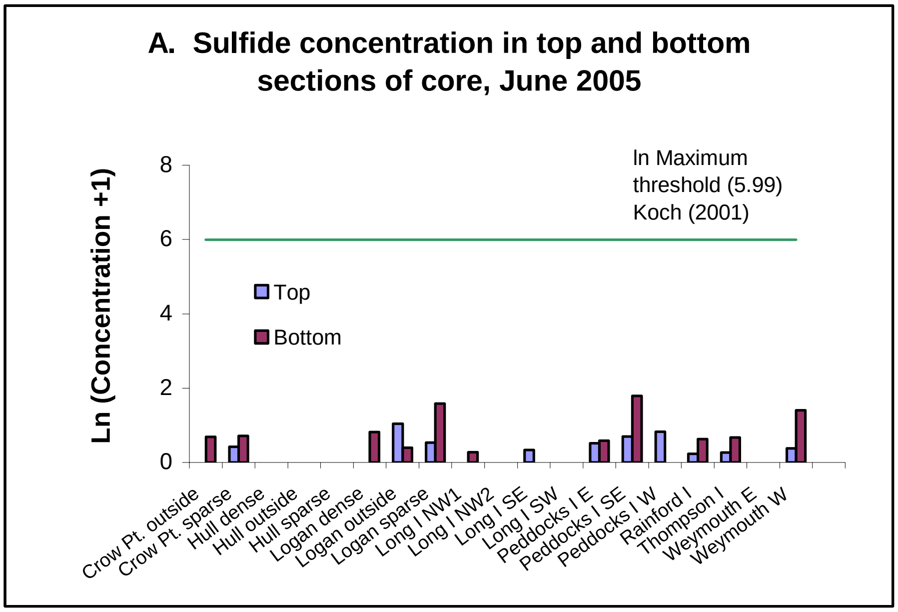


Figure 18. Percent silt/clay at successful (white bars) and failed (black bars) transplant sites, and existing beds (gray bars). Top (dashed) line is recommended maximum per Short model. Middle (solid) line is maximum found at our successful sites. Bottom (dotted) line is maximum recommended by Koch (2001).



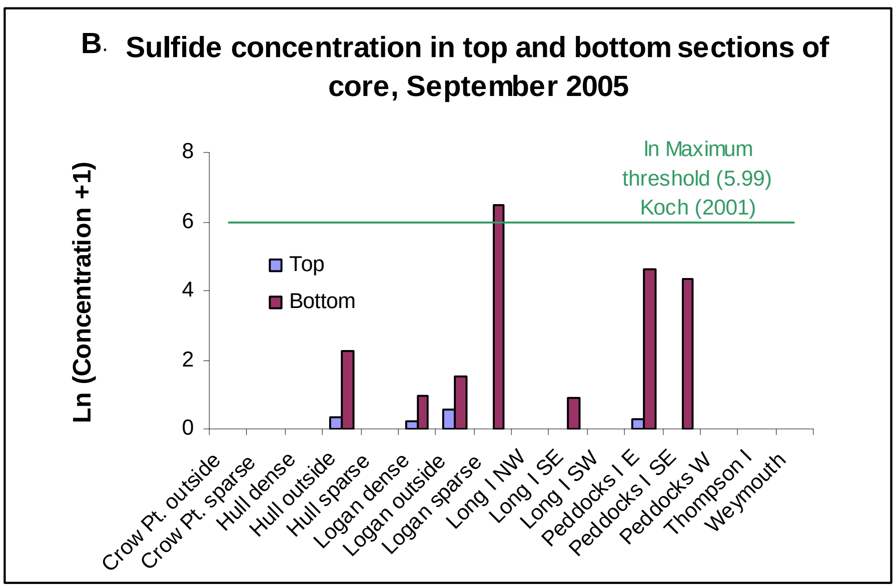
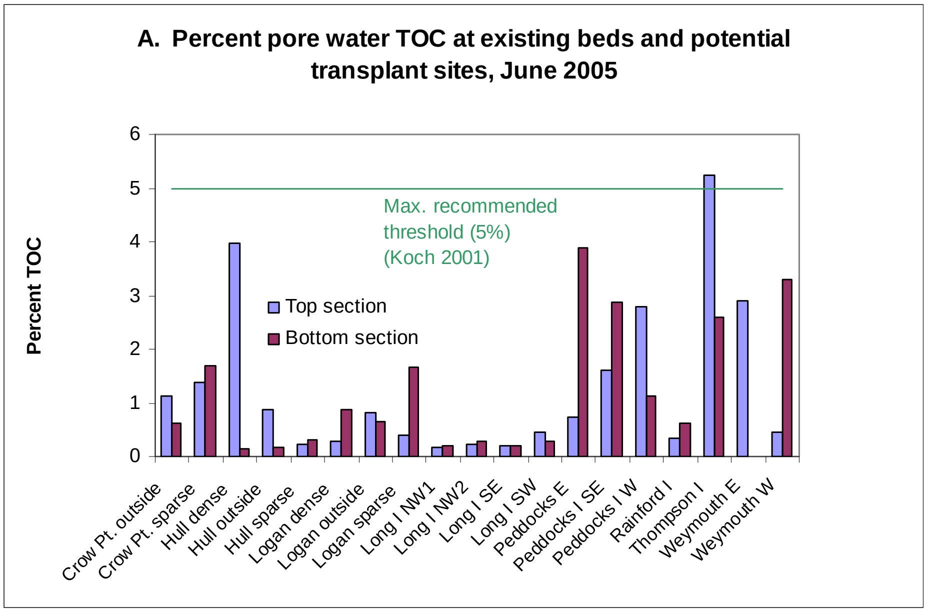


Figure 19. Porewater sulfide concentrations (converted to ln) at existing eelgrass beds and potential transplant sites in Boston Harbor in June (A) and September (B) 2005. In existing beds (Hull, Logan, Crow Pt.—see Figure 10), "dense" and "sparse" refer to a dense, central part of the bed and the sparse edges, respectively. "Outside" refers to just beyond the boundary of the bed where there is no eelgrass. "Top section" = upper 5 cm of the core; "Bottom section" = remainder of core (core length ranged from 9.4 - 17.5 cm due to collection techniques and sediment composition). Sites where concentration is zero either had too little porewater to test (typical of sandy/silty sediment) or tested below the detectable limit of sulfide, 0.21 µM. (The mean of replicate sample values was graphed when the data exhibited anomalously large differences.)



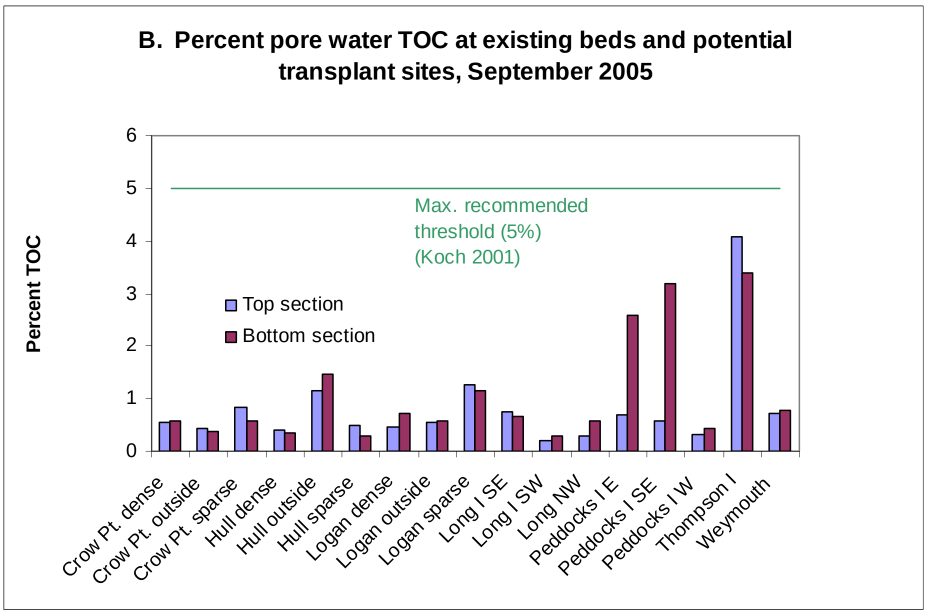


Figure 20. Percent Total Organic Carbon (TOC) at existing beds and potential transplant sites in Boston Harbor in June (A) and September (B) 2005. The length of the core from the Weymouth E site in June did not permit a bottom section analysis.

Water quality parameters were acceptable at all attempted transplant sites, minimizing macroalgal and epiphytic effects, and grain size composition was the only potential detrimental factor we found in common among failed sites. There were no other obvious similarities between Thompson, Rainford, and Weymouth sites that would account for the transplant failures there. For example, the Weymouth site is in a protected cove exposed to NW winds, whereas Thompson is more exposed, but to E winds; the Rainford coves have SW and SE exposure. Weymouth receives more ferry and other boat wakes, and although Peddocks SE also receives ferry wakes, its plantings have done very well. Rainford receives little in the way of ferry wakes, but experiences heavy weekend recreational boat traffic. The sediment at Thompson Island was more flocculent than at Weymouth. None of the sites had large numbers of epibenthic bioturbators when surveyed in 2004, nor did excavation appear to be a problem for transplants at Weymouth or Thompson. Rainford shoots disappeared over the winter, during the absence of monitoring, so we could not determine whether excavation played a role. While it is possible that other unknown factors contributed to eelgrass failure at all three sites, it is more likely that sediment quality may be responsible.

The exact mechanism by which high silt/clay content renders an area unsuitable for eelgrass transplant is unclear. Our sulfide analytical results (Figure 19) do not implicate sulfide toxicity per se as the cause for eelgrass decline and death, unless thresholds are less than Koch (2001) recommends (although in that case we might have expected Peddocks SE to do poorly). TOC levels were also acceptable at most sites. Sediment at established eelgrass beds can be rich in organics and have low redox potential without adversely affecting the plants (Smith et al. 1988; Klug 1980; Thayer et al. 1984). It is possible, however, that eelgrass transplants become stressed in reducing environments often found in very fine-grained sediment. Much of the sediment at our failed sites was black-colored with a shallow redox layer indicating anoxic conditions.

While eelgrass restoration programs have often used existing beds to determine baseline conditions for site selection, it is possible that transplants have different requirements than established beds. Seagrasses can ameliorate reducing conditions and resultant sulfide toxicity by releasing oxygen from their rhizome and root systems into the sediment (Terrados et al. 1999; Pedersen et al. 1998; Sand-Jensen et al. 1982; Smith et al. 1984; Lee and Dunton 2000). Oxygen is produced in the leaves through photosynthesis and delivered through the plant's lacunar system (Larkum et al 1989; Pedersen et al. 1998; Smith et al. 1984) to the roots to support respiration in these non-photosynthesizing structures (Goodman 1995; Zimmerman et al. 1989). When light and photosynthetic biomass are plentiful, the oxygen released by the roots is able to keep reducing conditions at a minimum, thus neutralizing the effects of high organic content (Koch et al. 2001; Lee and Dunton 2000; Brüchert and Platt 1996; Blackburn et al. 1994; Schlesinger 1991). In addition, if the sediment around the root zone is oxygenated, the plant does not have to continually send oxygen to the roots to maintain respiration in these structures. The supply of oxygen to the roots and surrounding sediment, where some diffuses, is therefore dependent on both the level of photosynthesis occurring in the leaves (Terrados et al. 1999; Smith et al. 1988; Nienhus 1983) and the demand of the roots for oxygen. If individual shoots, or even small clumps of eelgrass are transplanted into anoxic sediment, the net photosynthesizing biomass at the new site would be a fraction of that in the donor bed, thus making it more difficult for transplants to overcome an anoxic environment in very fine grained sediments. A study of *Phragmites australis*, an invasive salt marsh plant, found that severing rhizomes significantly lowered the photosynthetic rate of the plants, and that this effect was nearly double in anoxic vs. oxygenated sediment (Amsberry et al. 2000). If this effect is also true for eelgrass, severing the rhizomes during harvest would compound the already-diminished level of photosynthesis that occurs at a transplant site. The effort involved in attempting to keep roots oxygenated under these circumstances may stress the transplants to the point of death. Transplants, then, may need more oxygenated sediment than established beds until enough biomass is established to compensate for lower porewater oxygen in finer-grained sediments.

The prevalence of unsuitable sediment throughout much of Boston Harbor five years after the offshore outfall became operational (Figure 21) raises concerns about the future possibilities for eelgrass restoration in estuaries degraded by eutrophication. In areas where low flushing rates result in long-term deposition of organic matter, it may take years for sediment to recover enough to support eelgrass, even when water quality has improved. This issue will require further study as improvements are made to coastal water quality in other locations.

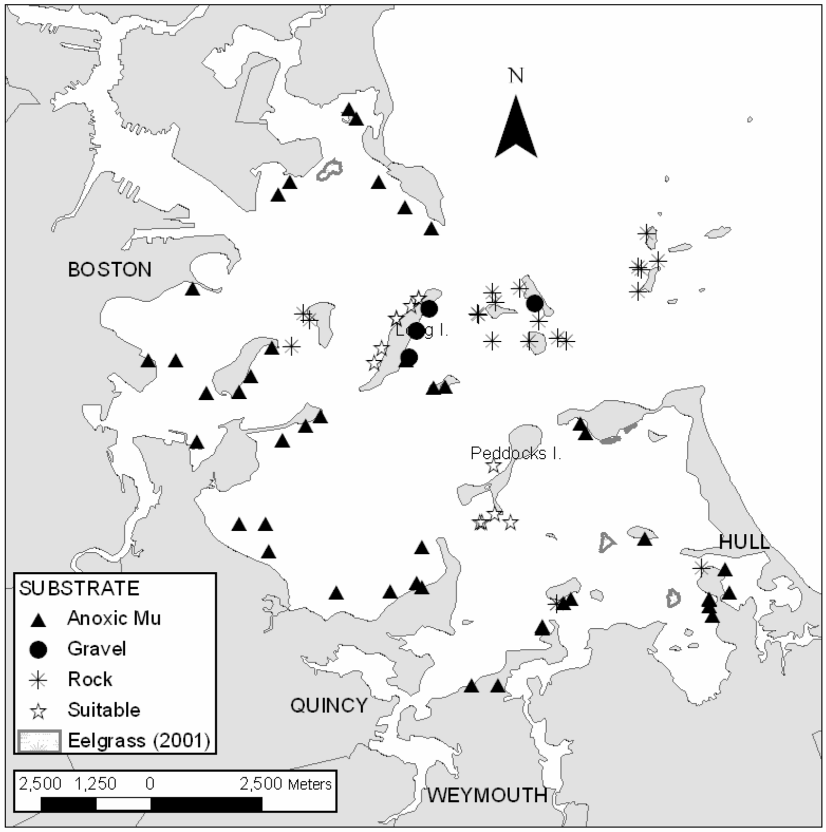


Figure 21. Sediment type observed in Boston Harbor. Data were gathered using an underwater camera, Ponar grab, or by divers taking cores. Note the prevalence of anoxic mud around the shoreline and the limited areas of suitable sediment.

Biological Monitoring

Epibenthic/demersal species abundance and diversity. From 2006 to 2007 Shannon diversity indices (H') for benthic and demersal fish and invertebrates increased at all Boston Harbor sites (Table 5 and Figure 22); by 2007 our 2-year old beds generated indices which exceeded those at Nahant and Hull. (There is no comparative data for the seed bed because it was first assessed in 2007.)

The Simpson diversity index (1-D) increased markedly at our planted sites; there was little change in reference and control sites. By 2007, indices at our planted sites exceeded those at reference beds.

Evenness, measured by Shannon's equitability (E_H) index, exhibited a similar pattern.

Overall, diversity indices for our planted sites were comparable to or exceeded those of natural beds and the Control site.

Total number of species (S) showed less variation than diversity between years at our planted sites. It did not change at Peddocks, but increased slightly at all other sites. Total number of species at planted sites approached or exceeded the healthy natural donor bed at Nahant and exceeded Hull and Control sites by 2007. Nahant, Hull, and Control site data also exhibited slight increases in species number across years. Mean number of individuals per m^2 (N) declined markedly at Peddocks and LIS 05 which was primarily due to greatly reduced numbers of *Mysis* spp. in 2007. Since *Mysis* spp. can number in the hundreds or thousands and greatly influence all indices, the data are reported in two ways in Table 5: with and without *Mysis* spp. A total list of epibenthic/demersal species is presented in Appendix A.

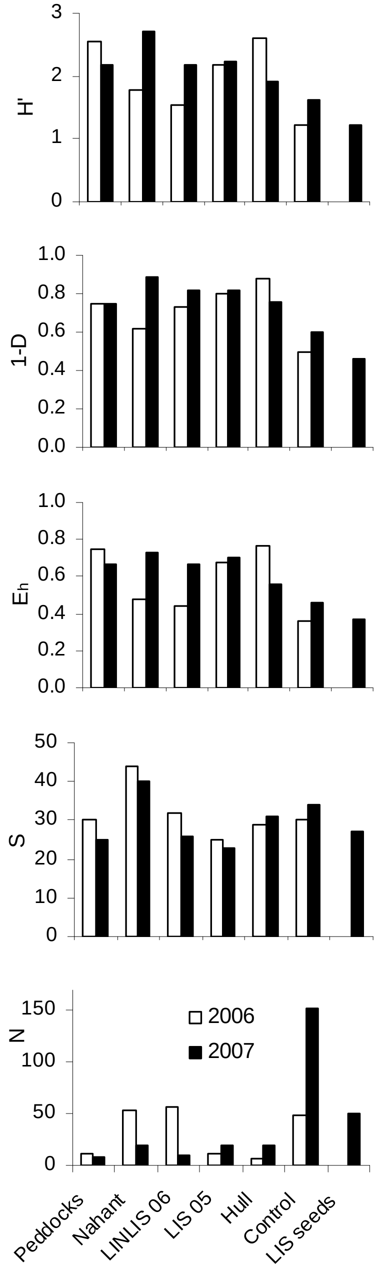
Benthic infaunal species abundance/diversity. There was a total of 71 species of infaunal invertebrates found at all the sites in 2006, and 69 in 2007 (Appendix B).

In 2006 *Pygosio elegans*, a spionid polychaete, was among the top 3 dominant species at all sites except Nahant, and at all sites in 2007 (Table 6 displays 2007 data). It comprised 33.7% and 55.3% of the total infauna in 2006 and 2007, respectively. All spionid polychaetes, combined, comprised 63% of the top 3 dominants. Control and LIS seed sites exhibited the largest number of individuals (N) in 2007 (Figure 23); they also had the highest dominance index, indicating that these N's were in large part due to the presence of just 2 species. This dominance is reflected in the lower evenness and diversity indices for these 2 sites. Highest number of species, evenness, and diversity were found at Nahant. These indices were slightly lower at our planted sites.

Measures of 3-D habitat function. Percent cover of algae and sessile invertebrates was negligible in almost all quadrats, and never exceeded 5%. Epiphytes also comprised immeasurable or minute weight fractions of above-ground biomass at all sites.

Table 5. Diversity indices of benthic and demersal samples at planted, reference and control sites. The LIS seeded site was not sampled until 2007. LIS 05 = Long Island South beds planted in 2005. LIN+LIS 06 represents combined data from Long Island South and North planted in 2006. Numbers in parentheses indicate the index with *Mysis* spp. excluded from analyses. There were no *Mysis* spp. seen at Nahant in 2006.

2006	Index			
Site	Shannon (H')	Pielou's evenness value J'	Total no. spp. (S)	Number individuals m^{-2} (N)
Peddocks	0.44 (1.51)	0.17 (.61)	13 (12)	366 (33)
LIS 05	1.09 (1.68)	0.40 (.64)	15 (14)	188 (55)
LIN06+LIS 06	.92 (1.41)	0.36 (.49)	12 (11)	174 (51)
Nahant	1.54	0.64	11	23
Hull	1.16 (1.02)	0.56 (.53)	8 (7)	59 (27)
Control	1.18 (.82)	0.51 (.37)	10 (9)	130 (89)
2007				
Peddocks	1.88 (1.72)	0.73 (.69)	13 (12)	29 (26)
LIS 05	1.87 (1.97)	0.68 (.73)	16 (15)	82 (49)
LIS06+LIN 06	1.67 (1.55)	0.60 (.57)	16 (15)	136 (89)
Nahant	1.55 (1.61)	0.59 (.63)	14 (13)	42 (23)
Hull	1.36 (1.31)	0.55 (.55)	12 (11)	131 (67)
Control	1.42 (1.39)	0.57 (.58)	12 (11)	122 (121)
LIS seeds	1.33 (1.13)	0.61 (.54)	9 (8)	117 (68)



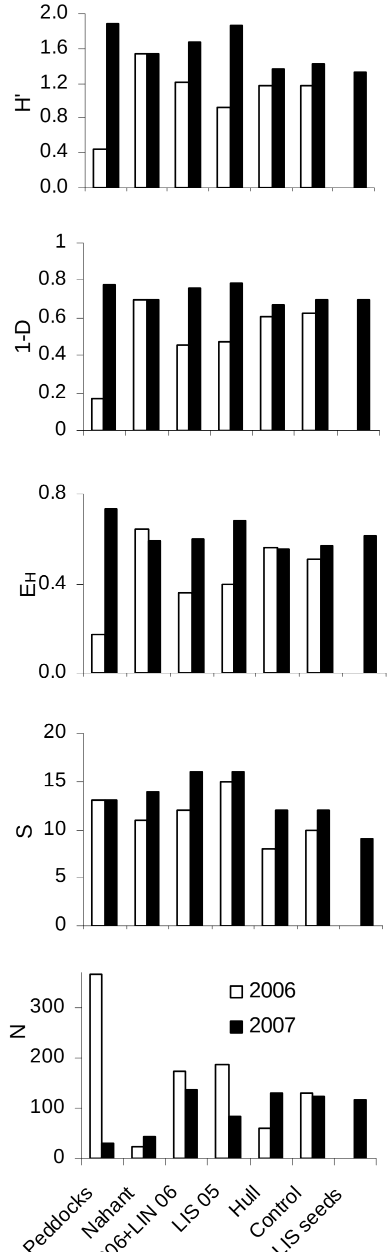


Figure 22. Indices of species diversity, abundance, evenness, and richness for epibenthic and demersal fish and invertebrates, 2006 and 2007. Sites on x-axis are: Peddocks, Nahant, LIS & LIN 06, LIS 05, Hull, Control, and LIS seeds. LIS seeds was not monitored until 2007. H' is Shannon diversity index; 1-D is Simpson diversity index; E_H is Shannon Equitablility; S is number of species found at site; N is mean number of individuals per m^2.

Figure 23. Indices of species diversity, abundance, evenness, and richness for benthic infauna, 2006 and 2007. Sites on x-axis are: Peddocks, Nahant, LIS & LIN 06, LIS 05, Hull, Control, and LIS seeds. LIS seeds was not monitored until 2007. H' is Shannon diversity index; 1-D is Simpson diversity index; E_H is Shannon Equitablility; S is number of species found at site; N is mean number of individuals per core.

Table 6. Top three numerical dominants with percent of total organisms of each species at each site in 2007. All are Annelids (polychaetes) except for *Leptocheirus pinguis (*Arthropoda/amphipoda). Index of dominance represents the percentage of total standing crops (N) contributed by the top 2 most numerous species (McNaughton 1967).

Site	Top 3 dominants (% of total (N) at site)	Family	Index of dominance
Peddocks	*Polydora cornuta (44.1%)*	*Spionidae*	
	Maldanidae spp (22.4%)	*Maldanidae*	66.5
	Pygospio elegans (6.6%)	*Spionidae*	
	Clymenella torquata (6.6%)	*Maldanidae*	
Nahant	*Exogone hebes (23.5%)*	*Syllidae*	
	Pygospio elegans (18.6%)	*Spionidae*	42.1
	Aricidea catherinae (7.4%)	*Paraonidae*	
	Maldanidae spp (7.4%)	*Maldanidae*	
LIN/S 06	*Pygospio elegans (33.9%)*	*Spionidae*	
	Polydora cornuta (20.1%)	*Spionidae*	54.0
	Spiophanes bombyx (16.9%)	*Spionidae*	
LIS 05	*Pygospio elegans (36.2%)*	*Spionidae*	
	Polydora cornuta (19.3%)	*Spionidae*	55.5
	Spiophanes bombyx (7.3%)	*Spionidae*	
	Tharyx acutus (7.3%)	*Cirratulidae*	
Hull	*Pygospio elegans(34.7%)*	*Spionidae*	
	Exogone hebes (32.7%)	*Syllidae*	67.4
	Polydora cornuta (12.1%)	*Spionidae*	
Control	*Pygospio elegans (62.2%)*	*Spionidae*	
	Leptocheirus pinguis (6.83%)	*Amphipoda*	69.0
	Polydora cornuta (6.7%)	*Spionidae*	
LIS seeds	*Pygospio elegans (72.7%)*	*Spionidae*	
	Spiophanes bombyx (6.7%)	*Spionidae*	79.4
	Exogone hebes (6.3%)	*Syllidae*	

In 2006, the healthy reference donor bed in Nahant exceeded our then-1 yr old beds in density, LAI, and biomass. In 2007, however, our two yr old beds were comparable to or exceeded Nahant, and surpassed Hull with respect to measures of 3-D habitat function. Peddocks equaled or exceeded Nahant with regard to all four measures: density, aboveground biomass, canopy height, and LAI (Figure 24). LIS 05 equaled Nahant in biomass, canopy height, and density. In 2007, our 1 yr old beds (LINLIS 06) exhibited comparable measures of structure to those of the 2005 plantings, when they were 1 yr old (2006 results). Significance of interactions is given in Table 7.

The one year old beds (LIN/LIS 06) did not yet exhibit the structure of the two year old beds, however, they are comparable to 2006 measurements at sites planted in 2005; if they continue to follow the growth pattern of older transplants, we can expect that, by their second year, they, too, will reach parity with the reference donor bed. Benthic infaunal composition was typical of healthy sand-mud sediment. Spionids, typically found more in non-complex habitats where a few opportunistic species do very well, were in fact more dominant at the Control and LIS seed sites (which were still patchy), and were least dominant at Nahant.

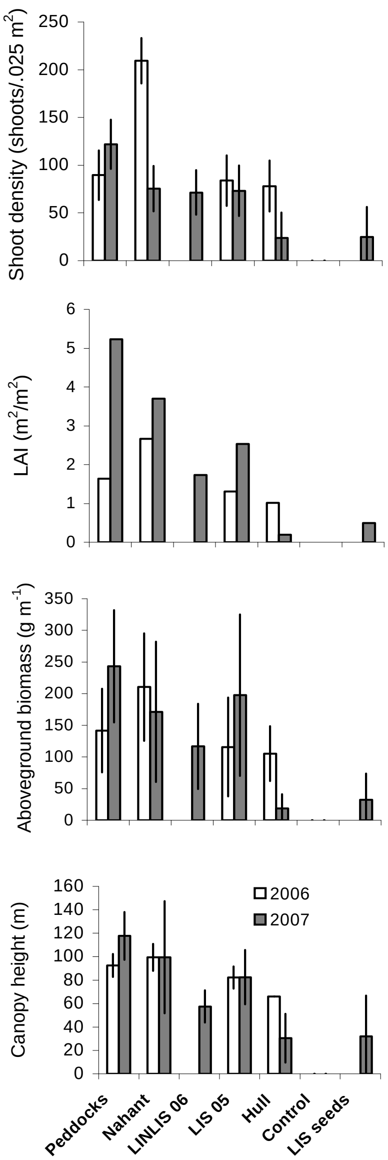


Figure 24. Measures of 3-D habitat function in areas monitored in 2006 and 2007. Control site = unvegetated. Nahant = healthy reference bed outside Boston Harbor; Hull = reference bed inside Boston Harbor. LIS seeds was seeded in fall 2005. LIS 05 and Peddocks were planted in 2005. LIN/LIS 06 were planted in 2006; data from these areas were combined. Error bars are ±SD. Because of the method used to calculate LAI, SD was not computed. Note: the drop in density at Nahant is likely due to the extraordinarily dense area randomly selected in2006, rather than a decrease in density of the bed as a whole in 2007.

The improvements in measures of both habitat structure and those of species abundance and diversity indicate that our successfully-planted beds are approaching or exceeding the habitat function of the healthy natural donor bed outside Boston Harbor. In contrast, habitat structure decreased in all four habitat measures at the pre-existing natural bed we monitored within Boston Harbor, i.e, Hull. Eelgrass at that site was patchier with leaves exhibiting more re-settled suspended sediment in 2007 than in 2006. Its location is not very well flushed and it may be suffering from localized water quality issues, possibly aggravated by its position in a mooring field. This indicates that a perceived general trend of improved suitability for eelgrass has not been uniform throughout Boston Harbor. It also emphasizes the need for careful site selection to locate those limited areas in previously degraded estuaries which may be conducive to restoration. The LIS seed bed values are diluted because the bed is still patchy; if zeroes are removed from the dataset the gap between it and the 2006 beds narrows.

Efficiency of harvest and planting methods. An efficiency analysis was conducted to evaluate harvesting rates and hand vs. frame planting (Table 8). Practiced divers harvested an average of 671 shoots/h. This number dropped to 450 shoots/h when volunteer divers participated, likely due to their inexperience. Hand-planting rate by *MarineFisheries* personnel was 390 shoots/person h (dive time plus boat helmsman), compared to 82 shoots/person h for frame planting which also included number of hours invested by dive, boat, and shore personnel. This difference is magnified (390 vs. 64) if time invested in stringing the frames is also included. However, if only dive hours expended in both planting techniques are counted, hand planting is less efficient than frame planting (441 shoots/h vs. 740 shoots/h, respectively). This gap narrows when dive time for retrieving frames is included (441 shoots/h vs. 542 shoots/h).

Table 7. Habitat structure: significance of interactions between sites monitored in 2006 (light gray), in 2007 (dark gray), and at each site between 2006 and 2007 (white). LIS seeds and LINLIS06 were only monitored in 2007; (*) denotes significant difference at P < 0.05; (**) denotes P< 0.01.

Shoot density

	Nahant	Hull	Peddocks	LIS 05	
Nahant	0.0001**	0.0022**	0.0067**	0.0022**	
Hull	0.0221*	0.0005**	0.9714	0.8624	
Peddocks	0.0134*	0.003**	0.0263*	0.9999	
LIS 05	0.9993	0.0499*	0.0447*	0.241	
LIS seeds	0.0373*	0.9982	0.0027**	0.0719	LIS seeds
LINLIS 06	0.9977	0.0318*	0.0154*	0.9999	0.0969

Aboveground biomass

	Nahant	Hull	Peddocks	LIS 05	
Nahant	0.4406	0.0933	0.5819	0.2452	
Hull	0.0065**	0.0004**	0.8239	0.9999	
Peddocks	0.6566	0.0021**	0.0209*	0.7769	
LIS 05	0.999	0.0112*	0.7506	0.0588	
LIS seeds	0.044*	0.999	0.0019**	0.008**	LIS seeds
LINLIS 06	0.9347	0.005**	0.0377*	0.4084	0.044**

Canopy height

	Nahant	Hull	Peddocks	LIS 05	
Nahant	0.512	0.0007**	0.8248	0.0353*	
Hull	0.018*	0.0012**	0.0009**	0.0005**	
Peddocks	0.9925	0.0031**	0.0139*	0.2802	
LIS 05	0.8957	0.0049**	0.0526	0.7608	
LIS seeds	0.1293	0.9999	0.0045**	0.0591	LIS seeds
LINLIS 06	0.5514	0.0402*	0.0043**	0.1738	0.7647

Table 8. Efficiency of different harvest and planting methods using trained *MarineFisheries* (MDMF) personnel and volunteers. Mean shoots per person-hour includes time invested by divers, boat handlers, and shoreside volunteers. Shoots/dive-h includes hours invested by divers only.

	Mean shoots per person hour					
	Harvest		Plant			
	MDMF only	MDMF + volunteer divers	MDMF (handplant)	MDMF+ shore volunteers (frames)	Shoots/dive h only (handplant)	Shoots/dive h only (frames)
Planting hours only	671	450	390	82	441	740
Includes time for stringing and retrieving frames				64		542

Table 9. Efficiency of single shoot vs. clump harvesting and planting (dive time only). Numbers are mean shoots per person-hour. Percent difference is between the 2 methods. HRM=horizontal rhizome method.

	Harvest		Plant	
	shoots/h	% difference	shoots/h	% difference
Single shoot/HRM	300*	271%	300	158%
Clump	814		475	

*exactly the number reported in Davis and Short (1997)

The "eelgrass clump" method proved much more efficient for both planting and harvesting than the single shoot/HRM methods (Table 9). Divers using the clump method were able to harvest 171% more shoots and plant 58% more shoots per person h than divers employing the single shoot/HRM methods. This analysis did not count time expended in bundling shoots for HRM, an unnecessary step for the clump method, which would increase the time differential between the two methods. The efficiency and apparent lack of negative impact on donor beds of the clump method (Leschen et al. 2006) provide justification for its use in areas where donor beds are sufficiently robust. The survival and eventual expansion of beds planted by the 2 methods should be studied further to determine if results differ significantly, but so far our planted plots show no evidence of a difference.

Hand planting can be accomplished efficiently by 2 or more experienced divers. On several occasions, 2 trained *MarineFisheries* divers transplanted 1000-3000 shoots (harvest and plant) in one day. While larger scale efforts can result in a greater numbers of shoots planted, such efforts also involve more coordination, divers, equipment, and boats.

Frame-planting was much less efficient than hand-planting, based on time invested, because of the number of steps involved. Volunteers are needed to string frames, sort and bundle shoots, and tie shoots in pairs onto the mesh of frames (25 pairs/frame) which are then deployed by divers. Frames must be retrieved at a future date and re-strung for re-use. Conversely, once shoots have been hand-planted, the task is completed (except for monitoring). However, frame planting provides a means for the non-diving public to be involved, and offers hands-on educational opportunities if within the goals of a restoration effort. Volunteers can also be employed in a hand-planting operation to bundle shoots (Short et al. 2002b; Sue Tuxbury, personal communication), although in our study area the distance between our harvest sites and a suitable shore base made this step prohibitively inefficient.

These instances highlight factors that must be considered in deciding which planting method and scale to use. The goals of the restoration program, available time, staff resources, including diving vs. shore-side volunteers, and tidal amplitude and resulting water depth in which volunteers would be working must be taken into account. In some coastal bays and rivers, much of this work can be done by snorkelers or even waders at low tide. This was never an option in Boston Harbor due to the steep tidal amplitude, short period of shallow depth available during low tide, and distance from shore; these factors limited our use of in-water volunteer workers to SCUBA divers.

Modeling of seed shoot movement. The paucity of suitable sediment in formerly eutrophic estuaries has implications for natural re-colonization of eelgrass, in addition to limiting possible restoration sites. GNOME™ model results showed it was improbable that seed shoots from existing bed locations would naturally disperse to the most suitable areas within Boston Harbor to grow new beds.

During the first simulation, hypothetical “shoots” were “spilled” at existing bed locations and spent 14 days adrift, but few shoots came near Long or Peddocks Islands, the only locations

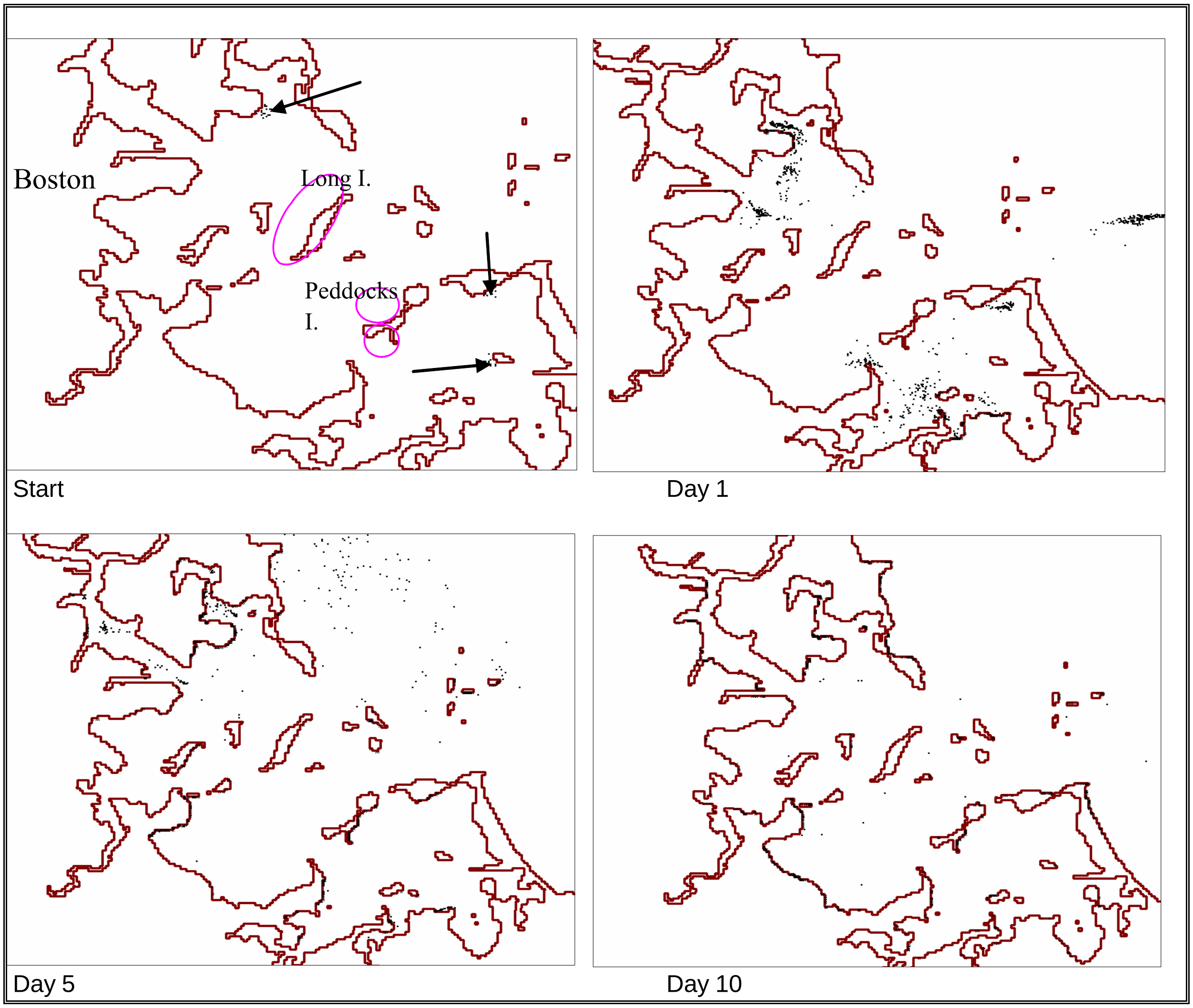


Figure 25. Selected output from GNOME™ model run for 14 days. "Shoots" (black dots) were "spilled" at remnant eelgrass beds (indicated by arrows) in Boston Harbor to ascertain if they would spread over areas identified in the site selection process as suitable for eelgrass (purple ovals).

where we found suitable sediment in the Harbor (Figure 25). According to the model output, the likelihood that floating reproductive shoots will pass over good eelgrass habitat is low. Furthermore, the probability that those few shoots which approach good sites actually sink and/or drop seeds there, and that those seeds go on to germinate and survive, decreases with each step. Since natural re-colonization appears unlikely, these results support the use of a restoration effort to "jump-start" the growth of eelgrass in Boston Harbor.

In contrast, when the simulation was re-run using our successful transplant sites as starting points, shoots were delivered throughout the Harbor, including a large number along the western coast of Long Island, in the Peddocks SE cove, and some in Portuguese Cove (Figure 26).

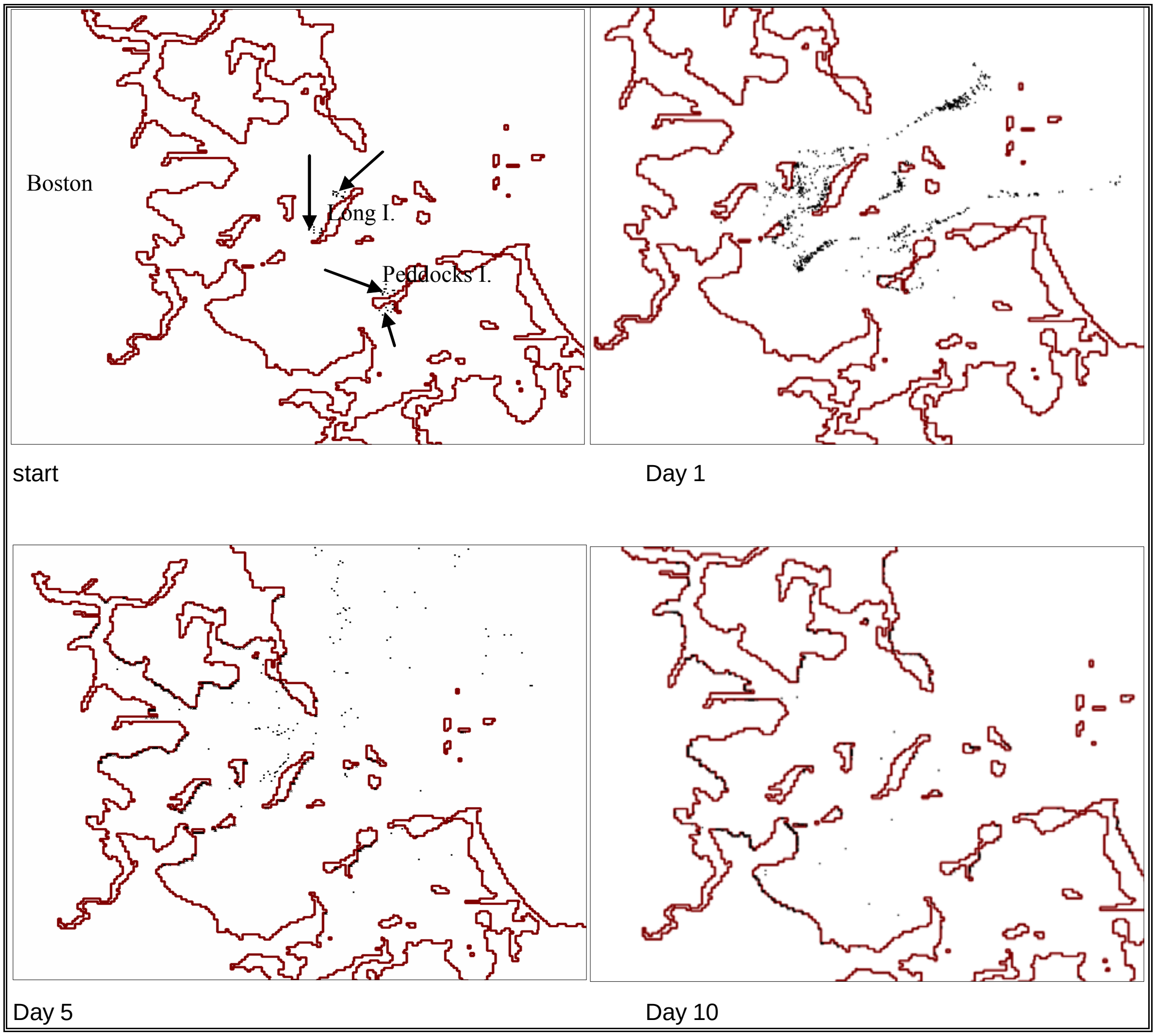


Figure 26. Selected output from GNOME™ model run for 14 days. "Shoots" (black dots) were "spilled" at our transplant sites (indicated by arrows) at Long and Peddocks Islands in Boston Harbor to ascertain if they would spread throughout those areas.

This result indicates that our planted beds are likely to "self-spread" throughout the transplant areas, thus increasing the efficiency of our restoration efforts. Possible empirical confirmation of this model result comes from our divers' observations that, in addition to expansion within the beds, both individual and small clumps of shoots grew scattered throughout the previously bare areas between plots, particularly at LIS. These shoots were found from a few to perhaps 20 m from the nearest plots with the furthest found beyond the range reported for seeds dropping from rooted plants (Orth et al. 1994; Harwell and Orth 2002a). This was a clear indication that 2005 and 2006 plantings were dispersing seed shoots and seeds in the area. The model output's display of a broad shoot distribution from our planted beds to points throughout the Harbor also increases the possibility that other, previously unidentified, small pockets of suitable sediment may be colonized by seeds from shoots originating in these locations. Though results are qualitative, use of GNOME™ can be useful to coastal managers in their decision-making about where or if to restore eelgrass. If initial site selection reveals a number of potentially suitable areas, GNOME™ can help steer resources toward areas that 1) are less likely to be naturally colonized, and 2) from which further self-spreading is likely.

Outreach. We received help during this Project from 155 volunteer shore workers and divers who provided 428 hours of assistance with harvesting and planting (Figure IVA.27). These included volunteers from several corporate groups, Odyssey High School, National Park Service, Boston Single Volunteers, Norfolk County House of Correction, New England Aquarium, Boston's Environmental Ambassadors to National Parks (BEAN) program, Genzyme, Clear Forest, State St. Corp., Boston University Marine Program, local dive clubs, and many individuals. Many *MarineFisheries*' divers also participated.

MarineFisheries' biologists gave presentations to staff at the New England Aquarium; to meetings of a Stellwagen Bank Sanctuary work group; the Massachusetts Bays Program; the Quincy Beaches and Coastal Commission; and also to the public on a catamaran tour of the Harbor sponsored by the National Park Service as part of its "Biodiversity Days". Other public outreach efforts included a presentation on a day-long biodiversity event for about 50 Earthwatch Institute employees who cruised Boston Harbor while learning about, and participating in, various research and restoration projects occurring there.

Our involvement with local school systems included Boston's Odyssey High School GIS class which used data from our research efforts as a real-life example to help them learn the mapping software. Members of Odyssey's after-school program completed the design of a logo which we used on t-shirts supplied to all volunteers. *MarineFisheries* staff also delivered a presentation to a career explorations class at Hull High School; to a group of Charlestown High School students who participated in the Courageous Sailing Program in Boston; and to Massachusetts Marine Educators at University of Massachusetts, Dartmouth. Children at the Marion Natural History Museum after school program also enjoyed learning about eelgrass from our staff. Eelgrass project personnel appeared on a Martha's Vineyard cable television program to talk about eelgrass and water quality with several other local biologists. Project activities and results were communicated through updates of the HubLine Eelgrass Restoration Project website and numerous news and magazine articles.

Figure 27. Some of the many volunteers that helped on this project.

Presentations were made to professional peers including a poster at the spring New England Estuarine Research Society meeting in Hull; a poster at the USEPA conference: “Celebrating aquatic habitat restoration in Massachusetts” in Ipswich in May 2007; an invited talk in a special section on urban estuaries at the September 2007 Annual American Fisheries Society meeting in SanFrancisco; oral and poster presentations at the Estuarine Research Federation conference in November 2007 in Providence, R.I.; talks at the annual meeting of eelgrass scientists and managers at EPA, Boston, and at the summer 2007 meeting of the Mass. Shellfish Officers Association on Martha’s Vineyard. A talk was also delivered at the Restore America’s Estuaries conference in October 2008.

Project staff participated in a multi-agency effort to harvest eelgrass from Gloucester Harbor in an area targeted for construction of a CSO pipeline. Approximately 7000 shoots were harvested by divers from *MarineFisheries*, EPA, and Metcalf & Eddy. Shoreside volunteers from CZM, MIT Seagrant, Winthrop Middle School, Gloucester Maritime Heritage Center, and other interested Gloucester citizens assisted with shoot bundling. These shoots were subsequently planted in Boston Harbor by *MarineFisheries* divers to augment restoration efforts there.

Conclusions

We successfully restored over 5 acres of eelgrass to a previously degraded estuary**,** Boston Harbor, by intensively focusing on site selection, with particular attention to sediment quality. High survival and expansion rates were recorded at 4 of 5 of our large-scale sites (the exception, Weymouth, would have been eliminated under new sediment guidelines). Our choice of planting locations was severely constrained by unsuitable sediment, which persisted throughout much of Boston Harbor even 5 years after elevated wastewater treatment and improved water quality were realized. These results have important implications for other estuaries where water quality improvement projects are undertaken. Such efforts may need to be combined with increasing flushing rates within these areas via dredging or other means in order to clear out accumulated depositional sediment that will impair eelgrass growth.

Acknowledgements

Dr. Ryan Davis assisted in the collation and evaluation of available Boston area environmental data sets during year #1 of the Eelgrass Restoration Project. A formal, comprehensive GIS analysis of all data layers was developed and initiated with the help of *MarineFisheries* Research Analyst Micah Dean. Other *MarineFisheries’* projects contributed data and/or SCUBA assistance: Coastal Lobster Investigations, Shellfish, and Environmental Impact Assessment. David Taylor and Ken Keay, MWRA kindly provided environmental data, and Charlie Costello, DEP contributed historical SAV distribution information. George Hampson and Pam Neubert provided helpful assistance with initial benthic infaunal investigative procedures.

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Appendix A. Species List - Benthic and demersal fish and invertebrates found at sites in Boston Harbor and Nahant.

Fish	**Invertebrates**
Cyclopterus lumpus	*Amphipod* spp.
Myoxocephalus aenaeus	*Cancer borealis*
Pholis gunnellus	*Cancer irroratus*
Pseudopleuronectes americanus	*Caprella* spp.
Sygnathus fuscus	*Carcinus maenus*
Tautogolabrus adspersus	*Crangon septemspinosa*
	Crepidula fornicate
	Echinaracnius parma
	Homarus americanus
	Libinia emarginata
	Littorina spp.
	*Laticidae (*Moon Shell*)* spp.
	Mysis spp.
	Mytilus edulis
	Pagurus spp.

Appendix B. Infaunal species observed in eelgrass sediment core analysis.
*** = 2006 only; ^ = 2007 only; no mark = both years.**

ANNELIDA
- Oligochaeta
  - Enchytraeidae
    - *Grania longiducta* Erseus & Lasserre, 1976*
    - Oligochaete sp. 1
    - Oligochaete sp. 2
- Polychaeta
  - Capitellidae
    - *Capitella capitata* Fabricius, 1780
    - *Capitella jonesi* Hartman, 1959
    - *Heteromastus filiformis* Claparède, 1864*
    - *Mediomastus ambiseta* Hartman, 1947
    - *Mediomastus californiensis*, Hartman 1944^
  - Cirratulidae
    - *Cirriformia grandis* Verrill, 1873
    - *Chaetozone hystricosus**
    - *Tharyx acutus* Webster & Benedict, 1887
  - Dorvilleidae
    - *Parougia caeca* Webster & Benedict, 1884*
  - Flabelligeridae
    - *Brada villosa* Rathke, 1843*
  - Hesionidae
    - *Microphthalmus pettiboneae* Riser, 2000
  - Lumbrineridae
    - *Lumbrineris tenuis*, Verrill, 1873^
  - Maldanidae
    - *Clymenella torquata* Leidy, 1855
    - Maldanidae sp. 1
  - Nephtyidae
    - *Nephtys buccera*^
    - *Nephtys caeca* Fabricius, 1780
    - *Nephtys discors* Ehlers, 1868*
    - *Nephtys incisa* Malmgren, 1865
    - *Nephtys picta* Ehlers, 1868
    - Nephtys sp. 1
  - Nereididae
    - *Nereis diversicolor* O.F. Müller, 1776
    - *Nereis succinea* Frey & Leuchart, 1847*
    - Nereidindae spp.
  - Orbiniidae
    - *Leitoscoloplos acutus* Verrill, 1873*
    - *Leitoscoloplos robustus* Verrill, 1873
    - *Scoloplos armiger* O.F. Müller, 1776*
  - Paraonidae
    - *Aricidea catherinae* Laubier, 1967
    - *Levinsenia gracilis* Tauber, 1879
  - Pectinariidae
    - *Pectinaria gouldi* Verrill, 1873*
  - Phyllocidae
    - *Eteone longa* Fabricius, 1780
    - *Eulalia viridis* Linnaeus, 1767*
    - *Pholoe minuta* Fabricius, 1780
    - *Phyllodoce mucosa* Oersted, 1843
    - Phyllodoce sp. 1
  - Polygordiidae
    - *Polygordius jouinae* Ramey, 2006
  - Polynoidae
    - *Harmothoe imbricata* Linnaeus, 1767
  - Spionidae
    - *Dipolydora quadrilobata*, Jacobi, 1883^
    - *Polycirrus eximius* Leidy, 1855
    - *Polydora cornuta* Bosc, 1802
    - *Prionospio steenstrupi* Malmgren, 1867*
    - *Pygospio elegans* Calparède, 1863
    - *Spiophanes bombyx* Calparède, 1870
    - *Spio limicola* Fabricius, 1785^
    - *Spio setosa* Verrill, 1873
    - *Streblospio benedicti* Webster, 1879
  - Syllidae
    - *Exogone hebes* Webster & Benedict, 1884
    - *Proceraea cornuta* Agassiz, 1862^
    - Syllid sp. 1*

ARTHROPODA
- Malacostraca
  - Ampeliscidae
    - *Ampelisca abdita* Mills, 1964
    - *Ampelisca vadorum* Mills 1963^
  - Aoridae
    - *Leptocheirus pinguis* Stimpson, 1853
    - *Microdeutopus anomalous*^
    - *Microdeutopus gryllotalpa* Costa, 1853*
    - *Unciola irrorata* Say, 1818
    - Unciola sp.
  - Cancridae
    - *Cancer borealis* Stimson, 1859^
    - *Cancer irroratus* Say, 1817*
  - Caprellidae
    - *Aeginina longicornis* Krøyer, 1842^
    - *Caprella unica* Mayer, 1903*
  - Corophiidae
    - Corophiidae sp.
    - *Crassicorophium bonelli* H.M. Edwards, 1830
    - *Crassicorophium crassicorne* Bruzelius, 1859
    - *Monocorophium sextonae* Crawford, 1937
  - Crangonidae
    - *Crangon septemspinosa* Fabricius, 1798
  - Dexaminidae
    - *Dexamine thea* Boeck, 1861^
  - Diastylidae
    - *Diastylis sculpta* Sars, 1871^
    - *Oxyurostylis smithi* Calman, 1912*
  - Idoteidae
    - *Idotea balthica* Pallus, 1772
    - *Idotea phospohorea* Harher 1873^
  - Isaeidae
    - *Photis pollex* Shoemaker, 1945^
  - Ischyroceridae
    - *Jassa marmorata* Holmes, 1903
  - Liljeborgiidae
    - *Listriella barnardi* Wigley, 1963*
    - *Listriella clymenellae* Mills, 1962^
  - Mysidae
    - *Mysis stenolepsis* Tyler, 1973^
    - *Neomysis americana* S.I. Smith, 1873*
  - Phoxocephalidae
    - *Eobrolgus spinosus* Holmes, 1903*
    - *Phoxocephalus holbolli* Kroyer, 1842

CHORDATA
- Ascidiacea
  - Stolidobranchia
    - *Molgula arenata* Stimpson, 1852*

MOLLUSCA
- Bivalvia
  - Lyonsiidae
    - *Lyonsia arenosa* Moller, 1842^
    - *Lyonsia hyalina* Conrad, 1831^
    - Lyonsia sp. 1
  - Myidae
    - *Mya arenaria* Linnaeus, 1758

Appendix B (*Continued*). Infaunal species observed in eelgrass sediment core analysis. * = 2006 only; ^ = 2007 only; no mark = both years.

Mytilidae
Mytilus edulis Linnaeus, 1758
Pharidae
Ensis directus Conrad, 1843*
Tellinidae
Tellina agilis Stimpson, 1857
Thraciidae
Thracia conradi Couthouy, 1839*
Gastropoda
Calptraeidae
Crepidula fornicata Linne, 1758^
Crepidula sp. 1*
Littorinidae
Lacuna vincta Montagu, 1803
Nassariidae
Ilyanassa trivittata Say, 1822
Phoronida
Phoronidae
Phoronis psammophila Cori, 1889 aka *Phoronis architecta* Andrews, 1890^

SIPUNCULA
Sipunculida
Sipunculidae
Phascolopsis gouldi Pourtales, 1851