DEVELOPMENT OF A PROTOCOL TO ASSESS THE RELATIVE HABITAT VALUES OF URBAN SHORELINES IN NEW YORK – NEW JERSEY HARBOR

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Development of a Protocol to Assess the Relative Habitat Values of Urban Shorelines in New York – New Jersey Harbor

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Cover photograph: Hard riprap shoreline south of Pier 1, Brooklyn Bridge Park during a low tide.

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Abstract

Habitat complexity is reduced when natural estuarine shorelines are replaced with concrete seawalls in highly urbanized regions. There is growing interest and investment in rehabilitating urbanized shorelines by adding physical habitat complexity to encourage establishment of diverse and resilient ecological communities. This is challenging, as multiple factors in addition to habitat availability operate across large scales to constrain ecosystem rehabilitation in urban estuaries. Design and management of shorelines to enhance their habitat and other ecological values should be based on using scientifically rigorous information to facilitate effective and efficient use of limited resources. This study was the initial step in the development of a protocol to provide a standardized and ecologically meaningful assessment of the relative habitat values of urban shorelines varying in physical habitat complexity across New York – New Jersey Harbor.

We developed a novel device with multiple colonization surfaces of standard dimensions, and a preliminary protocol manual to guide personnel in the construction and use of the device. In the subtidal zone of hard shorelines in the Harbor, mobile and sessile communities colonized mesh netting and hard settlement plates, respectively. Across all shorelines, mobile amphipods and encrusting algae were common, whilst isopods, shrimps, crabs and ascidians were common in some locations. Subtidal communities differed more between locations than between shorelines, varying in physical habitat complexity, within locations. Increased habitat complexity did not consistently favor any particular taxonomic group across shorelines, but there were notable differences in community structure between concrete seawall and riprap revetment shorelines in some locations. However, some components of the original colonization devices were not durable enough and loss of replicate samples prevented meaningful comparisons across all shoreline types.

Colonization devices were redesigned with stronger outer caging made from vinyl-coated steel, more secure settlement plate attachment and greater weight for anchorage to hard shorelines. Redesigned devices were successfully deployed on a hard shoreline subject to high water movement.

Intertidal surveys using quadrats, subtidal photoquadrat surveys, bivalve surveys using scouring pads and fish surveys using minnow traps were trialed, but did not provide additional useful information for comparing the relative habitat values of hard shorelines in New York – New Jersey Harbor. The colonization device and associated measurements of abiotic variables should be refined, as they show promise for facilitating standardized assessments that could inform the future design and management of hard estuarine shorelines in the Harbor.

Introduction

The important functions of natural estuarine shorelines include absorbing the erosive power of waves, filtering material suspended in water, trapping and stabilizing fine sediments, intercepting pollutants in runoff from land, processing detritus, cycling nutrients, and supporting both aquatic and riparian food webs (Kennish 2002, Currin et al. 2010, Barbier et al. 2011). Many of those functions depend on the activities of shoreline organisms. Shorelines provide a range of habitats that support both terrestrial and aquatic communities. Living and detrital vegetation (the later including 'wrack' redistributed by water movement) on the landward side of shorelines provides habitats for birds, mammals, reptiles and terrestrial invertebrates (e.g. Nagelkerken et al. 2008, Strayer and Findlay 2010). The types of animals and plants living in the intertidal and subtidal habitats on estuarine shorelines vary depending upon the size of the dominant substrate and whether aquatic vegetation is present. 'Soft' shorelines with fine unconsolidated sediment, such as sand or silt, are mainly inhabited by worms and crustaceans that burrow into the sediment (Day et al. 1989). Vegetation rooted in soft sediments can provide additional habitats, including important nursery habitat for young fish and roosting area for birds (e.g. Boesch and Turner 1984, Peterson et al. 2000, McKinney et al. 2006). Hard shorelines consisting of large consolidated rock, which limits the establishment of plants with roots and burrowing by animals, are mainly inhabited by mobile animals that seek refuge in crevices such as crabs, shrimps, amphipods, snails and worms, as well as sessile animals that attach to hard substrates as adults, such as barnacles, mussels, tunicates and bryozoans (Day et al. 1989, Butler 1995, Levinton et al. 2006). Encrusting algae also grow on hard substrates, but are readily scoured from unconsolidated sediments.

Much of the shoreline around New York – New Jersey Harbor was fine unconsolidated sediment prior to urbanization (Squires 1992, Sanderson 2009). However, as for other urban estuaries, most of the shoreline surrounding Manhattan was hardened to allow industry and construction along shorelines, protect human property from flooding, facilitate human access to the water and reduce erosion (Thompson et al. 2002, Charlier et al. 2005, Dugan et al. 2011, see Figure 1). Hardening of shorelines was historically done without considering impacts on shoreline ecology and many microhabitats and desirable ecological functions were undoubtedly lost when natural shorelines were hardened. Although it is infeasible to return many of the concrete seawalls in urban estuaries to unconsolidated sediment, it may be possible to add physical complexity to hardened shorelines using elements such as precast concrete, rocks or large wood. Hard shorelines with increased physical complexity may provide more diverse microhabitats to support more diverse and resilient communities (Airoldi et al. 2005, Bulleri and Chapman 2010, Chapman and Underwood 2011), that are able to perform a wider range of beneficial ecological functions than communities on vertical concrete seawalls.

There is growing interest and investment of resources to create urban estuarine shorelines with improved habitat value for native plants and animals (e.g. Currin et al. 2010, Goff 2010), including in New York – New Jersey Harbor (Johnson 2010). The *Hudson - Raritan Estuary Comprehensive Restoration Plan* identifies 'shorelines and shallows' as 'target ecosystems' requiring rehabilitation investment and suggests focusing rehabilitation initiatives on highly modified shorelines in New York – New Jersey Harbor (US Army Corps of Engineers and the Port Authority of New York and New Jersey 2009). Adding physical complexity to enhance habitat values for desirable biotic communities without conflicting with the other human uses of shorelines is complicated and involves collaboration between scientists, landscape architects, city planners, environmental managers and all levels of government. Ensuring that resources dedicated to shoreline rehabilitations are used effectively and efficiently requires clear articulation of the goals of each rehabilitation project and scientifically rigorous monitoring to assess whether these goals have been reached. Appropriate and adequate monitoring is required to learn from both ecosystem rehabilitation case study 'successes' and 'failures' in each region.

The main objective of the present study was to develop standardized survey methods that produce ecologically meaningful data for assessing the relative habitat value of urban shorelines that differ in physical habitat complexity within the New York – New Jersey Harbor. The study focused on development of colonization devices that allow a standardized measure of the diversity and abundances of subtidal invertebrate and algae, to estimate the relative habitat values of hard shorelines. The composition of subtidal communities are likely to reflect the habitat value of the shoreline, given that the adult animals and encrusting algae in such communities have limited mobility and need all of their requirements to be met within that shoreline habitat to persist. Colonization devices (i.e., settlement plates) have proven successful in detecting differences in sessile communities on natural shorelines and modified nearshore structures (Connell 2000, Connell 2001, Bulleri 2005). Past use of colonization devices to assess mobile invertebrate communities has been more commonly applied in streams (e.g. Hester and Dendy 1962, Meier et al. 1979, Nedeau et al. 2003) than in estuaries, but there are published examples of development of novel colonization devices for mobile invertebrates in estuaries (e.g. Atilla et al. 2005). Other methods used in past assessment of invertebrate and algal communities on natural and anthropogenic hard shorelines include quadrats or transects, photoquadrats, diver visual census, suction samplers and epibenthic scrapes (see Table A2). Although shorelines provide some species of fish, birds and mammals with critical resources during at least some life stages, quantifying the use of habitat by these transient users of shorelines is more difficult than for less mobile invertebrates and algae. A more detailed summary of the assessment of hard shorelines is provided in Appendix 1.

With a view to developing methods yielding response variables that can be used to assess the relative habitat values of hard shorelines differing in physical habitat complexity in New York – New Jersey Harbor, the hypotheses tested during this study were: i) diversity and/or abundance measures of intertidal invertebrate and algal communities are different between hard shorelines differing in physical habitat complexity, and these differences in intertidal community structure can be detected using surveys of quadrats; ii) colonization devices can be deployed on different hard shoreline types in New York – New Jersey Harbor and be colonized by subtidal communities after being submerged for four weeks or eight weeks; iii) the diversity and/or abundance measures of subtidal invertebrate and algal communities are different between hard shorelines differing in physical habitat complexity, and these differences in subtidal community structure can be detected using colonization devices and/or photoquadrat surveys; iv) the diversity and/or abundance measures of bivalves are different between hard shorelines differing in physical habitat complexity, and these differences in bivalves can be detected using scouring pads as colonization substrate; v) the abundances of fish are different between hard shorelines differing in physical habitat complexity, and differences in shoreline-associated fish communities can be detected using unbaited minnow traps; and, vi) the amounts of potential fish food are different between hard shorelines differing in physical habitat complexity, and these differences in food sources can be quantified by development of a fish food index for subtidal invertebrate and algal communities.

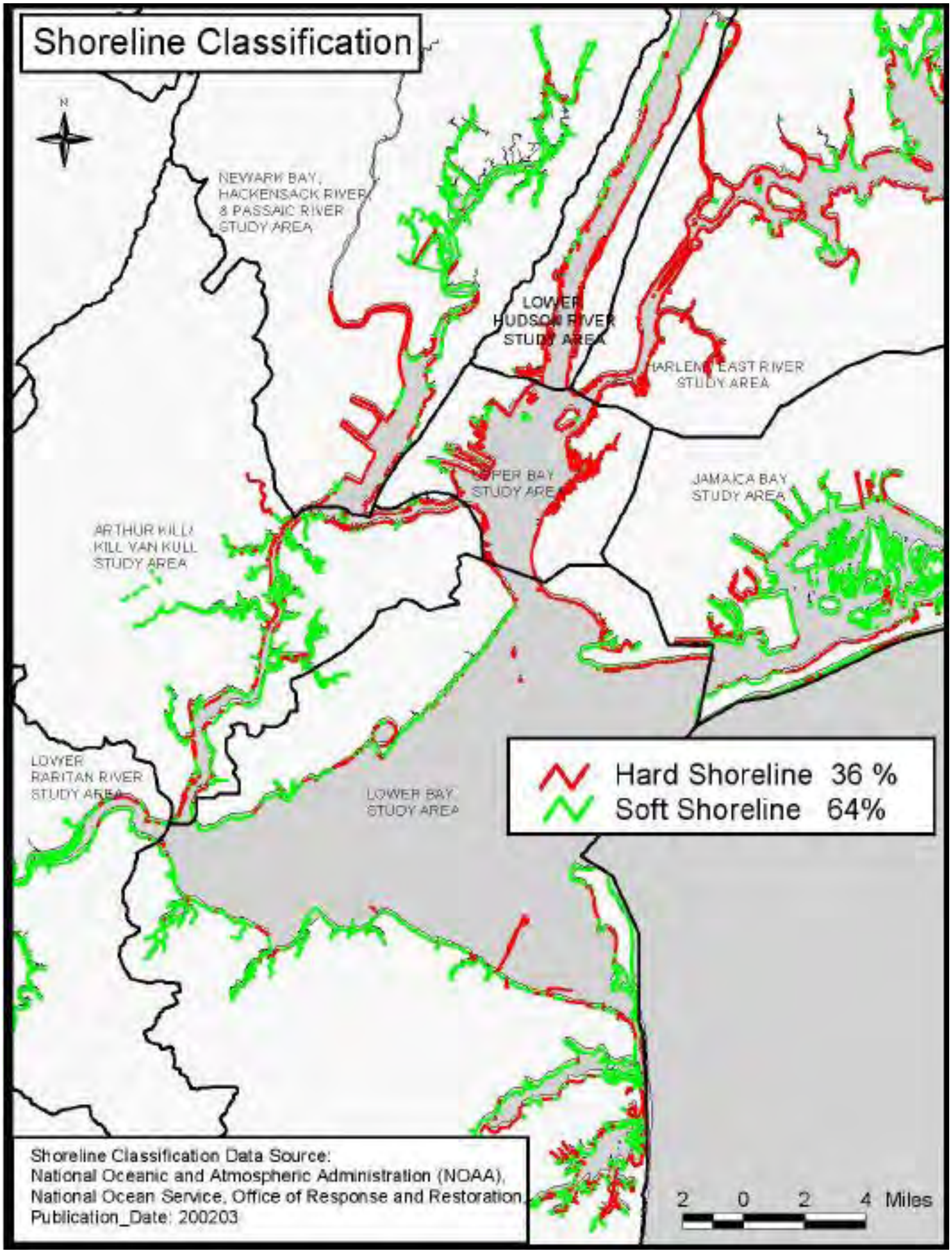


Figure 1: Locations of hard and soft shoreline within Hudson - Raritan estuary (from Bain et al. 2007). The present study was conducted on the shorelines surrounding Manhattan and the 'upper bay study area' (see Figure 3 for locations of study sites).

Methods

Shoreline sites

A thorough search of New York – New Jersey Harbor was done to identify which hard shoreline designs were most common and to ensure that representative hard shorelines chosen for the study covered a broad area across the Harbor. Given the variability in conditions across the Harbor, hard shorelines chosen for study in each location required a nearby reference shoreline, with all environmental variables except physical habitat complexity as similar as possible between paired shorelines within each location. The search involved examination of aerial maps of the shorelines around the entire Harbor, and visual assessments of shorelines during circumnavigation of Manhattan and travel from Governors Island to the north-eastern shoreline of Staten Island by boat. Ground-truthing was carried out to ensure that all the shorelines suitable for addressing the study objectives were accessible from the landward side.

The dominant hardened shoreline types found in New York – New Jersey Harbor were concrete seawall and riprap revetment (Figure 2a, b). There was also some shoreline built from large stone bricks (Figure 2c) and small stretches of gabion baskets (Figure 2d), but these were too uncommon to allow for replicate sites for sampling. Many shorelines appropriate for assessment using the methods detailed in this report were rejected for use in the present study, as they lacked nearby shorelines with different physical habitat complexity to serve as a reference. Site clusters were used to ensure that environmental conditions were as similar as possible between proximate groups of shorelines differing in physical habitat complexity. The site clusters used were Harlem River Park (Manhattan), southern end of Randall's Island (Manhattan, with seawall control at Astoria, Queens), West Harlem Piers (Manhattan), Brooklyn Bridge Park (Brooklyn), Liberty State Park (New Jersey, with seawall control on Ellis Island) and Fort Wadsworth (Staten Island) (Figure 3). Most site clusters included a riprap site with enhanced physical habitat complexity paired with a seawall site with minimal physical complexity. Although there was no suitable seawall site at Brooklyn Bridge Park, the riprap shoreline was used to examine the feasibility of deploying the colonization devices on a steep riprap shoreline with considerable boat traffic in the lower Harbor. To examine whether the survey methods were applicable to shoreline designs other than seawall and riprap, also included were a gabion basket shoreline at Harlem River Park and a stone brick shoreline at Randall's Island. Sites within clusters were no further than 1 km apart. Each shoreline site was ~100 m in length.

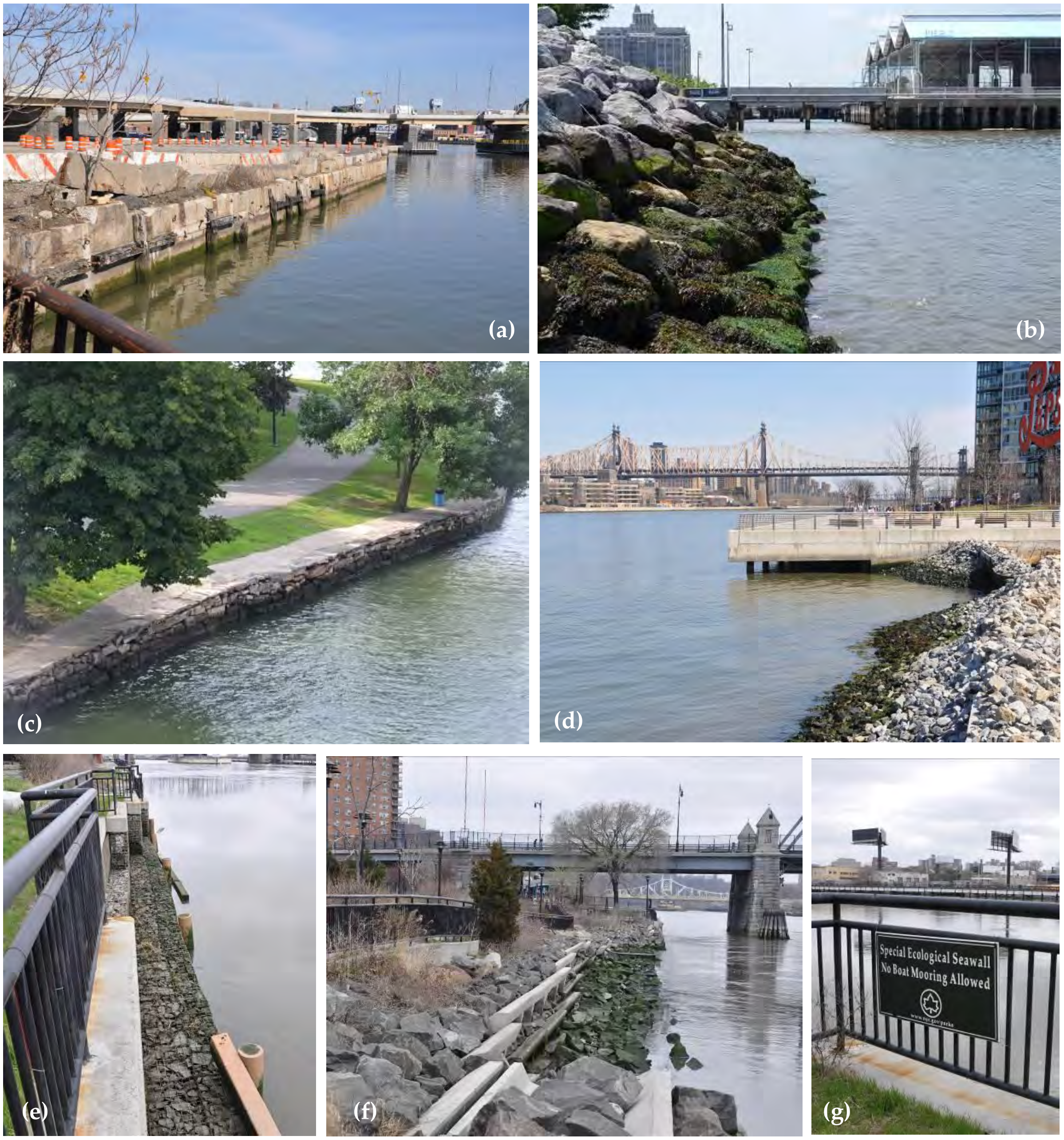


Figure 2: Representative engineered hard shoreline types from New York – New Jersey Harbor: (a) concrete seawall in Harlem River; (b) boulder riprap and pier during low tide at Brooklyn Bridge Park, East River; (c) stone wall at Randall's Island, East River; (d) boulder riprap and concrete pier with pilings at Gantry State Park, East River; (e) gabion baskets filled with cobbles and oyster shell, and (f) tiered boulder riprap at Harlem River Park, Harlem River. (g) The Harlem River Park shorelines have a New York City Parks sign indicating that they are 'Special Ecological Seawall'.

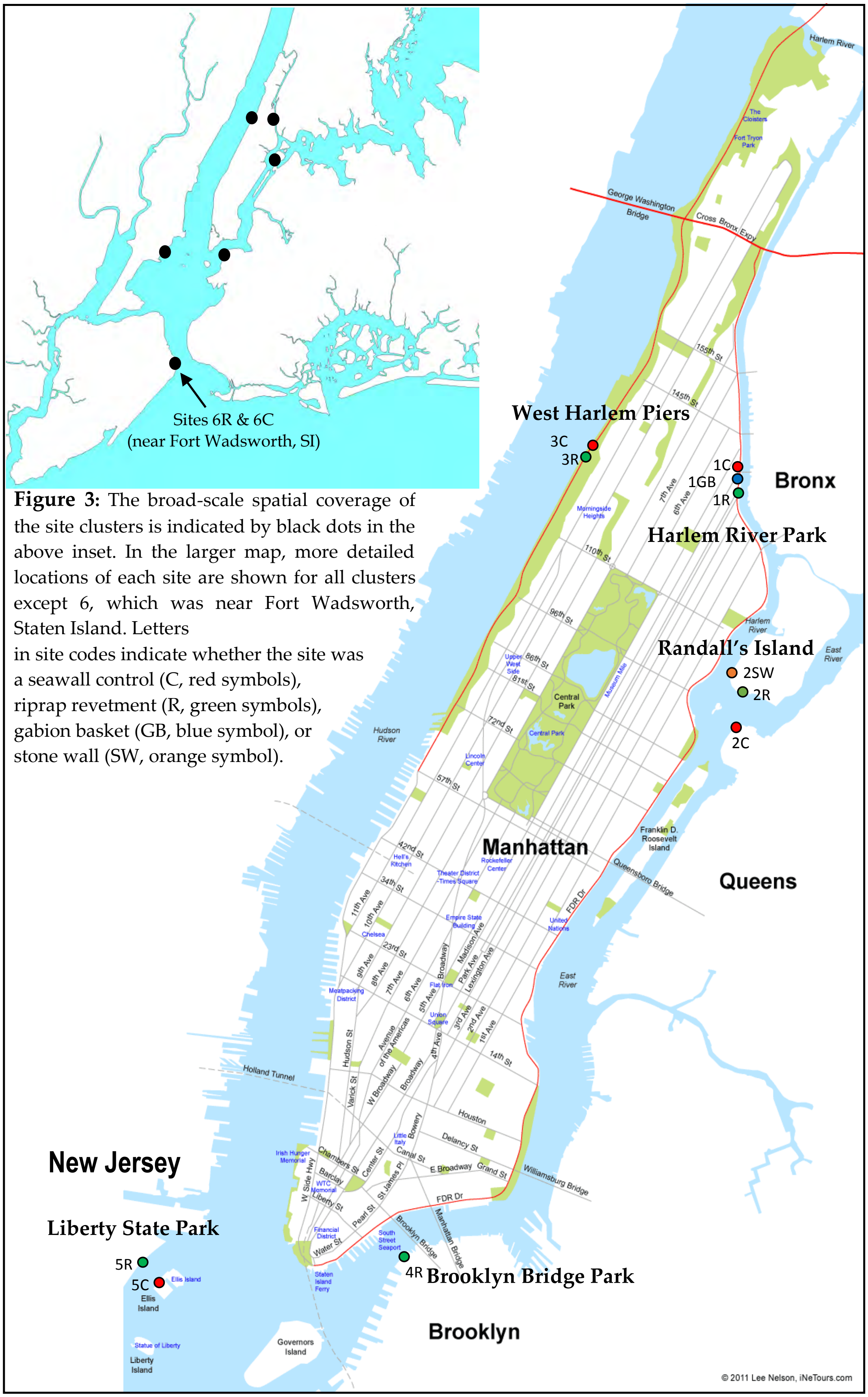


Figure 3: The broad-scale spatial coverage of the site clusters is indicated by black dots in the above inset. In the larger map, more detailed locations of each site are shown for all clusters except 6, which was near Fort Wadsworth, Staten Island. Letters in site codes indicate whether the site was a seawall control (C, red symbols), riprap revetment (R, green symbols), gabion basket (GB, blue symbol), or stone wall (SW, orange symbol).

Site descriptions: assessment of physical habitat structure, abiotic conditions and primary food resources

There was minimal habitat complexity along vertical seawalls, but the method of Strayer et al. (2012) was used to quantify differences in the sizes of hard substrate on riprap shorelines at each location. Briefly, at each shoreline, substrate particle size was estimated by categorizing the sediment by size at 100 regularly spaced intervals (e.g. 1 m intervals across 100 m) along a transect placed parallel to, and immediately above, the low-tide water level. The scores for particle size categories were: 0 = silt or clay, 1 = sand, 2 = sand to the size of a marble (16 mm), 3 = size of a marble to a tennis ball (64 mm), 4 = size of a tennis ball to a basketball (250 mm), 5 = size of a basketball to 1-m diameter, 6 = 1–4-m diameter, and 7 = larger than 4-m diameter. The 100 scores were averaged to provide a single estimate of the average particle size at each riprap site. Qualitative characterizations, photographs, cross-sectional and plan diagrams, and notes of prominent features likely to influence the habitat value of each shoreline were noted. See Reid et al. (2015) for an example of a completed site characterization field sheet.

As they were likely to vary and influence subtidal community structure at different locations across New York – New Jersey Harbor, abiotic conditions including nutrient concentrations were measured to assist the interpretation of the relative degree of influence of physical habitat complexity of shorelines on the structure of biotic communities. At each site, water temperature and light reaching colonization devices were measured continuously whilst they were submerged, using one HOBO data logger (Onset Corporation, Bourne MA) attached to the top of a randomly selected device. Mean and maximum irradiance were calculated for data collected between 6 am and 7 pm, eastern daylight time. Between July 21 and July 29, 2014, a water quality meter (Multi-probe, YSI, Yellow Springs OH) was used to measure salinity, conductivity and dissolved oxygen at each shoreline. Water samples were also collected and filtered in the field between those times, to determine concentrations of nutrients (i.e., ammonium, nitrate, nitrite, phosphate, silicon dioxide). See Reid et al. (2015) for a more detailed description of collection of water samples for water quality analyses. Water samples were processed in the Marine Biology laboratory of Lamont-Doherty Earth Observatory, using a four-channel autoanalyzer to measure nutrients.

Three parameters for which samples were collected were not available for this report: total suspended sediments, chl-a, and oxygen isotope ratios. These measurements are in process at the Lamont-Doherty Environmental Tracers Laboratory and will be delivered as an addendum upon completion. Samples for suspended sediment and chl-a analysis were lost as a result of a freezer failure, and are being resampled. Measurement of the project's oxygen isotope samples are waiting for the resolution of an inter-lab calibration exercise.

A Principal Component Analysis (PCA) was used to determine groupings of shorelines based on averages for salinity, dissolved oxygen, and concentrations of ammonium, nitrate, nitrite, phosphate and silicon dioxide. The linear relationship between salinity and conductivity was positive and highly significant (linear regression: adjusted $R^2 = 0.996$, $F_{1,53} = 14920$, $p < 0.001$), so conductivity was not included in the PCA. Temperature and daily irradiance were not included in the PCA, as they were not recorded across all shorelines. All statistical analyses were conducted using R 3.1.2 (R Development Core Team 2006).

Nutrient Sampling

Water samples for nutrient concentration measurements were collected at each site and measured at the Lamont-Doherty Earth Observatory Marine Biology Laboratory. Elevated nutrient levels have been a central concern in the New York – New Jersey Harbor area for many decades, both as indicators of sewage fluxes *per se*, and as harbingers of eutrophication. Biologically available inorganic nitrogen (nitrate, nitrite, ammonia) enters the system primarily from waste processing plants, dispersed runoff and areal deposition. Phosphorus enters with wastewater effluents, dispersed runoff and by leaching from mineral sediments. We have not seen a rigorous parsing of nutrient supply along these several pathways, but "rule of thumb" estimates put over half the supply in New York Harbor as coming from wastewater treatment facilities. Untreated human waste is also a major indirect source of inorganic nutrients, as dissolved or particulate organic matter is cycled *in situ* by bacterial decomposition.

Untreated waste entry into the New York – New Jersey Harbor is primarily through combined sewer overflows (CSOs): sewer systems that double as storm drains. During heavy rains CSOs partially bypass treatment plants, carrying untreated waste into the harbor. About 2/3 of New York City is served by CSOs, passing about 114 million cubic meters per year of untreated outflow into the Harbor. The past 25 years has seen major efforts to address CSOs, and the City is currently formulating and implementing Long Term Control Plans for its CSOs. The NYC Department of Environmental Protection claims that implementation of CSO best practices (increased capacity holding tanks, green surfacing, etc.) has nearly tripled the combined sewage capture rate in the past 35 years, http://www.nyc.gov/html/dep/html/stormwater/combined_sewer_overflow.shtml.) Significant progress on CSO best practices, green infrastructure and tertiary wastewater treatment notwithstanding, NYC DEP estimates for New York City alone run into many billions of dollars to remove organic matter from its point-source effluent stream.

Non-point-source nutrient fluxes have historically resulted mainly from the use of agricultural fertilizers, residential use of fertilizers and untreated sewage (e.g., from older septic systems). A 2010 New York State law bans phosphates from dishwasher detergents and residential fertilizers, but leaves the largest dispersed source, agricultural fertilizers, untouched (see http://www.dec.ny.gov/chemical/67239.html). Thus, the release of biologically available inorganic nutrients and organic matter to the harbor is an ongoing problem, and monitoring of nutrient levels remains an important metric of progress toward a healthy aquatic environment locally.

Nutrient Sampling Protocol

For this study we tested a simple nutrient sampling protocol that has been adapted by Lamont-Doherty's Secondary School Field Research Program for use by high school science teachers and their students in the Hudson Estuary. The materials required are inexpensive, widely available and durable. In our experience a novice trainee will typically use two or three filters whilst learning the loading procedure for the filter holder. Otherwise, the procedure is quite robust and straightforward, even for inexperienced field personnel.

Equipment and Supplies:

1. Two 60-mL syringes with 25 mm filter holders
2. Several 25 mm filters. For nutrients only, most laboratory-grade filters will work. If the water is turbid, as it often is in New York Harbor, Millipore "prefilters" work well. If the filters are being saved for chlorophyll analysis, filters with a nominal pore size less than 1 micron are recommended.
3. 60 mL test tubes (2/sample site)
4. microcentrifuge tubes (2/sample site)
5. Tweezers
6. Cooler ("6-pack" size works well)
7. Ice packs, frozen.
8. Permanent marker (e.g. "Sharpie")
9. Data sheets

Methods:

Record the: site name, site ID (if relevant), latitude, longitude, date, time, sampler's name, site conditions in a field notebook or on data sheets.

1. Use a marker to record site information, date, and time on two 50 ml test tubes; label them as replicates (e.g.: "A" and "B").
2. Wearing disposable lab gloves (nitrile or vinyl), use the tweezers to load the filter onto the filter holder without contaminating the filter (e.g. through contact with your bare fingers).
3. Attach the filter holder to the 60-ml syringe.
4. Extract the plunger from the syringe barrel and rinse the syringe 3 times with water from the location to be sampled.
5. Fill the syringe a 4^{th} time and insert the plunger into the barrel.
6. Apply a firm, smooth pressure on the plunger to force water through the filter holder and into each of the two test tubes.
7. Seal the test tubes and place them in the cooler. The filtered samples should be kept in the dark and as cold as is practical.
8. If measurements are to be made on the suspended sediments:
    a. Use tweezers to remove each filter and place it in a small cryo-tube. Be careful not to contaminate or tear the filter. Fold the filter on itself with the sediment sample inside the fold.
    b. Seal the small tubes and add those to the cooler.
    c. Record the amount of water that was filtered in collection of each sediment sample.
9. Unless measurements are going to be made within 24 hours, water samples and filters should be frozen as soon as practical and kept frozen until approximately 12 hours before measurement.

Measurement

There are a number of laboratories in the New York – New Jersey Harbor region that can measure samples for nutrient concentrations. For this study, we used the Marine Biology Laboratory and the Lamont-Doherty Earth Observatory. That lab can be reached through: Ray Sambrotto, 11 Marine Biology, LDEO, 61 Route 9W, Palisades, NY 10964.

Intertidal community assessment

Most past studies comparing the relative habitat value of engineered and natural hard shorelines have been conducted in the intertidal zone (see Table A2). However, the intertidal zones of most vertical seawall shorelines in New York – New Jersey Harbor are not readily accessible from the landward side. This limits the applicability of intertidal surveys in a protocol where it is proposed that seawall controls be used to assess the relative habitat values of other shoreline designs. Nonetheless, it was desirable to determine whether intertidal surveys may be useful in future hard shoreline habitat assessments in the region. In June 2014, qualitative assessments of the intertidal community were conducted at Harlem River Park, West Harlem Piers, Brooklyn Bridge and Liberty State Park riprap shorelines. The presence of biota were recorded from five quadrats (25 cm × 25 cm) placed randomly across the mid-shore during low tide, within the constraint of ensuring that quadrats were separated from each other by at least 5 m.

Colonization device design

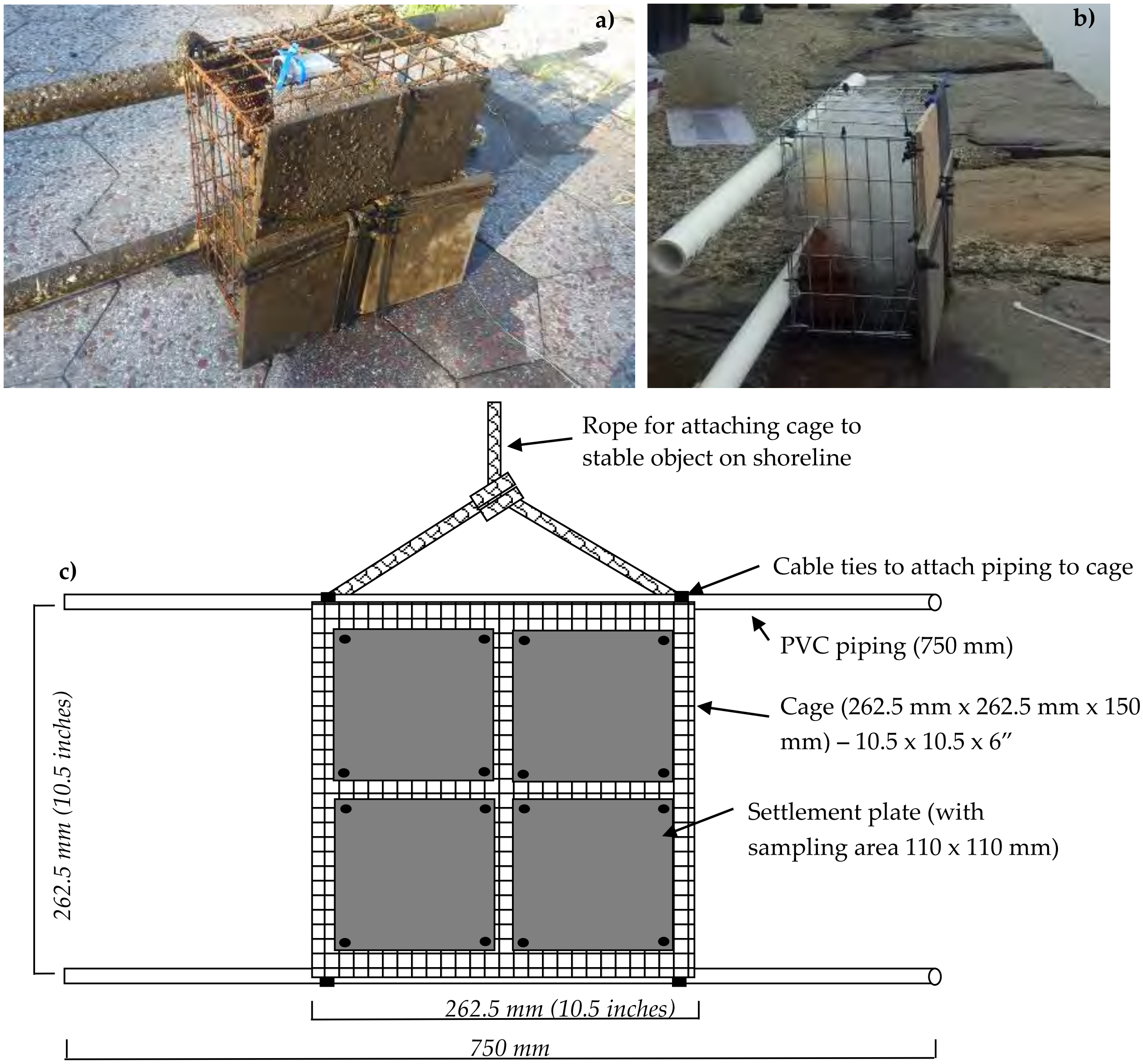
To allow comparison amongst a broad range of different shoreline types, standardized measurements must be obtained regardless of differences in physical habitat complexity between shorelines. This required the development of an approach that allowed sampling on nearly vertical hard shorelines, which differ from environments with unconsolidated sediments and low slopes where most estuarine surveys are conducted. Surveys involving entry into the water along near vertical shorelines can be hazardous, so methods were designed to minimize the need for entry of personnel into the water. A novel colonization device was designed for sampling of subtidal mobile and sessile invertebrate communities on hard shorelines ranging in physical habitat complexity and steepness, including vertical seawalls (Figure 4).

Colonization devices were designed to allow both mobile and sessile subtidal estuarine biota to colonize a standardized surface area that was submerged over a standardized duration at each shoreline. To allow adequate time for animals from the surrounding shoreline to colonize and grow to identifiable forms, most devices were deployed for eight weeks from early June to early August, 2014. At each shoreline, five cages (Foxy-Mate® low profile crab trap, 10.5 × 10.5 × 6”) containing bunched plastic mesh netting (mesh size of 4 × 4 mm, sheet dimensions of 750 × 2100 mm) and a brick with ten circular holes (brick dimensions of 191 × 90 × 56 mm, hole diameters ~ 20 mm) were used to sample mobile invertebrates. Polyvinylchloride plastic (PVC) piping attached to the shoreward side of cages provided stability when the colonization devices were sitting against the shoreline, countering rotational forces of waves acting to overturn submerged cages. Colonization devices were deployed in the subtidal zone, with sampling surfaces at least 0.5 m below spring low tide water level (*sensu* Levinton et al. 2006). Additional devices (five at both seawall and riprap shorelines) were deployed at West Harlem Piers for four weeks, early July to early August, to determine whether this time was sufficient for communities to colonize devices and for sessile biota to grow into identifiable forms. Devices were deployed at each shoreline during low spring tides to allow placement at depths that would ensure they remained submerged. The positioning of replicate colonization devices was randomized across the 100 m long sites, within the constraint of ensuring that devices were at least five meters apart.

The assumption of using colonization devices is that biota are most likely to arrive from the microhabitats immediately surrounding each device, and those biota that colonize and are best adapted to the conditions on the shoreline where each colonization device is deployed are those most likely to persist on that device. If these assumptions hold true, the community on colonization devices should be a reliable proxy of the community on the shoreline where each device was deployed. Colonization devices are not intended for use where it is desirable to obtain an inventory of all invertebrate and algal taxa occurring on shorelines.

Mobile invertebrate community assessment

Upon retrieval of colonization devices, the bricks, mesh netting and associated colonists were removed from cages and washed to remove all animals (see Reid et al. 2015 for detailed description of colonization device retrieval and processing of mobile invertebrate samples in the field). Briefly, all animals retained on a 500-μm sieve, used to remove fine sediment from samples, were preserved in 70% ethanol prior to identification in the laboratory. Mobile invertebrates were identified to the lowest possible taxonomic resolution that could be reliably achieved by all team members using dissecting microscopes, the taxonomic keys in Pollock (1998) and the taxonomic lists from previous reports on the Hudson benthic fauna (see Table A1). Personnel responsible for identifying mobile animals all had experience in the identification of invertebrates from estuaries subject to large human disturbances in north-eastern USA. A random selection of 10% of the mobile samples were cross-checked by the team member with the most extensive experience in identifying invertebrates from New York – New Jersey Harbor (i.e., J. Levinton), for quality control to facilitate consistent identification of taxa from mobile samples.



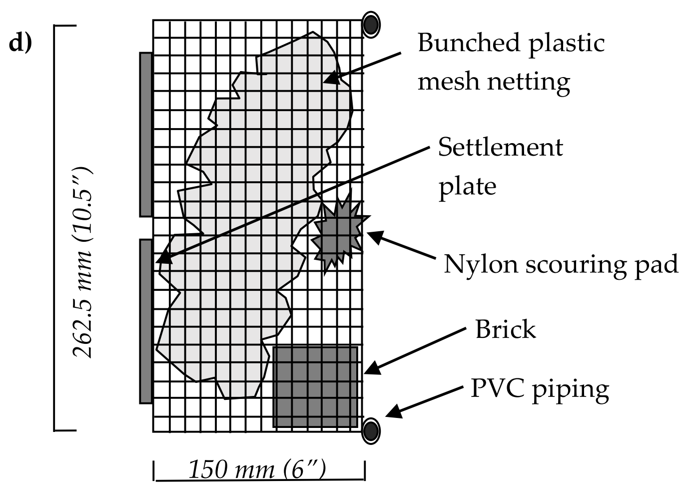


Figure 1: Design of colonization devices to sample communities using hard shorelines as habitat. a) Front and b) side views of the device. c) Diagram of the front view of the device (i.e., showing the side that faces away from the shoreline when device is deployed), where tiles (four, one each of acrylic, unglazed ceramic, slate and wood) were attached to a cage (using plastic cable ties) to allow colonization by sessile invertebrates that occupy the water column as larvae. Most devices were submerged over eight weeks, with attachment to a stable anchorage point on the shoreline using rope and PVC piping attached to the cage for stability. b) Diagram of the side view of the device showing bunched plastic mesh netting and scouring pad, which were used to facilitate colonization by mobile invertebrates and bivalve recruits, respectively. A brick was also included to both stabilize the device and to provide additional surfaces for colonization.

Sessile invertebrate community assessment

The relative abundance and spatial distribution of sessile invertebrates and algae were measured through the deployment of settlement plates attached to the outside of cages using plastic cable ties (Figure 4). These artificial substrata are widely used for measuring recruitment of sessile species and have been used in past assessments of the relative habitat values of hard shorelines (Connell 2000, 2001). Previous studies within the Hudson estuary have used plates constructed from a variety of materials, including unglazed ceramic (Levinton et al. 2006). For the present study, one each of an acrylic, unglazed ceramic, slate and wood (i.e., untreated oak) plate were used on each colonization device. Use of these plates was a compromise between allowing the assessment of the effects of differences in settlement plate materials, ensuring adequate replication of plate materials used in past similar studies (i.e., acrylic and unglazed ceramic), and including a range of materials that are representative of those that may be used in present and future hardened shorelines (i.e., stone and wood). The surface of plates were roughened with fine-grained sandpaper to facilitate bioadhesion of larvae of taxa such as ascidians, bryozoan and sponges (e.g. Bixler and Bhushan 2012). Colonization devices were designed to hold settlement plates parallel to the shoreline and with weight distribution that made them most stable when sitting on the shoreline with settlement plates facing away from the shoreline. This design meant that settlement plates were accessible to colonists in the water column and were oriented in the same plane as the shoreline where they were deployed.

Upon retrieval of colonization devices, high resolution photographs were taken with a digital camera (Nikon D3200, Tokyo) mounted on a forensic stand (Quadra-Pod™, Missouri) positioned at a standardized distance from each settlement plate (see Reid et al. 2015). Abundances of sessile organisms on 105 mm x 105 mm of the center of each settlement plate were assessed using a count of 100 stratified random points (i.e., 25 randomly positioned points in each quarter of each photograph) across each photograph taken of the plate surface (Glasby 1999, Knott et al. 2004). Overlays with stratified random points were created using Mathematica[1] (Wolfram Research Inc. 2014).

[1] Colm Kelleher (New York University) created the appropriate Mathematica coding during the present study

Point counts allowed counting of taxa occurring as individuals (e.g., barnacles) or percent cover for colonial species, such as colonial ascidians. Taxa were identified using the keys in Pollock (1998). Personnel responsible for identifying sessile taxa all had experience in the identification of sessile communities from estuaries subject to large human disturbances in north-eastern USA. Given that identification of sessile communities in photographs proved more challenging than identification of preserved mobile animals, a random selection of 20% of the images with sessile communities on settlement plates were cross-checked by personnel involved in sessile community identifications (i.e., J. Levinton, E. Bone and M. Thurman). This was done for quality control and to facilitate consistent identification of sessile taxa.

The loss of replicate mobile and sessile samples precluded the use of univariate analyses to compare taxon richness and abundance between shorelines varying in physical habitat complexity in many locations and also use of planned two-way ANOVAs to test for differences in those response variables associated with the predictor variables shoreline type and location. Univariate analyses were possible only for locations where there were at least three replicate samples retrieved from both the seawall control shoreline and a shoreline with more physical habitat complexity (i.e., riprap, gabion basket or stone wall). Two-sample t-tests were used to determine whether there were any differences in taxon richness or total abundances of mobile biota between nearby shorelines differing in structural habitat complexity. Two-sample t-tests were used to determine whether there were any differences in taxon richness, algal cover or invertebrate cover of sessile communities on settlement plates between nearby shorelines differing in structural habitat complexity.

Multidimensional scaling of fourth-root transformed multivariate data was used to create ordinations displaying the relative Bray-Curtis dissimilarities of mobile communities, and sessile communities on acrylic, unglazed ceramic, slate and wood settlement plates. Ordinations were used to examine whether dissimilarities in the structures of comparable communities between shorelines were attributable to differences in locations and/or physical habitat complexity. Similarly, multidimensional scaling ordinations were used to determine whether the sessile communities on different settlement plate materials were relatively similar within each location, compared to the similarity in community structure between locations. For ordinations comparing sessile communities on different settlement plate materials, only data from riprap and seawall shorelines were used, and separate ordinations were created for locations with fresh water influence (i.e., Harlem River Park and West Harlem Piers) and locations with predominantly marine influence (i.e., Randall's Island, Brooklyn Bridge Park and Liberty State Park), to reduce 'noise' owing to the effect of different water sources on sessile community structure. Fourth-root transformations were applied prior to calculating dissimilarity matrices for use in all ordinations, to reduce the influence of the most abundant taxa on the ordinations (Clarke and Warwick 2001). The application of this transformation allows less abundant taxa to have more influence on the ordination, with the drawback of making it increasingly unlikely that the two-dimensional ordination will have low stress (Clarke and Warwick 2001). Samples that had no biota where not included in ordinations that display all other replicate samples. Given that the two-dimensional ordinations displaying all replicate samples had relatively high stress (i.e., ≥ 0.15, see Clarke 1993), two-dimensional ordinations based on average community structure at each shoreline were also created. These ordinations had relatively low stress (i.e., ≤ 0.11), indicating that they display configurations more closely resembling actual dissimilarities between shorelines than the ordinations with all analogous samples represented (Clarke 1993). All statistical analyses were conducted using R 3.1.2 (R Development Core Team 2006).

Bivalve assessment

Given the importance of bivalve communities in the stabilization, productivity and filtration of water along shorelines (Strayer and Findlay 2010), the assessment of these communities was of specific interest. Some bivalves, such as oysters, may recruit to vertical tiles (Levinton et al. 2006), but other bivalves may settle on other three-dimensional substrates. The relative abundance of bivalve (e.g. mussel, oyster) recruits to each sampling device was assessed using nylon scouring pads. Nylon scourers (e.g. Tuffy™) have proven useful in assessing the recruitment of bivalves to hard substrata (Menge et al. 1994, Navarrete et al. 2008). Although the larvae of the eastern oyster *Crassostrea virginica* settle on oyster shell (Michener and Kenny 1991) and concrete (Anderson 1996), scourers can also provide favorable microhabitats for this species. Given that scourers are more easily acquired, manipulated and deployed than either oyster shells or concrete surfaces, they are more suited to use in a cost-effective and readily repeatable protocol. One nylon scourer was tied to the inside of the cage of each colonization device using a plastic cable tie. Upon retrieval, each scourer was placed within a sample bag to which 70% ethanol was added. Scourers were dismantled in the laboratory to remove and identify bivalves, and other taxa, using the same methods as for mobile invertebrate samples.

Photoquadrats of shorelines

The premise for using colonization devices in ecosystem assessment is that animals and algae are most likely to colonize from the microhabitats directly surrounding each device and will persist in devices only if local conditions are suitable, so that the communities on devices should provide a reliable proxy measure of the communities that inhabit the shorelines where they were deployed. However, validation is required to ensure that the communities occurring in colonization devices represent those on adjacent hard estuarine shorelines. With a view to comparing communities on settlement plates to those on hard substrate of shorelines, underwater photographs were taken of the subtidal region of a subset of riprap shorelines. Whilst standing in the water at low tide, a GoPro® Hero 4 camera (California) with waterproof housing was used to take photographs of the surfaces of boulders, at the same depth as colonization devices were deployed. A customized frame was constructed to ensure that the camera was at a standardized distance from the substrate and a scale (i.e. plastic ruler) was visible within all captured images. This allowed determination of the surface area represented within each image, so they could be assessed using the same methods as outlined above for sessile communities in photographs of settlement plates (i.e., grid point counts).

Small fish community assessment

Small fish communities were surveyed using five replicate unbaited minnow traps at each shoreline. These were regularly spaced across the subtidal zone of each shoreline (i.e., approximately 20 m apart from each other) and left submerged overnight. In the field, each fish was identified and their length measured, before returning them to the water. Large fish that were unable to enter minnow traps were not assessed. These fish are likely to be transitory across a variety of estuarine habitats and their presence or absence on a shoreline during a single survey may not be useful for determining the local habitat suitability for each species. These larger fish are also more difficult to sample than smaller fish, requiring larger nets, which is not consistent with the development of a cost-effective and readily repeatable protocol.

Figure 5: Deployment of unbaited minnow traps in the subtidal zone to capture small fish along a riprap shoreline near Fort Wadsworth, Staten Island.

Fish food index

It was anticipated that including comprehensive surveys of highly mobile fish populations (particularly larger bodied fish) would limit the use of a habitat assessment protocol by management agencies with limited resources. However, fish are important and highly valued components of estuarine communities, with ecological, economic and recreational roles. Therefore, it is desirable to obtain some measure of whether resources used by fish differ between hard shorelines varying in physical habitat complexity. Species trait analyses, particularly examination of feeding preferences of resident fauna, have been used to provide additional layers of information in past studies of estuarine communities (e.g. Lohrer et al. 2006, Reid et al. 2011). A thorough literature review was done to compile information on the relative likelihood of invertebrate and algae taxa which colonized devices during the present study to be useful as fish food[2]. ‘Fish food index’ values were assigned to each taxon based on whether it was: unlikely to be eaten by fish (0); eaten by fish, but not a preferred food source (1); or, preferentially eaten by fish (2). It was then possible to calculate a relative fish food index score for mobile samples and sessile communities on different settlement plate materials for each shoreline: by multiplying the abundance of each taxon by its assigned fish food index value, and adding the scores for all taxa. The mean fish food index scores calculated for each type of sample (i.e., mobile samples, acrylic settlement plates, ceramic settlement plates, slate settlement plates and wood settlement plates) were then added and compared to the abundances of small fish captured in colonization devices at each shoreline, using linear regression. All statistical analyses were conducted using R 3.1.2 (R Development Core Team 2006).

[2] Hillary Smith and Alex Fox conducted literature reviews for information about the feeding preferences of fish likely to occur in New York – New Jersey Harbor.

Refining methods

As this study was focused on developing a protocol, methods were continually refined as they were being tested in the field. The progression of steps to develop a scientifically sound protocol to assess shoreline habitat values is shown in Figure 6.

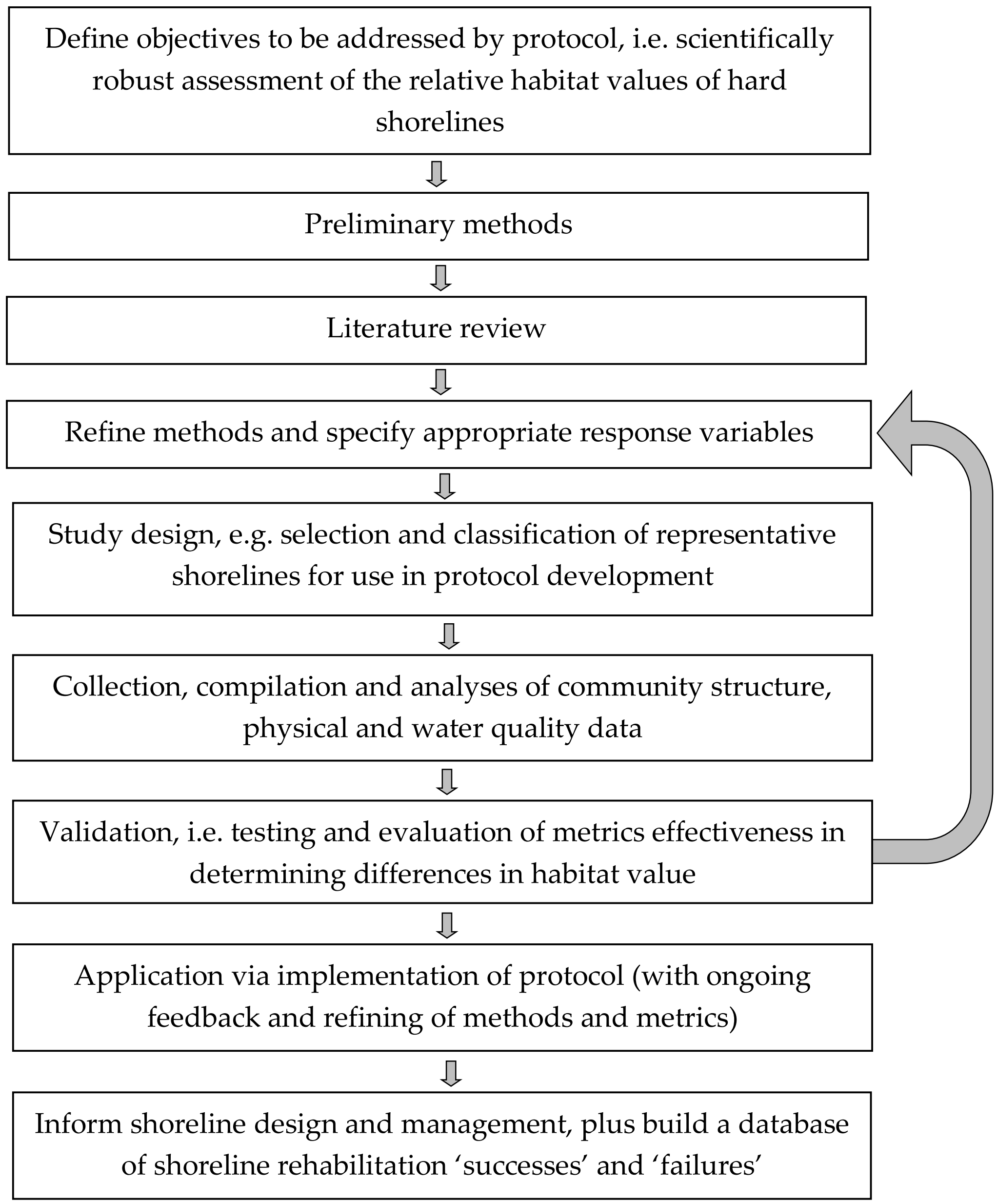


Figure 6: Flow diagram of sequential stages in development of the habitat assessment protocol (modified from Gibson et al. 2000).

Results

Site descriptions: assessment of physical habitat structure, abiotic conditions and primary food resources

There was ~100 m of shoreline with similar physical complexity to the study site on either side of the vertical concrete seawall at Harlem River Park. There were some large metal bolts and decayed wooden planks attached to the seawall. The nearby riprap shoreline was primarily large boulders (0.25–1 m diameter, Figure 10), with ~50 m of shoreline with similar physical complexity on either side of the surveyed shoreline. That shoreline was tiered, with three parallel concrete barriers along the shoreline holding boulders in place above the low tide water level. The vertical gabion basket shoreline (Figure 2d) also had ~50 m of physically similar shoreline on either side of the surveyed shoreline.

The vertical concrete seawall used as a control for Randall's Island shorelines was on the Astoria side of the channel directly across from the Island, with physically similar shoreline to that surveyed extending for ~100 m on either side. On Randall's Island, the riprap shoreline was mainly large boulders, but smaller substrates also occurred in the gaps between these boulders (Figure 7). Physically similar shoreline extended for at least 250 m on either side of the Randall's Island riprap shoreline. The vertical stone wall shoreline on Randall's Island (Figure 2c) had large gaps up to ~0.1 m between stones of varying size, up to ~1 m wide and ~0.5 m high. This type of shoreline extended for at least 200 m on either side of the shoreline that was surveyed during the present study.

At West Harlem Piers, the vertical seawall had a plastic veneer covering concrete shoreline which was adjacent to piers. Physically similar shoreline extended for ~50 m south and ~150 m north of the West Harlem Piers seawall. The riprap shoreline at West Harlem Piers was located ~100 m south of the end of the vertical seawall and had mainly large boulders (Figure 7). Physically similar riprap shoreline extended south from the shoreline that was surveyed for several kilometers.

The Brooklyn Bridge Park riprap shoreline had larger hard substrates than analogous shorelines in other locations (i.e., most often categorized as between 1 and 4 m, Figure 7) and occurred within a small cove between Piers 1 and 2 (Figure 2b). On the northern side of the cove there were pier pilings and riprap shoreline extending for ~100 m, then a concrete pier. On the southern side, the riprap shoreline extended for ~70 m down to the concrete shoreline adjacent to Pier 2 (underneath this pier was pilings and complete shade).

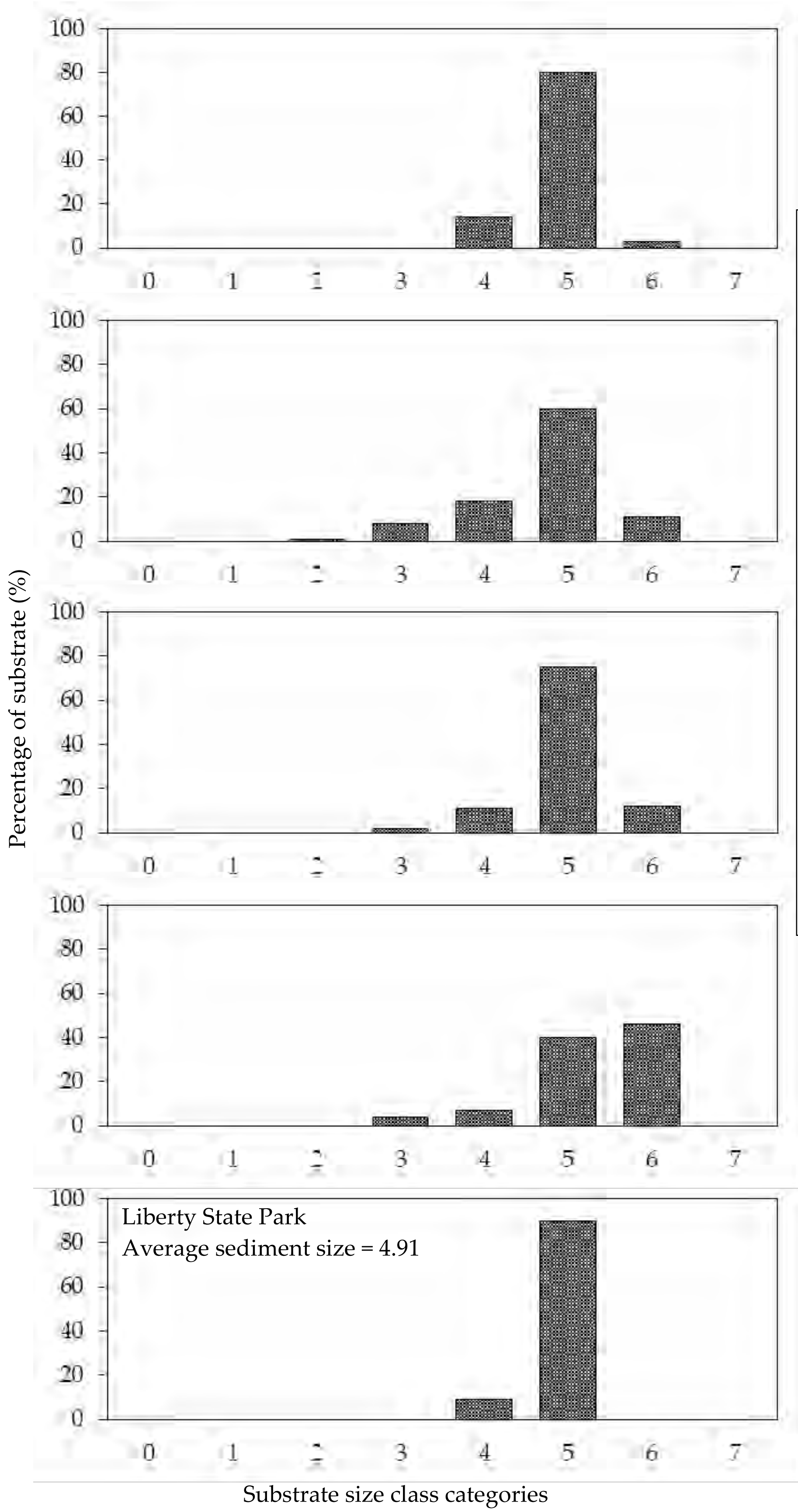


Figure 7: Proportions of sediment size categories at riprap shorelines. Sediment size categories are: 0 = silt or clay, 1 = sand, 2 = sand to the size of a marble (16 mm), 3 = size of a marble to a tennis ball (64 mm), 4 = size of a tennis ball to a basketball (250 mm), 5 = size of a basketball to 1 m diameter, 6 =

Much of the shoreline along Liberty State Park was riprap similar in physical complexity to the shoreline surveyed in that location, extending for over 300 m north and 700 m south of the riprap shoreline used during the present study. That shoreline mainly comprised larger boulders (Figure 7). The vertical concrete seawall used as a control for the Liberty State Park riprap shoreline was on Ellis Island, which had physically similar shoreline on all sides. Unlike the eastern side of the island, the western side where seawall was surveyed during the present study had minimal boat traffic.

Nutrient analyses

As the current study is focused on protocol development, and not intended to be a survey of shoreline conditions, nutrient sampling was limited to a single set of samples for each location. Characterization of the conditions at the sites would require time series and, ideally, sampling of waters in the area surrounding the sites. A proper characterization would especially require sampling during both wet and dry conditions, as data from other Harbor projects indicates that with each freshet nutrient concentrations spike, often by an order of magnitude, and then return to "dry" levels over about a 6-day timeframe.

The shoreline monitoring values fall generally within historical ranges (Table 1) and indicate that harbor waters are nutrient replete: primary production is not limited by any of the major nutrient concentrations. Without a more focused study, it is not possible to say exactly what the limitation is, but light limitation in the turbid harbor conditions is a likely candidate. Displaying the data in nitrate-phosphate phase space is clarifying (Figure 8). Most samples fall in regimes with N:P slopes between 11.5 and 14 to one. These are well within ratios typical for estuarine production (photosynthesis) and consumption (respiration). However, the Randall's Island samples exhibit extremely high phosphate levels and lie along an N:P slope of about 4.5 to 1. The likely source of this exogenous phosphate is the set of 7 Tier 1 CSO wastewater treatment plants feeding effluent into waters immediately east of the Island. These waters wash over Randall's Island each tidal cycle, but to establish whether they are the phosphate source would require further sampling along the tidal range.

Silicate is relatively high throughout the study area (Figure 9), indicating that the harbor is also replete in available silicate. Since diatom production often proceeds faster than carbonate based phytoplankton, this is further evidence that primary production in the harbor is not nutrient limited. Consideration of the samples in silicate:nitrate (Figure 9a) and silicate:phosphate (Figure 9b) phase spaces confirms that the Randall's Island environment is anomalous. There is a significant excess of silicate above the levels supported by the nitrate levels there and phosphate levels well above those supported by silicate, when compared with the balance of harbor samples.

In most cases, the values for salinity and dissolved oxygen were similar between seawalls and riprap shorelines within each location (Table 1). Salinity was relatively low at Harlem River Park and West Harlem Piers shorelines, which have much fresh water contributed from upstream watersheds (Table 1). Randall's Island, Brooklyn Bridge and Liberty State Park shorelines had mainly marine water sources and relatively high salinity. Dissolved oxygen concentrations were lowest at shorelines on the north-eastern side of Manhattan (i.e., Harlem River Park and Randall's Island), whereas shorelines at West Harlem Piers (north-western side of Manhattan, on the Hudson River) and the lower Harbor shorelines (i.e., Brooklyn Bridge Park and Liberty State Park) had relatively high dissolved oxygen concentrations (Table 1). Mean water temperature was between 73˚F and 76˚F at all shorelines, but with larger ranges in temperature on the north-eastern side of Manhattan than at West Harlem Piers and the lower Harbor (Table 1). The highest mean daily irradiances (i.e., > 500 lumen ft^{-2}) occurred at riprap shorelines, whereas mean daily irradiance at all seawalls, where data were available, was < 350 lumen ft^{-2}.

In some locations there were notable differences in nutrient concentrations between nearby shorelines (Table 1). At Harlem River Park, nutrient concentrations were consistently lower at the gabion basket shoreline than at the seawall and riprap shoreline. Concentrations of nitrate, nitrite, phosphate and silicon dioxide were all higher at the riprap and stone wall shorelines on Randall's Island than at the nearby Astoria seawall shoreline that was used as a control for Randall's Island shorelines. At West Harlem Piers, all nutrient concentrations were higher at the seawall than at the riprap shoreline. Compared to other locations, the concentrations of most nutrients were relatively similar between shorelines near Liberty State Park, except for higher ammonium concentration at the seawall on Ellis Island than at riprap shoreline along the park (Table 1).

Shorelines with predominantly marine influence and high salinities occur in the upper right-hand of the Principal Component Analysis ordination (Figure 10), with shorelines at Randall's Island separated from Brooklyn Bridge Park and Liberty State Park shorelines owing to their lower dissolved oxygen concentrations. The ordination separated the fresh water influenced shorelines based on their nutrient concentrations: gabion basket shoreline at Harlem River Park and riprap shoreline at West

Harlem Piers had similar low nutrient concentrations, whilst the riprap shoreline at Harlem River Park and seawalls at both Harlem River Park and West Harlem Piers had similar high nutrient concentrations (Figure 10). It is strongly recommended that samples be filtered in the collection process. We did, nonetheless, test a set of unfiltered samples (red dots in Figure 7). These samples were kept dark and cold during the field excursion and then frozen until 12 hours before measurement. Their nutrient values are plausible, and fit well within the ranges of the overall sample set. However, one has to be aware that in-sample photosynthesis and respiration would likely move the samples along the observed regime slopes, so the results do not indicate that filtering can be omitted from the field protocol.

Riprap shorelines retained more debris (i.e., large wood and plastic rubbish) than seawalls. This was a visible sign of human disturbance from watershed-scale processes, as most rubbish was presumably washed onto shorelines rather than being deposited directly on shorelines.

Table 1. Nutrient concentrations (μmol kg^{-1}) for samples taken at each location in the shoreline monitoring protocol development

Location	SampleID	Phosphate	Ammonia	Nitrite	Nitrate	Silicate	NPlusN
Harlem River	HRPRR1	3.203	19.978	1.717	30.198	23.999	31.915
Harlem River	HRPSW1	2.903	15.188	1.311	26.174	30.254	27.485
Harlem River	HRPSW2	2.801	15.403	1.346	26.391	31.281	27.737
Harlem River	HRPGB1	1.339	9.64	0.324	7.289	6.889	7.613
Harlem River	HRPGB2	2.047	14.116	0.741	14.755	14.221	15.496
West Harlem	WHRR1	1.423	11.574	0.421	8.209	7.019	8.63
West Harlem	WHRR2	1.599	12.821	0.686	14.506	14.242	15.192
West Harlem	WHSW1	2.722	21.192	1.257	23.414	24.044	24.671
West Harlem	WHSW2	2.925	18.952	1.347	25.339	28.103	26.686
Randall's Island	RISW1	3.573	14.492	1.361	14.829	24.006	16.19
Randall's Island	RISW2	4.783	17.686	1.857	21.093	31.462	22.95
Randall's Island	RIRR1	4.838	14.488	1.803	20.288	33.117	22.091
Randall's Island	RIRR2	4.602	14.85	1.669	18.627	31.598	20.296
Ellis Island	EISW1	3.121	20.666	1.624	18.701	24.741	20.325
Ellis Island	EISW2	2.997	21.919	1.588	19.406	24.612	20.994
B'klyn Bridge Pk	BB1	3.47	13.803	1.587	24.448	29.42	26.035
B'klyn Bridge Pk	BB2	2.523	8.264	0.776	13.762	15.232	14.538
Liberty State Pk	LSPRR1	3.202	11.756	1.235	21.123	22.74	22.358
Liberty State Pk	LSPRR2	3.297	11.917	1.332	22.351	24.472	23.683
Astoria	ASTSW1	2.675	14.388	1.068	11.935	14.975	13.003
Astoria	ASTSW2	2.592	13.912	0.983	10.861	13.66	11.844

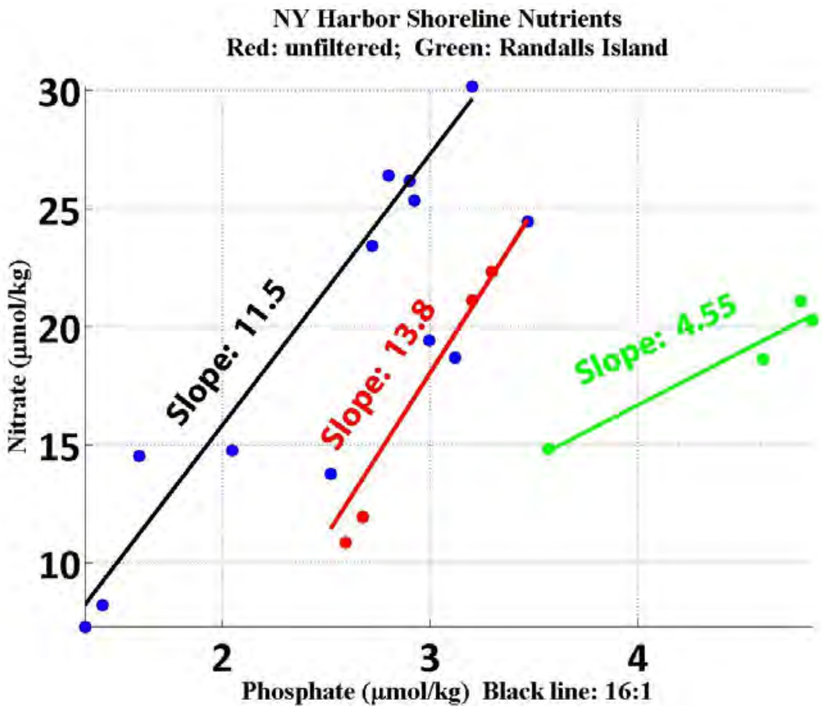
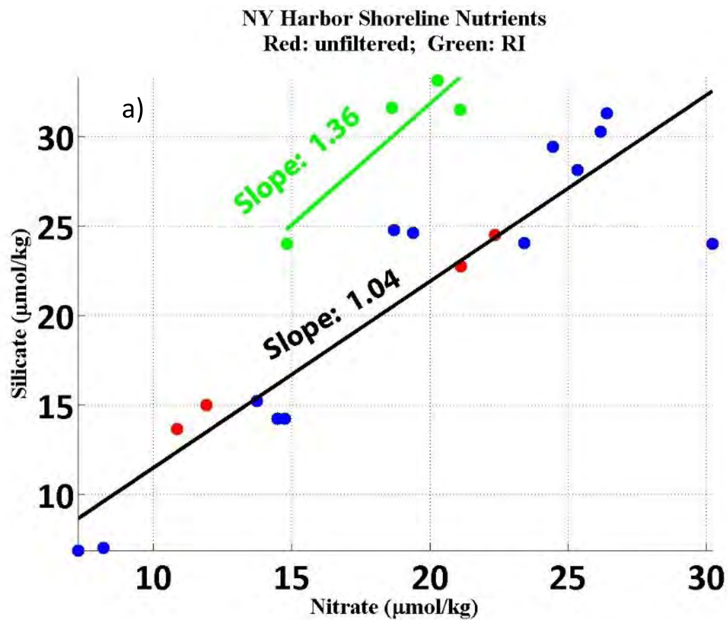


Figure 8. Concentrations of nitrate vs. phosphate in water samples collected at shoreline monitoring protocol test sites. Most samples fall along regimes dominated by biological production and consumption cycles. Randall's Island samples (green) indicate very high levels of exogenous phosphate. Unfiltered samples (red) do not show obvious anomalies. All samples were stored dark and frozen.



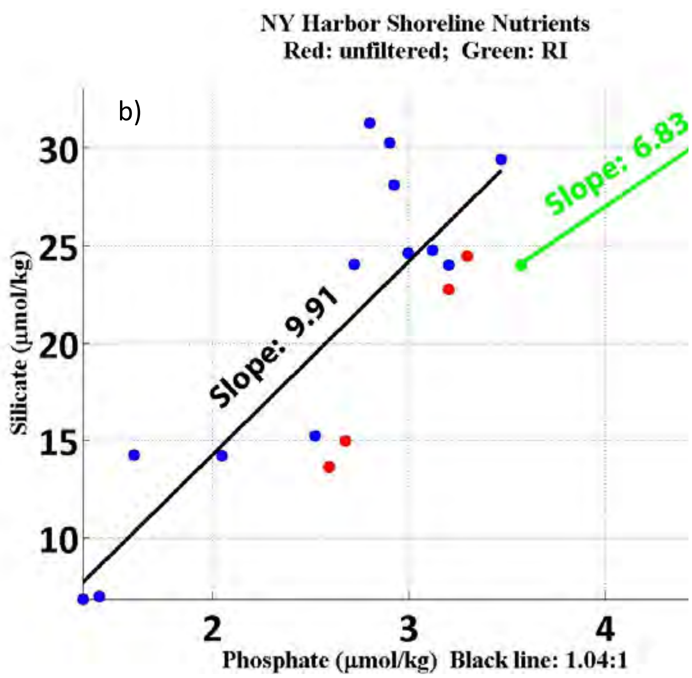


Figure 9. Concentrations of silicate vs. nitrate in water samples collected at shoreline monitoring protocol test sites. Most samples fall within typical estuarine ratios of silicate to nitrate. Samples from Randall's Island lie on a regime that is either high in silicate or low in nitrate.

Table 2: Abiotic conditions at shorelines surrounding New York – New Jersey Harbor. For most abiotic variables, the mean ± standard error is shown. For water temperature the mean is shown, with ranges in parentheses. For irradiance the mean is shown, with maximum in parentheses, calculated from values recorded hourly between 6 am and 7 pm, eastern daylight time.

Site code*	Salinity (ppt)	Conductivity ($mScm^{-1}$)	Dissolved oxygen (mgL^{-1})	Ammonium, NH_4 (μM)	Nitrate, NO_3 (μM)	Nitrite, NO_2 (μM)	Phosphate, PO_4 (μM)	Silicon dioxide, SiO_2 (μM)	Water temperature (˚F)	Daily irradiance ($lumenft^{-2}$)
1C	11.6 ± 0.0	19.4 ± 0.0	4.9 ± 0.1	15.3 ± 0.1	26.3 ± 0.1	1.33 ± 0.01	2.85 ± 0.04	30.8 ± 0.4	Not recorded	Not recorded
1R	12.0 ± 0.0	19.9 ± 0.0	4.9 ± 0.1	20.0	30.2	1.72	3.20	24.0	75 (67 – 92)	511 (1984)
1GB	11.5 ± 0.0	19.2 ± 0.0	4.9 ± 0.0	11.9 ± 1.6	11.0 ± 2.6	0.53 ± 0.15	1.69 ± 0.25	10.6 ± 2.6	75 (73 – 77)	92 (480)
2C	22.3 ± 0.0	34.3 ± 0.0	4.5 ± 0.0	14.2 ± 0.2	11.4 ± 0.4	1.03 ± 0.03	2.63 ± 0.03	14.3 ± 0.5	73 (71 – 89)	141 (576)
2R	23.1 ± 0.0	35.4 ± 0.4	4.2 ± 0.0	14.7 ± 0.1	19.5 ± 0.6	1.74 ± 0.05	4.72 ± 0.08	32.4 ± 0.5	73 (62 – 87)	1104 (13312)
2SW	22.7 ± 0.1	35.4 ± 0.3	4.5 ± 0.2	16.1 ± 1.1	18.0 ± 2.2	1.61 ± 0.18	4.18 ± 0.43	27.7 ± 2.6	Not recorded	Not recorded
3C	12.2 ± 0.0	20.1 ± 0.1	5.6 ± 0.1	20.1 ± 0.8	24.4 ± 0.7	1.30 ± 0.03	2.82 ± 0.07	26.1 ± 1.4	76 (74 – 79)	345 (2688)
3R	12.0 ± 0.0	19.9 ± 0.1	6.1 ± 0.1	12.2 ± 0.4	11.4 ± 2.2	0.55 ± 0.09	1.51 ± 0.06	10.6 ± 2.6	Not recorded	Not recorded
4R	20.0 ± 0.3	30.5 ± 0.3	5.2 ± 0.1	11.0 ± 2.0	19.1 ± 3.8	1.18 ± 0.29	3.00 ± 0.33	22.3 ± 5.0	74 (73 – 76)	534 (2944)
5C	22.1 ± 0.0	34.4 ± 0.2	4.9 ± 0.1	21.3 ± 0.4	19.1 ± 0.3	1.61 ± 0.01	3.06 ± 0.04	24.7 ± 0.1	74 (73 – 75)	114 (576)
5R	22.6 ± 0.1	34.8 ± 0.1	5.5 ± 0.1	11.8 ± 0.1	21.7 ± 0.4	1.28 ± 0.03	3.25 ± 0.03	23.6 ± 0.6	74 (73 -77)	567 (2304)

*Numbers in site codes indicate: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park. Letters indicate whether the shoreline was seawall control (C), riprap (R), gabion basket (GB) or stone wall (SW).

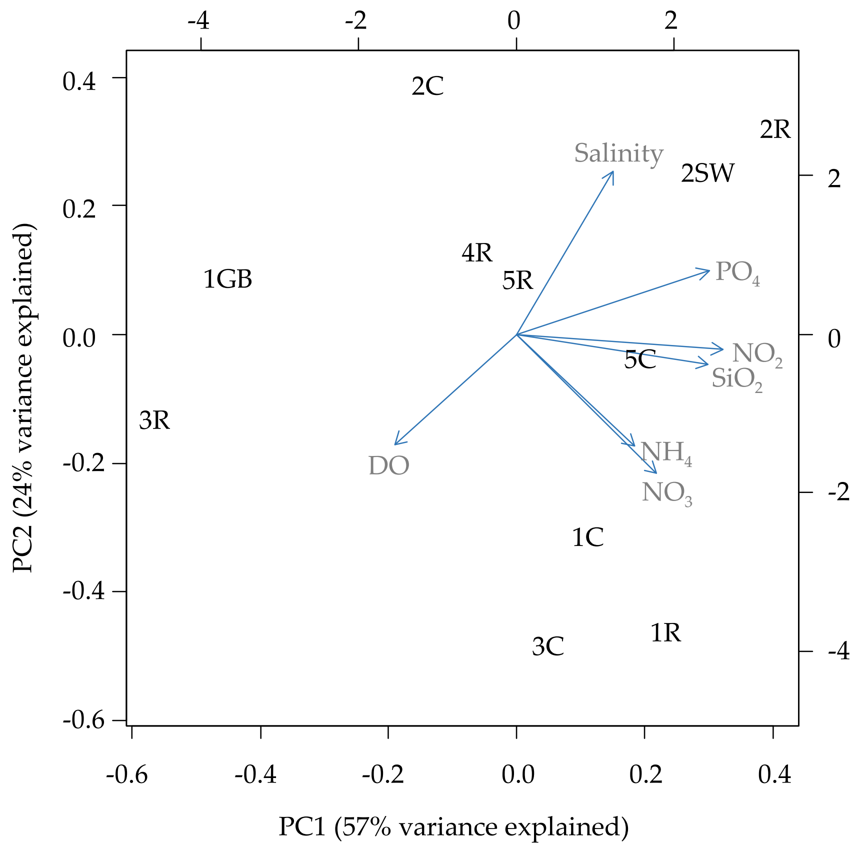


Figure 10: Principal Component Analysis of the abiotic conditions at hard shorelines in New York – New Jersey Harbor. Numbers in site codes indicate: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park. Letters in site codes indicate whether the shoreline was seawall control (C), riprap (R), gabion basket (GB) or stone wall (SW).

Intertidal Community Assessment

There was low biotic diversity in the intertidal zone of most shorelines surveyed. Often the rocks within the intertidal zone on the riprap shorelines around Manhattan (i.e., Harlem River Park, Randall's Island and West Harlem Piers, Figure 11) were covered in fine sediment, without identifiable biota on their upper surfaces. There was greater biotic diversity in the intertidal zone of riprap shorelines in the lower Harbor (i.e., Brooklyn Bridge Park, Liberty State Park and Fort Wadsworth), where there was less fine sediment and more macroalgal cover on rocks. Where visible from the landward side, the intertidal zone of most vertical seawall shorelines in both the upper and lower Harbor appeared to support very low biotic diversity. However, it was not possible to access and quantify the intertidal communities on seawalls, precluding use of these shorelines as references for assessment of the relative habitat values of the intertidal zones of nearby riprap shorelines. Intertidal community assessments were discontinued owing to: i) the difficulty in quantifying communities on seawall shorelines as references for riprap shorelines; ii) the apparent dominance of factors other than local shoreline physical complexity (e.g., sedimentation, salinity) in determining intertidal diversity; and iii) very low diversity of intertidal communities on shorelines in the upper Harbor, regardless of their physical habitat complexity.

Figure 11: A quadrat (0.25 m x 0.25 m) placed in the intertidal zone of the riprap shoreline at Harlem River Park. As with other shorelines around Manhattan, the intertidal zone was covered in fine deposited sediment, without identifiable animals or algae on the upper surfaces of rocks.

Constraints on colonization device utility in New York – New Jersey Harbor

Mobile invertebrates colonized plastic mesh netting and bricks. Sessile invertebrates and algae colonized settlement plates and bricks. The colonization devices worked largely as anticipated at Liberty State Park, where the seawall shoreline on the western side of Ellis Island was protected from strong currents, waves and boat traffic, and the hard riprap shoreline extended deep enough to allow devices to be placed below the low tide swash zone. However, across all shorelines the outer cages (i.e., galvanized steel crab traps) quickly rusted when left submerged in the field and were not suitable for repeated use. The cages were not durable enough to withstand the conditions at Brooklyn Bridge Park, where some devices were found crushed after eight weeks (Figure 12a). Across all shorelines, replicate settlement plates were frequently dislodged from the cable ties used to attach them to crab traps. Loss of replicate samples often precluded the use of univariate analyses when comparing the diversities and abundances of communities occurring on different shorelines (Table 3).

No data were obtained from the colonization devices deployed at those shorelines near Fort Wadsworth. Colonization devices along the seawall were removed during maintenance of the grounds surrounding Fort Richmond. In any case, the seawall did not extend far below low tide water level and colonization devices were subject to intense wave action in the swash zone during low tide. The devices were not designed for deployment on such shorelines. All devices deployed at the riprap shoreline near Fort Wadsworth were washed ashore (Figure 12b-d). That shoreline had low gradient and was influenced by strong currents flowing through The Narrows, as well as presumably experiencing larger waves during storms than more protected shorelines.

The damage to cages and dislodgement of settlement plates indicated that water movement was more powerful than anticipated. Devices were presumably most damaged during a storm that created large waves in the Harbor in early July, as Hurricane Arthur swept up the east coast. Devices would have been subject to less turbulence if it was possible to submerge them well below the swash zone. However, it was necessary to deploy colonization devices close to the low-tide water level at most shorelines, owing to the hardened shorelines not extending far below this depth. Immediately below the hardened portion of shorelines, the substrate was usually silt and this was readily suspended in the water column by water turbulence. This suspended sediment was deposited as a thin layer on settlement plates. Although fine sediment did not entirely smother the plates, it may have affected the ability of some taxa to colonize the hardened surface. This phenomena was reflective of that occurring along shorelines in the Harbor, rather than being unique to settlement plates used on colonization devices.

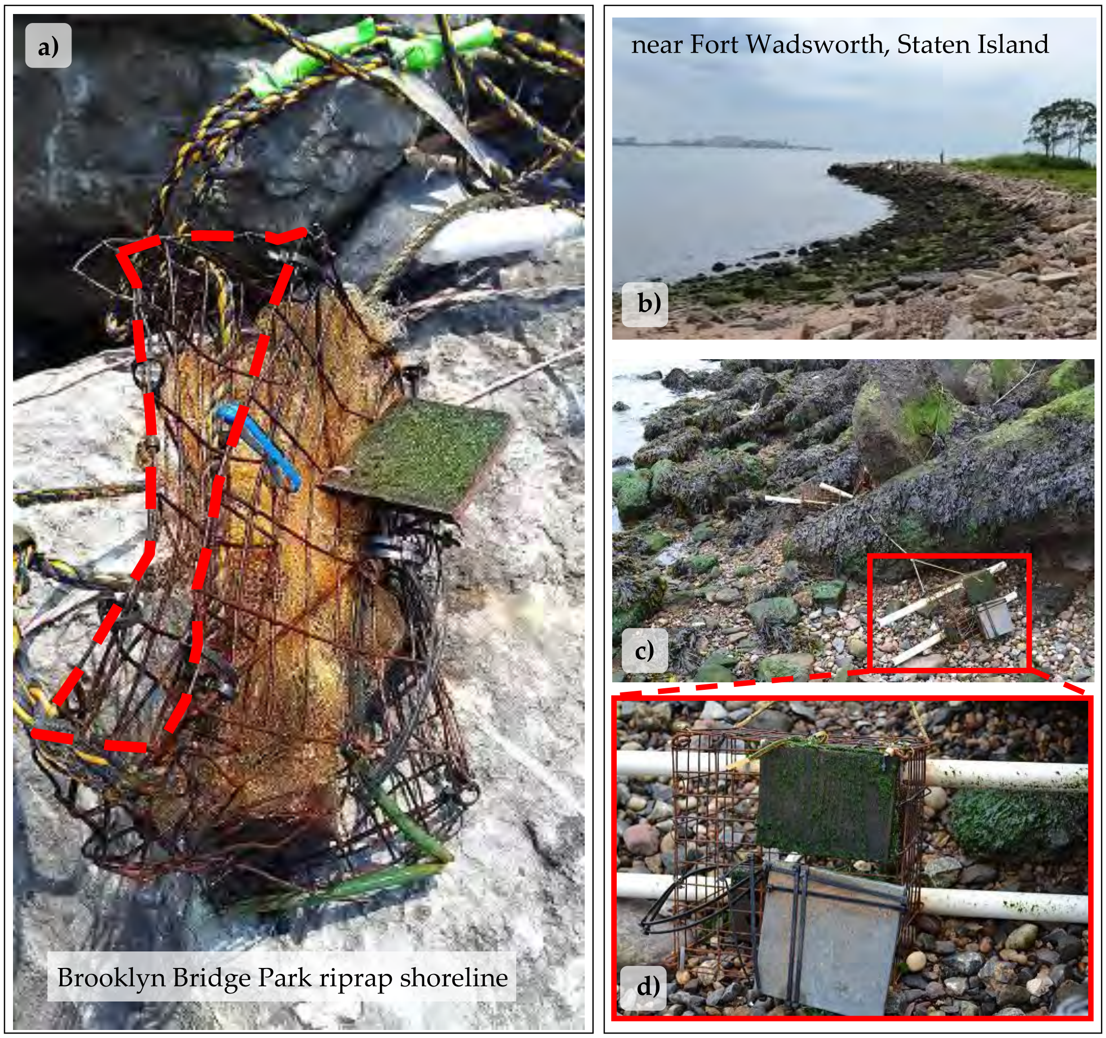


Figure 12: Damaged colonization devices. a) High wave energy at the Brooklyn Bridge Park riprap shoreline crushed some cages and dislodged settlement plates. The red dashed line highlights the modified profile of the previously rectangular caging made from a stainless steel crab trap (see Figure 4b for original profile). A broken wood settlement plate with green macroalgae cover is tenuously attached to the caging by one remaining cable tie, but all other settlement plates were dislodged. b) The riprap shoreline at Fort Wadsworth had relatively low gradient and fast flowing water moving through The Narrows. At that shoreline, (c) colonization devices were washed ashore and (d) settlement plates were dislodged from cable ties attached to metal cages (e.g. acrylic and ceramic plates were dislodged from the left-hand side of the device shown).

Table 3: Numbers of replicate mobile invertebrate samples and settlement plates retrieved from colonization devices. Grey shading indicates where univariate analyses between shorelines were possible (i.e., two sample t-tests were used to compare taxon richness and abundances where ≥ 3 replicates were available from both the seawall control shoreline and at least one of the riprap, gabion basket or stone wall shorelines within a location).

Location	Shoreline type	Site code	Mobile samples	Acrylic plates	Ceramic plates	Slate plates	Wood plates
Harlem River Park	Seawall control	1C	**3**	**3**	2	2	**3**
	Gabion basket	1GB	**4**	**4**	2	2	**3**
	Riprap	1R	2	1	1	1	1
Randall's Island	Seawall control	2C	1	0	0	0	0
	Riprap	2R	3	1	1	0	2
	Stone wall	2SW	1	1	2	0	2
West Harlem Piers	Seawall control	3C	**5**	**5**	**4**	5	**5**
	Riprap	3R	**4**	**3**	**3**	1	**3**
Brooklyn Bridge	Riprap	4R	2	2	1	0	2
Liberty State Park	Seawall control	5C	**3**	**5**	**5**	**5**	**5**
	Riprap	5R	**3**	**5**	**4**	**4**	**5**
Fort Wadsworth	Seawall control	6C	0	0	0	0	0
	Riprap	6R	0	0	0	0	0

Mobile invertebrate community assessment

Amphipods, isopods, caprellids, shrimp, crabs, mussels and worms colonized the plastic mesh netting and bricks in colonization devices. The plastic mesh netting also facilitated the accumulation of detritus and sediment, which potentially provided food and additional microhabitat for mobile invertebrates. The mean number of individual mobile invertebrates (Table 3) and number of taxa (Figure 13) colonizing devices were higher at riprap shorelines than at seawall control shorelines at all locations except Liberty State Park. However, the number of mobile invertebrates was highly variable between the colonization devices deployed along the same shoreline, and taxon richness was low across both control and riprap shorelines (i.e., only 17 mobile invertebrate taxa captured during the present study).

Harlem River Park

At Harlem River Park, there were consistently close to 200 individual mobile invertebrates in three colonization devices deployed on concrete seawall (Table 3). Conversely, on the riprap shoreline the number of individual mobile invertebrates was highly variable between the two retrieved colonization devices: 180 in one device and 548 in the other. The mean number of taxa in mobile invertebrate samples was 5.0 ± 0.9 on seawalls, slightly lower than the 6.0 ± 1.0 taxa on riprap shorelines (Figure 13). However, all of the mobile taxa captured on riprap shoreline were also captured on seawall at Harlem River Park. On the seawall, amphipods were the most common order in two of the colonization devices (>80% of total), but over 75% of the mobile animals were isopods (mainly *Idotea* sp.) in one of the colonization devices (Figure 14a). Amphipods had relatively high abundances in both of the devices deployed on riprap shoreline, but >80% of the invertebrates in the more densely colonized device were isopods (i.e., *Idotea* sp.). The amphipods *Amphithoe valida*, *Microdeutopus gryllotalpa* and *Amphithoe* sp. were the taxa most often captured in the colonization devices deployed at both the seawall and riprap shoreline at Harlem River Park (Figure 15b).

As for the riprap shoreline at Harlem River Park, the abundances of mobile invertebrates from colonization devices deployed along the gabion basket shoreline were highly variable (i.e., ranging from 112 to 479) and similar to those on the seawall (t = –1.15, df = 3.3, P = 0.33, Table 3). The numbers of taxa were also similar at the seawall and gabion basket shorelines (t = –1.41, df = 3.2, P

= 0.25). The mobile invertebrate community structure at the gabion basket shoreline was similar to that at the nearby seawall and riprap shorelines at Harlem River Park (Figure 16, Figure 17).

Randall's Island

Owing to the fast current that damaged colonization devices along the northern Astoria seawall shoreline, only one mobile invertebrate sample was retrieved as a control for comparison to the Randall's Island riprap and stone wall shorelines. There were only 64 mobile invertebrates collected in that seawall sample (Table 4), mainly consisting of the amphipod *A. valida* as well as five other taxa (Figure 15). In contrast, two of the mobile samples from the Randall's Island riprap shoreline had over 700 individuals, whilst the third had 177 individuals. Seven of the mobile taxa (amphipods *M. gryllotalpa* and *Jassa marmorata*, crabs *Hemigrapsus sanguineus*, shrimp *Palaemontes* spp., isopods *Idotea* sp. and *Sphaeroma quadridentatum*, and an unidentified amphipod) captured in the three colonization devices retrieved from the riprap shoreline were not captured in the lone device from the seawall control. However, none of those taxa were captured across all three devices retrieved from the riprap shoreline and all taxa except the amphipods occurred in low abundance even in colonization devices deployed at the riprap shorelines (Figure 15). Meaningful comparisons of the mobile invertebrate communities occurring on Randall's Island shorelines were not possible, given the lack of replicate samples from the seawall and stone wall shorelines, and the patchy distribution of populations of mobile invertebrates on the riprap shoreline.

West Harlem Piers

Most of the replicate mobile invertebrate samples were retrieved from the colonization devices deployed at West Harlem Piers shorelines. The mean abundance of mobile animals was higher in colonization devices from the riprap shoreline than the seawall (Table 3), but not significantly so ($t = -1.55$, $df = 7$, $P = 0.16$). Amphipods, crabs and shrimp were commonly captured on both the seawall and riprap shoreline (Figure 15). Taxon richness of mobile was similar at riprap and seawall shorelines at West Harlem Piers ($t = -0.75$, $df = 7$, $P = 0.48$, Figure 15). However, the crab *Carcinus maenas* (2 individuals) and isopod *S. quadridentatum* (4 individuals) were captured at the seawall, but not at the riprap shoreline. Also, there were significantly more of the exotic crab *H. sanguineus* captured at the riprap shoreline than the seawall shoreline ($t = -3.15$, $df = 6.8$, $P = 0.02$).

Liberty State Park

At Liberty State Park, the number of mobile invertebrates captured in colonization devices was similar at the seawall and riprap shoreline ($t = 1.58$, $df = 2.1$, $P = 0.25$), although the mean number of individuals at the seawall was over three times that at the riprap shoreline (Table 4). The taxon richness in mobile samples was also similar for seawall and riprap shorelines ($t = 2.41$, $df = 4$, $P = 0.07$, Figure 14). Most of the mobile invertebrates captured at the Liberty State Park riprap shoreline were the amphipod *A. valida* (Figure 14), which were also relatively common on the seawall. The amphipods *Corophium volutator*, *Gammarus* sp. and *Amphithoe* sp. were also relatively commonly captured on the seawall (Figure 14b). However, there was no significant difference in the abundances of those taxa between the seawall and riprap shoreline, owing to their patchy distribution across colonization devices.

Table 4: Total number of mobile invertebrates (means ± standard errors) captured in colonization devices retrieved from each hard shoreline in New York – New Jersey Harbor

Location	Shoreline type	Site code	No. of replicate mobile samples	No. of mobile invertebrates
Harlem River Park	Seawall control	1C	3	202 ± 17
	Riprap	1R	2	364 ± 184
	Gabion basket	1GB	4	298 ± 81
Randall's Island	Seawall control	2C	1	64
	Riprap	2R	3	554 ± 143
	Stone wall	2SW	1	580
West Harlem Piers	Seawall control	3C	5	143 ± 32
	Riprap	3R	4	204 ± 16
Brooklyn Bridge	Riprap	4R	2	306 ± 104
Liberty State Park	Seawall control	5C	3	275 ± 118
	Riprap	5R	3	86 ± 15

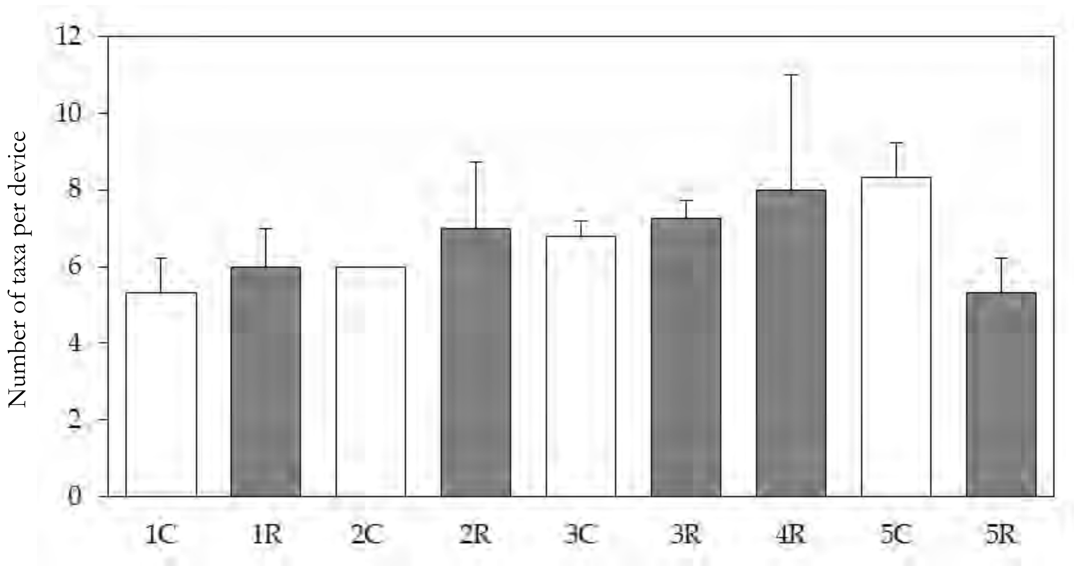


Figure 13: Taxon richness (means + standard errors) of mobile invertebrate communities captured in colonization devices deployed at seawall control (C, white bars) and riprap (R, grey bars) shorelines. Numbers in site codes on the horizontal axis indicate: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park.

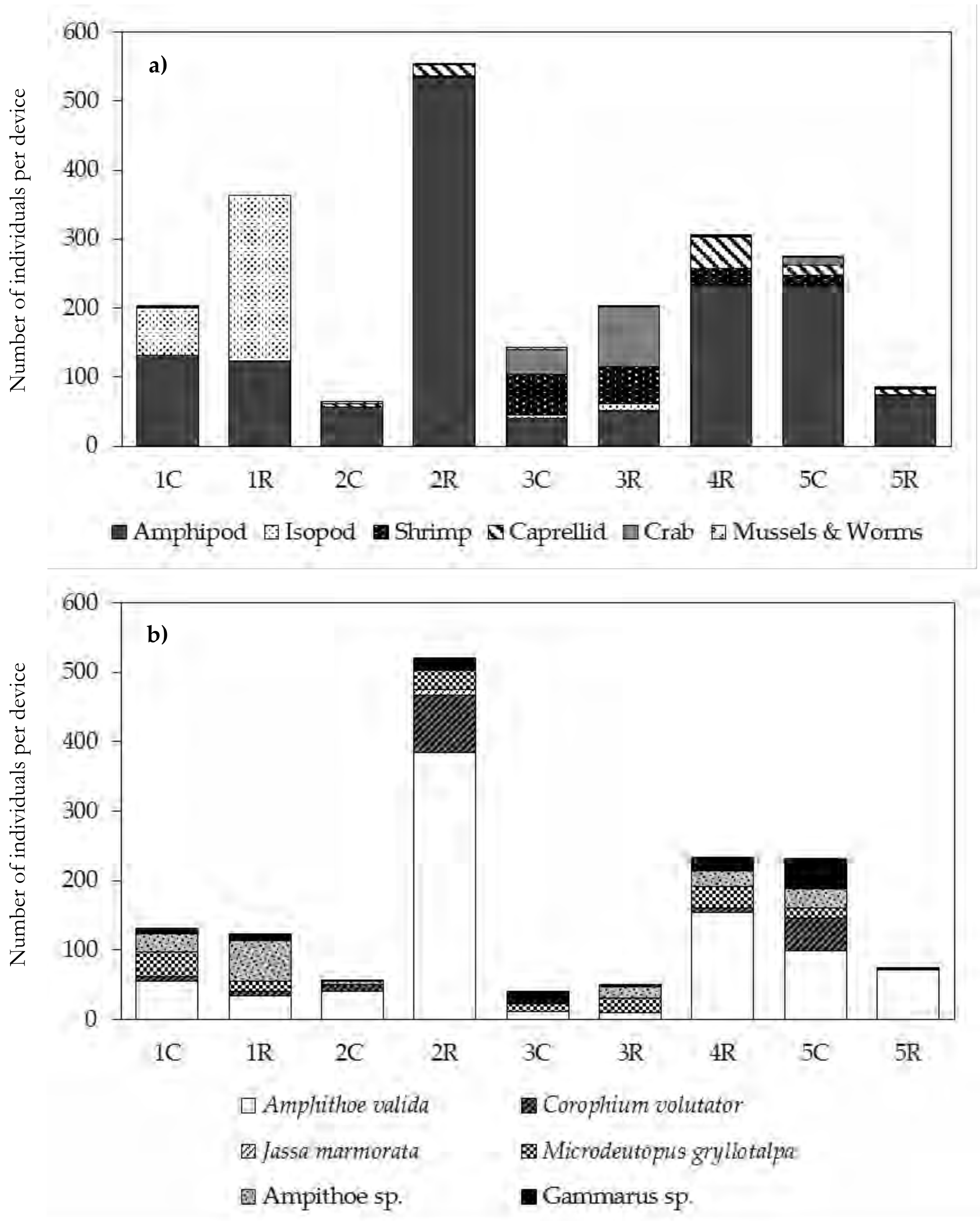


Figure 14: a) Mean numbers of mobile invertebrates and b) amphipods captured in colonization devices at seawall control (C) and riprap (R) shorelines. Numbers in site codes on the horizontal axes indicate: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park.

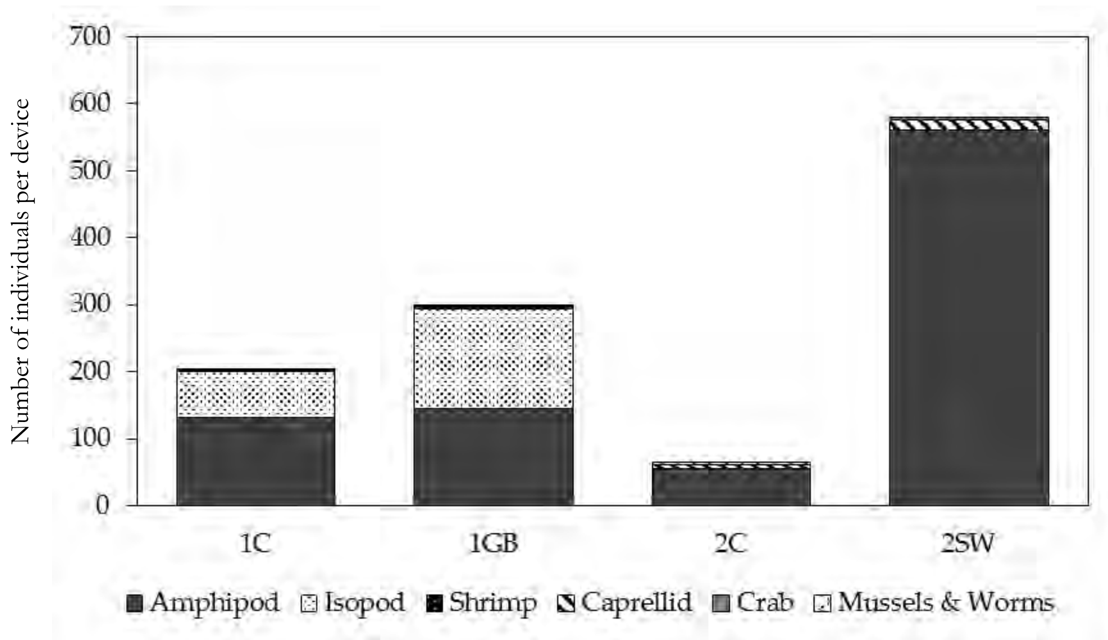


Figure 15: Mean numbers of mobile invertebrates from gabion basket shoreline at Harlem River Park (1GB), stone wall shoreline at Randall's Island (2SW) and nearby seawall controls (C).

Mobile invertebrate samples tended to be more dissimilar between locations than between shoreline types within each location (Figure 16a). However, the multidimensional scaling ordination of all replicate samples had a relatively high stress (0.15) and should be interpreted with caution. Those locations with more marine influence (i.e., Randall's Island, Brooklyn Bridge Park and Liberty State Park) appear on the left-hand side of the ordination, whilst those with more fresh water influence (i.e., Harlem River Park and West Harlem Piers) appear on the right-hand side of the ordination (Figure 16a). Multidimensional scaling based on the average community structure of mobile samples from each shoreline confirmed that the communities on shorelines within each location tended to be more similar to each other than to those on shorelines at other locations (Figure 16b), with the low stress (0.08) indicating that the ordination displays a configuration closely resembling actual dissimilarities. At the locations influenced by fresh water, the mobile communities on seawall and riprap shorelines were similar to each other (Figure 16b). There was greater dissimilarity between mobile invertebrate communities at seawall and riprap shorelines at locations with more marine influence. However, the directional shift in community structure represented in the ordination was not consistent for marine locations: riprap appears below and to the left of the seawall for Randall's Island shorelines, but riprap appears above and to the right of the seawall for Liberty State Park shorelines (Figure 17b). This reflects greater abundance and diversity of mobile invertebrates at the riprap shoreline at Randall's Island than the nearby seawall at Astoria, but the inverse occurring at Liberty State Park (Figure 15).

For quality control, 10% of the mobile invertebrate community samples were cross-checked by experienced personnel. Identifications were consistent between personnel for 97.76% of cases, with a 2.24% error rate (i.e., >97% of the mobile invertebrates in samples were identified consistently by the two personnel). To facilitate consistency in identifications, voucher collections with both specimens in vials and photographs were created (see Reid et al. 2015), personnel provided feedback when inconsistencies in identifications occurred and all taxa were identified to the taxonomic level that all personnel could consistently achieve.

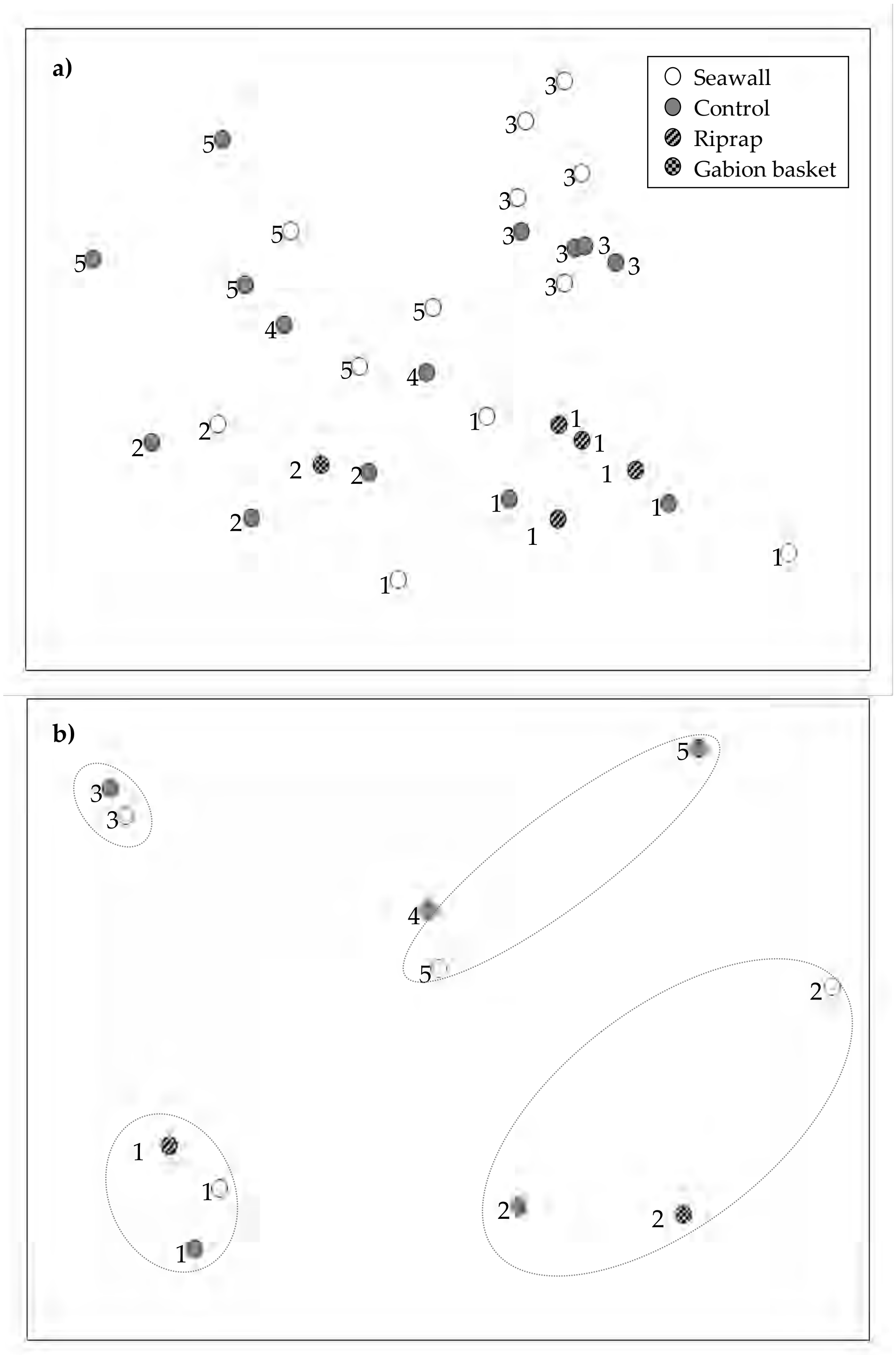


Figure 16: Multidimensional scaling ordinations of mobile invertebrate communities which colonized devices: a) all replicate samples (stress = 0.15) and b) all shorelines, based on average community structure calculated across samples from each shoreline (stress = 0.08). Numbering alongside symbols indicates the shoreline was at: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park. Lettering alongside symbols indicates gabion basket (G) and stone wall (S) shorelines. Ellipses enclose symbols representing clusters of shorelines which occurred within each location.

Sessile community assessment

Sessile communities including filamentous algae, macroalgae, ascidians, bryozoans, tube-building amphipods, barnacles, hydroids and sponges colonized acrylic, unglazed ceramic, slate and wood settlement plates (Figure 17). Only 12 identifiable taxa colonized settlement plates and the taxon richness on each plate was low, i.e. usually < 2 for each settlement plate material at each shoreline, except those at Liberty State Park. Polychaete tubes (probably species such as *Polydora ligni*) were also present in the sediment matrix on some settlement plates, but it was not possible to consistently distinguish from digital images of the plates whether tubes contained live animals, so those tubes were not included in calculation of sessile community abundances or taxon richness.

Settlement plates were often dislodged from the plastic cable ties used to attach them to cages (Figure 13, Table 2). The loss of replication precluded using univariate analyses to compare taxon richness and abundances for sessile communities from seawall and riprap shorelines in most locations. There were some contradictory patterns in the two locations where the numbers of replicate settlement plates retrieved were sufficient to allow univariate analyses. At West Harlem Piers (in the upper New York – New Jersey Harbor, with considerable fresh water influence from the Hudson River), taxon richness and algal cover were general lower on settlement plates retrieved from riprap than those from the seawall shoreline. Conversely, at Liberty State Park (in the lower New York – New Jersey Harbor, with minimal fresh water influence), taxon richness and algal cover were general higher on settlement plates retrieved from riprap than those from the seawall shoreline. At both of those locations, invertebrate cover was generally lower on settlement plates retrieved from riprap than those from the seawall shoreline.

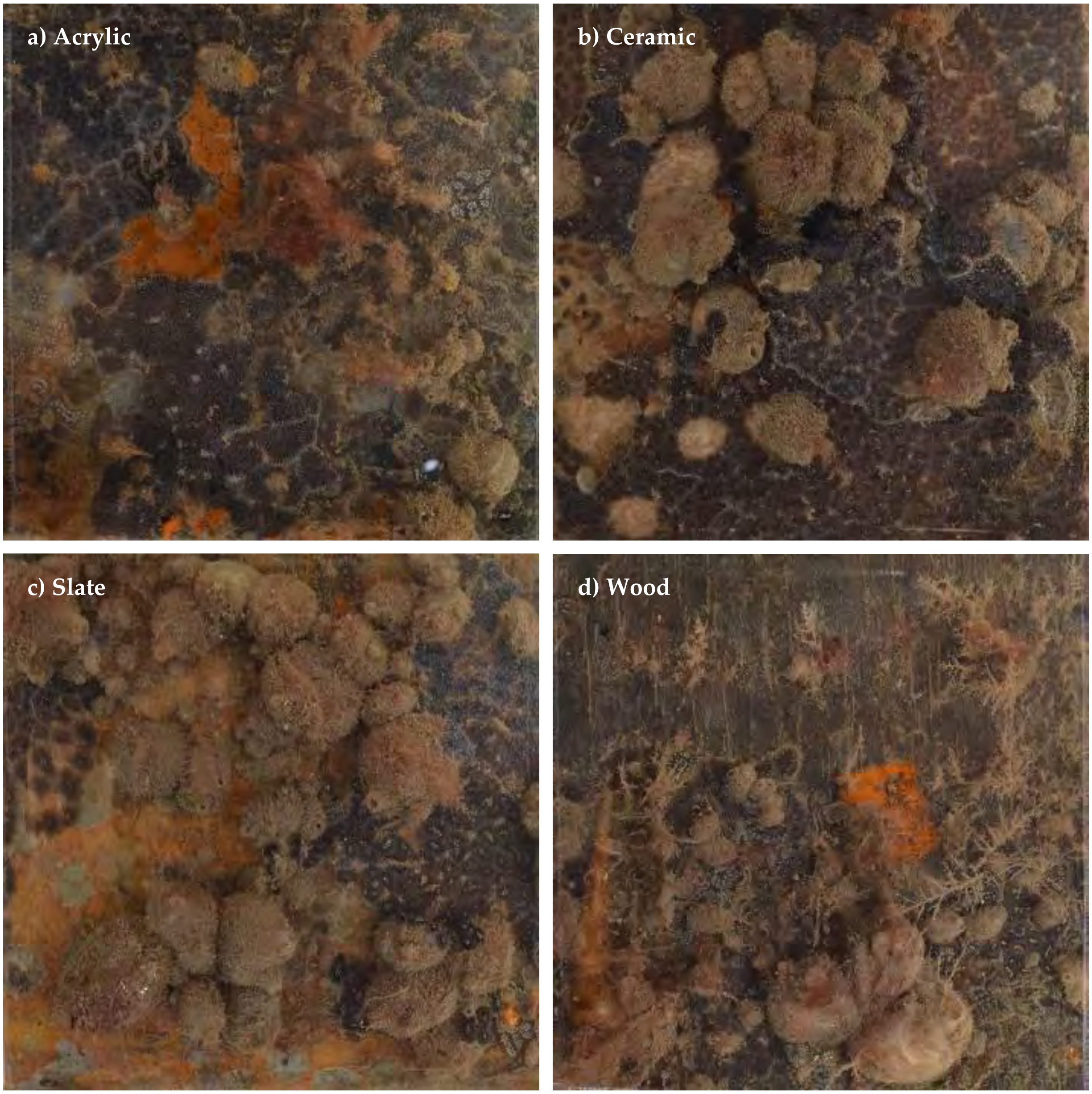


Figure 17: Images of sessile communities on settlement plates that were retrieved after deployment in the subtidal zone of the Liberty State Park seawall control shoreline (located on the southwestern side of Ellis Island) over eight weeks. Each panel shows the community occurring within a 10.5 cm x 10.5 cm area in the center of the settlement plate.

i) Acrylic settlement plates

Harlem River Park

At Harlem River Park, the only acrylic settlement plate retrieved from the riprap shoreline had not been colonized by any identifiable biota (Table 5). The loss of replication prevented meaningful comparison of sessile communities at the riprap and seawall shoreline for acrylic settlement plates, but comparisons between communities at seawall and gabion basket shorelines were possible. There was no significant difference in taxon richness on acrylic settlement plates retrieved from seawall and gabion basket shorelines ($t = -0.16$, $df = 3.27$, $P = 0.88$). Algal cover on acrylic settlement plates at Harlem River Park was highly variable: although mean algal cover from gabion basket shoreline was over double that on seawall, there was no significant difference in the algal cover between those shoreline types ($t = -1.16$, $df = 4.37$, $P = 0.30$). Green filamentous algae dominated at the seawall (Figure 19), whereas three of the four acrylic settlement plates from the gabion basket shoreline had >90% red filamentous algae cover (Figure 20). Invertebrate cover was low at both seawall and gabion basket shorelines, with no significant difference between the shoreline types ($t = 0.238$, $df = 4.73$, $P = 0.82$, Table 5). The sessile communities which colonized acrylic settlement plates from the gabion basket shoreline at Harlem River Park were more similar to those on West Harlem Piers shorelines (particularly the seawall) than those on the nearby seawall (Figure 21).

Randall's Island

Only two acrylic settlement plates were retrieved from Randall's Island shorelines: no biota occurred on 98% and 100% of the surface of settlement plates from the riprap and stone wall shorelines, respectively (Table 5, Figure 20). The remaining 2% of the acrylic settlement plate at the riprap shoreline was covered by the ascidian *Molgula manhattensis*. No acrylic settlement plates were retrieved from the seawall control for Randall's Island, which was deployed on the northern shoreline of Astoria immediately across the channel from Randall's Island.

West Harlem Piers

At West Harlem Piers, the high cover of red filamentous algal on acrylic settlement plates retrieved from the seawall was significantly higher than that on plates from the riprap shoreline (t =

10.01, df = 3.13, $P < 0.01$, Table 5, Figure 19). There were no significant differences in the low taxon richness (t = 0.49, df = 2.74, $P = 0.66$) and invertebrate cover (t = 0.11, df = 4.33, $P = 0.91$), between the seawall and riprap shoreline (Table 5). In comparison to other locations, the sessile communities on acrylic settlement plates were relatively similar between the seawall and riprap shoreline at West Harlem Piers (Figure 21).

Liberty State Park

At Liberty State Park, the taxon richness of sessile communities on acrylic settlement plates was higher than that at other locations (Table 5). Ascidians were the dominant taxon on acrylic settlement plates from the seawall and the most common invertebrates on acrylic settlement plates from the riprap shoreline, whilst red and green filamentous algae were also common on the plates from the riprap shoreline (Figure 19). There was no significant difference in the taxon richness (t = –0.52, df = 8, $P = 0.62$), but algal cover was higher (t = -4.62, df = 6.94, $P < 0.01$), and invertebrate cover was lower (t = 2.55, df = 5.76, $P = 0.045$), on acrylic settlement plates from the riprap shoreline than from seawall shoreline at Liberty State Park (Table 5, Figure 19). Despite these differences between the nearby shorelines, the community structure on acrylic settlement plates retrieved from seawall and riprap shoreline at Liberty State Park were similar, in comparison to the dissimilar community structures occurring on acrylic settlement plates retrieved from other locations (Figure 21).

Table 5: Total percent cover (means ± standard error) of sessile algae and macroinvertebrates on acrylic settlement plates deployed on hard shorelines in New York – New Jersey Harbor

Location	Shoreline type	Site code	No. of replicates	Taxon richness	Algae cover (%)	Invertebrate cover (%)
Harlem River Park	Seawall control	1C	3	1.3 ± 0.9	29.0 ± 29.0	3.3 ± 1.8
	Riprap	1R	1	0.0	0.0	0.0
	Gabion basket	1GB	4	1.5 ± 0.5	73.0 ± 24.4	2.8 ± 1.7
Randall's Island	Seawall control	2C	0	–	–	–
	Riprap	2R	1	1.0	0.0	2.0
	Stone wall	2SW	1	0.0	0.0	0.0
West Harlem Piers	Seawall control	3C	5	1.8 ± 0.4	90.2 ± 3.8	7.8 ± 4.3
	Riprap	3R	3	1.3 ± 0.9	7.3 ± 7.3	7.0 ± 5.6
Brooklyn Bridge	Riprap	4R	2	1.5 ± 0.5	37.0 ± 37.0	14.0 ± 12.0
Liberty State Park	Seawall control	5C	5	4.2 ± 0.6	9.6 ± 8.4	56.8 ± 11.5
	Riprap	5R	5	4.6 ± 0.5	56.0 ± 5.5	24.2 ± 5.6

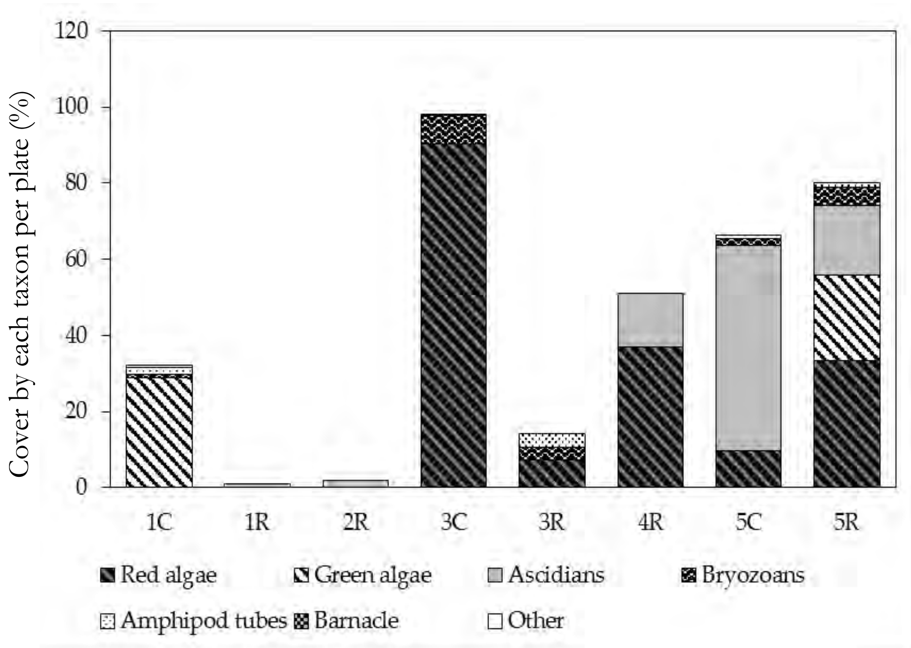


Figure 19: Mean percentage cover of sessile taxa from acrylic settlement plates at each seawall control (C) and riprap (R) shoreline. Numbers in site codes on the horizontal axis indicate: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park.

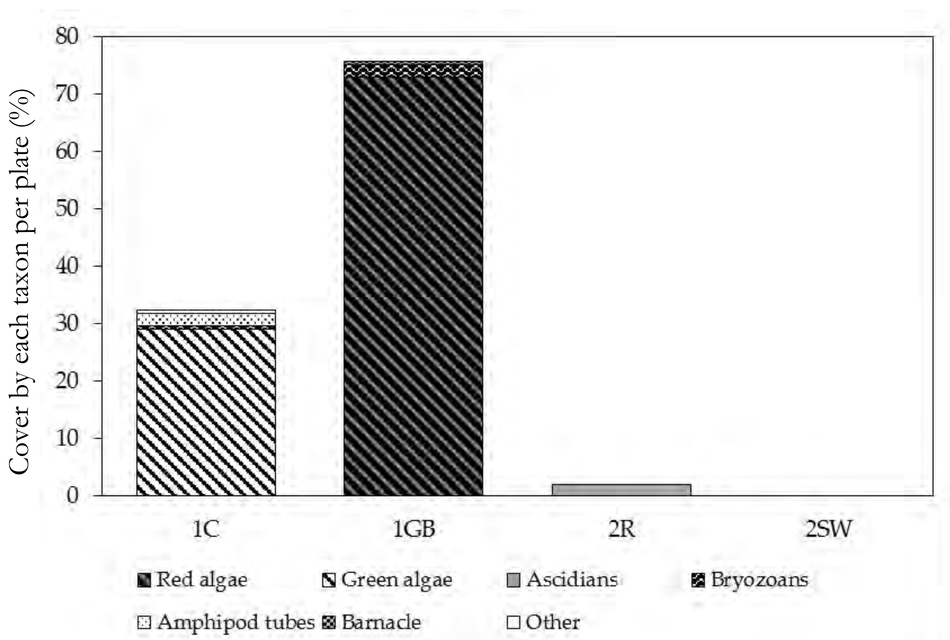


Figure 20: Mean percentage cover of sessile taxa on acrylic settlement plates from gabion basket shoreline (1GB) and nearby seawall control (1C) at Harlem River Park, and stone wall shoreline (2SW) and nearby riprap (2R) at Randall's Island. The riprap shoreline is shown for comparison, because no data were available from the seawall control at Randall's Island.

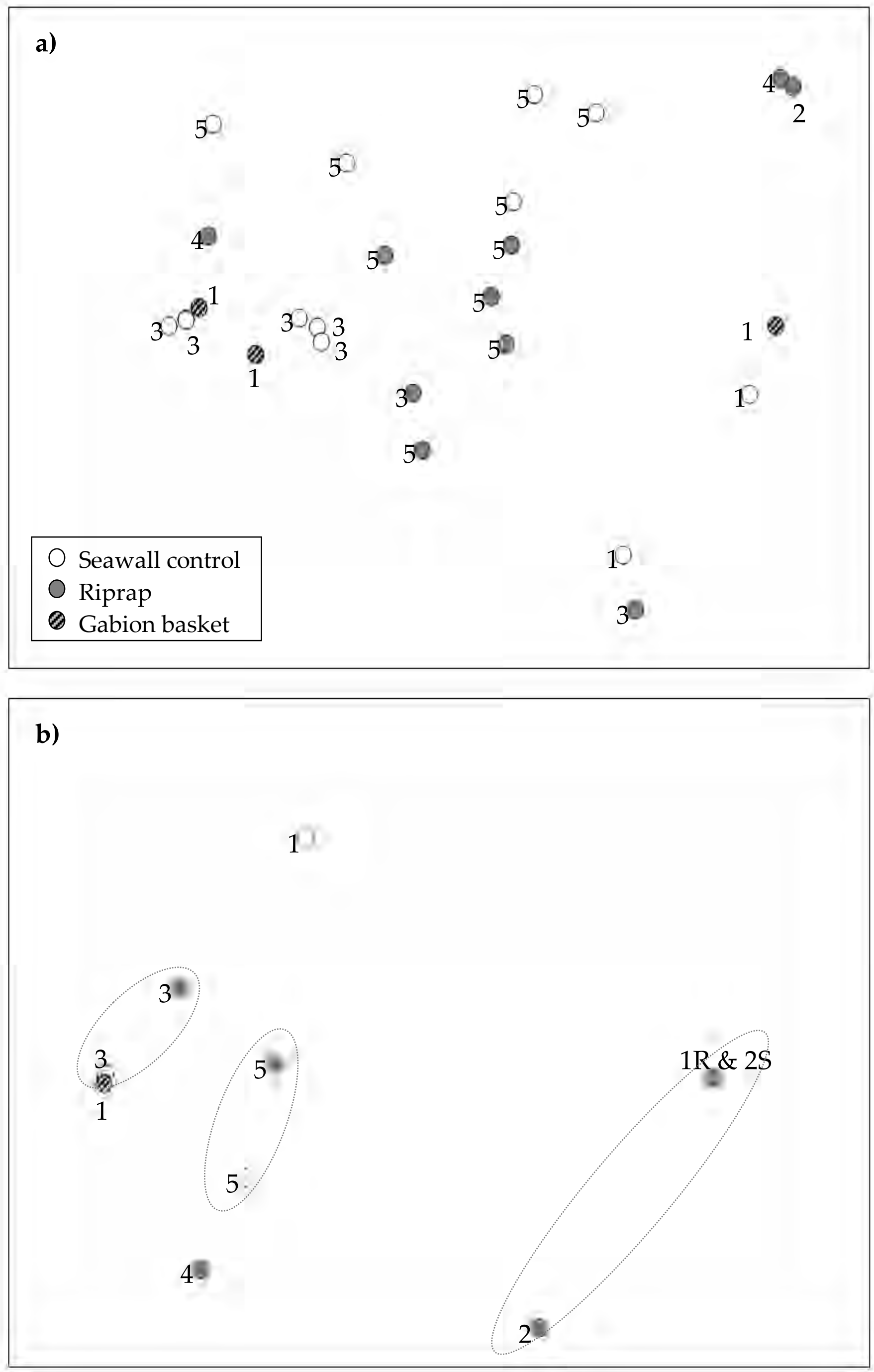


Figure 21: Multidimensional scaling ordination of communities colonizing acrylic settlement plates across shorelines: a) all replicate acrylic settlement plates (stress = 0.19) and b) all shorelines, based on average community structure across settlement plates from each shoreline (stress = 0.09). Numbering indicates the shoreline was at: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park. Lettering alongside symbols indicates gabion basket (G) and stone wall (S) shorelines. Ellipses enclose symbols representing clusters of shorelines that occurred within each location. The communities that colonized acrylic settlement plates were comparatively dissimilar between Harlem River Park shorelines and symbols representing those shorelines are not enclosed in an ellipse.

ii) Ceramic settlement plates

Harlem River Park

At Harlem River Park, the only ceramic settlement plate retrieved from the riprap shoreline had not been colonized by any identifiable biota (Table 6). The loss of replication precluded statistical analyses to compare sessile communities across shoreline types. There was 97% cover by green filamentous algae on one of the ceramic settlement plates from the seawall shoreline, but the other plate at that shoreline and the two at the gabion basket shoreline were largely devoid of identifiable biota (Figure 22, Figure 23). The community structure on ceramic settlement plates from different Harlem River Park shorelines were dissimilar to each other, in comparison to the similar communities occurring between nearby shorelines within other locations (Figure 24).

Randall's Island

No ceramic settlement plates were retrieved from the seawall shoreline on the northern Astoria shoreline (Randall's Island control) and only one ceramic settlement plate with low algae and invertebrate cover was retrieved from the riprap shoreline at Randall's Island (Table 6). One of the ceramic settlement plates retrieved from the stone wall shoreline had no identifiable biota, whilst the other had some red (17%) and green (3%) algal cover (Figure 23). The sessile community on ceramic settlement plates at Randall's Island shorelines was similar to those from other lower Harbor shorelines and West Harlem Piers shorelines (Figure 24).

West Harlem Piers

At West Harlem Piers, the high cover of red filamentous algal on ceramic settlement plates retrieved from the seawall was significantly higher than that on plates from the riprap shoreline (t = 7.59, df = 2.17, $P = 0.01$, Table 6, Figure 22). There were no significant differences in the low taxon richness (t = 0.23, df = 2.76, $P = 0.83$) and invertebrate cover (t = 0.81, df = 3.90, $P = 0.47$), between the seawall and riprap shoreline (Table 6). Community structure on ceramic settlement plates retrieved from West Harlem Pier shorelines was similar to those from lower Harbor shorelines (Figure 24).

Liberty State Park

At Liberty State Park, the taxon richness of sessile communities on ceramic settlement plates was higher than that at other locations (Table 6). Ascidians were the dominant taxon on ceramic settlement plates from the seawall and most common invertebrate taxa on plates from the riprap shoreline, whilst red filamentous algae was also common on the plates from the riprap shoreline (Figure 22). There was no significant difference in the taxon richness (t = 1.21, df = 7, $P = 0.27$) or algal cover (t = –1.63, df = 3.59, $P = 0.19$), but invertebrate cover was lower on the riprap than the seawall shoreline (t = 6.53, df = 6.77, $P < 0.01$) for ceramic settlement plates retrieved from Liberty State Park (Table 6, Figure 22). In comparison to other locations, the community structures on ceramic settlement plates retrieved from seawall and riprap shoreline at Liberty State Park were similar to each other (Figure 24).

Table 6: Total percent cover (means ± standard errors) of sessile algae and macroinvertebrates on ceramic settlement plates deployed on hard shorelines in New York – New Jersey Harbor

Location	Shoreline type	Site code	No. of replicates	Taxon richness	Algae cover (%)	Invertebrate cover (%)
Harlem River Park	Seawall control	1C	2	0.5 ± 0.5	46.5 ± 46.5	0.0
	Riprap	1R	1	0.0	0.0	0.0
	Gabion basket	1GB	2	1.5 ± 0.5	0.0	2.5 ± 0.5
Randall's Island	Seawall control	2C	0	–	–	–
	Riprap	2R	1	3.0	6.0	3.0
	Stone wall	2SW	2	1.0 ± 1.0	10.0 ± 10.0	0.0
West Harlem Piers	Seawall control	3C	4	1.5 ± 0.3	96.3 ± 2.3	2.3 ± 1.4
	Riprap	3R	3	1.3 ± 0.7	15.0 ± 9.6	1.0 ± 0.6
Brooklyn Bridge	Riprap	4R	1	2.0	70.0	0.0
Liberty State Park	Seawall control	5C	5	4.6 ± 0.9	11.0 ± 6.0	68.4 ± 6.6
	Riprap	5R	4	3.0 ± 0.9	43.5 ± 19.1	15.5 ± 4.7

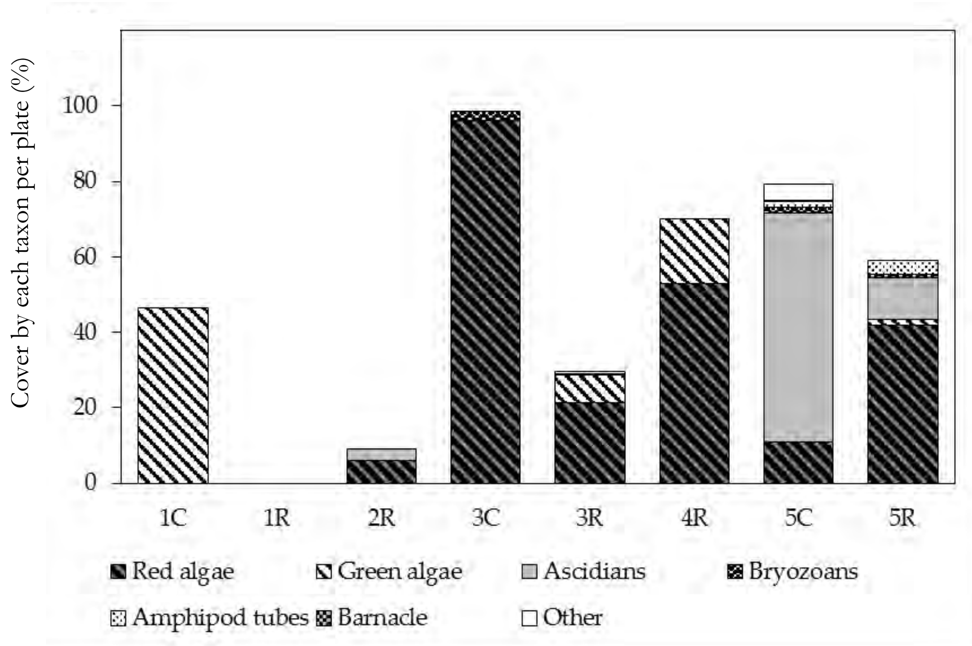


Figure 22: Mean numbers of sessile taxa from ceramic settlement plates at each seawall control (C) and riprap (R) shoreline. Numbers in site codes on the horizontal axis indicate: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park.

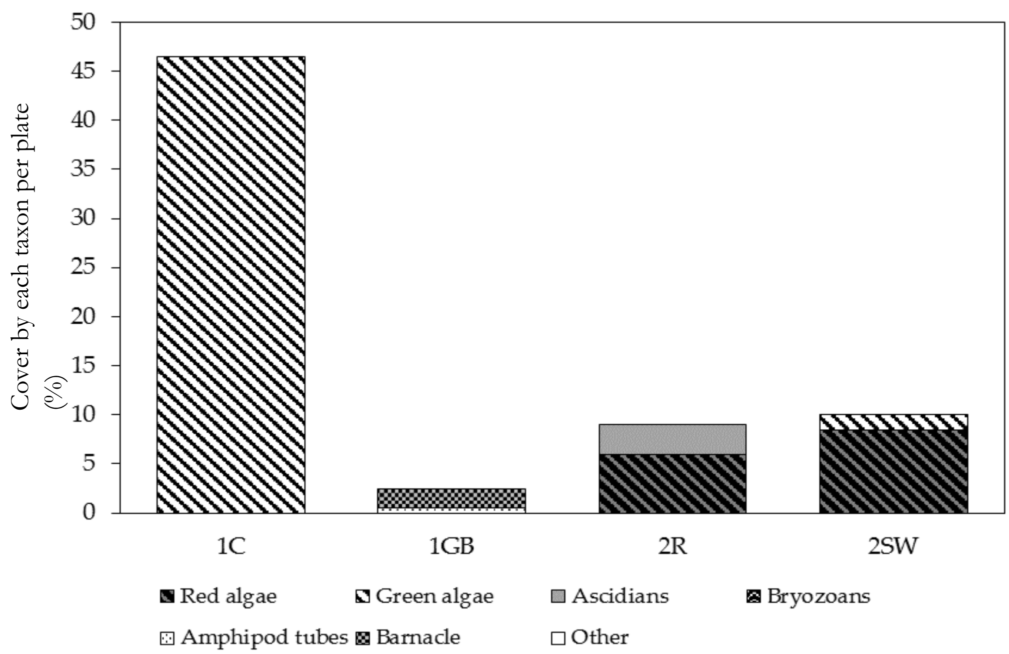


Figure 23: Mean percentage cover of sessile taxa on ceramic settlement plates from gabion basket shoreline (1GB) and nearby seawall control (1C) at Harlem River Park, and stone wall shoreline (2SW) and nearby riprap (2R) at Randall's Island. The riprap shoreline is shown for comparison, because no data were available from the seawall control at Randall's Island.

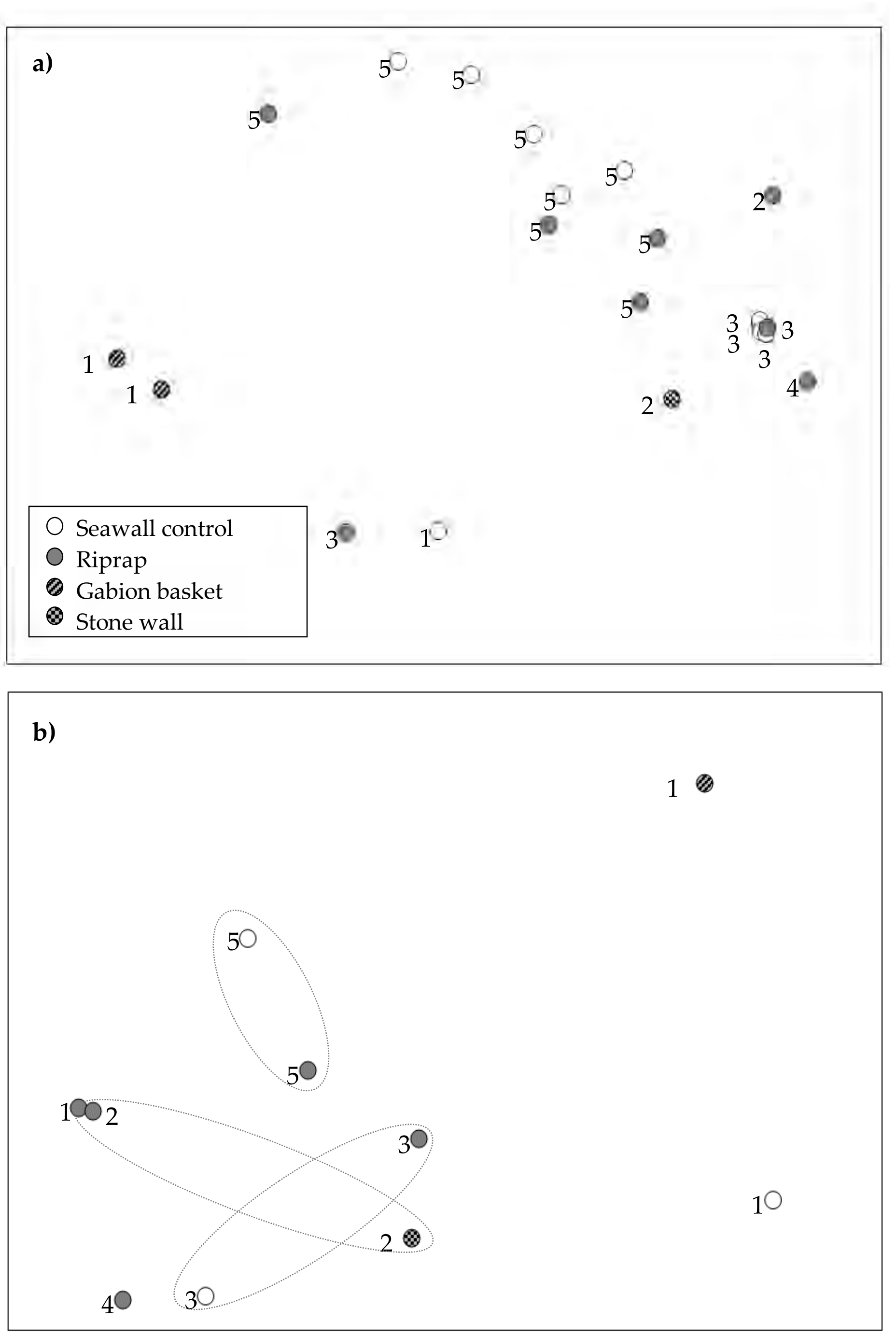


Figure 24: Multidimensional scaling ordination of communities which colonized ceramic settlement plates across shorelines: a) all replicate ceramic settlement plates (stress = 0.14) and b) all shorelines, based on average community structure across ceramic settlement plates from each shoreline (stress = 0.08). Numbering indicates the shoreline was at: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park. Lettering alongside symbols indicates gabion basket (G) and stone wall (S) shorelines. Ellipses enclose symbols representing clusters of shorelines which occurred within each location. The communities that colonized ceramic settlement plates were comparatively dissimilar between Harlem River Park shorelines and symbols representing those shorelines are not enclosed in an ellipse.

iii) Slate settlement plates

Harlem River Park

At Harlem River Park, the only slate settlement plate retrieved from the riprap shoreline had not been colonized by any identifiable biota (Table 7). The cover of sessile communities on slate settlement plates retrieved from the seawall and gabion basket shorelines also was sparse (i.e., ≤ 3%, Figure 26). The sparsely colonized slate settlement plates retrieved from shorelines at Harlem River Park were dissimilar to most of the communities on slate settlement plates from West Harlem Piers and Liberty State Park, although some of the slate settlement plates from those locations also had few colonists (Figure 27a).

Randall's Island

No slate settlement plates were retrieved from any shoreline at Randall's Island (Table 7).

West Harlem Piers

At West Harlem Piers, there was consistently high cover (i.e., ≥ 95%) by filamentous red algae and low cover by *Membranipora membranacea* on the slate settlement plates retrieved from the seawall (Table 7, Figure 25). The only slate settlement plate recovered from the riprap shoreline had low cover by red filamentous algae, green macroalgae and tube-building amphipods (Table 7).

Liberty State Park

At Liberty State Park, the taxon richness of sessile communities on slate settlement plates was higher than that at other locations, with higher taxon richness on the riprap shoreline than the seawall shoreline ($t = -2.94$, $df = 5.06$, $P = 0.03$, Table 7). On seawall, the most common taxa on slate settlement plates were ascidians, red filamentous algae and barnacles (Figure 25). Red and green filamentous algae, ascidians and tube-building amphipods were all common on slate settlement plates retrieved from the riprap shoreline (Figure 25). Algal cover was higher ($t = -2.46$, df = 6.41, $P = 0.047$) and invertebrate cover was lower ($t = 2.71$, df = 6.99, $P = 0.03$) on slate settlement plates retrieved from the riprap shoreline than those on the seawall shoreline (Table 7). The communities on slate settlement plates retrieved from seawall and riprap shorelines at Liberty State Park were dissimilar to each other, compared to communities on analogous shorelines at West Harlem Piers (Figure 27). That pattern was owing to large differences in algal cover but not invertebrates at West Harlem Piers, compared to the significant differences in both algae and invertebrate cover at Liberty State Park (Figure 25).

Table 7: Total percent cover (means ± standard errors) of sessile algae and macroinvertebrates on slate settlement plates deployed on hard shorelines in New York – New Jersey Harbor

Location	Shoreline type	Site code	No. of replicates	Taxon richness	Algae cover (%)	Invertebrate cover (%)
Harlem River Park	Seawall control	1C	2	1.0 ± 0.0	2.5 ± 2.5	1.5 ± 1.5
	Riprap	1R	1	0.0	0.0	0.0
	Gabion basket	1GB	2	1.0 ± 1.0	0.0	1.5 ± 1.5
Randall's Island	Seawall control	2C	0	-	-	-
	Riprap	2R	0	-	-	-
	Stone wall	2SW	0	-	-	-
West Harlem Piers	Seawall control	3C	5	1.6 ± 0.2	97.6 ± 0.9	1.4 ± 0.7
	Riprap	3R	1	3.0	6.0	1.0
Brooklyn Bridge	Riprap	4R	0	-	-	-
Liberty State Park	Seawall control	5C	5	2.6 ± 0.2	18.2 ± 13.9	65.8 ± 11.4
	Riprap	5R	4	4.0 ± 0.4	58.5 ± 8.7	25.5 ± 9.5

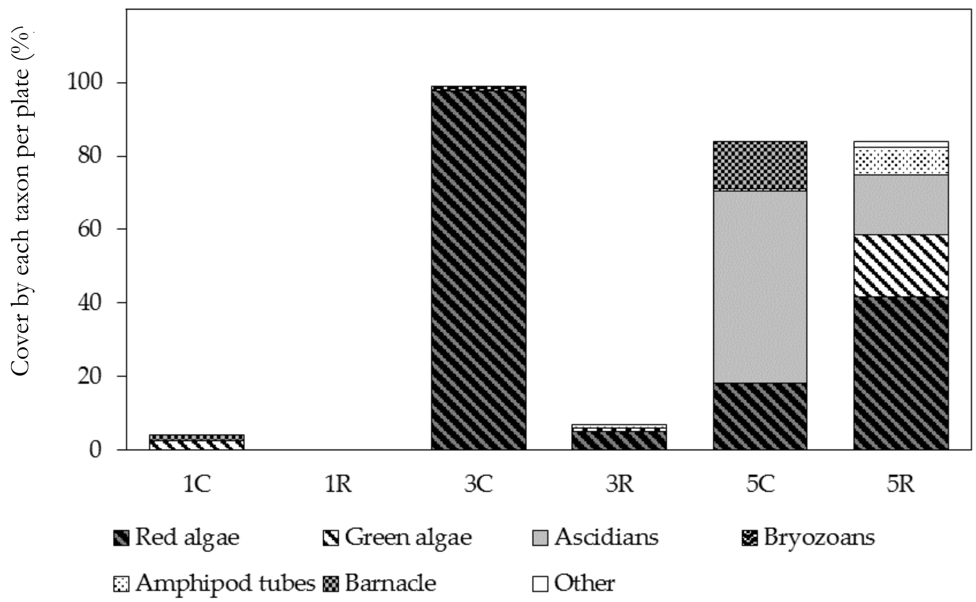


Figure 25: Mean numbers of sessile taxa from slate settlement plates at each seawall control (C) and riprap (R) shoreline. Numbers in site codes on the horizontal axis indicate: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park.

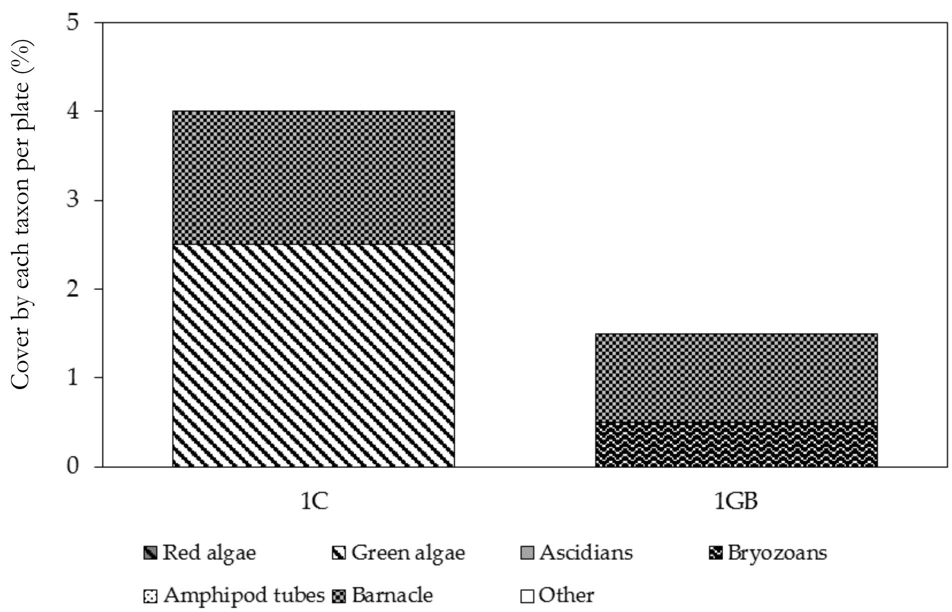


Figure 26: Mean percentage cover of sessile taxa on slate settlement plates from gabion basket shoreline (1GB) and nearby seawall control (1C) at Harlem River Park.

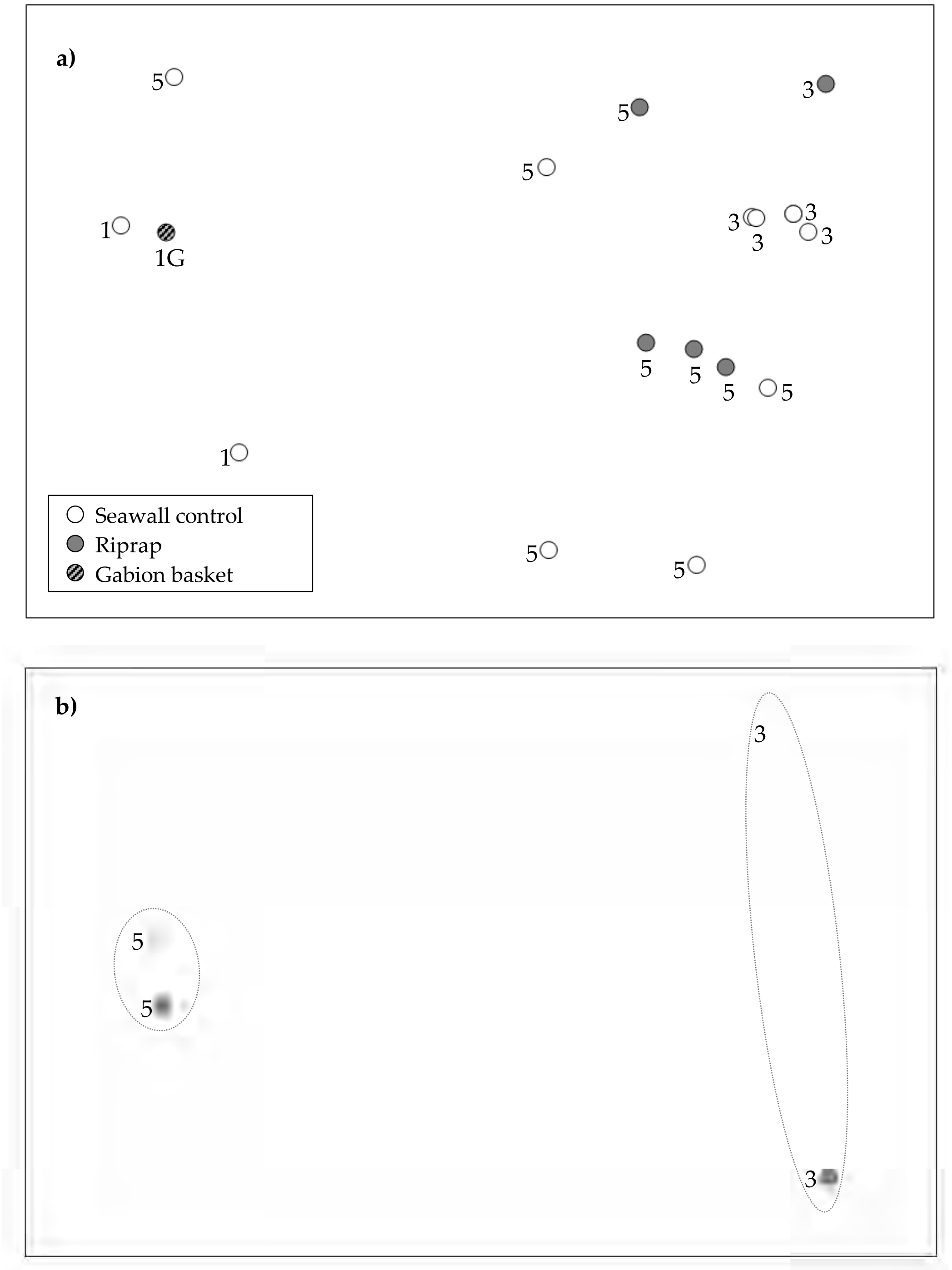


Figure 27: **Multidimensional scaling ordination of communities that colonized slate settlement plates across shorelines: a) all replicate settlement plates (stress = 0.18) and b) West Harlem Piers and Liberty State Park shorelines, based on average community structure across slate settlement plates from each shoreline (stress < 0.01). Numbering indicates the shoreline was at: 1) Harlem River Park; 3) West Harlem Piers; and, 5) Liberty State Park. Randall's Island shorelines were no included in the lower ordination, as slate settlement plates were sparsely colonized at that location. Lettering alongside symbols indicates gabion basket (G) shorelines. Ellipses enclose symbols representing clusters of shorelines that occurred within each location.**

iv) Wood settlement plates

Harlem River Park

At Harlem River Park, the lone wood settlement plate retrieved from the riprap shoreline had been colonized only by one pink sponge (Table 8). The loss of replication prevented meaningful comparison of sessile communities on wood settlement plates from the riprap and seawall shoreline, but comparisons between communities from seawall and gabion basket shorelines were possible (Figure 29). There was no significant difference in taxon richness on wood settlement plates retrieved from seawall and gabion basket shorelines ($t = -1.00$, $df = 4$, $P = 0.37$). Algal cover on wood settlement plates from the gabion basket shoreline was highly variable and although there was no algal cover on any plate from the seawall, there was no significant difference in the algal cover between those shoreline types ($t = -1.00$, $df = 2$, $P = 0.42$). Invertebrate cover was low at both seawall and gabion basket shorelines, with no significant difference between the shoreline types ($t = 0.22$, $df = 4$, $P = 0.83$, Table 8). The sessile communities which colonized wood settlement plates from the gabion basket shoreline at Harlem River Park were more similar to those on the West Harlem Piers seawall than the nearby seawall (Figure 30).

Randall's Island

No wood settlement plates were retrieved from the seawall on the northern Astoria shoreline, which was the control for Randall's Island shorelines (Table 8). The communities on wood settlement plates from Randall's Island were highly variable within each shoreline type. From the riprap shoreline, one of the wood settlement plates had 12% cover by ascidians whilst the other had 27% cover by green filamentous algae, neither having any other identifiable biota. From the stone wall shoreline, one wood settlement plate had no identifiable biota, whilst the other had 98% red filamentous algal cover (Figure 29). Using average values for each shoreline, the communities from wood settlement plates from Randall's Island riprap and stone wall shorelines were dissimilar from each other, compared to the similarity in community structure for nearby shorelines in other locations (Figure 30).

West Harlem Piers

At West Harlem Piers, the high cover of algae on wood settlement plates retrieved from the seawall was significantly higher than that on plates from the riprap shoreline ($t = 3.84$, $df = 4.09$, $P =$

0.02, Table 8, Figure 28). Invertebrate cover was not significantly different on wood settlement plates retrieved from seawall and riprap shorelines ($t = 2.48$, $df = 4$, $P = 0.07$), despite no invertebrates colonizing those plates retrieved from the riprap shoreline (Table 8). There were no significant differences in the low taxon richness on wood settlement plates retrieved from seawall and riprap shorelines ($t = 2.20$, $df = 2.37$, $P = 0.14$). The high algal cover for the seawall and sparse communities for the riprap shoreline contributed to dissimilarity between the communities on wood settlement plates from these shorelines at West Harlem Piers (Figure 30).

Liberty State Park

At Liberty State Park, the taxon richness of sessile communities on wood settlement plates was higher than that at other locations (Table 8). Ascidians and red algae were the dominant taxa on wood settlement plates from the seawall, and these taxa plus green filamentous algae were common on the plates from the riprap shoreline (Figure 28). There were no significant differences in taxon richness ($t = 1.84$, $df = 6.53$, $P = 0.11$) or invertebrate cover ($t = 1.90$, $df = 4.86$, $P = 0.12$), but algal cover was higher ($t = –6.78$, $df = 7.69$, $P < 0.01$) on wood settlement plates from the riprap shoreline than from the seawall shoreline at Liberty State Park (Table 8, Figure 28). Despite these differences between the nearby shorelines, the community structure on wood settlement plates retrieved from seawall and riprap shoreline at Liberty State Park were similar, in comparison to the dissimilar community structures occurring on wood settlement plates retrieved from other locations (Figure 30).

Table 8: Total percent cover (means ± standard errors) of sessile algae and macroinvertebrates on wood settlement plates deployed on hard shorelines in New York – New Jersey Harbor

Location	Shoreline type	Site code	No. of replicates	Taxon richness	Algae cover (%)	Invertebrate cover (%)
Harlem River Park	Seawall control	1C	3	2.0 ± 0.0	0.0	9.3 ± 3.2
	Riprap	1R	1	1.0	0.0	1.0
	Gabion basket	1GB	3	2.3 ± 0.3	28.7 ± 28.7	8.3 ± 3.2
Randall's Island	Seawall control	2C	0	-	-	-
	Riprap	2R	2	1.5 ± 0.5	13.5 ± 13.5	6.0 ± 6.0
	Stone wall	2SW	2	0.5 ± 0.5	49.0 ± 49.0	0.0
West Harlem Piers	Seawall control	3C	5	2.2 ± 0.2	73.8 ± 18.6	6.6 ± 2.7
	Riprap	3R	3	0.7 ± 0.7	2.0 ± 2.0	0.0
Brooklyn Bridge	Riprap	4R	2	1.0 ± 0.0	30.0 ± 30.0	1.5 ± 1.5
Liberty State Park	Seawall control	5C	5	4.6 ± 0.7	21.2 ± 5.7	31.0 ± 10.1
	Riprap	5R	5	3.0 ± 0.4	71.4 ± 4.7	10.8 ± 3.3

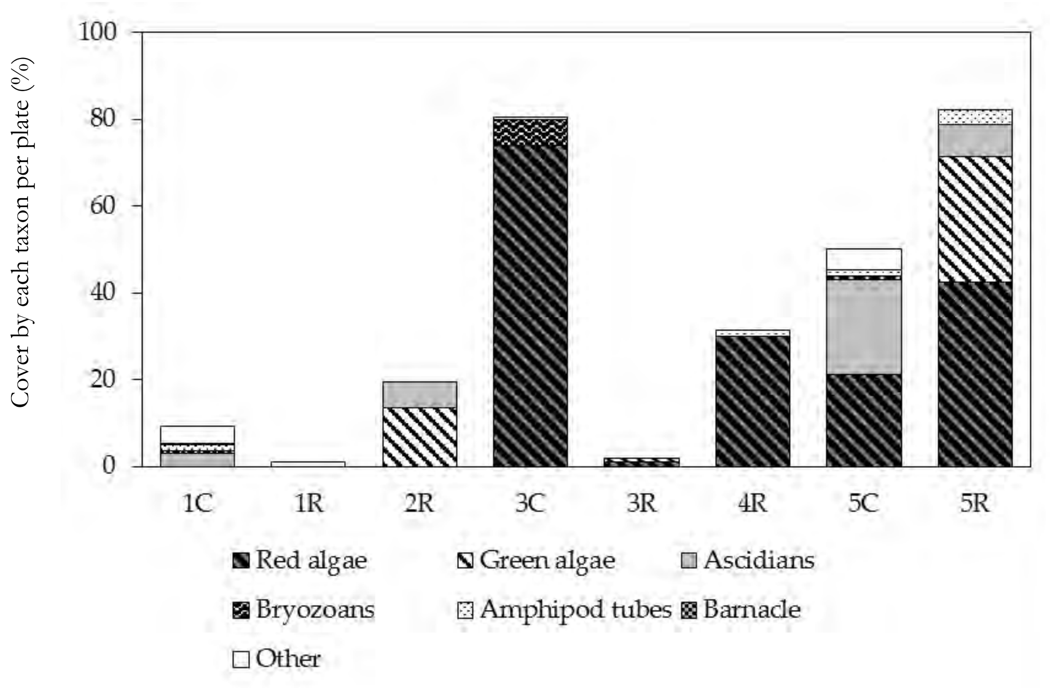


Figure 28: Mean numbers of sessile taxa from wood settlement plates at each seawall control (C) and riprap (R) shoreline. Numbers in site codes on the horizontal axis indicate: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park.

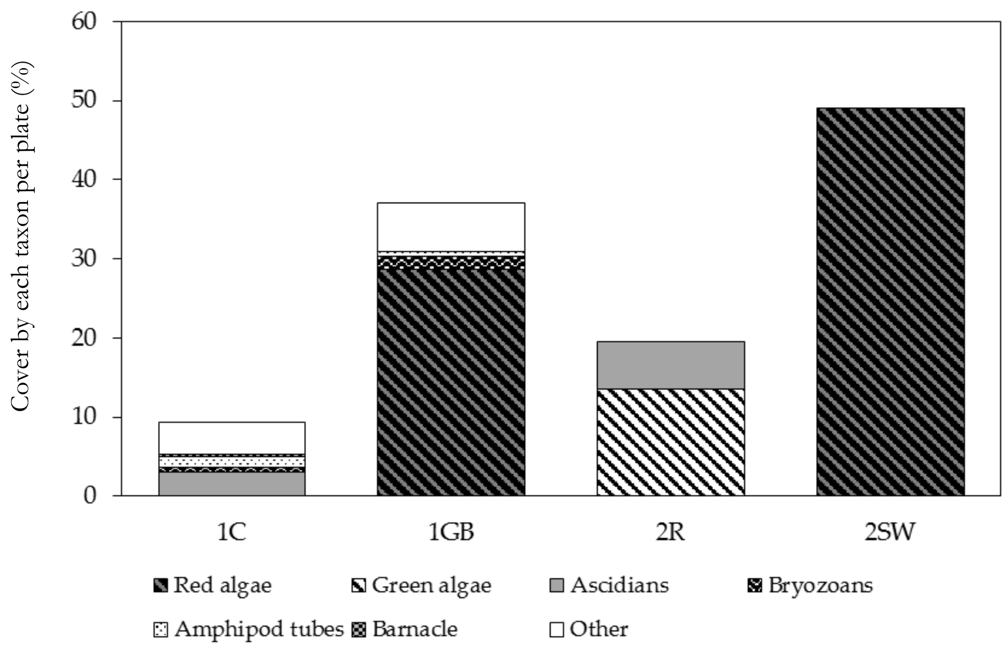


Figure 29: Mean percentage cover of sessile taxa on wood settlement plates from gabion basket shoreline (1GB) and nearby seawall control (1C) at Harlem River Park, and stone wall shoreline (2SW) and nearby riprap (2R) at Randall's Island. The riprap shoreline is shown for comparison, because no data were available from the seawall control at Randall's Island.

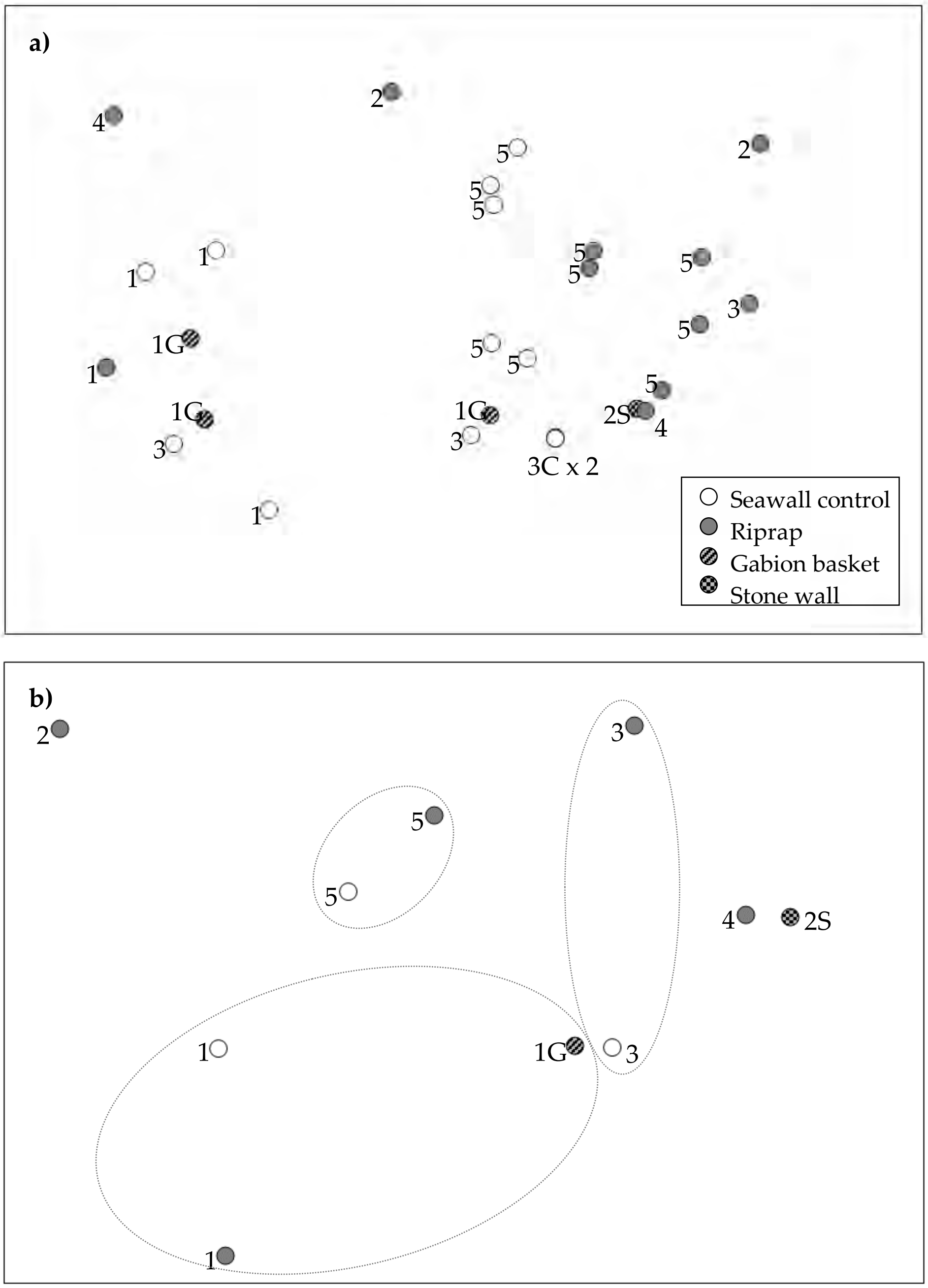


Figure 30: Multidimensional scaling ordination of communities that colonized wood settlement plates across shorelines: a) all replicate wood settlement plates (stress = 0.18) and b) all shorelines, based on average community structure across wood settlement plates from each shoreline (stress = 0.11). Numbering indicates the shoreline was at: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park. Lettering alongside symbols indicates gabion basket (G) and stone wall (S) shorelines. Ellipses enclose symbols representing clusters of shorelines that occurred within each location. The communities that colonized wood settlement plates were comparatively dissimilar between Randall's Island shorelines and symbols representing those shorelines are not enclosed in an ellipse.

Multidimensional scaling ordinations demonstrated that the structures of sessile communities were consistently more dissimilar between shorelines than between different settlement plate materials deployed within each location, for both fresh water-influenced and predominantly marine-influenced locations (Figure 31).

For quality control, 20% of the sessile community identifications were cross-checked by experienced personnel (i.e., 22 of the 107 photographs of communities on settlement plates were assessed by two personnel). Identifications were consistent between personnel for 97.1% of cases (i.e., 2136 of the 2200 points that were assessed on photographs were identified consistently by the two personnel). To facilitate consistency in identifications, photographic guides were created (see Reid et al. 2015), personnel provided feedback when inconsistencies in identifications occurred and all taxa were identified to the taxonomic level that all personnel could consistently achieve.

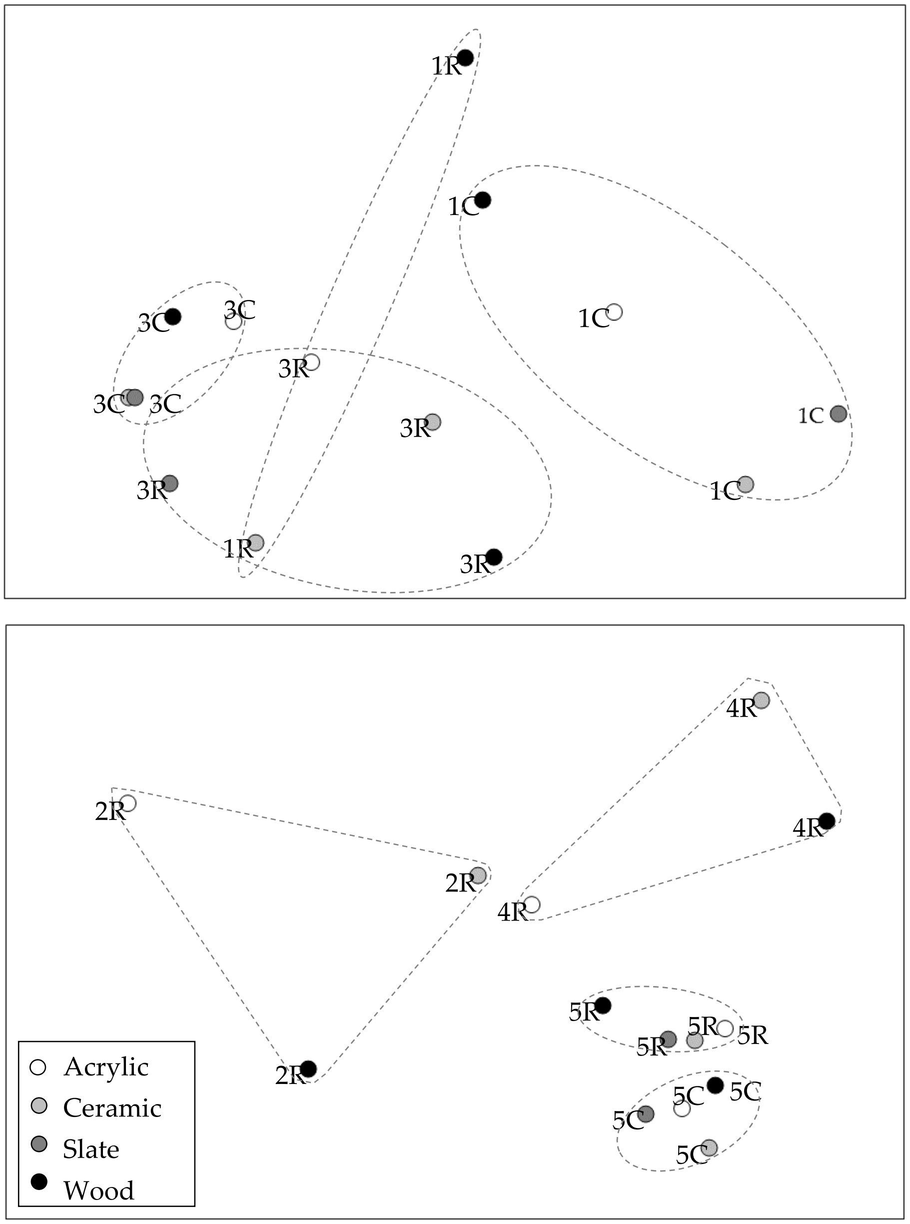


Figure 31: Multidimensional scaling ordination of communities that colonized settlement plates made from different materials (i.e., acrylic, unglazed ceramic, slate and wood) across shorelines with (a) fresh water influence (stress = 0.12) and (b) predominantly marine influenced (stress = 0.10). Numbering indicates the shoreline was at: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park. Lettering alongside symbols indicates seawall control (C) and riprap (R) shorelines. Dashed lines enclose symbols representing the communities occurring on different settlement plate materials within each shoreline.

Colonization of devices after four weeks

Colonization devices were deployed at West Harlem Piers shorelines over four weeks to examine whether this timing was sufficient to allow establishment of mobile and sessile communities that could be used in assessing relative habitat values of hard shorelines. Devices that were deployed for four and eight weeks were retrieved from the shoreline on the same day, 7 August, 2014. At the seawall shoreline, the mean abundance of mobile invertebrates in each colonization device was higher for devices deployed for four weeks than it was for devices deployed for eight weeks, largely owing to particularly high abundances of shrimp in devices deployed for four weeks (Figure 31). The amphipod *Corophium volutator* was the only mobile taxon in colonization devices deployed for four weeks that was not present in the colonization devices deployed for eight weeks, whereas *Carcinus maenas* (invasive crab), *Geukensia demissa* (mussel) and polychaete worms were present in colonization devices deployed for eight weeks but not those deployed for four weeks.

At the riprap shoreline, the mean abundance of mobile invertebrates in each colonization device was lower in devices deployed for four weeks than it was in devices deployed for eight weeks (Figure 31). As for the seawall shoreline, *C. volutator* was the only mobile taxon present in colonization devices deployed for four weeks that wasn't present in the colonization devices deployed for eight weeks. *Geukensia demissa* was present in colonization devices deployed for eight weeks, but not those deployed for four weeks.

At the seawall shoreline, settlement plates deployed for four weeks had many more tube-building amphipods than plates deployed for eight weeks (Figure 32a, Figure 33). Much more red filamentous algae had grown on settlement plates deployed for eight weeks than on those deployed for four weeks. Some bryozoans had also become established on those plates deployed for eight weeks, but were rarely identified on plates deployed for four weeks (Figure 32a). At the riprap shoreline, the differences in the structure of sessile communities from settlement plates deployed for four weeks or eight weeks were not as marked as those differences occurring for plates from seawall (Figure 32b). Cover of settlement plates that were deployed on riprap shoreline for four weeks were generally low, except for >40% cover by red filamentous algae on ceramic plates. Settlement plates deployed for eight weeks on riprap shorelines tended to have more filamentous algae than those plates deployed for four weeks on the same shoreline, but far less algal cover than that on settlement plates deployed for eight weeks on the seawall (Figure 32b).

There were more amphipods and crabs in colonization devices from riprap shoreline than in devices from seawall, regardless of whether the devices were deployed for four or eight weeks (Table 8). However, whether more or less isopods, shrimps, mussels and red algae had colonized the devices deployed on riprap shorelines than devices deployed on seawalls depended upon the duration that the devices had been deployed (Table 8).

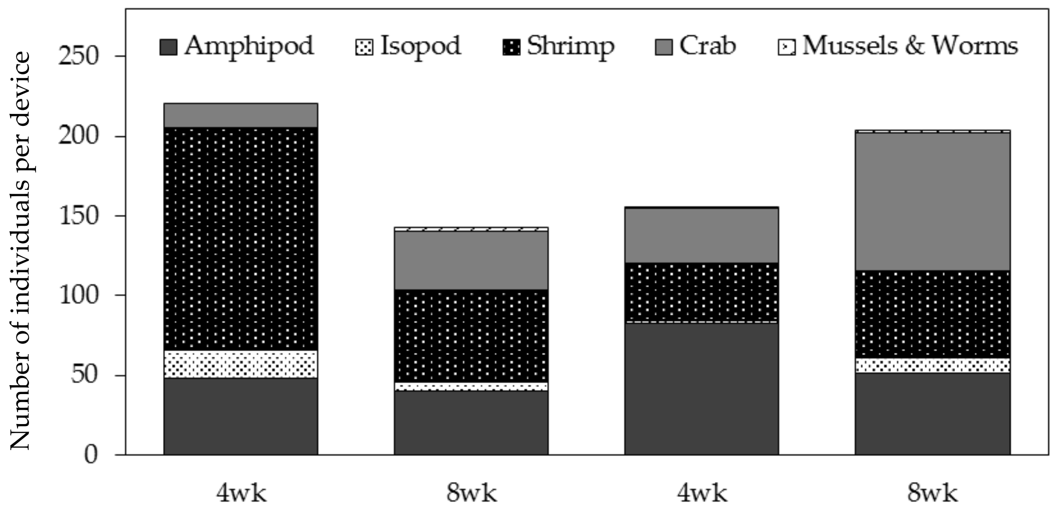


Figure 32: Mean numbers of mobile invertebrate taxa captured in colonization devices deployed at West Harlem Piers seawall control (left) and riprap shorelines (right) over four weeks or eight weeks.

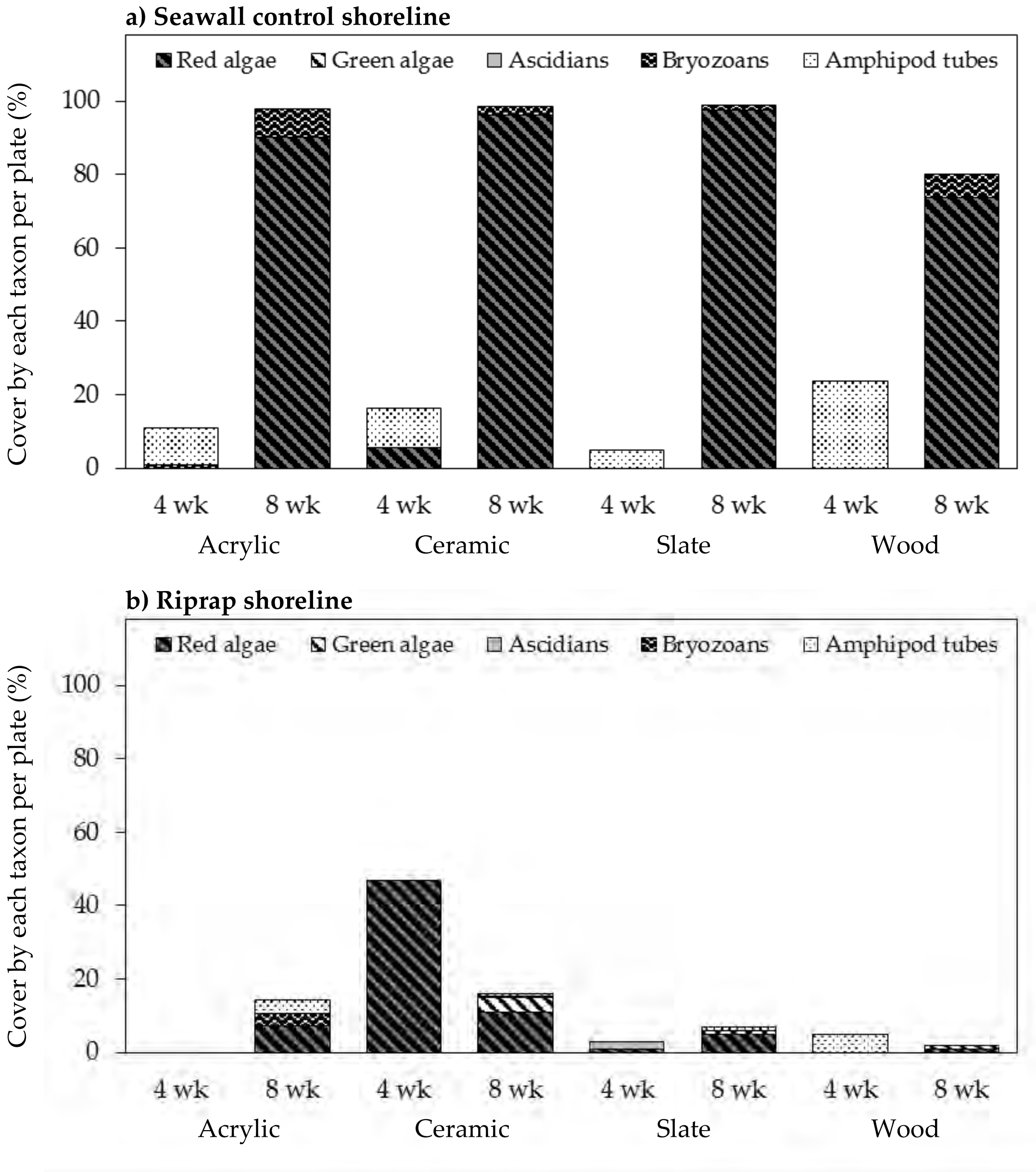


Figure 33: Mean numbers of taxa captured on acrylic, unglazed ceramic, slate or wood settlement plates deployed at a) seawall control or b) riprap shoreline at West Harlem Piers over four weeks or eight weeks.

Figure 34: Images of sessile communities on a) acrylic, b) unglazed ceramic and c) wood settlement plates that were deployed in the subtidal zone of the West Harlem Piers seawall shoreline for either four or eight weeks. All settlement plates were retrieved on the same day in summer. The surface area shown in each panel was taken from 10.5 cm x 10.5 cm of each settlement plate.

Table 9: Differences in the relative abundances of different taxa from colonization devices deployed on seawall and riprap shorelines at West Harlem Piers for either four or eight weeks. All samples were retrieved on the same day. Symbols indicate whether the abundance of each taxa in devices from the shoreline on the left of symbol was at least double (>>>), more than 150% (>>), more than 110% (>) or approximately equal to (=) that from devices deployed for the same duration on the shoreline to the right of symbol.

	Four weeks	Eight weeks	General pattern same at four and eight weeks
Amphipod	Riprap >> Seawall	Riprap > Seawall	Yes
Isopod	Seawall >>> Riprap	Riprap >> Seawall	No
Shrimp	Seawall >>> Riprap	Seawall = Riprap	No
Crab	Riprap >> Seawall	Riprap >> Seawall	Yes
Mussels	None present	Seawall > Riprap	No
Red algae*	Riprap >>> Seawall	Seawall >>> Riprap	No

*All other sessile taxa had relatively low abundances after both four and eight weeks.

Bivalve assessment

Across all shorelines there were low numbers of bivalves per scouring pad, i.e. only 24 bivalves had colonized the 31 scouring pads retrieved from colonization devices (Figure 34). Rather than mainly facilitating colonization by bivalves, the communities in scouring pads were dominated by amphipods and polychaete worms. The low abundances of bivalves contributed to contradictory patterns for comparisons of bivalve abundance between seawalls and riprap shoreline: bivalves were more common in scouring pads from seawalls than riprap shorelines at Harlem River Park and West Harlem Piers, but less common in scouring pads from seawalls than riprap shorelines at Randall's Island and Liberty State Park.

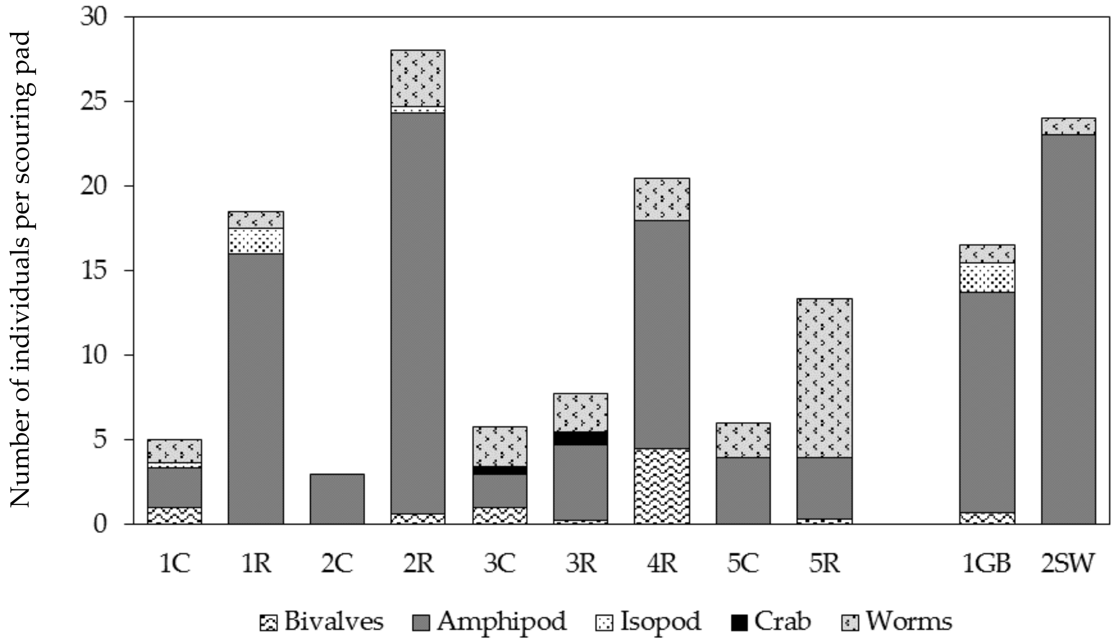


Figure 35: Mean numbers of individuals of different invertebrate taxa captured in scouring pads deployed at seawall control (C), riprap (R), gabion basket (GB) and stone wall (SW) shorelines. Numbers in site codes on the horizontal axes indicate: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park.

Photoquadrats of shorelines

Photoquadrats of shorelines were discontinued as they were not of consistently high enough resolution to enable reliable identification of communities, owing to the high turbidity of water (Figure 36).

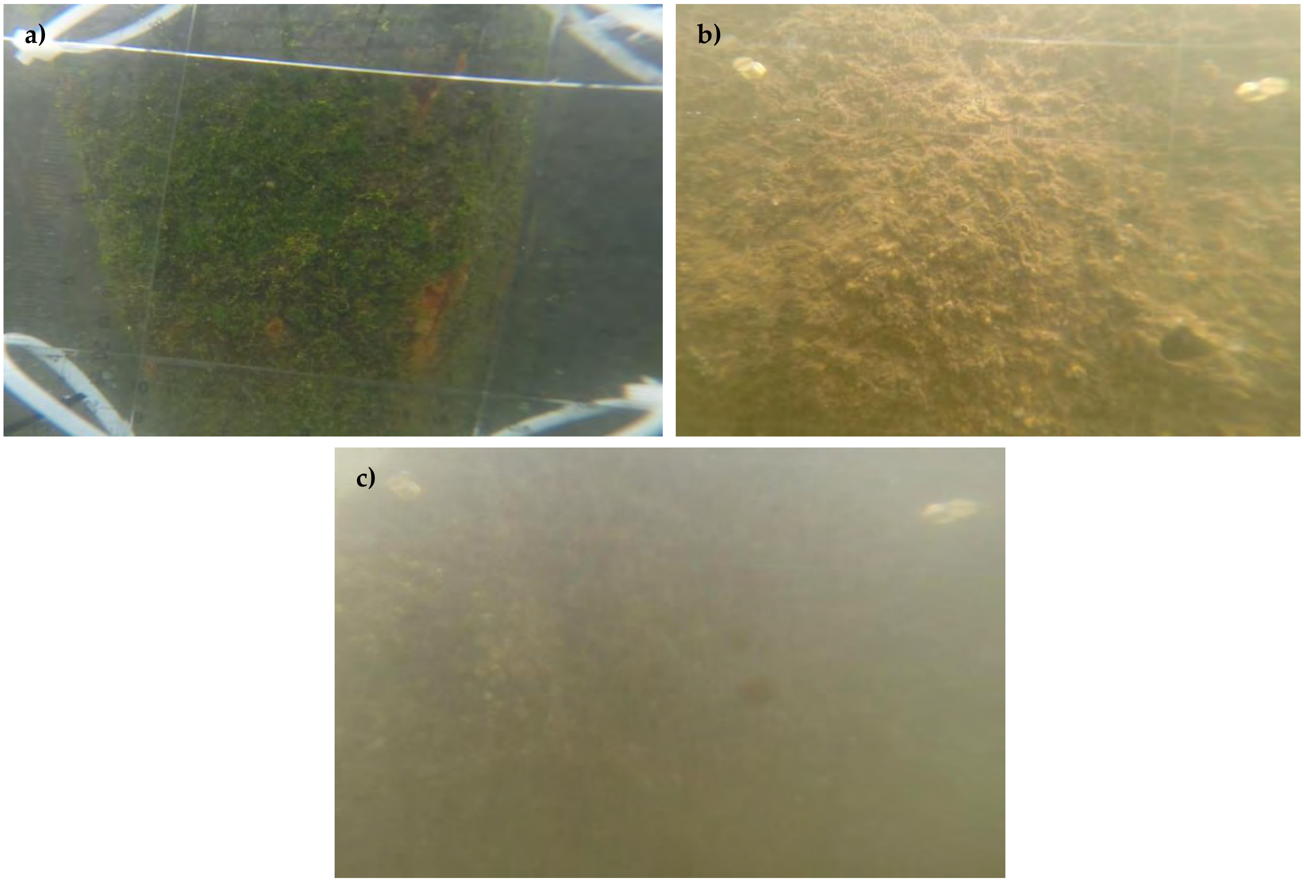


Figure 36: Photographs of subtidal hard substrate on shorelines, taken with a GoPro camera mounted on a frame which allowed images to be captured whilst either standing in the water or by lowering camera from the top of vertical seawalls. The edges of plastic rulers that were a component of the frames can be seen in panel a): these rulers allowed accurate measurements of the size of images. a) In the lower Harbor, relatively high resolution images could be captured. However, turbidity was typically high in the upper Harbor. b) Some images taken in the upper Harbor were of high enough resolution to allow identification of some taxa, but not complete confidence that all cryptic taxa could be identified, c) whilst other images were of such low resolution that no taxa could be confidently identified from them. In comparison, retrieval of settlement plates allows for consistently high resolution images to be captured (see Figure 18 and Figure 33).

Fish community assessment

Only six fish were captured in the 47 minnow traps retrieved across shorelines: two *Paralichthys dentatus*, two *Opsanus tau*, a *Syngnathus fuscus* and an *Anguilla rostrata* (Table 10, Figure 37). However, although they were not designed to capture fish, many small fish (i.e., < 100 mm in body length) were captured in the plastic mesh netting of colonization devices (Figure 38). The most common fish species captured in colonization devices was *O. tau*. Also captured were *Gobiesox strumosus*, *Tautoga onitis* and *Gobiosoma* sp.

Table 10: Fish captured in minnow traps deployed over 24 hours at shorelines in New York – New Jersey Harbor. No fish were captured at Harlem River Park shorelines (1C, 1R, 1GB), Randall's Island stone wall (2SW), West Harlem Piers riprap (3R). No fish traps were deployed at the Randall's Island seawall (2C).

Location	Shoreline type	Site code	Species	Length
Randall's Island	Riprap	2R	Summer flounder x 2 (*Paralichthys dentatus*)	46 mm, 72 mm
West Harlem Piers	Seawall control	3C	Oyster toadfish (*Opsanus tau*)	80 mm
Brooklyn Bridge Park	Riprap	4R	Oyster toadfish (*Opsanus tau*)	68 mm
Liberty State Park	Seawall control	5C	Northern pipefish (*Syngnathus fuscus*)	152 mm
	Riprap	5R	American eel (*Anguilla rostrata*)	450 mm

Figure 37: Summer flounder (*Paralichthys dentatus*), Northern pipefish (*Syngnathus fuscus*) and American eel (*Anguilla rostrata*) captured in minnow traps deployed at Randall's Island riprap, Ellis Island (Liberty State Park seawall control) and Liberty State Park riprap shorelines, respectively.

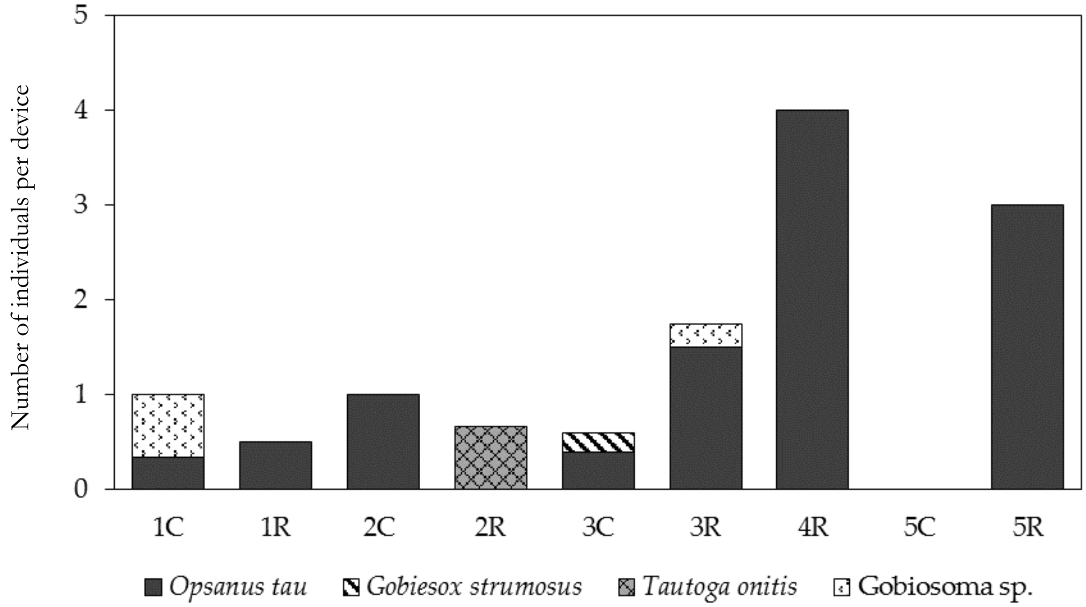


Figure 38: Mean number of fish captured in colonization devices at seawall control (C) and riprap (R) shorelines. Numbers in site codes on the horizontal axis indicate: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park.

Fish food index

Published information on the feeding specificity of fish likely to occur in New York – New Jersey Harbor only allowed for fish food index values to be assigned at Order level. The lack of specific information about dietary preferences by fish for invertebrate and algal taxa identified during the present study resulted in many taxa being assigned to the generic category of 'eaten by fish, but not a preferred food source' (Table 11). Calculation of fish food index scores for each shoreline were crude given the lack of species-specific information about fish feeding preferences and the multiplication of assigned fish food index values by abundances of each taxon, rather than biomass (which would take additional allocation of resources to measure accurately). The fish food index scores for mobile communities followed patterns for abundances, with higher mean values at riprap shorelines than seawall shorelines at all locations except Liberty State Park (Figure 39). At most locations, loss of replicate settlement plates prevented meaningful comparisons between riprap and seawall shorelines for fish food index scores based on sessile communities: patterns were contradictory where there was adequate replication, at West Harlem Piers Park and Liberty State Park (Figure 39). A regression indicated that there was no linear relationship between the total fish food index score at each shoreline and the numbers of small fish captured in colonization devices (Figure 40).

Table 11: Fish food index values assigned to taxa identified from colonization devices deployed on hard shorelines in New York – New Jersey Harbor. Values were assigned to each taxa based on whether they were: unlikely to be eaten by fish (0); eaten by fish, but not a preferred food source (1); or, preferentially eaten by fish (2).

	Taxa	Fish food index value
Mobile invertebrates	Amphipod (i.e., *Amphithoe* spp., *Corophium volutator, Gammarus* sp., *Jassa marmorata* and *Microdeutopus*)	1
	Isopod (i.e., *Idotea* sp. and *Sphaeroma quadridentatum*)	1
	Caprellid (i.e., *Paracaprella tenuis*)	1
	Shrimp (i.e., *Palaemonetes* sp.)	1
	Crab (i.e., *Carcinus maenus*, *Hemigrapsus sanguineus* and Xanthidae sp.	1
	Mussel (i.e., *Geukensia demissa* and *Mytilus edulis*)	1
	Polychaete worm	2
Sessile invertebrates and algae	Green and red filamentous algae	1
	Green macroalgae	1
	Tube-building amphipod	1
	Solitary ascidian (i.e., *Molgula manhattensis*)	1
	Colonial ascidian (i.e., *Botrylloides violaceus* and *Botryllus* spp.)	0
	Barnacle (i.e., *Amphibalanus* spp.)	0
	Hydroid	0
	Bryozoan (i.e., *Membranipora membranacea*)	1
	Sponge	0

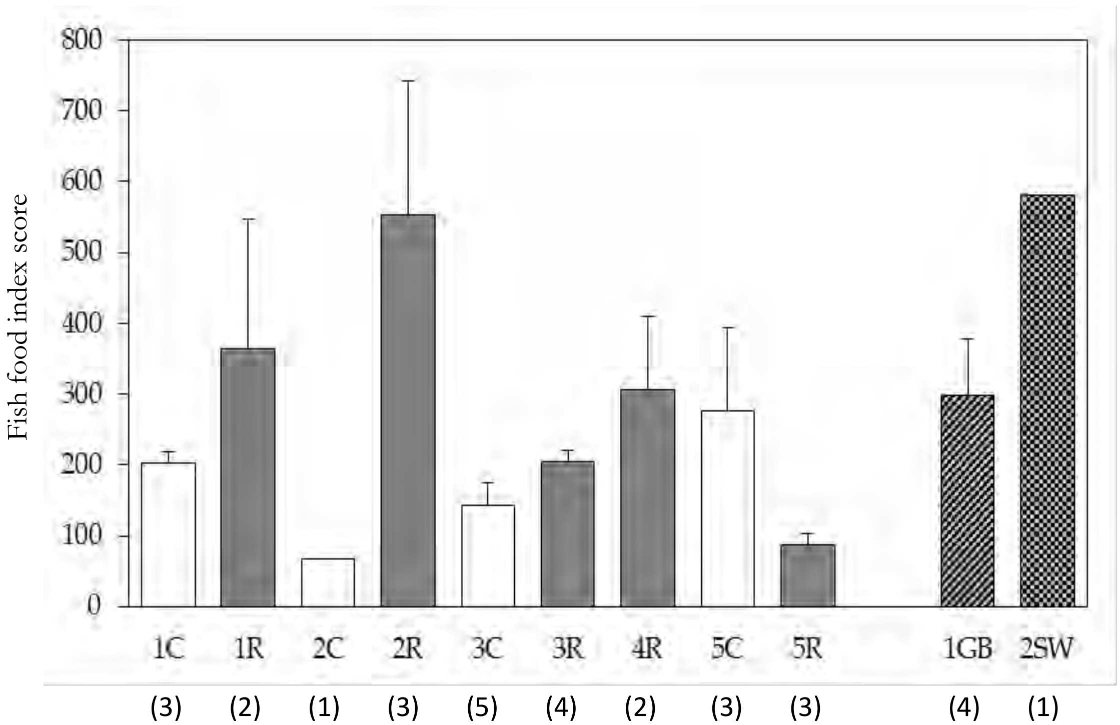


Figure 39: Fish food index scores (mean + standard error) for mobile invertebrate communities captured in colonization devices at seawall control (C, white bars) and riprap (R, grey bars) shorelines. Numbers in site codes on the horizontal axis indicate: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park. The numbers of replicate colonization devices retrieved at each shoreline are shown in parentheses.

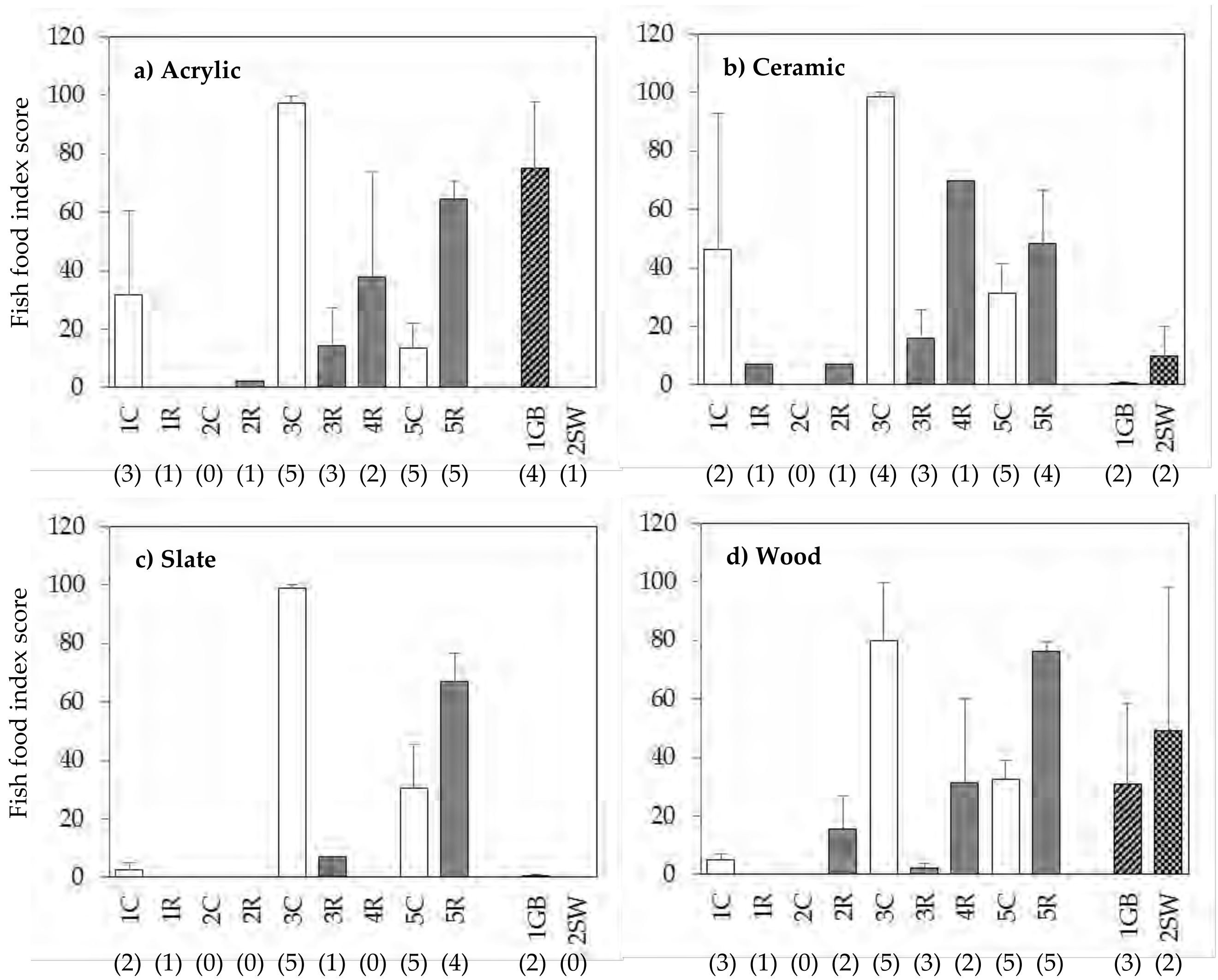


Figure 40: Fish food index scores (mean + standard error) for sessile communities that colonized (a) acrylic, (b) unglazed ceramic, (c) slate or (d) wood settlement plates at seawall control (C, white bars), riprap (R, grey bars), gabion basket (GB) and stone wall (SW) shorelines. Numbers in site codes on the horizontal axes indicate: 1) Harlem River Park; 2) Randall's Island; 3) West Harlem Piers; 4) Brooklyn Bridge Park; and, 5) Liberty State Park. The numbers of replicate colonization devices retrieved at each shoreline are shown in parentheses.

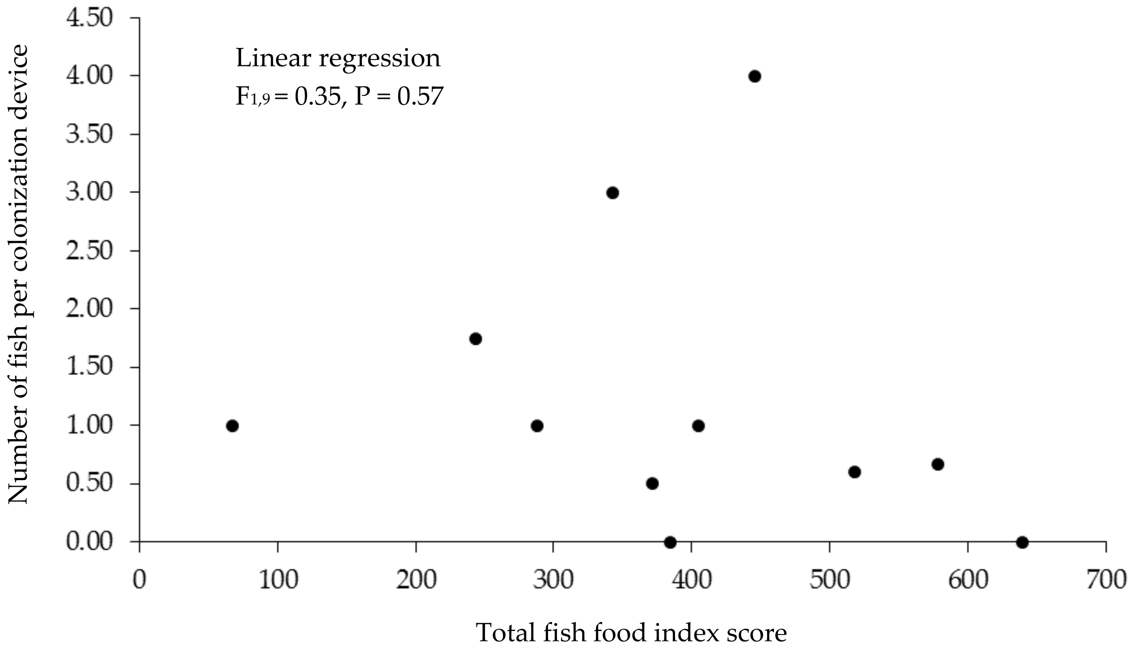


Figure 41: Regression between mean total fish food index score and the mean number of small fish captured in colonization devices deployed on hard shorelines in New York – New Jersey Harbor.

Redesigned colonization device

Colonization devices were redesigned (Figure 41), to avoid problems that were identified with the original colonization devices:

i) Galvanizes steel outer caging that was crushed whilst deployed at Brooklyn Bridge Park riprap shoreline and quickly rusted at all shorelines was replaced with more durable vinyl-coated steel mesh;

ii) Plastic cable ties used to attach settlement plates were replaced with hinged panels of vinyl-coated steel mesh to secure settlement plates to devices; and,

iii) Additional bricks were added to colonization devices to anchor them to shorelines, reducing the likelihood of devices being washed ashore (as occurred with the original colonization devices deployed at Fort Wadsworth riprap shoreline) or rotated during storms.

Materials that successfully enabled colonization by mobile and sessile communities (i.e., plastic mesh netting and settlement plates, respectively) were retained in the redesigned colonization devices. Five redesigned colonization devices were deployed at the Brooklyn Bridge Park riprap shoreline over eight weeks, from early September to early November, 2014. The communities that colonized redesigned devices were not assessed; rather, the objective of deploying those devices was to test their durability. The Brooklyn Bridge Park shoreline was chosen for testing the redesigned devices, as it was where the original devices were most damaged (Figure 13). Upon retrieval, none of the redesigned colonization devices had any obvious damage to outer vinyl-coated steel mesh caging and all settlement plates had been retained.

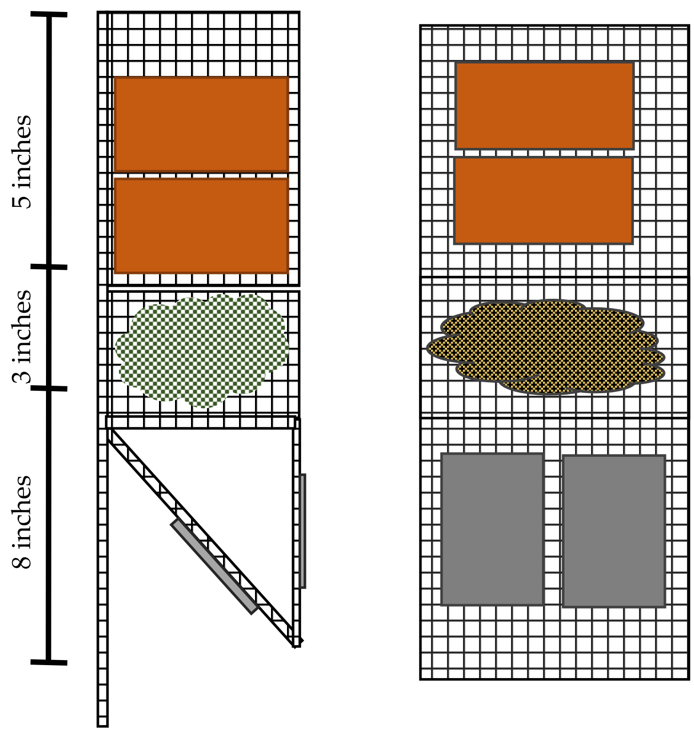


Figure 42: Redesigned colonization device with durable outer caging made from vinyl-coated steel mesh; side view (L) and top view (R). The upper five-inch-high compartment has four bricks to provide weight for anchoring devices on hard shorelines. The middle three-inch-high component has plastic mesh netting for colonization by the mobile invertebrate community. At the bottom of devices, settlement plates are secured in place by eight-inch-high hinged covers made from vinyl-coated steel mesh. Holes are cut from the mesh to provide a clear area for colonization by sessile biota. See Figure 4 to compare the original device used during the present study to the redesigned device. A more complete description of the redesigned device is provided in Reid et al. 2015.

Discussion

Quantitative assessments of subtidal invertebrate and algal communities using a novel device with colonization surfaces of standard surface area that were deployed over a standard duration showed promise for inclusion in a repeatable protocol to determine the relative habitat values of urban shorelines differing in physical habitat complexity in New York – New Jersey Harbor. Novel colonization devices were able to be deployed and retrieved from the subtidal zone of different urban shoreline types (i.e., concrete seawall, riprap revetment, gabion basket and stone wall). Over eight weeks, submerged devices allowed colonization by subtidal mobile invertebrate communities on plastic mesh netting, and sessile invertebrate and encrusting algae communities on settlement plates made from different hard materials (i.e., acrylic, unglazed ceramic, slate and wood). Conversely, qualitative assessments using quadrats demonstrated that in some regions of the Harbor the diversities of intertidal invertebrate and algal communities were primarily constrained by factors other than local shoreline physical habitat complexity: for example, sedimentation of hard shorelines in the upper Harbor likely reduced their habitat value. Underwater photoquadrat surveys were not useful for assessing shoreline habitat values in New York – New Jersey Harbor, owing to highly turbid water. Comprehensive fish surveys require investment of resources exceeding those typically available for repeated ecosystem monitoring, owing to high temporal variability in the abundances of highly mobile fish populations along estuarine shorelines. More detail regarding each of the surveys trialed during the present study is provided below: italicized text below subheadings provide a synopsis of the validity of each hypotheses specified in the introduction to this report.

Community diversity in the subtidal zone of hard shorelines in New York – New Jersey Harbor was generally low. Gibson et al. (2000) suggest that the relatively low richness of epibenthic communities, particularly in temperate coastal marine waters of the USA, may limit the sensitivity of this measure and hinder the use of such communities in ecosystem assessments. However, assessment of both the sessile and mobile community is desirable for determining the relative habitat values of hard shorelines, given their prominence and role in providing beneficial ecosystem functions on such shorelines (e.g. Butler 1995, Levinton et al. 2006). Despite their low diversities, epibenthic communities may be more sensitive than infaunal communities to certain anthropogenic stresses (Gibson et al. 2000). Further, given that adult stages of sessile communities are sedentary, the presence of particular taxa is a definitive sign that the surrounding environment had suitable conditions for their survival from the time that they attached to hard substrates as larvae to the time that they were collected as adults (Gibson et al. 2000). Also, the low diversity of benthic communities in New York – New Jersey Harbor is not unique to the communities associated with hard shorelines. Rather, it is consistent with previous studies of other benthic invertebrate communities in the region (e.g. Crawford et al. 1994, Goto and Wallace 2009), owing to the multiple anthropogenic disturbances in the highly urbanized and industrialized estuary. There were no significant differences in the taxon richness of sampled mobile or sessile communities between nearby seawall and riprap shorelines, with the exception of higher richness on slate settlement plates deployed at Liberty State Park riprap shoreline than on analogous plates deployed at the nearby seawall. The abundances of some taxa in colonization devices were different between nearby seawall and riprap shorelines at both West Harlem Piers and Liberty State Park. However, these differences were not consistent between the two locations and univariate analyses between communities from seawall and riprap shorelines were not possible for other locations, owing to loss of replicate samples.

Multivariate analyses of the data from colonization devices consistently demonstrated that communities were more dissimilar between locations in New York – New Jersey Harbor than between shorelines differing in physical habitat complexity within each location. Differences in invertebrate community structure in the lower and upper Harbor was consistent with previous studies conducted nearby (e.g. Strayer and Malcom 2007) and other studies conducted across estuarine salinity gradients (e.g. Barnes and Ellwood 2012, Josefson and Goke 2013, Wetzel et al. 2014). During the present study, locations receiving most water from marine sources in the lower Harbor (i.e., Brooklyn Bridge Park and Liberty State Park) and on Randall's Island tended to have relatively similar communities to each other, as did those locations with more fresh water contributed from upstream (i.e., Harlem River Park and West Harlem Piers, on the Hudson River and Harlem River, respectively).

Differences in subtidal community structure occurring across New York – New Jersey Harbor, regardless of shoreline physical habitat complexity, mean that the type of community that can become established after shoreline rehabilitations will be location-specific. Future comparisons of the relative habitat values of shorelines will be most meaningful between those shorelines in the same 'ecoregion': this requires defining the boundaries of ecoregions across the Harbor within which hard shoreline communities are likely to be similar and building a database of the community structures occurring on hard shorelines differing in physical habitat complexity in each ecoregion. Delineating ecoregion boundaries was beyond the scope of the present study. However, the boundaries in Bain et al. (2007) provides a guide to how the Harbor may be separated into different regions for ecological assessments (Figure 1); although the present study demonstrates that Harlem River should be classified as occurring within a distinct ecoregion or merged with the 'Lower Hudson River' region of Bain et al. (2007), owing to the influence of fresh water from upstream on local shoreline ecology.

Types of shorelines presently in New York – New Jersey Harbor

Approximately 36% of the shoreline across the Hudson – Raritan Estuary is predominantly composed of hard man-made structures, and these types of shorelines are particularly common in the lower Hudson River, Harlem River, East River and Upper New York Harbor (i.e., in the region covered during the present study) (Bain et al. 2007, Figure 1). Approximately 66% and 87% of shorelines are hardened in the Lower Hudson River and Upper Bay, respectively (US Army Corps of Engineers and The Port Authority of New York and New Jersey 2009). Bain et al. (2007) suggested focusing on areas with extensive built shorelines for future rehabilitation initiatives, with a long-term ecosystem target to 'restore all available shoreline and shallows sites in the following areas: Lower Hudson, Upper Bay, and Harlem – East River – Long Island Sound' by 2050. Emphasis on the word 'available' has been added. It was envisaged that the number of shoreline sites available for rehabilitation in the region would be low, and at those sites rehabilitation targets should include revegetating riparian zones, reconfiguring intertidal zones with stable slopes and facilitating the illumination of shallow littoral zones (Bain et al. 2007). Those rehabilitation targets stated by Bain et al. (2007) were adopted in the *Hudson – Raritan Estuary Comprehensive Restoration Plan*, which further states that there are 737 kilometers of man-made shoreline without piers and 221 kilometers of shallow littoral zone along shorelines in the estuary that could be targeted for rehabilitation (US Army Corps of Engineers and The Port Authority of New York and New Jersey 2009). Considerable investment will be required to rehabilitate all of this shoreline. Rehabilitation of shorelines to increase their habitat value for desirable native communities may be facilitated by increasing the physical complexity of hard shorelines or, where feasible, returning shorelines to predominantly unconsolidated sediments, as well as rehabilitating the remaining shorelines with unconsolidated 'soft' sediments. However, in the highly urbanized and industrialized New York – New Jersey Harbor there are constraints operating across multiple scales to hinder shoreline rehabilitation efforts with ecological goals.

During initial qualitative searches for hard shorelines suitable for use in the present study, most hard shorelines around Manhattan and in the lower Harbor were broadly categorized as either vertical concrete seawall with minimal physical habitat complexity or riprap revetments made from boulders, which provided more diverse microhabitats. There were particularly long reaches of riprap shoreline along the New Jersey and northern Manhattan shorelines, with the New Jersey shoreline across from Manhattan also having large reaches of predominantly unconsolidated sediment (consistent with previous surveys of Bain et al. 2007, Figure 1). Although stone walls (such as those occurring along Randall's Island western shoreline, Figure 2c) are similar to concrete seawalls insofar as they are hard and vertical, they have crevices between the large stones, which may provide additional microhabitats for some intertidal and subtidal taxa. The addition of increased physical habitat complexity, including small holes and crevices, to seawalls in the highly urbanized Sydney Harbour provided novel microhabitat for algae, and mobile and sessile invertebrates (Browne and Chapman 2011). There were also some shorelines with piers or decommissioned pier pilings scattered around New York – New Jersey Harbor. The shading under piers limits primary productivity of benthic communities (e.g. Kennish 2002, Stutes et al. 2006) and fish abundances (Able et al. 1998, Duffy-Anderson et al. 2003), but pilings without shading from piers provide valuable habitat for fish in New York – New Jersey Harbor (Able et al. 1998).

Along the Harlem River Park shoreline there were reaches of tiered riprap and gabion basket filled with natural hard substrates, with New York City Parks signs indicating that the shoreline was 'Special Ecological Seawall' (Figure 2e, f, g). The rehabilitation of the foreshore along Harlem River Park involved a multimillion dollar collaboration between landscape architects, city planners, marine engineers, marine biologists, environmental artists and the local community (Johnson 2010). It is proposed that creation of a bike path, esplanade, restrooms, green space and murals along the Manhattan bank of the Harlem River will improve public access and empower the local community to care for the river and surrounds (Johnson 2010). This 'Designing the Edge' project will be a useful case study in the rehabilitation of New York City shorelines for ecological, aesthetic and community-building values, if monitored appropriately and adequately. In addition to cobbles, the gabion basket shoreline at Harlem River Park also had oyster shell that may facilitate colonization by oyster larvae (Michener and Kenny 1991) on the shoreline, also one of the explicit goals of the ambitious 'Billion Oyster Project' in New York – New Jersey Harbor. The New York Harbor School currently uses oyster shell in submerged cages to encourage settlement of oyster larvae and to monitor oyster populations in the Harbor (New York Harbor School 2014). Oyster shells could be contained within the redesigned colonization device that was used during the present study, allowing assessment of oyster populations on shorelines where establishing such populations is stated as an objective of shoreline rehabilitation. However, adding oyster shells to the devices was not attempted during the present study. To guide appropriate shoreline management actions, allow tailoring of shoreline monitoring techniques and facilitate learning lessons from all future shoreline rehabilitations, the goals of each rehabilitation project should be explicitly stated prior to implementation. Future hard shoreline rehabilitations may include multiple natural materials (e.g. wood, rocks, oyster shell), as well as man-made elements (e.g. precast concrete, rubber tires) to add physical habitat complexity (e.g. Goff 2010, Johnson 2010, Dafforn et al. 2015).

Variability in water quality across New York – New Jersey Harbor

There has been large investment in improving the water quality of New York – New Jersey Harbor, with evidence of considerable improvement over the past 100 years of testing (NYC DEP 2012). Colonization devices were deployed during summer, when temperate estuaries support relatively high biotic productivity. Mean water temperature during the study was 73 to 76˚F across shorelines. Water temperatures were more variable on those shorelines in north-eastern Manhattan (i.e., Harlem River Park and Randall's Island), possibly owing to those shorelines receiving water from tidal currents both flowing north from the Narrows and south through the Long Island sound during different tidal phases (Figure 43). All shorelines in the lower Harbor and East River would have received most water from marine sources, as reflected in their high salinities compared to shorelines along the Hudson River and Harlem River. As mentioned above, the communities occurring on shorelines varied between ecoregions. Differences in the salinity between locations would have exerted considerable influence on subtidal community structure, as has been shown in previous estuarine studies (e.g. Strayer and Malcom 2007, Barnes and Ellwood 2012, Josefson and Goke 2013, Wetzel et al. 2014).

Shading may also influenced subtidal community structure and the 'illumination of shallows' is one of the specific rehabilitation goals suggested for shorelines in the *Hudson – Raritan Estuary Comprehensive Restoration Plan* (US Army Corps of Engineers and The Port Authority of New York and New Jersey 2009). During the present study, mean irradiance at seawalls was generally lower than at riprap shorelines. The extent and duration of shading during the day depended upon the aspect of the shoreline and height of obstructions that could intercept light. The west-facing seawall on Ellis Island (control for Liberty State Park riprap shoreline) and the north-facing seawall at Astoria, which both had walls extending at least three meters above deployed colonization devices, had the lowest mean irradiances. The effects of such shading operate through multiple mechanisms, especially by limiting the amounts of primary production and creation of microhabitats by submerged plants and algae (e.g. Glasby 1999, Garrison et al. 2005, Stutes et al. 2006, Thom et al. 2008). Where shading is complete, such as under piers, the numbers of fish (e.g. Able et al. 1998) and feeding efficiency of visual predators (e.g. Benfield and Minello 1996) may also be lower than in analogous unshaded habitats. During the present study, differential shading between shorelines appeared to have a large influence on macroalgal growth in some locations, particularly at Liberty State Park where algal cover was much higher on the relatively unshaded riprap shoreline than the shaded seawall.

The availability of nutrients also affects algal growth and there were notable differences in the concentrations of nitrate and phosphate between nearby shorelines. For example, at Harlem River Park, nutrient concentrations were lower at the gabion basket shoreline than at the seawall or riprap shoreline; similarly, concentrations of all nutrients were higher at the seawall shoreline than the riprap shoreline for West Harlem Piers. These localized differences in nutrient concentrations could well drive differences in primary productivity and food supply between shorelines within locations, with the pending additional analyses allowing additional examination of this potential effect.

The nutrient measures across all shorelines supported the general assertion that primary production within the harbor is not nutrient-limited (e.g. Levinton and Waldman 2006). However, there were some interesting anomalies, most notable of which was that the samples taken from around Randall's Island were extremely high in phosphates and exhibited a P:N ration higher than that typically shown in unperturbed systems. A possible reason for this finding is the location of the Tier 1 CSO wastewater treatment plant that discharge effluent into the waters east of the Island (e.g. Muñoz and Panero 2008)

Increased fine sediment loads owing to human land use intensification in upstream watersheds is a major pollutant of estuarine ecosystems (e.g. Airoldi 2003, Thrush et al. 2004, Reid et al. 2011). Sedimentation can detrimentally affect estuarine invertebrates via direct smothering, reduced surface-habitat heterogeneity, changes in fluxes of oxygen and nutrients, reduced autotrophic production, and/or impaired feeding and growth (e.g. Wood and Armitage 1997). Fine sediment deposition may detrimentally affect benthic communities on both unconsolidated sediments (e.g. Thrush et al. 2004, Reid et al. 2011) and hard substrates (e.g. Airoldi 2003). Deposition of fine sediments on the hard shorelines in New York – New Jersey Harbor is exacerbated by the close juxtaposition of hard and soft substrates; which may limit the opportunities for establishment of taxa with different habitat requirements, as hard substrates prevent colonization by burrowing taxa and suspended fine sediments hinder attachment and feeding of some taxa adapted to living on hard substrates. The hard shorelines used during the present study usually did not extend far below the low tide water level and were immediately above predominantly silt or sand substrates. This fine sediment was readily suspended in the water column by water turbulence and then deposited on the hardened portion of shorelines. Martin et al. (2005) found that, in general, low crested coastal defense structures which provided the most benefits for surrounding biota were those that extended furthest away from the shoreline. More study would be required to determine whether extending the hardened portion of shorelines to greater depths would facilitate colonization by more diverse communities adapted to living on hard substrates in New York – New Jersey Harbor.

Measuring the relative power of water movement owing to tidal currents, storm waves and boat traffic was not attempted during the present study. However, water movement may significantly influence subtidal community structure in estuaries, particularly owing to its ability to transport fine sediment, scour biota and overturn rocks to open new habitats and reset succession (Sousa 1979, Wernberg and Connell 2008). During the present study, the relative mass lost from submerged gypsum balls (also called 'clod cards') was considered as a relatively cost-effective way of estimating the relative erosive power of water movement (Petticrew and Kalff 1991). However, dissolution may be influenced by salinity and temperature, as well as water movement (Porter et al. 2000), so is difficult to calibrate for comparisons of estuarine shorelines with large differences in salinity between surveyed sites. Further, there may have been unintended effects of the dissolved material from gypsum balls on shoreline communities. Strayer et al. (2012) demonstrate that 'exposure' to water movement is an important physical characteristic of shorelines which influences shoreline community structure. They suggest that given its importance, the development of a faster, easier, more reliable and standardized way of measuring this variable than using existing techniques should be a priority (Strayer et al. 2012). If an appropriate method can be identified, it would be beneficial to quantify relative water movement during future assessments of hard shorelines in New York – New Jersey Harbor.

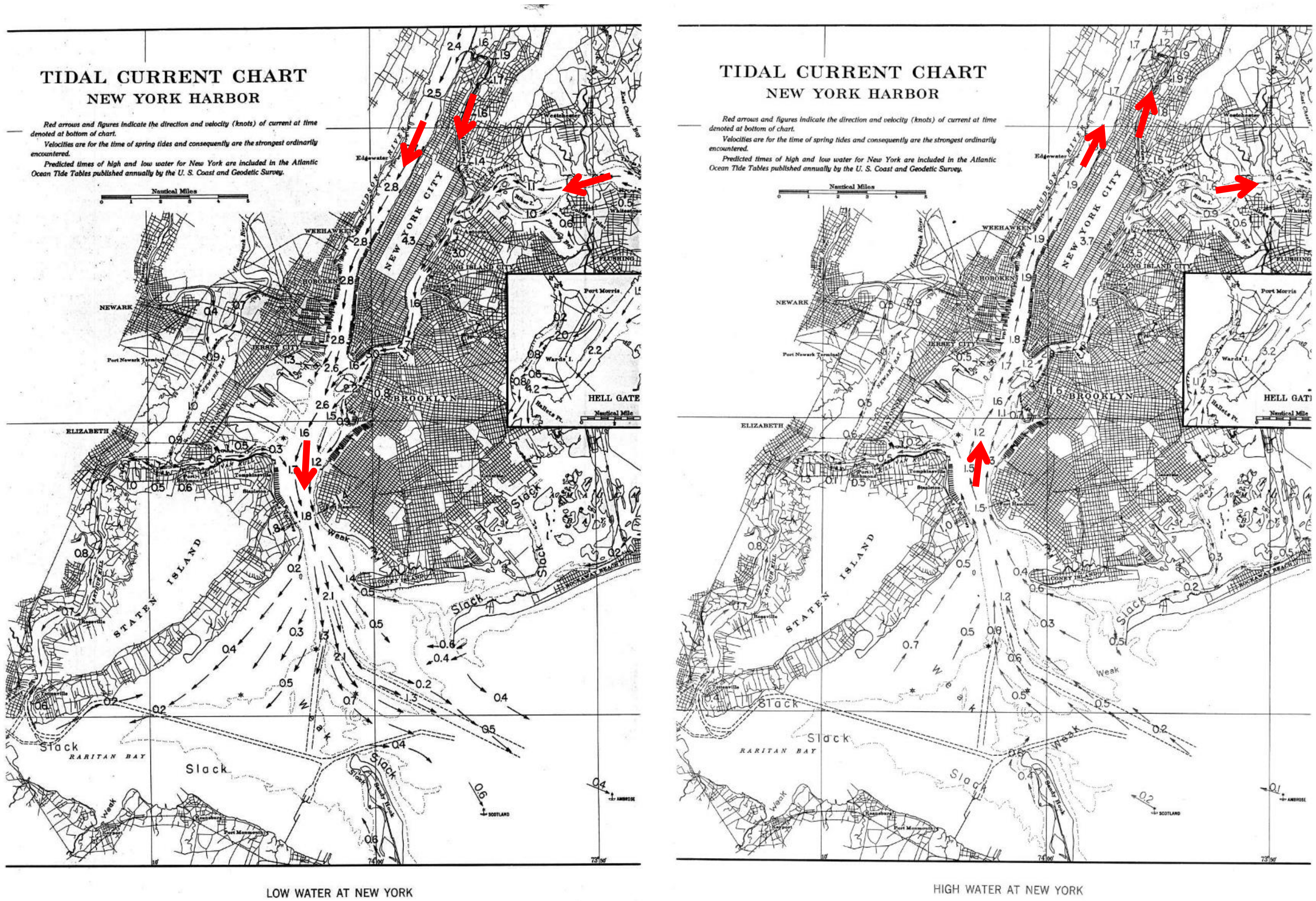


Figure 43: Direction and relative power of water currents in New York – New Jersey Harbor at a) low tide and b) high tide (from Eldridge and Eldridge 2009, with red arrows added to emphasize direction of water flow and changes in water sources across the Harbor during different tidal cycles). All of New York – New Jersey Harbor is influenced by tides, but water sources vary depending upon location in the Harbor and the tidal cycle. Major sources of water come from the Hudson River, through The Narrows and through Long Island Sound.

Colonization device performance, constraints and redesign

Colonization devices were able to be deployed on the dominant hard shoreline types in New York – New Jersey Harbor. Colonization devices were colonized by subtidal communities after being submerged for four or eight weeks, with the community structure differing between devices submerged for different durations.

The original colonization device (Figure 4) was deployed on vertical seawall, stone wall, gabion basket and riprap shorelines. Colonization devices were designed to allow retrieval of a subset of the communities on hard shorelines which would be a reliable proxy for the community on the shoreline, rather than for surveys to provide an inventory of all species occurring on shorelines. It was assumed that colonization devices deployed on shorelines with depauperate communities would be colonized by fewer taxa than colonization devices deployed on shorelines with higher richness. It was also assumed that the relative abundances of each taxon on colonization devices would reflect the relative abundances of that taxon on the shoreline where the device was deployed. Plastic mesh netting allowed colonization by amphipods, isopods, caprellids, shrimps, crabs, mussels and worms. Settlement plates were colonized by filamentous algae, macroalgae, ascidians, bryozoans, barnacles, amphipods, worms, hydroids and sponges. The material used for settlement plates did not obviously influence sessile community structure: i.e., similar communities occurred on acrylic, unglazed ceramic, slate and wood settlement plates at each shoreline. In future assessments, the use of readily obtainable and inexpensive settlement plate materials is preferable. However, the redesigned device allows for the use of different settlement plate materials and more specialized settlement plate materials may be appropriate to survey sessile populations of particular interest for future shoreline rehabilitations.

The colonization devices (Figure 42) can theoretically be used on most hard estuarine shorelines, regardless of their physical habitat complexity and slope, if they extend deep enough to allow deployment at subtidal depths that ensure devices remain just below the highly turbulent swash zone during all tides. The device performed largely as anticipated on vertical seawalls where they extended deep enough to allow devices to be deployed under the low tide swash zone (i.e., Ellis Island, which was the Liberty State Park control shoreline) or were protected from turbulent water movement (i.e., West Harlem Piers). However, limitations in the original design of colonization devices and where they could be effectively deployed were also identified whilst testing them in the field. An inherent risk with using colonization devices, as opposed to direct collection of samples during a single visit, is that no data can be recovered if the devices are damaged or removed from the subtidal zone of shorelines during deployment. Stainless steel crab traps and plastic cable ties were not robust enough to securely support the other components of the colonization device. Crab traps rusted across all locations and some were crushed at Brooklyn Bridge Park. Settlement plates were frequently dislodged from the cable ties used to attach them to the crab traps.

Colonization devices are not suitable for use on shorelines subject to powerful water movement, such as the strong tidal currents along the northern Astoria shoreline and The Narrows, and large waves at the base of the Fort Wadsworth seawall. Constant turbulence is not conducive to colonization of those surfaces provided on the colonization device. The original device was relatively light-weight and susceptible to being washed ashore on low gradient shorelines with large waves, as occurred at the Fort Wadsworth riprap shoreline. Devices were presumably most susceptible to being moved and damaged during large storms, such as that occurring in early July as Hurricane Arthur swept up the east coast of the USA. The hurricane was offshore as it passed New York City, but created large waves in the Harbor. Although colonization devices were not washed ashore on other shorelines, it was obvious from the twisted ropes used to attach them to the shoreline that some devices had also been rotated during storms. At Brooklyn Bridge Park, the damage to cages suggests that there was much water turbulence at the shoreline (Figure 13).

To improve their durability and reduce their susceptibility to being moved by water turbulence, colonization devices were redesigned with stronger vinyl-coated steel mesh outer caging, more secure settlement plate attachment and more weight (Figure 42). The redesigned device still includes those materials that facilitated colonization of subtidal communities during deployment of the original colonization devices (i.e., plastic mesh netting and hard settlement plates). Redesigned colonization devices were successfully deployed, with no obvious damage to caging or loss of settlement plates, at Brooklyn Bridge Park. Use of the redesigned device in future assessments will reduce the loss of replicate samples that hindered assessment during the present study. All materials used to construct colonization devices are readily available, and a manual detailing construction, deployment and retrieval of devices was developed during the present study, to facilitate the ability of different organizations to perform standardized assessments using the colonization devices (see Reid et al. 2015).

Given that the study was conducted near New York City, there was also the potential for direct human interference with colonization devices left in the field. Fortunately, evidence of vandalism of the colonization devices was rare: one rope at Harlem River Park appeared to have been cut to remove a device deployed on the gabion basket shoreline, but there was no obvious evidence of tampering with any other device. Direct human disturbance of shorelines also occurred from boat wakes. The shoreline at Harlem River Park was particularly susceptible to such disturbance: the Harlem River channel was narrow and large tour boats frequently created large waves which washed along shorelines and suspended fine sediments. Brooklyn Bridge Park in the lower Harbor also had large wakes from boat traffic, but there was more dispersal of the power of wakes before they reached the shoreline, owing to the more open channel than that in the Harlem River. Previous studies have shown that boat wakes can influence subtidal communities, mainly by redistribution of sediments, but also potentially owing to increased turbidities, reduced larval supply, changes in resource availability and/or scouring of biota from substrates (e.g. Garrad and Hey 1987, Bishop 2007). This may hinder establishment of some populations on the rehabilitated shorelines along Harlem River Park, and future rehabilitations subject to regular disturbance from large boat wakes.

A deployment duration of four weeks was trialed, with a view to reducing the duration that colonization devices were left in the field, and therefore reducing damage and loss. The desire for a brief deployment must be balanced against the desire to provide enough time for adequate colonization, so that the communities that colonization devices are reliable proxies for the communities occurring on shorelines. It is probable that estuarine shorelines contain a mosaic of microhabitats at different successional stages, because they are subject to frequent pulse disturbances from waves and seasonal shifts in dominance by different taxa (e.g. Nichols 1985, Jewett et al. 2005). It was assumed that the dominant community on shorelines would be that occurring at later successional stages: members of the Advisory Committee for the present study suggested that the communities on colonization devices should reflect shoreline community development at later succession stages, but questioned whether this was likely to be possible with devices deployed over relatively short durations.

Four weeks was adequate for most mobile taxa to colonize, although taxa such as worms and mussels did not colonize devices deployed for only four weeks. However, not all sessile taxa that had grown into identifiable forms after eight weeks deployment (e.g. bryozoans) were identified on settlement plates deployed for four weeks. Comparison of data collected from colonization devices deployed for four weeks and eight weeks showed that the temporal dynamics of colonization by subtidal populations differed between seawall and riprap shorelines. There were large shifts in dominant taxa depending upon the duration over which they were deployed and the physical complexity of the shoreline on which they were deployed. Deployment for eight weeks allowed algal growth to dominate sessile communities, which reflected the later succession community occurring on shorelines. Given this variability in colonization dynamics depending upon differences in physical habitat complexity of shorelines, more work is required to determine the optimal duration over which colonization devices should be deployed. Sessile community structure varies throughout the year (e.g. Jewett et al. 2005, Levinton et al. 2006), so to allow meaningful comparison between shorelines they should be monitored at the same time of year. Ideally, the communities retrieved in colonization devices should be reliably close proxies for the communities occurring on the shorelines. To confirm that the communities from colonization devices reflect the communities on shorelines, they should be compared with the communities collected using alternative methods (e.g. suction sampling, photoquadrats) on shorelines where such comparisons are feasible.

Frequent direct human access to shorelines presents challenges and constraints to shoreline rehabilitation and monitoring, but also opportunity for education about ecological values of shorelines. Signs near the Brooklyn Bridge Park riprap shoreline provided information about the potential ecological values of Harbor shorelines. That shoreline has much human traffic and educational signs in such high-use areas may reach a large audience. Further, whilst conducting fieldwork during this study, we encountered widespread public interest in both the colonization devices and the types of animals living along shorelines in New York – New Jersey Harbor.

Mobile invertebrate communities

The diversities of subtidal invertebrate and algal communities were not consistently different, but there were differences in the abundances of some taxa in some locations, for comparisons between hard shorelines differing in physical habitat complexity in New York – New Jersey Harbor. The ability to detect differences in diversities and abundances of subtidal communities using colonization devices was hindered by loss of replicate samples.

Although mobile communities were highly variable and patchy, even between colonization devices deployed on the same shoreline, multidimensional scaling analyses of five replicate colonization devices (or less) from each shoreline were able to be used to distinguish between mobile invertebrate communities of hard shorelines in different locations. Multivariate ordinations consistently grouped similar communities from nearby shorelines and separated communities with dissimilar communities across different locations. Unfortunately, the loss of replicate mobile invertebrate samples made it impossible to meaningfully assess differences in taxon richness and abundances between the seawall and riprap shoreline in each location. Mean taxon richness and abundances were higher at riprap shorelines than at seawalls at all locations except Liberty State Park. However, at the two locations for which univariate analyses were possible (i.e. West Harlem Piers and Liberty State Park), there were no significant differences in taxon richness or abundances for the mobile communities at seawall and riprap shorelines. All mobile invertebrate samples from colonization devices deployed on shorelines had low diversity, regardless of the structural habitat complexity of the shoreline on which they were deployed. Low diversity of benthic communities may limit the ability to detect differences in shoreline communities (Gibson et al. 2000). However, such low diversities are not unique to colonization devices, rather they are typical of communities in estuaries subject to multiple human press disturbance from urbanization and industrialization (e.g. Crawford et al. 1994, Goto and Wallace 2009).

No mobile taxon was consistently favored by the greater habitat complexity on riprap shorelines than seawalls during the present study, although other studies have shown that enhanced physical habitat complexity along shorelines may facilitate establishment of desirable populations unable to persist on less complex shorelines (e.g. Chapman 2003, Bulleri and Chapman 2004, Lawless and Seitz 2014). With careful planning and implementation, it may be possible to create nearshore habitats which support the establishment of desirable native taxa, such as the provision of frayed net material to successfully increase abundances of protected seahorses in the highly urbanized Sydney

Harbour (Hellyer et al. 2011). A caveat is that the pool of native taxa which can become re-established along rehabilitated shorelines in New York – New Jersey Harbor is lower than in estuaries surrounded by less human development. Physical habitat complexity should not be added indiscriminately. In addition to supporting establishment of desirable taxa, addition of novel habitats along shorelines can facilitate establishment of invasive taxa (e.g. Bulleri and Airoldi 2005, Glasby et al. 2007, Bulleri and Chapman 2010, Airoldi and Bulleri 2011). During the present study, there were significantly more of the exotic Japanese shore crab (*H. sanguineus*) captured from the riprap shoreline than from the seawall shoreline at West Harlem Piers. More extensive surveys would be required to identify the full suite of desirable native taxa which may be favored by increasing shoreline habitat complexity in the Harbor, being mindful of those constraints other than physical habitat complexity on each taxon. Future shoreline rehabilitations should have explicitly stated goals to facilitate targeted habitat enhancements, so that the planning and implementation of rehabilitation actions favor establishment of desirable native taxa over less desirable exotic taxa.

Sessile communities

As for mobile communities, the assessment of the sessile communities from settlement plates detected distinct communities between locations, but did not provide evidence that the additional physical habitat complexity on riprap shorelines in comparison to seawalls consistently favored any particular taxa. Sessile communities on acrylic, unglazed ceramic, slate and wood settlement plates were dominated by red and green filamentous algae and ascidians. Bryozoans, tube-building amphipods, barnacles, hydroids and sponges also colonized settlement plates. Only 12 identifiable taxa colonized settlement plates and the taxon richness on each plate was low, i.e. mean taxon richness was usually < 2 for each settlement plate material at each shoreline, except those at Liberty State Park.

Contradictory patterns for taxon richness and algal cover occurred at the two locations where the numbers of replicate settlement plates were sufficient to allow univariate analyses. At West Harlem Piers (in the upper New York – New Jersey Harbor, with considerable fresh water influence from the Hudson River), taxon richness and algal cover were generally lower on settlement plates retrieved from riprap than those from the seawall shoreline. Conversely, at Liberty State Park (in the lower New York – New Jersey Harbor, with minimal fresh water influence), taxon richness and algal cover were generally higher on settlement plates retrieved from riprap than those from the seawall shoreline.

Nutrient concentrations and light are positively associated with algal growth, and likely contributed to the different patterns of algal dominance for riprap and seawall shorelines at West Harlem Piers and Liberty State Park. At West Harlem Piers, higher nitrate and phosphate concentrations occurred at the seawall than the riprap shoreline, whilst irradiance was relatively high along the seawall (not recorded at the nearby riprap shoreline, but given the slope of the shoreline irradiance was higher at the riprap shoreline than the seawall). Conversely, similar nutrient concentrations occurred along seawall and riprap shoreline at Liberty State Park, but there was less light available at the seawall than at the riprap shoreline, which would have limited the productivity of encrusting algae and likely contributed to observed differences in overall subtidal community structure. Given the high nutrient levels throughout the study site, we conclude the light, and not inorganic nutrient availability, is the likely limiting factor in algal growth differences.

Hard surfaces that are in the shallow subtidal zone or are well lighted are often dominated by algae, with macroalgal communities comprising various species of red, green or brown algae (e.g. Schiel 1990, Sanderson et al. 1997). Changes in the cover of different taxa of sessile algae on settlement plates have proven useful in assessments of communities on hardened urbanized shorelines (e.g. Connell and Glasby 1999, Bulleri et al. 2005). In urbanized estuaries, algae are often limited to ephemeral taxa such as *Ulva* spp., *Enteromorpha* spp. and other filamentous green algae, as well as other small colonies of red and brown algal species (e.g. Glasby and Connell 2001). However, results have contrasted between studies in different regions with different algal species that vary in their sensitivity to habitat modification: both more (Bulleri et al. 2005) and less (Bulleri and Chapman 2004) encrusting algae has been shown to occur on those shorelines with more physical habitat complexity, in comparison to seawalls.

At both West Harlem Piers and Liberty State Park invertebrate cover was generally lower on settlement plates retrieved from riprap than those from the seawall shoreline. In Sydney Harbor, Australia, the densities of sessile taxa and overall sessile community structure were similar on natural sandstone and sandstone seawall, in comparison to sessile communities on pilings and pontoons: sandstone substrates tended to have relatively low densities of invertebrates and were dominated by coralline algae (Connell and Glasby 1999). Coralline algae also dominated the sessile epibiotic community on a natural reef in Sydney Harbor, although not as completely as on natural reef in the nearby Botany Bay, and whilst some sessile animal taxa (including ascidians) were less common on natural reefs than on concrete breakwalls, the reverse pattern was true for other sessile animal taxa (including oysters and sponges) (Knott et al. 2004).

Given their role in 'biofouling' of economically important infrastructure, there is much more specific information about most sessile invertebrate taxa than most mobile invertebrate taxa from subtidal estuarine communities. All of the sessile invertebrates commonly identified during the present study were hardy taxa that tolerate human disturbance. The colonial ascidian *Botryllus schlosseri* was the most common sessile animal on settlement plates. *B. schlosseri* is a cosmopolitan species found on all continents except Antarctica, as it can tolerate harsh abiotic conditions and has a life history that is well adapted to invading and becoming established in man-made habitats (Stoner 2002, Jewett et al. 2005). Although *B. schlosseri* occurs in more natural environments in Europe, it is largely confined to highly modified environments elsewhere (Stoner 2002). These habitat associations and genetic studies provide evidence that *B. schlosseri* probably originates from the Mediterranean, and was likely to have been introduced to the East Coast of the USA by shipping traffic around the 1830s (Stoner 2002, Jewett et al. 2005). *B. schlosseri* can usurp space on hard substrates, restricting recruitment of other taxa to adjacent unoccupied areas (Cordell et al. 2013). The colonial ascidian *Botrylloides violaceus* was the next most common sessile animal on settlement plates. As for *B. schlosseri*, *B. violaceus* is a widespread non-native invader, which is more common in highly modified environments than in natural environments (Simkanin et al. 2012, Cordell et al. 2013). The solidarity ascidian *Molgula manhattensis* was the next most common sessile animal on settlement plates. The geographic origins of *M. manhattensis* are unknown, with a possible origin on either the western or eastern side of the North Atlantic Ocean (Jewett et al. 2005). It is now found worldwide, able to tolerate harsh abiotic conditions and establish in highly modified environments (Jewett et al. 2005). The most common non-ascidian animal on settlement plates was the bryozoan *Membranipora membranacea*: a relatively recently arrived invasive species, with the ability to spread rapidly owing to the lack of competitors and predators in introduced habitats (Berman et al. 1992). Although not as common as the above taxa, barnacles (*Amphibalanus* sp.), hydroids, sponges and polychaete worms also colonized settlement plates. *Amphibalanus* sp. have been collected from highly polluted estuaries around the world (e.g. Ho et al. 2009, Farrapeira et al. 2010, Gall et al. 2013). Hydroid and sponges are common in estuaries and some taxa tolerate pollution (Boero 1984, Turque et al. 2010). Polychaetes are typically abundant in benthic communities, with some taxa highly tolerant of pollutants, including heavy metals, organic contaminants and pesticides (e.g. Dean 2008). During the present study, the polychaete *Polydora ligni* built tubes on settlement plates, but it was not included in assessments of abundances, as it was difficult to distinguish tubes containing live worms from uninhabited sediment in photographs. The presence of *P. ligni* has previously been used as an indicator of estuarine pollution (Dean 2008). Dominance of sessile communities on settlement plates deployed in New York – New Jersey Harbor by this 'rogues gallery' of hardy taxa indicates that the ambition to facilitate establishment of desirable ecological communities should be tempered with the reality that many factors operating across multiple scales will hinder such efforts.

At each shoreline, similar sessile communities colonized settlement plates, regardless of whether they were made from acrylic, unglazed ceramic, slate or wood. This contrasts with past studies showing that communities on wood settlement plates were significantly different from those on sandstone or concrete plates (Glasby 2000). It has also been demonstrated that the settlement of some taxa is influenced by the color of surfaces available for colonization (Swain et al. 2006). The orientation, shading and sedimentation of settlement plates can also influence the structure of communities which colonize them (Glasby 2000, Irving and Connell 2002). The redesigned colonization devices can hold four separate settlement plates and allow colonization of both sides of each plate: two plates are positioned on the most exposed side of the device, with another two plates positioned in a different orientation and under the first two plates (see Reid et al. 2015). Different orientations of settlement plates within each colonization device create variable exposure of colonization surfaces to sedimentation and shading, as would occur in different microhabitats on three-dimensional shorelines. This may facilitate colonization by more diverse sessile communities, where they exist, and provide greater sensitivity for detecting differences in taxon richness associated with increased physical complexity of shorelines. However, this remains to be tested, as no communities have yet been identified from the redesigned colonization devices.

Intertidal surveys

Physical habitat complexity was not the major factor limiting diversities and abundances of intertidal communities on some shorelines in New York – New Jersey Harbor, with very depauperate intertidal communities occurring on riprap shorelines in the upper Harbor that were covered in large amounts of deposited fine sediment. Surveys of intertidal communities using quadrats was not useful for detecting differences in the relative habitat values of hard shorelines differing in physical habitat complexity in New York – New Jersey Harbor.

Most past studies comparing the relative habitat value of seawalls and natural rock shorelines have been conducted in the intertidal zone: providing evidence of consistent differences in community structure associated with differences in physical habitat complexity and types of material in those shorelines (e.g. Moreira et al. 2006, Chapman and Bulleri 2003, Bulleri and Chapman 2004, Bulleri et al. 2005). However, in New York – New Jersey Harbor intertidal communities were depauperate on some shorelines, regardless of their physical habitat complexity. In the upper Harbor, much deposited fine sediment in intertidal zones of shorelines likely limited their suitability for taxa otherwise adapted to live in such habitat. Further, the intertidal zone of most vertical seawall shorelines is not easily accessible from the landward side, making the use of quadrats to quantify intertidal communities difficult. It is unlikely that assessments of intertidal communities will be useful for inclusion in a readily repeatable protocol to assess the relative habitat values of the hard shorelines in New York – New Jersey Harbor.

Bivalve surveys

No differences in the diversities or abundances of bivalves associated with the physical habitat complexity of shorelines were detected using scouring pads to assess their recruitment in New York – New Jersey Harbor.

Plastic scouring pads are inexpensive and easily obtained, and past studies have shown that they are preferentially colonized by some bivalve species that do not readily attach to hard settlement plates (e.g. Menge et al. 1994, Underwood and Chapman 2006, Navarrete et al. 2008). However, during the present study, rather than facilitating colonization by bivalves, the communities in scouring pads were dominated by amphipods and polychaete worms. Few bivalves colonized scouring pads. It is unlikely that scouring pads will be useful to sample bivalves as part of a readily repeatable protocol to assess the relative habitat values of the hard shorelines in New York – New Jersey Harbor, although more bivalves may have colonized the scouring pads at other times of the year than when the present study was done.

Subtidal photoquadrats

Photoquadrat surveys did not allow detection of differences in the diversities and/or abundances of subtidal invertebrate and algal communities between hard shorelines differing in physical habitat complexity in New York – New Jersey Harbor, owing to highly turbid waters preventing the capture of high resolution underwater photographs.

Much work has been dedicated to the development of using underwater photoquadrats to conduct surveys of macrobiota on hard substrate. In some environments, photoquadrat surveys can be practical, rapid, inexpensive and provide more information than direct observation or hand-collected biomass samples (e.g. Bohnsack 1979, Van Rein et al. 2009). For example, photoquadrats are commonly used to survey coral habitats in clear tropical waters (e.g. Molloy et al. 2013). They have also been successfully used in urban estuaries to assess the relative habitat values of different human-made habitats for epibenthic communities (Connell and Glasby 1999, Glasby 1999, Knott et al. 2004). However, photoquadrats are feasible only in locations where water clarity allows high resolution images to be captured. The highly turbid waters in New York – New Jersey Harbor prevented capturing underwater photographs of high enough resolution to enable reliable identification of epibenthic communities. It is unlikely that photoquadrats of shoreline substrates will be useful for inclusion in a readily repeatable protocol to assess the relative habitat values of the hard shorelines in New York – New Jersey Harbor.

Fish surveys

No differences in diversities or abundances of fish associated with the physical habitat complexity of shorelines were detected using unbaited minnow traps in New York – New Jersey Harbor.

Owing to numerous abiotic factors and fish behaviors, including variability in river flows, plus the importance of migratory and transient species, there is much intra- and interannual variation in the structure of fish communities associated with nearshore habitat in New York – New Jersey Harbor (Hurst et al. 2004). During the present study, few fish were captured in unbaited minnow traps deployed on hard shorelines in New York – New Jersey Harbor (i.e., six individuals captured across

47 minnow traps). Many more small fish were captured in the plastic mesh netting of colonization devices (i.e., 38 individuals, each < 100 mm, captured across 31 colonization devices). Fish taxa captured on shorelines during the present study were oyster toadfish (*Opsanus tau*), Gobiosoma sp., tautog (*Tautoga onitis*), skilletfish (*Gobiesox strumosus*), summer flounder (*Paralichthys dentatus*), American eel (*Anguilla rostrata*) and northern pipefish (*Syngnathus fuscus*). Many more taxa have been captured along the nearshore zone in New York – New Jersey Harbor during surveys conducted for over 20 years by The River Project (i.e., 50 species, see Table A5).

The eleven fish species specified by Bain et al. (2007) as being related to ecosystem rehabilitation goals and having societal or management value in the Hudson – Raritan estuary include four species captured on hard shorelines during the present study (i.e., *O. tau*, *T. onitis*, P. *dentatus* and *A. rostrata*). Although *O. tau* are relatively common in other estuaries, its abundance has been reduced in the Hudson – Raritan estuary, and it may be desirable to apply management actions to increase its abundance owing to its potential role in reducing predation of oysters by crabs (Bain et al. 2007). Little is known about the habits of juvenile *O. tau* (Bain et al. 2007). *T. onitis* and *P. dentatus* are popular sport and food fish that inhabit shallow estuarine waters as juveniles (Bain et al. 2007). Adult *T. onitis* continue to preferentially live around structured habitats, such as oyster beds, rocky reefs or man-made structures (Bain et al. 2007). *A. rostrata* is catadromous and spawns in the Sargasso Sea before moving to estuaries as glass eels: males may remain in estuaries, whilst females spend most of their adult life in fresh water (Bain et al. 2007). *A. rostrata* prefer habitats enabling shelter in the dark, although its habitat uses are poorly known in the Hudson – Raritan estuary (Bain et al. 2007).

The presence of juvenile fish along shorelines may indicate that hard shorelines can act as nursery habitat for some fish taxa. More comprehensive surveys would be required to determine how fish use hard shoreline habitats and respond to the designs of additional habitat complexity that are feasible to use in New York – New Jersey Harbor. Pilings without shading from piers provide valuable habitat for several fish taxa in New York – New Jersey Harbor, whilst *A. rostrata* is distinct in preferring shaded habitat under piers (Able et al. 1998). Comprehensive fish surveys along hard shorelines would require considerable effort, which likely precludes their inclusion in a readily repeatable protocol. Comprehensive fish surveys should be performed at a subset of shorelines alongside the use of colonization devices, to determine whether particular subtidal communities in devices are consistently associated with particular fish taxa.

Fish food index

No differences in the amounts of potential fish food associated with the physical habitat complexity of shorelines were detected by development of a fish food index for subtidal invertebrate and algal communities in New York – New Jersey Harbor.

Published information on the feeding preferences of those fish likely to occur in New York – New Jersey Harbor only allowed for fish food index values to be assigned at broad taxonomic levels for invertebrates and algae. The lack of specific information about preferences of fish to eat invertebrate and algaeidentified during this study resulted in most taxa being assigned to the generic category of 'eaten by fish, but not a preferred food source' (Table 11). Further, calculation of fish food index scores for each shoreline were crude given that assigned fish food index values were multiplied by abundances of each taxa, rather than biomass (which would be preferable, but would require additional allocation of resources to measure accurately), as well as the lack of species-specific information about fish feeding preferences. Given the difficulty in detecting differences in taxon richness for epibiotic communities with low diversities, it may prove fruitful to develop functional measures to facilitate detection of differences between such communities (Gibson et al. 2000). However, determining the appropriate suite of functional measures and/or species traits that will be useful in detecting relative habitat values on hard shorelines will require considerable further study. Trophic relationships are amongst the most commonly studied functional measures in aquatic ecosystems, but species-specific information is still lacking.

Restoration lessons from freshwater ecosystems and the 'Estuarine Quality Paradox'

The focus of the present study was the development of readily repeatable and ecologically meaningful methods to assess the relative habitat values of urban shorelines differing in physical habitat complexity in New York – New Jersey Harbor. Novel colonization devices were developed, as they can readily be replicated for shoreline assessments by different organizations, can safely be deployed on vertical or near vertical shorelines, and provide a standardized measure of the subtidal invertebrate and encrusting algal communities on hard shorelines (see Reid et al. 2015). There are several assumptions when using colonization devices deployed in subtidal habitats to assess relative habitat values of hard shorelines in highly urbanized and industrialized estuaries: i) 'relative habitat value' refers to the habitat values for taxa able to reach the shoreline and cannot account for taxa that could inhabit the shoreline, but are unable to reach it owing to barriers to connectivity or other broad-scale constraints; ii) accurate assessment of subtidal communities on different shorelines at one point in time reliably estimates the relative habitat value of shorelines for communities that change seasonally and successionally; and, iii) the community which colonizes each device deployed on a hard shoreline provides a reliable proxy measure of the subtidal community occurring on the immediate surrounding shoreline at the time of deployment. To be useful, assessments of the communities on colonization devices should enable detection of differences in communities between shorelines, where such differences occur.

It is also commonly assumed that rehabilitation of physical habitat complexity will lead to reestablishment of biological communities (Palmer et al. 1997, Bond and Lake 2003). Unfortunately, in New York – New Jersey Harbor there are multiple constraints on shoreline rehabilitation. Given that more work has been dedicated to the rehabilitation of streams than estuarine shorelines, it may be instructive to consider which of those lessons learned in streams are applicable to estuarine shoreline rehabilitations. The 'Field of Dreams hypothesis', based on the catchphrase from the 1989 movie of the same name, i.e. "if you build it, they will come", suggests that if local habitat is rehabilitated, natural communities will move into the habitat and become reestablished (Palmer et al. 1997). Habitat rehabilitation is associated with enhanced biotic diversity in some, but not all, freshwater ecosystems (Palmer et al. 1997). In urban streams the constraints to reestablishment of biotic diversity in rehabilitated reaches include limited spatial extent of restoration, lack of connectivity to similar habitats that may provide sources of colonists, legacies from past intensive land use and negative effects of invasive taxa (e.g. Palmer et al. 1997, Bond and Lake 2003, Violin et al. 2011). The 'urban stream syndrome' describes the ecological degradation of streams flowing through urbanized watersheds, characterized by reduced taxon richness and increased dominance of communities by hardy taxa (Walsh et al. 2005). Whilst the water quality in New York – New Jersey Harbor has improved over the past century (NY DEC 2012), it is still subject to high magnitudes of disturbance from urbanization and industrialization, as well as legacies from past human activities. Previous studies have shown that, even where it is possible to reduce human disturbances and improve local habitat, past environmental conditions continue to influence present-day communities in streams (Harding et al. 1998). Although these constraints have been explored more in freshwater ecosystems than estuaries, they may be more severe in highly modified estuaries than in upstream ecosystems where it is possible to reconfigure the entire in-stream and riparian habitat along relatively narrow channels and apply whole watershed management to reduce impacts above such channels. A further complication in estuarine environments is the 'Estuarine Quality Paradox' (Elliot and Quintino 2007): because estuarine shorelines are naturally highly variable physicochemically and urban shorelines are further disturbed by multiple anthropogenic stressors, the biota on these shorelines may be adapted to stress and demonstrate limited response to changes in the physical complexity of local shoreline habitat. Determining the suite of indicators likely to provide useful information for determination of shoreline habitat quality in estuaries is likely to be challenging.

Addressing the challenges of shoreline rehabilitation and monitoring in urbanized estuaries requires that realistic goals (see Palmer et al. 1997, Ehrenfeld 2000) for all future rehabilitations be clearly articulated prior to applying management actions, and adequate monitoring be conducted post-rehabilitation. Learning from both 'successful' and 'failed' actions will facilitate the effective and efficient use of resources devoted to shoreline management, but will be possible only if rehabilitation goals are stated *a priori*. Depending on the stated goals, it may be appropriate to monitor shoreline rehabilitations using a suite of methods for assessments of ecological values of hard shorelines, e.g. monitoring birds or mammal use of shorelines, in addition to invertebrate, algae and fish. Bain (2011) provides an example of how to set clear and measurable fish community targets after identifying barriers which can be overcome by applying management actions, with a case study from the New York – New Jersey Harbor. Further, in addition to ecological goals, shoreline rehabilitation or revitalization may have specific aesthetic and educational goals to make shorelines more accessible and attractive for human use.

Recommendations

One of the common objectives of ecosystem rehabilitations is to provide habitat complexity to facilitate establishment of diverse and resilient communities (Palmer et al. 1997). Ecological monitoring is required to determine whether shoreline physical habitat complexity favors particular desirable taxa in highly urbanized and industrialized estuaries. This study was the initial step in the development of a protocol (Reid et al. 2015) to allow assessment of the relative habitat values of hard shorelines varying in physical habitat complexity in New York – New Jersey Harbor for subtidal communities. Devices designed to allow colonization of standardized surface areas over a standard duration of deployment were able to be used on different hard shoreline types in New York – New Jersey Harbor. Colonization devices were colonized by subtidal mobile and sessile invertebrate communities from hard shorelines, allowing a standardized assessment of these communities in the Harbor.

Whilst it was promising that the data from colonization devices was able to be used to consistently detect differences in the community structure between locations (i.e., colonization device able to be used to detect differences in shoreline communities owing to differences in local conditions and resources), it was not possible to demonstrate that more physical habitat complexity on riprap shorelines consistently favors the establishment of more diverse communities than those occurring on seawall (i.e., unclear whether colonization device can be used to detect differences in nearby shoreline communities, where conditions and resources are relatively similar between shorelines, except for differences in shoreline physical habitat complexity, or if such differences in nearby hard shoreline communities exist). The loss of replicate samples hindered the ability to test for differences in community structure between shorelines varying in physical habitat complexity within each location. The colonization device was redesigned to be more robust and reduce losses of replicate samples in the future. But, more work is required to:

i) Determine whether communities that colonize devices are a reliable proxy for communities on shorelines, i.e. the taxon richness and relative abundances of each taxon on colonization devices reflects taxon richness and relative abundances of each taxon on the shoreline where the device was deployed. This will required cross-validation with methods that directly sample subtidal communities from hard shorelines (e.g. suction samplers, photoquadrats and/or scrapes, see Table A2) in locations where multiple methods can be applied. Validation need not occur in New York – New Jersey Harbor, where access to the subtidal region of vertical shorelines and highly turbid water constrain the use of alternative methods. If it is proven that the communities on colonization devices are not a reliable proxy for communities on shorelines, they should not be used for future shoreline habitat assessments;

ii) Determine the optimal materials for use in devices. Plastic mesh netting was colonized by mobile taxa, but different mesh sizes and other materials could be trialed. Although the material used to make settlement plates was not associated with differences in sessile community structure during the present study, a standardized material should be chosen and used in comparable surveys. Unglazed ceramic is a strong candidate for future use as the standardized settlement plate material: it is readily available, was deployed successfully during the present study and past related studies, and is physically similar to the hard substrate on shorelines in New York – New Jersey Harbor. The vinyl-coated steel caging can contain a range of materials and use of unique materials may be warranted for monitoring some shorelines (e.g. oyster shell may be added to monitor shorelines rehabilitated specifically to encourage establishment of oyster populations), but this would preclude comparing data to those studies using standard materials in devices (e.g. plastic mesh netting, unglazed ceramic settlement plates and bricks). Also, it is desirable that no animals escape from devices during retrieval, which would be facilitated by designing netting which could be cinched around the mobile sample just prior to retrieval;

iii) Determine the optimal duration and time of year to deploy colonization devices to facilitate retrieval of communities that are reliable proxies of the communities occurring on hard shorelines;

iv) Determine the response variables useful for detecting differences in communities inhabiting hard shorelines varying in physical habitat complexity, where such differences exist. Given the low taxonomic diversity of epibiota inhabiting hard substrates it may be useful to develop functional measures (e.g. trait analyses, percentage of suspension feeders), in addition to traditionally used response variables such as taxon richness, diversity indices, abundances of indicator taxa, multivariate analyses of community structure, etc. (Gibson et al. 2000). The response variables ultimately used in the protocol should be selected based on their contribution to the assessment of relative habitat values of hard estuarine shorelines, whether their collection can be standardized and reliably collected with minimal training or specialized equipment, the cost of their collection, and ease of interpretation;

v) Determine the replication of colonization devices needed on each shoreline to detect differences in communities inhabiting hard shorelines varying in physical habitat complexity, where such differences exist, using power analyses of a full dataset of assessments;

vi) Build databases with information about the communities occurring on hard shorelines in different ecoregions across New York – New Jersey Harbor. This will facilitate determination of the types of taxa that may be favored by shoreline rehabilitations in different locations across the Harbor. Such information is necessary to guide addition of habitat complexity that will aid reestablishment of desirable taxa in favor of undesirable taxa;

vii) Develop standardized methods to monitor other variables that may benefit from hard shoreline rehabilitations, including ecological benefits (e.g. habitat for fish and/or birds), as well as social benefits (e.g. increased amenity values for local communities), and

viii) Ensure that all future hard shoreline rehabilitations are guided by clearly articulated and realistic goals, with appropriate and adequate monitoring to determine whether goals are addressed.

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Appendix 1: Literature review

Review of the effects of stabilization on the habitat value of shorelines in highly urbanized estuaries

Natural estuarine shorelines support biotic communities that perform a range of important ecological functions (Kennish 2002, Barbier et al. 2011, Strayer 2012). However, the structure and function of shorelines are altered by human engineering (Moschella et al. 2005, Bulleri and Chapman 2010). Human populations are attracted to estuaries in high densities and in these highly urbanized regions much of the shoreline has been engineered for stability and to improve human access. In the New York–New Jersey harbor estuary, the majority of shoreline has been stabilized over the past two centuries (Squires 1992, City of New York 2013). Further, the use of hard coastal defense structures is predicted to increase in response to forecast rises in sea level and increased magnitude of coastal storms owing to climate change (Bulleri and Chapman 2010), despite the recognition that soft infrastructure will be more sustainable solution in the face of future storm surge and sea level rise (Coleman 2012). In developing traditional, hardened shoreline stabilization techniques, potential effects on the shoreline's ecology were not a primary concern in the design and materials used (Chapman and Underwood 2011); rather, the major considerations have been in ensuring protection of coastal land using sound engineering principles (Chasten et al. 1993, de Pippo 2006). Over the past 40 years, however, there has been increasing interest in restoring aquatic ecosystems (e.g. Bohn and Kershner 2002, Elliot et al. 2007), coinciding with much of the existing shoreline stabilization infrastructure now requiring maintenance or rebuilding. As shorelines are replaced, new ecologically enhanced designs may be viable in some situations. Considerably more estuarine research has been devoted to restoration of soft-sediment ecosystems than those with hard substrates; however, it is well-recognized that in many locations including those that are heavily developed, returning hardened shorelines to unconsolidated sediment is not a feasible solution (Chapman and Blockley 2009, Bulleri and Chapman 2010, Browne and Chapman 2011). Increasing the habitat complexity and/or using more natural materials on hardened shorelines, as an alternative measure to full restoration, may also prove beneficial to natural ecological communities (Airoldi et al. 2005, Bulleri and Chapman 2010, Chapman and Underwood 2011).

Ecologically enhanced shoreline designs require additional resources beyond those needed in traditional stabilization designs (Airoldi et al. 2005, NYC Parks and Recreation and Metropolitan Waterfront Alliance 2010, de Pippo 2006, Chapman and Underwood 2011). Given the lack of monitoring of these enhanced shorelines, it is unclear whether this investment is justified in terms of ecological benefits. It is also recognized that shoreline communities in highly urbanized estuaries will be influenced by a wide range of variables that operate across different spatial and temporal scales, and may not directly influenced by local habitat availability (e.g. Airoldi et al. 2005, Elliot and Quintino 2007). Although data on specific lengths of individual shoreline enhancements are minimal in most locations, many structural modifications such as seawalls, jetties, groynes and riprap tend to be constructed at a localized scale (10s–100s of m to 1 km; e.g. Williams and Thom 2001, Airoldi et al. 2005, de Pippo 2006, Friends of the San Juan 2010, Borsje et al. 2011). However, the potential benefits for estuarine communities in using ecologically enhanced designs and more natural materials to improve local and meso-scale habitat availability in shoreline restorations are rarely evaluated in a robust scientific manner; this lack of data is particularly evident in highly urbanized regions where shorelines are influenced by multiple human impacts, such as the New York–New Jersey Harbor Estuary. The aim of this review, therefore, is to provide an overview of: the important ecological functions of temperate estuarine shorelines; effects of urbanization on estuarine ecosystems; the factors likely to influence shoreline communities that are not directly related to local habitat structure; the types of organisms that occur on hardened shorelines and consideration of whether these are likely to be influenced by local habitat structure; the types of surveying methods used to measure these groups of organisms, and past assessments of relative habitat value of natural and engineered shorelines. This review was done to ensure that the latest scientific knowledge was utilized in the development of a habitat assessment protocol for hardened shorelines in the New York–New Jersey Harbor Estuary region.

Physical and chemical factors likely to influence estuary shoreline habitats

In developing biological information, it is imperative that the physical and chemical habitat be carefully measured and documented. Information such as salinity, depth, sediment grain size, and water quality (including pH, temperature, DO, nutrients, and toxicants) is essential to proper classification of the waters for comparison and to the potential subsequent investigation of possible causes of degradation. (Gibson et al. 2000). The New York Harbor is a unique physical setting, flushed by tidal exchange with the New York Bight and Long Island Sound and fed by tributaries of the Hudson and Raritan estuaries (Geyer et al. 2006). Knowledge of the sources of chemical and particulate constituents of the aqueous and sedimentary environments is important to assess underlying causes of the presence or disappearance of taxa. These sources can be assessed through chemical and isotopic composition (Chilrud 1996).

The New York Harbor and lower Hudson Estuary have a long history of biologically important contamination (Ayres et al. 1986, Ayres et al. 1988, Lee et al. 1982, Bopp et al. 1989, Gottholm et al. 1993, Brosnan 1991, Clark et al. 1992, Phillips et al. 1996, Rod et al. 1989). The chemical state of the aqueous environment has a first-order impact on plant and animal colonization and growth, and any ecological study of built or natural shorelines should include sampling for basic water chemistry: nutrients, temperature, salinity and dissolved oxygen at the least. Nutrient loading, in particular, is a critical factor in the ecological health of Harbor systems (O'Shea and Brosnan, 2000) which has been an intense focus of publicly-funded research (Brosnan and O'Shea 1996 a, b) and remediation. Heavy nutrient loads leads to algal blooms and hypoxia, which can inhibit the colonization of shorelines by certain taxa, and open ecological niches to others, including at the micro-biological level (Findlay and Sinsabaugh 2006, Findlay et al. 1998, Clark et al. 1995, Cole et al. 2006). Nutrient loading in the New York-New Jersey Harbor area is known to be principally from municipal point sources and secondarily from tributaries, which carry their own municipal loadings as well as fertilizer runoff from suburban lawns and farms (Lampman et al. 1999, Malone et al. 1982). Nonetheless, the relationship between nutrient inputs to surrounding watersheds and carbon cycle dynamics in the river is not well understood (Arrigoni et al. 2008).

The shore zone (the region closely adjoining the shoreline in which strong and direct interactions tightly link the terrestrial ecosystem to the aquatic ecosystem, and vice versa; Strayer and Findlay 2010) is one of the most active and valuable ecosystems of the world. Its characteristically heterogeneous physical structure (resulting from the close juxtaposition of land, water, and air; and the sculpting power of currents, waves, and wind) and high availability of water, light, and allochthonous organic matter (wrack) often result in high biodiversity, primary production, and nutrient processing on both the land and water side of the shore zone. Specifically, shore zones often contain specialized species of plants, algae, birds, fishes, and invertebrates that are scarce or absent outside of shore zones (Bertness 1999, Thompson et al. 2002, McLachlan and Brown 2006, Airoldi and Beck 2007, Strayer and Findlay 2010). Local biodiversity of both these specialized species and more generalist species on both the land and water side of the shore zone is often high (Obrdlik et al. 1995, Nilsson and Svedmark 2002, Strayer et al. 2012, 2014a) and is affected by the physical complexity (Jenkins and Wheatley 1998, Pollock et al. 1998, Jennings et al. 1999, Barwick 2004, Moschella et al. 2005, Brauns et al. 2007, Strayer et al. 2012, 2014b), hydrological and tidal regimes (Keddy and Reznicek 1986, Hill et al. 1998, Naiman et al. 2005, Strayer and Findlay 2010), elevation and bathymetry (Strayer and Smith 2000, Bulleri et al. 2005), exposure and disturbance regime (Kennedy and Bruno 2000, Brown and McLachlan 2002, Strayer et al. 2012, 2014a), inputs of organic matter (Backlund 1945, Tolley and Christian 1999, Minchinton, 2002, Rossi and Underwood 2002), inorganic nutrients (Bertness et al. 2002, Kraufvelin et al. 2006, Chambers et al. 2008), grain size of soils and sediments (Barton and Hynes 1978, McLachlan 1983), and biogeographic region of the shore zone (Strayer and Findlay 2010).

Estuarine biota associated with hard shoreline habitats

In coastal environments, habitats comprised of hard substrata are one of a range of substrate types that harbor unique associated assemblages of invertebrates, fish and algae. Hard substrata habitats may be naturally occurring (e.g. rocks, coral reefs) or artificial (e.g. breakwaters, jetties, seawalls), with a growing number of studies suggesting that the origin and composition of hard substrata can have profound effects on the resident assemblages (e.g. Connell and Glasby 1999, Connell 2000). As urban coastlines become increasingly developed, concerns have been raised about the effects of artificial substrata on natural assemblages of organisms (Bacchiocchi & Airoldi 2003, Bulleri et al. 2000, Chapman & Bulleri 2003, Connell & Glasby 1999, Davis et al. 2002). A necessary initial step to assessing the effects of stabilized shorelines on coastal ecosystems is to compare assemblages of organisms that occur on natural shorelines and stabilized shorelines.

Assemblages occupying marine hard substrata have been the focus of intensive study and are known to provide important ecosystem services such as substrate stabilization (e.g. Meyer et al. 2008), water filtration (Dame 1984), and habitat (Coen & Luckenbach 2000, Luckenbach et al. 2005) and food provisioning (Able et al. 1999, Conover and Hurst 2002). Moreover, they support assemblages of organisms that can be extremely diverse and are often unique along a particular stretch of the coastline. If these populations are to persist along shorelines, then the conditions and resources in that habitat need to be suitable, or continual recruitment from a nearby source population needs to be maintained. These factors are dependent on the life histories and dispersal abilities of each taxon utilizing the shoreline, such that shoreline modification may have differential effects on taxa within assemblages. Groups of organisms utilizing hard shorelines are diverse not only in the species composition but also in their life histories and their degree of occupancy of the actual structure of the shoreline. The degree to which shoreline modifications may affect community structure may depend in part on the residency patterns of species within these communities, on their dispersal capabilities and the location and condition of source populations, and on the trophic interactions between species. For example, we may expect larger fish and wading birds that may be transient or migratory to be less affected by localized structural changes to hardened shorelines than sessile organisms that spend their entire adult life attached to the shoreline structure. However, where sessile organisms on hard substrata also represent important sources of food for more transient organisms, changes to biota as a result of shoreline modification can have more lasting impacts on community structure. Larvae and spores of rocky substrate organisms are typically planktonic; therefore, recruitment may also be strongly influenced by factors outside the estuary (e.g. Gibson et al. 2000).

Primary occupants of coastal hard substrata often comprise sessile invertebrate taxa, which spend a portion of their life-cycle fixed to the substrate. These organisms are often spatially dominant, particularly in the low subtidal and shaded environments (e.g. Jackson 1977, Keough 1983, 1984, Sebens 1986), recruiting to available space on hard substrate habitats following a mobile larval phase. In assemblages of sessile invertebrates on hard substrata, organisms are commonly organized by functional groups of species that utilize and affect their environment similarly (Woodin and Jackson 1979) and may be based by feeding type (e.g. filter feeder, suspension feeder) or by body plan (e.g. solitary or colonial) (Jackson 1977, Woodin and Jackson 1979). Hard substrata communities are successional, often beginning with species that are strong recruiters and leading to dominance by strong competitors and long-lived species (Greene and Schoener 1982, Keough 1983, Breitburg 1985). The seasonal recruitment of many hard substrata species also leads to seasonal patterns in assemblages. Furthermore, a consequence of the biphase life cycle of these sessile species, where they spend a large proportion of time as planktonic larvae, is that community development may also be influenced by characteristics of the open water column; for example, flow rates, water quality variables and the location of other populations and sources of larvae. It is important, then, to consider the possible influences of factors both proximal and remote to the shoreline itself in examining shoreline assemblage development and composition.

Few studies have evaluated hard substrata communities in the local waters of the New York–New Jersey harbor, with the report by Levinton et al. (2006) providing the most extensive data to date from the lower Hudson River. This study, which monitored recruitment from late spring to early fall over two years (2002–3), identified 31 invertebrate taxa within hard substrata assemblages, 19 of which were sessile invertebrates, including colonial tunicates (e.g. *Botryllus schlosseri*), solitary tunicates (e.g. *Molgula manhattensis*), bryozoans (e.g. *Membranipora membranacea, Bugula simplex*), bivalves (e.g. *Mytilus edulis, Crassostrea virginica*), and barnacles (*Balanus eburneus, Balanus improvisus*). Typical mobile species that are commonly associated with sessile assemblages on hard substrata, including those in the lower Hudson, include gammarid and caprellid amphipod and predatory nereid polychaete worms. Other mobile species utilizing these environments included mud worms living in sediment that accumulated on the hard substrata and gastropods grazing on macroalgae and suspension feeding crustaceans (Levinton et al. 2006, Table 1). Water quality within the lower Hudson was also determined to have improved markedly over the past several decades, allowing more sensitive organisms, such as the commercially and ecologically important eastern oyster *Crassostrea virginica*, to colonize the river. Successful recruitment of *C. virginica* spat depends upon many factors including light (Michener and Kenny 1991), tidal elevation (Bartol and Mann 1997) and substrate complexity (Coen & Luckenbach 2000, Luckenbach et al. 2005). Eastern oysters are of particular interest in this region because of their commercial value, but they also contribute to filtering sediment from the water column and attracting fish species (Levinton et al. 2006).

Hard surfaces that are in the shallow subtidal or have an abundance of light penetrating the water column may instead be dominated by algal species, with macroalgal communities comprising various species of red, green or brown algae (e.g. Foster 1990, Schiel 1990, Sanderson et al. 1997). These macroalgal communities, the most prominent examples of which are the kelp forests of cool-water temperate upwelling regions across Australia, New Zealand, North and South America and Japan, provide a wealth of ecosystem services, including primary production, habitat and food provisioning (e.g. Dayton 1985, Steneck et al. 2002, Estes et al. 2004). However, communities dominated by kelp, *Sargassum* spp. and other macroalgae, as well as coralline species, are typically more likely to occur on natural rocky shore substrates (e.g. Glasby et al. 2007, Benedetti-Cecchi et al. 2001). In contrast, anthropogenic activities may reduce the cover of these macroalgal species (Benedetti-Cecchi et al. 2001) and algal species recruiting to hard substrata in urban systems are often limited to ephemeral taxa such as *Ulva* spp., *Enteromorpha* spp. and other filamentous green algae, as well as other small colonies of red and brown algal species (e.g. Glasby and Connell 2001). The use of artificial substrata in shoreline modifications may also facilitate the spread of invasive algal species along developed coasts (Bulleri and Airoldi 2005).

The complex communities that develop on hard subtidal substrata may also support smaller resident fish species, with more transient mobile fish potentially using the assemblages for food and short-term habitat. Whilst negative associations with shoreline alterations and available subtidal structural habitat have been assessed for some fish communities (Bilkovic et al. 2006), the majority of work on shoreline communities has focused on invertebrate assemblage dynamics (see Table 2).

Table A1. Species list from Levinton et al. (2006) study in the lower Hudson. Species listed were found on ceramic settlement tiles in 2001.

Species	Common Name
Microciona prolifera	Read Beard Sponge
Haliclona loosanoffi	Loosanoff's Haliclona
Aurelia aurita	Moon Jelly
Mnemiopsis leidyi	Comb Jelly
Obelia geniculata	Knotted Thread Hydroid
Gonothyrea loveni	Hydroid
Campanularia flexuosa	Hydroid
Euplana gracilis	Flatworm
Bugula neritina	Bushy Bryozoan
Membranipora membranacea	Lacy Bryozoan
Botryllus schlosseri	Golden Star Tunicate
Molgula manhattensis	Common Sea Grape
Botrylloides violaceus	Orange tunicate*
Polydora ligni	Mud Worm
Amphitrite ornata	Ornate Terebellid Worm
Spirorbis spp.	Hard Tube Worm
Nereis sp.	Nereid worms
Nephtys incisa	Common Painted worm
Hydroides dianthus	Carnation Worm
Crepidula fornicata	Slipper Limpet
Crepidula plana	Eastern White Slipper Shell
Mytilus edulis	Blue Mussel
Crassostrea virginica	Eastern Oyster
Balanus improvisus	Bay barnacle
Balanus eburneus	Ivory barnacle
Jassa marmorata	Tube-building amphipod
Gammarus spp.	Scuds
Caprella penantis	Skeleton shrimp
Phloxichilidium femoratum	Sea spider
Idotea metallica	Isopod
Palaemonetes pugio	Grass shrimp
Rithropanopeus harrisii	Harris Mud Crab
Libinia emarginata	Spider Crab
Callinectes sapidus	Blue Crab
Opsanus tau	Oyster Toadfish
Microciona prolifera	Redbeard sponge

Important functions of the shore zones of temperate estuaries

Primary production by aquatic, wetland, and terrestrial plants in the shore zone can be very high (Wetzel 1990, Naiman et al. 2005), and supports local food webs as well as being exported to and used in adjacent ecosystems. Organic matter from this local production, as well as wrack and driftwood carried in from nearby ecosystems and retained by shore zones, is processed in the shore zone, supports food webs, and provides physical structure in shore zones (Backlund 1945, Colombini and Chelazzi 2003, Harris et al. 2014). The high heterogeneity of the shore zone, with its dramatic physical, chemical, and biological contrasts over very short distances, allows redox-sensitive or coupled biogeochemical processes such as sulfate reduction and nitrification–denitrification (Juutinen et al. 2003, Kankaala et al. 2004, Hirota et al., 2007), as well as providing favorable habitat for species that require multiple habitats to complete their life cycles (Strayer and Findlay 2010). The physical (beaches) and biological (algae, rooted vegetation, oyster reefs) structures of the shore zone help to dissipate the energy from waves and currents that impinge on the shore zone (Coops et al. 1996, Strayer and Findlay 2010, Scyphers et al. 2011). Finally, natural shore zones often are important corridors for the dispersal of plants and animals (Jansson et al. 2005). Further information on these and other ecological functions provided by shore zones is available in the general reviews of Bertness (1999), Brown and McLachlan (2002), Thompson et al. (2002), Naiman et al. (2005), McLachlan and Brown (2006), Airoldi and Beck (2007), National Research Council (2007), and Strayer and Findlay (2010).

Effects of urbanization on estuarine shore zones

Humans often alter shore zones to protect human life and property, and to facilitate activities such as shipping, construction of roads and buildings, water-based recreation, and so on. These alterations often produce a characteristic set of changes in the shore zones of urban estuaries, including narrowing and stabilization of shore zones (Tockner and Stanford 2002, Airoldi and Beck 2007, Winn et al. 2005, Miller et al. 2006, Fujii and Raffaelli 2008), changes to the natural hydrological regime (Nilsson et al. 2005), shortening and simplification of the shoreline (Sedell and Froggatt 1984, Tockner and Stanford 2002, Miller et al. 2006), hardening and steepening of the shoreline (Miller 2005, Airoldi and Beck 2007, Strayer et al. 2012), tidying of the shore zone (e.g., removing wrack and driftwood, cutting vegetation; Malm et al. 2004), increasing inputs of physical energy to the shore zone (as a result of increased boat traffic, dredging, and building out into the channel; Strayer and Findlay 2010), pollution by a wide range of substances including xenobiotics (Strayer and Findlay 2010), disturbance from recreational use of the shore zone (Asplund 2000, Pinn and Rodgers 2005, Davenport and Davenport 2006), introduction of nonnative species (Hill et al. 1998, Airoldi and Beck, 2007), climate change (including local heat island effects) (Strayer and Findlay 2010, Kirwin and McGonigal 2013), and construction of buildings and impervious surfaces in and near the shore zone, thereby fragmenting remaining shore zone habitats (Strayer and Findlay 2010).

These changes in turn probably cause a characteristic set of changes to ecological functioning of urbanized shore zones, although an "urban shore zone syndrome" (by analogy to the "urban stream syndrome" of Walsh et al. 2005) has not been systematically described. Nevertheless, the following effects on ecological functioning have been demonstrated in some cases or seem likely. Activities such as narrowing and tidying of the shore zone, shortening and simplification of the shoreline, and increased disturbance from recreation, should decrease local biodiversity and rates of many biogeochemical processes. Changes to hydrological and disturbance regimes, pollution, climate change, and species invasions could substantially change biodiversity and biogeochemical cycling in unpredictable ways, depending on the details of these changes. Steepening shorelines and increasing physical energy inputs to shorelines may select for a more disturbance-resistant biota. Increased cover by impervious surfaces will degrade rates of biogeochemical processes and habitat quality for species, as well as causing rapid runoff of water and pollutants after storms, potentially leading to local erosion or toxicity in the shore zone. Many of these activities should substantially reduce the effectiveness of urban shore zones as dispersal corridors. The net effect of all of these changes on the ecological functioning of urban shore zones may be large and often complex, and should vary from city to city as a function of the specific mix of human activities around that city, and the ecological setting of the city.

Table A2: Methods used in previous studies to compare the relative habitat value of natural and engineered hard shorelines or breakwaters in urbanized estuaries or along coastlines for different aquatic community types. Note: methods were designed for use in addressing the specific research questions of each study, not for use in a repeatable protocol.

Habitat	Community	Method	Location	Reference
Intertidal	Algae and invertebrates	Quadrats or belt transects	Sydney, Australia	Chapman 2003, Chapman and Bulleri 2003, Bulleri et al. 2005, Bulleri 2005b, Chapman 2006, Green et al. 2012
			Northeastern Italy	Bacchiocchi and Airoldi 2003
			Northwestern Italy	Bulleri and Chapman 2004
			Catalan coast, Spain	Gacia et al. 2007
			Denmark, Italy, Spain and UK	Moschella et al. 2005
			Victoria Harbor, Hong Kong	Lam et al. 2009
			Vizhinjam Bay, India	Ravinesh and Bijukumar 2013
			San Diego, California, USA	Davis et al. 2002
			Southern California, USA	Pister 2009
		Quadrats positioned within experimental clearings	Sydney, Australia	Bulleri 2005a, Bulleri 2005b
		Settlement plates	Sydney, Australia	Bulleri 2005a
	Molluscs	Quadrats	Sydney, Australia	Chapman 2006
	Limpets	Quadrats	Sydney, Australia	Moreira et al. 2006
Intertidal mussel beds	Sessile and mobile invertebrates	Scraped from defined area	Sydney, Australia	People 2006
Subtidal	Sessile epibiota	Photoquadrats	Sydney, Australia	Connell and Glasby 1999, Glasby 1999, Knott et al. 2004
			Dubai, United Arab Emirates	Burt et al. 2011

Table A2 cont.

Habitat	Community	Method	Location	Reference
		Settlement plates	Sydney, Australia	Connell 2000, Connell 2001
		Scraped from defined area	Weser estuary, Germany	Wetzel et al. 2014
	Benthic infauna	Suction sampler	Chesapeake Bay, Virginia, USA	Lawless and Seitz 2014
		Corers and grabs	Italy, Spain and UK	Martin et al. 2005
	Hydroids	Diver visual census	Iberian Peninsula, Spain	Megina et al. 2013
	Fish and sea urchins	Diver visual census	Northeastern Italy	Guidetti et al. 2005
	Fish	Diver visual census	Southwestern Italy	Guidetti 2004
		Diver visual census	San Diego, California, USA	Davis et al. 2002
		Enclosure nets and divers	Puget Sound, Washington, USA	Toft et al. 2007
Tidal freshwater	Fish, invertebrates	Electrofishing (fish), D-net and cores (invertebrates)	Hudson River, New York, USA	Strayer et al. 2012

Ecological functions of urbanized shore zones in temperate estuaries

Given the wide range of shoreline types in urban estuaries, and the wide range in ecological functioning of different shore types, it is natural to ask which shore types best perform different ecological functions. Unfortunately, we do not yet have replicated, empirical studies of a wide range of ecological functions measured over a wide range of shore types in urban estuaries around the world, so we cannot yet confidently answer this question. In addition, the ultimate "value" of a particular bit of shore depends on the setting into which it is placed, and the values that different stakeholders place on the mix of ecological functions that a bit of shore zone provides (see Strayer and Findlay 2010 for a more detailed discussion of these points). Despite these caveats, Tables 3 and 4 present very preliminary attempts to summarize the likely value of different shore types for different ecological functions. These should be regarded as hypotheses rather than established facts.

Generally, it seems likely that biodiversity and biogeochemical processes are highest in shore types that are wide and flat (rather than narrow and steep), physically complex and heterogeneous, physically continuous (rather than fragmented by inhospitable habitat like smooth, vertical steel or concrete), well vegetated (preferably with native plants), are capable of retaining wrack and driftwood (which are not then "cleaned up"), where recreational use is modest and/or localized, and where the physical energy inputs and hydrologic regime have not been dramatically altered. It is worth noting than many built shore types (e.g., bulkheads) score poorly on many or all of these criteria.

Table A3. Preliminary assessment of the relative provision of various ecosystem services provided by different kinds of marine shorelines (modified from National Research Council, 2007).

	Shoreline type										
Ecosystem services	Sandy beaches	Sand dunes	Mudflats	Marshes and mangroves	Seagrasses and macroalgae	Bluffs	Bulkheads and seawalls	Revetments	Groins	Breakwaters and sills	Planted marshes /mangroves
Fish habitat	+	+	++	+++	+++	+	+	++	+	+++	+++
Mollusk habitat	+++	+	+++	+++	+++	+	++	++	++	++	+++
Crustacean habitat	++	+	++	+++	+++	+	0	+	+	+	+++
Turtle habitat	+++	++	0	+	++	+	0	0	0	0	+
Bird habitat	++	+++	+++	+++	+	+++	0				+++
Nutrient processing	++	+	++	+++	+++	+	+	+	+	+	+++
Food production	+	+	++	+++	+++	++	+	+	+	+	+++
Wave attenuation	++	+++	+	++	++	+++	+	++	++	+++	++
Sediment stabilization	0	+++	++	+++	+++	0			++	+++	+++
Gas regulation	+	0	+	+++	+++	0	0	0	0	0	+++
Biodiversity	++	+++	++	+++	+++	+++	+	+	+	+	+++
Recreation	+++	+++	+	+++	+++	++	++	+	++	++	+++
Raw materials	+++	+++	+	+++	+++	+++	+	+	+	+	+++
Aesthetic value	+++	+++	++	+++	+++	+++	0	0	0	0	+++

Table A4. Hypothesized ability of different kinds of freshwater shore zones to provide ecological functions (Strayer and Findlay 2010).

	Shoreline type						
Ecosystem functions	Unvegetated mud flat	Unvegetated sand beach	Cobble or bedrock	Vegetated sand or mud	Marsh	Riprapped revetment	Steel or concrete bulkhead
Habitat for aquatic plants	0	0	+	+++	+++	+	0
Habitat for aquatic invertebrates	++	++	+++	+++	+++	++	+
Habitat for fishes	+	+	++	+++	+++	++	+
Habitat for birds	++	++	++	+++	+++	++	0
Energy dissipation	++	++	++	+++	+++	++	+
Primary production	+	+	+	+++	+++	+	0
Retention or decomposition of organic matter	++	++	++	+++	+++	++	0
Nutrient transformation	++	++	++	+++	+++	+	0
Biotic dispersal	++	++	++	+++	+++	++	+

Prevalence and types of shoreline hardening in urban estuaries

Hardening of shorelines, for coastal protection and commercial and recreational purposes, is prevalent in urban estuaries worldwide on sheltered and open coastlines, with a variety of armoring efforts evident in both on-shore and off-shore structures (overview in Dugan et al. 2011). Types of shoreline protection techniques include those that are installed to reduce wave energy reaching the shore, thus providing erosional protection for coastal areas and those that directly modify or harden the shoreline. Examples of the former include detached offshore breakwaters; structures that run parallel to shore and reduce wave energy reaching the shore, as well as groins and low-crest reef breakwaters (Chasten et al. 1993). These offshore structures can impound sediment on the shoreward side of the structure and reduce inshore water movement, with corollary effects on benthic and mobile assemblages. Examples of the latter type of shoreline armoring include seawalls and bulkheads that are designed to redirect and deflect waves and storm surges from coastal areas and to enable moorage of vessels adjacent to land (Mulvihill et al. 1980, Williams and Thom 2001). Seawalls are usually vertical or steeply curved structures composed of hard materials such as timber, concrete or stone with their foundations generally built from the seafloor. These seawalls differ from bulkheads in that the latter are usually built above the mean high water level, with the exception being in some sheltered estuaries and tidal shorelines (Dugan et al. 2011). Thus, seawalls and bulkheads may both provide additional hard substrata as habitat within sheltered estuarine and tidal areas. The scope and extent of shoreline hardening is also extensive across urban areas globally. For example, according to data generated by the EC CORINE program, up to 55% of coastline in European Union are stabilized, with a further 19% of shorelines assessed as having erosional problems and likely to be stabilized as a protective measure (Airoldi et al. 2005). Hard substrate defense structures such as breakwaters, groynes, seawalls and other rock armored structures are an increasingly common feature of the coastal landscape in intertidal and shallow subtidal environments. The MESSINA project reports that around 20% of the European Union coastline is severely affected by erosion, with a wide range of hard engineering methods used offshore (e.g. breakwaters, barrages), low shore (e.g. groynes, revetments), upper shore (e.g. sea walls, revetments) and behind the shore (cliff strengthening, dune building) (de Pippo 2006). Further discussion of the general types of shoreline stabilization methods can be found in Dugan et al (2011), and the U.S. Army Corp of Engineers (2004) provides details on the engineering principles and design guidelines.

In the US, several comprehensive localized assessments and inventories of shoreline modification have been undertaken, with detailed information available for several counties in Washington State. A 2009 assessment of 400 miles of shoreline at the confluence of Puget Sound, Georgia Strait and Strait de Fuca identified shoreline modification as a top threat to marine ecosystems in the region, with shoreline armoring (ranging from marinas and jetties to docks, groins and armored beaches) covering over 18 linear miles of shoreline, with an average of 4 shoreline modifications per mile (Friends of the San Juans 2010). In some counties in Washington State (e.g. King County), armoring covers up to 75–90% of coastlines in Puget Sound, ~29% of shoreline is stabilized, with 1.7 miles newly armored each year (Canning and Shipman 1995). It is estimated that 32% of intertidal and 73% of subaerial wetlands around Puget Sound have been lost to hardening since 1980 (Bortleson et al. 1980). In Chesapeake Bay, an estimated 342 km of tidal shoreline has been altered with riprap (stone revetments and retaining walls (bulkheads) (Bilkovic et al. 2006). Further assessments of regional shoreline modification efforts are found in Zabawa and Ostrom (1982, Chesapeake Bay) and Zelo and Shipman (2000, Puget Sound).

In the New York–New Jersey harbor estuary, surge from coastal storms are recognized as the most significant climate-related risk to coastal areas and parklands in the coming years (City of New York 2013). Although no recent comprehensive inventory of shoreline hardening has been completed within the estuary, coastal protection initiatives outlined by the City of New York to mitigate future storm impacts include hardening or otherwise modifying shorelines, reinforcing or redesigning bulkheads, retrofitting or hardening waterfront park facilities (City of New York 2013). Thus, examination of the potential ecological impacts of shoreline modification is key to developing effective coastal protection and storm mitigation that also minimizes negative impacts on habitat quality and ecological communities.

Assessment of shoreline macroinvertebrate communities

A wide range of methodologies have been developed for ecological assessments of aquatic ecosystems (e.g. Borja and Dauer 2008). Choosing the appropriate suite of measures for each study depends upon the habitat being assessed (e.g. hardened estuarine shorelines), response variables of interest to address research questions (e.g. community structures of biota in response to relative habitat availability of different shoreline types) and the scale at which the ecosystem responds to those attributes being tested (e.g. reach-scale). Aquatic macroinvertebrates are frequently used in ecological assessments. In the US, bioassessment protocols using macroinvertebrates have been developed for wadeable streams and rivers (e.g. Plafkin et al. 1989, Barbour et al. 1999, Houston et al. 2002), and estuaries (e.g. Gibson et al. 2000, Eaton 2001, Llanso et al. 2002, Pelletier et al. 2010). These biological assessments have been integrated into more traditional chemical and physical assessments of ecosystem condition (Gibson et al. 2000, Downes et al. 2002, Borja and Dauer 2008). Macroinvertebrates communities are typically relatively diverse and consist of representatives from different phyla that utilize different habitats, have different life histories and feeding habits, and occupy multiple trophic levels (Hauer and Resh 2006, Strayer 2012). This variability in traits makes macroinvertebrate communities useful for ecological assessments, as it causes different taxa to vary in their sensitivity to ecosystem disturbances (e.g. Conlon 1994, Gaston et al. 1998, Feldman et al. 2000, Downes et al. 2002). Owing to them being relatively sedentary, macroinvertebrates reflect the cumulative effects of human and natural disturbances that influence the local ecosystem over multiple timescales (Gibson et al. 2000). Given their proven utility in ecological assessments and the relatively ease and cost effectiveness of sampling macroinvertebrate communities, there is a large database of information about the effects of disturbances on varying taxonomic groups (Gibson et al. 2000, Pinto et al. 2008). By comparing the structure of macroinvertebrate communities between sites of a given ecosystem type within a bioregion it is possible to determine their relative condition (Gibson et al. 2000, Downes et al. 2002). Bioassessments with macroinvertebrates are recognized as being useful for detecting generalized impairments to ecological condition and vital for comprehensive water resource protection and management (Gibson et al. 2000). Measurement of these communities is also most appropriate for directly determining the relative habitat value of shorelines differing in structural complexity and materials for different macroinvertebrate groups (e.g. Connell and Glasby 1999, Diaz et al. 2004, Chapman and Underwood 2011).

Surveys to assess the relative habitat value of estuarine shorelines should include the sessile (i.e., that component of the epibenthos that attaches to the exposed side of large hard substrates) and mobile macroinvertebrate communities that commonly inhabit these shorelines. Standardized samples must be collected to enable accurate comparison of the condition of different habitat patches by examining the structure of macroinvertebrate communities. Nets of various designs are frequently used when sampling macroinvertebrates from streams (e.g. Houston et al. 2002, Hauer and Resh 2006). In stream reaches with fine sediment and macrophytes, the substrate can be disturbed with a hand-held net to capture the full suite of resident macroinvertebrates (e.g. Barbour et al. 1999, Hauer and Resh 2006). A commonly used method to capture stream macroinvertebrates that preferentially inhabit the underside of larger substrates involves positioning a net downstream of an area of streambed that is vigorous 'kicked' to overturn cobbles (Hauer and Resh 2006). Similarly, the operator may use their hands and/or a hand-held brush to dislodge animals from cobble substrates (e.g. Surber sampler, Hauer and Resh 2006) or large wood (e.g. Johnson et al. 2003). Use of hand-held nets to capture macroinvertebrates provides semi-quantitative samples, given the difficulty in ensuring that the dimensions of habitat from which each sample is collected are precisely equal using these nets

(Lenat 1988). Many assessments of soft-sediment estuarine communities are conducted by collecting benthic grabs or cores of standard dimensions and examining the infauna sieved from unconsolidated fine sediment (e.g. Gibson et al. 2000). Sleds and trawls can be used to sample epibenthos living on the upper surface of soft substrates (Rozas and Minello 1997, Rees et al. 2009). Suction samplers have also successfully been used to sample macroinvertebrates from soft and hard substrates, up to the size of boulders (e.g. Gale and Thompson 1975, Boulton 1985, Heck et al. 1995, Rees et al. 2009, Lawless and Seitz 2014). These methods for collecting samples from estuarine ecosystems provide quantitative data if the exact dimensions of habitat that are sampled can be determined, although often this varies somewhat between samples. However, none of the aforementioned methods of collection are appropriate for surveys of sessile invertebrate communities that are attached to the surface of hard substrates. Further, the above methods would not provide standardized quantitative samples of mobile macroinvertebrates collected from hardened shorelines that differ in habitat structure, from being highly accessible two-dimensional seawalls to highly structurally complex three-dimensional habitats that include large boulder and wood elements where the underside of substrate is inaccessible.

The use of epibenthos living on sediments or structures in ecological assessments is less developed and not as established as the use of infaunal benthic macroinvertebrates from soft sediments or fish (Gibson et al. 2000). However, there is interest in the development of these communities for ecological assessment, as they may require fewer resources than for assessment of infaunal macroinvertebrates, which exist within three dimensional habitat matrices and are generally more diverse than sessile communities, and highly mobile and therefore highly variable fish communities (Gibson et al. 2000). Other potential advantages of using epibenthos in ecological assessments include that the sedentary nature of the community ensures that it can be used as a measure of the conditions and resources of the local habitat over an extended period of time (Rees et al. 2009), and the relative ease of identification of crustaceans, molluscs and echinoderms (Gibson et al. 2000). Epibenthos are known to be sensitive to substrate material (Connell 2001) and structural habitat complexity at different spatial scales (Bulleri and Chapman 2004, Borsje et al. 2011), and therefore are potentially useful in assessment of relative habitat value of shorelines differing in design and types of material.

Belt transects or quadrats of standard area can be used in surveys of the sessile invertebrate communities that occur on the visible surface of large hard substrates. This is a common approach in intertidal habitats (e.g. Connell 1961, Underwood and Chapman 1996, Bulleri and Chapman 2004), and is also feasible for use in surveying sessile communities in subtidal habitats (e.g. Schiel et al. 1995, Glasby 1999, Simkanin et al. 2012). However, conducting underwater surveys whilst SCUBA diving is difficult where there are strong currents and highly turbid water. These conditions occur in much of the New York – New Jersey Harbor estuary (Strayer 2012). Photoquadrats of shorelines reduce the amount of diving time, but the need for specialized training and equipment, logistics and associated costs with using SCUBA divers likely precludes their use in a readily repeatable protocol for ecological assessment. The limitations imposed by using SCUBA divers may be overcome if suitably high resolution images of shorelines can be captured using cameras that are remotely operated by personnel on shorelines. Improvements in the quality of underwater photography have led to increased use of photoquadrats in assessments of subtidal communities (e.g. Connell and Glasby 1999, Smith and Witman 1999, Knott et al. 2004, Preskitt et al. 2004, Burt et al. 2011). Remotely operated cameras have the potential to produce high resolution images for the assessment of subtidal community structure, although their utility for use on hard substratum is presently underdeveloped (Roberts et al. 1994, Rees et al. 2009, Van Rein et al. 2009). Such images need to be of high enough resolution to enable accurate and precise identification of the sessile community. The communities captured in digital photographs are assessed by using digitized grids to estimate the abundances and proportion of area covered by different sessile taxa (e.g. Connell and Glasby 1999, Burt et al 2011). This method has been shown to be effective at reliably estimating the actual abundances of organisms within these assemblages (Benedetti-Cecchi et al. 1996, Fraschetti et al. 2001, Knott et al. 2004, Trygonis and Sini 2012). For ongoing use as a component of a protocol a method to capture high resolution images must be developed that is readily repeatable, able to collect standardized images and cost-effective in terms of equipment and labor.

Another common and established method for assessing the sessile communities on hard substrates in marine ecosystems is the use of settlement plates of standard area (e.g. Keough 1998, Connell 2000, Levinton et al. 2006, Perkol-Finkel 2008). Settlement plates deployed in subtidal habitats are colonized by the mobile planktonic stages of marine biota, which can be retrieved after these larval stages have developed into their identifiable adult forms. To allow sufficient colonization and development time to enable assessment of community structure, settlement plates are deployed typically for durations adequate to allow colonization and growth of a complex assemblage to identifiable stages. These durations can range from around 4 to 8 weeks, depending on the system (habitat, depth, light, substratum), with communities subject to similar conditions and lengths of colonization usually tending to become more similar over time (e.g. Breitburg 1985, Anderson and Underwood 1994, Glasby 1998, Levinton et al. 2006, Underwood and Chapman 2006, but see Chapman 2007). Although the larval dispersal distances of different taxa vary, much of the recruitment into sessile invertebrate communities may occur from nearby populations (e.g. Jackson 1986, Swearer et al. 2002, Shanks 2009). In addition, although local sessile invertebrate community structure is influenced by the arrival of planktonic recruits from the water column, it also depends upon post-settlement survival of these recruits (Connell 1985, Caley et al. 1996, Fraschetti et al. 2002). There is likely to be variation between different hard shoreline types in the strength of processes that influence post-settlement survival, such as physical disturbances, access to refuges, relative predation rates and competitive ability of each population in each habitat (e.g. Breitburg 1985, Connell 1985, Underwood and Fairweather 1989, Walters and Wethey 1996, Sams and Keough 2007).

Sessile biota can be identified directly from plates or from photographs of the colonization surface of plates. To be useful in the assessment of the habitat value of different shorelines, the communities that colonize settlement plates need to be representative of those occurring on each shoreline where they are deployed. In Sydney Harbour, differences in the structure of algae and invertebrate communities colonizing natural rocky shores and engineered seawalls were evident, regardless of whether the community were assessed using experimentally cleared areas directly on the substrate or by introducing vacant sandstone plates (Bulleri 2005a). Assessments of shoreline habitat quality commonly focus on benthic infaunal assemblages (e.g. Bain et al. 2000, Bremner et al. 2003, Bradley 2011). However, in the New York–New Jersey harbor estuary, settlement plates have been used to demonstrate that a diverse suite of taxa are able to quickly settle and form high coverage on this primary space in a matter of weeks (Levinton et al. 2006) A recent study detailing colonization of hard substrata in the lower Hudson showed that the number of species recruiting to artificial settlement tiles over the spring–fall seasons increased linearly over time (Levinton et al. 2006), with 31 taxa present in total, comprising 19 sessile species and 11 mobile species after one month's settlement time. If the communities on settlement plates can provide a robust proxy of the communities on the local shorelines where they are deployed, the advantages of using settlement plates over *in situ* photographs for conducting standardized surveys of subtidal sessile invertebrate communities include that settlement plates are cost-effective, the ability to retrieve plates allows for very high resolution photographs to be taken, and entry into the water is not necessary. Conversely, the disadvantages of using settlement plates in preference to *in situ* photographs are that whilst deployed, the plates may be damaged by large storm or vandalism, the scale of the processes that most influence colonization of plates and local shoreline habitat may not directly coincide, and the early-successional epibenthic communities on settlement plates may not be representative of later succession communities actually occurring along shorelines (Bulleri 2005b). As a first step in the deciding upon the most appropriate method to be used for assessment of sessile communities in a readily repeatable protocol, cross-validation is required to determine whether similar communities can be identified from settlement plates, photoquadrats captured using remotely operated cameras and photoquadrats captured using SCUBA divers.

Similar to the use of settlement plates that allow colonization by sessile invertebrates on hard substrate, different designs of artificial substrate can be used to allow colonization by other invertebrate groups for use in ecological assessments. Whereas two-dimensional plates can be used to allow colonization of most sessile invertebrates, more complex three-dimensional substrate is required to allow colonization by fauna that preferentially inhabit crevices between hard substrates. Scouring pads can be preferentially colonized by some bivalve species that do not readily attach to hard settlement plates (e.g. Menge et al. 1994, Underwood and Chapman 2006, Navarrete et al. 2008). Devices with artificial substrates designed to facilitate colonization by mobile biota have been used in surveys of streams (e.g. Hester–Dendy multiplate substrate samples: Hester and Dendy 1962, Meier et al. 1979, Nedeau et al. 2003) and estuaries (e.g. Anderson and Underwood 1994, Atilla et al. 2005). Artificial substrate samplers all work on the same principle that by providing a uniform surface area for colonization of resident fauna, a standard sample can be collected (e.g. Beak et al. 1973). These types of samplers have proven effective in a range of aquatic habitats, including when used to monitor cryptic fauna on rocky shores of coasts and estuaries (e.g. Myers and Southgate 1980, Vinuesa et al. 2011).

The limitations of using macroinvertebrate communities in ecological assessments include the need for operators to have some taxonomic expertise (Gibson et al. 2000) and the chance of relatively high rates of error in identification and enumeration for species-level identification (Downes et al. 2002). However, useful resources are available to aid in the identification of estuarine invertebrates of north-eastern USA (e.g. Gosner 1978, Pollock 1998, Martinez 1999). In ecological assessments it is desirable to examine multiple communities (Gibson et al. 2000), as the influence of changes in conditions and resources between sites is likely to vary depending on the life history and traits of organisms. Use of diverse communities provides a more robust assessment of ecosystem condition than using a single community, but this must be reconciled with additional costs. Also, it is important to choose ecosystem attributes with adequate background information to allow interpretation. Besides macroinvertebrates, the communities most often used in ecological assessments are plankton, aquatic vegetation and fish (Gibson et al. 2000, Downes et al. 2002, Borja and Dauer 2008). Plankton communities are influenced by conditions and resources in the water column, and are unlikely to be directly influenced by shoreline habitat structure. The diversity of aquatic vegetation is limited on hardened shorelines where the roots of vascular plates are precluded (Strayer et al. 2012), but sessile algae colonize hard surfaces and percent cover can be used in ecological assessments (Gibson et al. 2000). Changes in the cover of different taxa of sessile algae on settlement plates have proven useful in assessments of communities on hardened urbanized shorelines (e.g. Connell and Glasby 1999, Bulleri et al. 2005). Some fish preferentially inhabit shorelines with increased structural complexity (e.g. Gorman and Karr 1978, White et al. 2009) and more natural materials (e.g. Able et al. 1998), so are potentially useful in the assessment of the relative habitat value of different hardened shoreline designs.

Assessment of fish communities associated with local shoreline habitat structure

Fish are important and highly valued components of estuarine communities, with ecological, economic and recreational roles. The abundance and diversity of fish communities is the most broadly understood indicator of ecosystem condition by the general public (Gibson et al. 2000). Fish are important components of estuarine food webs, relatively sensitive to habitat disturbances and may actively avoid less desirable habitats, and are long-lived and therefore the continued presence of a population may be reflective of long-term habitat suitability. However, given their ability to move between habitats, the temporal and spatial variability of fish populations can be very high and require a large sampling effort to accurately characterize community structure (Gibson et al. 2000). The New York–New Jersey Harbor has a high diversity of fishes, with few species expected in the bioregion being absent from these local waters (Berg and Levinton 1985, Bain 2011). However, the shoreline community is highly biased towards fish that are considered to predominantly inhabit open pelagic waters, rather than being directly dependent on shoreline habitat structure (Bain 2011). The fish populations most likely to be influenced by local habitat structure on hard shorelines are relatively small-bodied, such as Cunner (*Tautogolabrus adspersus*), boxfish (Ostraciidae) or pufferfishes (Tetradontidae), or juveniles of larger species such as snapper (Lutjanidae) or basses (Serranidae). Table 5 lists fish species caught by staff at The River Project at Piers 26 and 40 in lower Manhattan from 1988 to 2011, and see Steimle et al. 2000 for a detailed description of fish habitats and diets in the lower Hudson River estuary. Given that most of the resident fish utilizing shorelines are small-bodied, the use of passive minnow traps is more appropriate than larger nets such as seines, which are suitable in open water or along gently sloping shorelines with soft sediment. Further, although they are commonly used in estuaries, seine nets and trawls are known to have low efficiency and require considerable effort to compensate for high variability in catches of small fish (Rozas and Minello 1997). The use of those techniques is not compatible with a readily repeatable protocol for use on hardened shorelines. Enclosure devices are useful for sampling fish from a defined area in shallow estuarine habitats with fine substrates (Rozas and Minello 1997), but are not designed for use on shorelines that are near-vertical and/or have predominantly large rocky substrates. Hand-held nets are unlikely to adequately sample fish along those hardened shorelines where access is difficult owing to steep gradients. Further, fish can avoid capture in nets by using inaccessible crevices between immovable hard substrates on some shorelines, but not others. A previous assessment of fish communities in the NY–NJ Harbor Estuary (Duffy-Anderson et al. 2003) used unbaited benthic traps set on the bottom for 24 hours. This method may capture more species that utilize the sandy bottom habitats, rather than those that may directly associate with the hardened shoreline. Further, potential for variable bias in catch efficiency between sites directly related to the 'treatment' being measured (e.g. habitat complexity of shorelines) must be avoided (Rozas and Minello 1997).

Passive fish capture methods, such as use of minnow traps, can be applied equally across locations that vary in habitat complexity. A consistent catch effort between sites can be ensured by deploying replicate minnow traps placed directly along the shoreline for a consistent duration (e.g. 12, 24 hours). Traps used for the assessment of local habitat should not be baited, as this likely attracts fish from surrounding habitats. Minnow traps have frequently been used for estimating fish population characteristics in lakes (e.g. Tonn and Magnuson 1982, He and Lodge 1990, Macrae and Jackson 2006) and to reliably assess the effect of subtidal habitat on estuarine fish population structure in North Carolina saltmarshes (Irlandi and Crawford 1997). However, there is some differential catch efficiency among species using minnow traps, as shown in Canadian lakes (Jackson and Harvey 1997) and shallow estuarine habitats of Virginia (Layman and Smith 2001). During initial development of the shoreline habitat assessment protocol, minnow traps will be tested to ensure that they capture those fish confirmed to be present along shorelines by divers (see Toft et al. 2007), remotely operated underwater filming and/or by groups conducting regular surveys of fish communities in the Harbor (e.g. The River Project). Given that fish populations are highly spatially and temporally variable, future iterations of the protocol will need to consider the sampling effort required to ensure that the estimation of fish community structure is sufficiently accurate and precise to allow meaningful comparison between shorelines.

Table A5. List of fish species caught at Piers 26 and 40 from 1988 to 2011, courtesy of The River Project.

Note: Fish were caught in baited or unbaited minnow traps set from the pier on the mud bottom, unless marked with symbols:

+ Caught by rod and reel.

++ Caught with dip net.

+++ Caught midwater in minnow trap

FAMILY	COMMON NAME	GENUS	SPECIES
Anguillidae - freshwater eels	eel, American	*Anguilla*	*rostrata*
Congridae - conger eels	eel, conger	*Conger*	*oceanicus*
Clupeidae - herring	herring, Atlantic	*Clupea*	*harengus*
Gadidae - codfishes	tom cod, Atlantic	*Microgadus*	*tomcod*
	Pollock	*Pollachius*	*virens*
	hake, spotted	*Urophycis*	*regia*
	hake, white	*Urophycis*	*tenuis*
Batrachoididae - toadfishes	toadfish, oyster	*Opsanus*	*tau*
Belonidae - needlefishes	needlefish, Atlantic	*Strongylura*	*marina*
Cyprinodontidae - killfishes	killifish, eastern banded	*Fundulus*	*diaphanus diaphanus*
	mummichog	*Fundulus*	*heteroclitus*
Atherinopsidae - new world silversides	silverside, Atlantic	*Menidia*	*menidia*
Gasterosteidae – sticklebacks	stickleback, fourspine	*Apeltes*	*quadracus*
	stickleback, threespine	*Gasterosteus*	*aculeatus*
Syngnathidae - pipefishes	seahorse, lined	*Hippocampus*	*erectus*
	pipefish, northern	*Syngnathus*	*fuscus*
Triglidae - sea robins	sea robin, northern	*Prionotus*	*carolinus*
	sea robin, striped	*Prionotus*	*evolans*
Cottidae - sculpins	grubby	*Myoxocephalus*	*aenaeus*
	sculpin, longhorn	*Myoxocephalus*	*octodecemspinosus*
Moronidae - temperate basses	perch, white	*Morone*	*americana*
	bass, striped	*Morone*	*saxatilis*
Serranidae - sea basses	sea bass, black	*Centropristis*	*striata*
	grouper, gag	*Mycteroperca*	*microlepis**
Pomatomidae - bluefishes	bluefish	*Pomatomus*	*saltatrix*
Rachycentridae - cobias	cobia	*Rachycentron*	*canadum++**
Carangidae - jacks	jack, crevalle	*Caranx*	*hippos*
Lutjanidae - snappers	snapper, gray	*Lutjanus*	*griseus**
Sparidae - porgies	scup, (porgy)	*Stenotomus*	*chrysops*
	sheepshead	*Archosargus*	*probatocephalus**
Sciaenidae - drums	perch, silver	*Bairdiella*	*chrysoura*
	weakfish	*Cynoscion*	*regalis*
	spot (Lafayette)	*Leiostomus*	*xanthurus+*

Table A5 cont.

FAMLY	COMMON NAME	GENUS	SPECIES
Chaetodontidae - butterflyfishes	butterflyfish, foureye	*Chaetodan*	*capistratus**
	butterflyfish, spotfin	*Chaetodan*	*ocellatus**
Labridae – wrasses	tautog (blackfish)	*Tautoga*	*onitis*
	cunner (bergall, chogy)	*Tautogolabrus*	*adspersus*
Pholidae - gunnels	gunnel, rock	*Pholis*	*gunnellus*
Blenniidae - combtooth blennies	blenny,feather	*Hypsoblennius*	*hentz*
Gobiesocidae - clingfishes	skilletfish	*Gobiesox*	*strumosus**
Gobiidae - gobies	goby, naked	*Gobiosoma*	*bosc*
	goby, seaboard	*Gobiosoma*	*ginsburgi*
Scombridae - mackerels	mackerel, Atlantic	*Scomber*	*scombrus*
Stromateidae - butterfishes	butterfish	*Peprilus*	*triacanthus*
Bothidae - lefteye flounders	flounder, summer (fluke)	*Paralichthys*	*dentatus*
	windowpane	*Scophthalmus*	*aquosus*
Pleuronectidae - righteye flounders	flounder, winter	*Pleuronectes*	*americanus*
Balistidae - leatherjackets	filefish, orange	*Aluterus*	*schoepfi*
Ostraciidae - boxfishes	trunkfish, spotted	*Lactophrys*	*bicaudalis**
Tetraodontidae - puffers	puffer, northern	*Sphoeroides*	*maculatus*

Previous assessments of the relative habitat value of urbanized shore zones in estuaries

There is mounting evidence that the epibiota and fish communities inhabiting human engineered structures differ from those on habitats that have been replaced. To date, the region where the most comprehensive suite of studies assessing the relative habitat value of natural and engineered substrates for algae and invertebrates is in the estuaries surrounding Sydney, Australia. This region is highly urbanized and approximately half of shoreline is stabilized by seawall in the most highly developed harbor. Studies have utilized photoquadrats of subtidal epibiota (Connell and Glasby 1999, Glasby 1999, Knott et al. 2004), experimental settlement plates (Connell 2000, Connell 2001), or surveys of quadrats in the interidal zone to assess components of algae and invertebrate communities (Chapman 2003, Chapman and Bulleri 2003, Bulleri et al. 2005, Bulleri 2005b, Chapman 2006). Direct surveys of sandstone reefs and sandstone seawalls showed that they had similar coralline algae dominated subtidal epibenthic communities, which were distinct from communities on concrete pilings and pontoons (Connell and Glasby 1999). The epibenthic communities on wooden pilings were more similar to those on concrete than sandstone (Connell and Glasby 1999, Glasby 1999). An examination of assemblages associated with a common substrate, mussel beds, nevertheless found significant differences in the species composition and diversity of communities in mussels associated with artificial substrata such as pontoons, pilings and seawalls, and those associated with natural rocky substrata (People 2006), suggesting that primary substrate composition can have a lasting effect on epifaunal assemblages. Further evidence that susbtrate differences are important is shown by the finding that seawall habitats in the Sydney region are insufficient at supporting viable populations of intertidal limpets, with egg masses less likely to be found on these materials than on rocky shores (Moreira et al. 2006), and that the materials used in artificial boulder reefs supported very different assemblages than on naturally occurring sandstone (Green et al. 2012). Assessing the secondary effects of substrate, experimental plates submerged for seven months until mid-summer were also useful for detecting the differences in epibenthic communities on rocky reefs, concrete pilings and concrete pontoons (Connell 2000, Connell 2001), regardless of whether the plates were made from concrete or sandstone (Connell 2000). Along vertical surfaces of intertidal zones diverse algae and invertebrate communities differed between sandstone seawalls and rocky shores at mid- and high-shore, but not low-shore (Chapman and Bulleri 2003, Bulleri et al. 2005). In a preliminary study, seawall was found to support similar species to rocky shores, with most differences in community structure owing to differences in densities, rather than differential presence/absence, between the shore types (Chapman and Bulleri 2003). More intensive surveys showed that those species particularly sensitive to the differences in habitat types included a tubeworm that consistently occurred more often on seawalls, and an encrusting alga that occurred more on rocky shores (Bulleri et al. 2005). Further detailed surveys to determine presence/absence showed that rocky shores had more rare taxa than seawalls, and although seawalls had similar algae and sessile animal taxa to rocky shores, they lacked approximately 50% of mobile animal taxa (Chapman 2003). At mid-shore, common intertidal molluscs were equally likely to be found on natural hard habitats and seawalls, but rare species preferentially inhabited the microhabitats on natural habitats (Chapman 2006). Surveys of areas cleared at the beginning of experiments showed that the intertidal algae and invertebrate communities on sandstone seawalls differed from those on rocky shores, from early successional stages and with persistent differences over two years (Bulleri 2005b). Visual censuses along belt transects whilst SCUBA diving in Sydney Harbor and surrounding estuaries showed that similar fish species occurred around marinas (constructed of wooden jetties and pylons, or floating pontoons and concrete pylons), swimming enclosures (wooden pilings and jetties, with interior enclosed by metal bars or netting) and natural rocky reefs (Clynick et al. 2008). However, abundances of some fish species differed between the artificial and natural habitats (Clynick et al. 2008).

Studies in other regions of the world have generally shown varying levels of sensitivity in benthic fauna to changes in habitat structure on natural and engineered hard shorelines. In northern Italy, transect surveys were used to show that the diversity of intertidal macroinvertebrate and algae communities was relatively low on human engineered groynes and breakwaters, possibly owing to harvesting of abundant mussels and the need for frequent maintenance of the coastal defense structures, in addition to effects of habitat structure (Bacchiocchi and Airoldi 2003). Also in northern Italy, quadrats were used for intertidal surveys that were conducted to demonstrate that the multivariate community structure of algae and invertebrates on stone seawalls around marinas was largely distinct from that on more structurally complex natural rocky shores and boulders breakwaters (Bulleri and Chapman 2004). Further, variability in community structure was lower on seawalls than on the other shorelines, providing supporting evidence that low complexity and heterogeneity of microhabitats did influence invertebrates and algae, although seawalls were also less exposed to waves than the other shorelines and this likely also influenced community structure. Univariate analyses demonstrated that seawalls had lower taxon richness than the other two hard shorelines, were dominated by encrusting algae, consistently lacked a common limpet and generally had low densities of erect algae (Bulleri and Chapman 2004). In south-western Italy along the Mediterranean coast, visual census of fish assemblages associated with breakwaters showed significant differences between those associated with adjacent sandy habitats, possibly as a result of the addition of novel rocky substrate habitat (Guidetti 2004).

On coastlines of Italy, Spain, UK and Denmark, surveys of intertidal epibiota using quadrats showed that algae and invertebrate communities on coastal defense structures (seawalls, rock groynes, offshore breakwaters or jetties) were qualitatively similar but less diverse than those on natural rocky shores (Moschella et al. 2005, Gacia et al. 2007). In Dubai, assessments of photoquadrats along transects were used to show that concrete breakwaters had different subtidal algae and invertebrate communities than rocky-reefs, although communities on the breakwaters became more similar to those on reefs with increasing age (Burt et al. 2011). A comprehensive assessment of both intertidal and subtidal infaunal and mobile fauna assemblages associated with low crested coastal defence structures (LCS) across the DELOS project area (Spain, Italy, UK) found consistent decreases in species diversity in all study areas with the presence of LCS (Martin et al. 2005). Moreover, these effects were more pronounced on the landward side of the LCS, with potential causal mechanisms including changes in hydrological regimes and sediment movement. The possible mechanisms contributing to differences in community structure between natural and engineered shorelines are discussed further in reviews by Thompson et al. (2002), Chapman et al. (2009), and Bulleri and Chapman (2010).

Most studies of the effects of shoreline modification in the United States have been conducted along the Pacific coastline, where the ultimate focus is often on assessing the effects of shoreline structures and armoring on the provision of habitat for and populations of important local fish species (e.g. Thom et al. 1994, Simenstad et al. 2004, Toft et al. 2007, 2013). In California, emergent intertidal invertebrate and algal communities were examined at both riprap and coastal sites using regular points within 0.25 m^2 quadrats placed on parallel alongshore transect lines, with differences in vertical zonation structure found between the site types. Fish communities were also assessed across sites and several fish species were found to preferentially inhabit riprap made from granite boulders, which has more habitat complexity than featureless seawalls (Davis et al. 2002). However, given that riprap lacks tidal pools and often does not extend to great subtidal depths, it does not contain the full suite of habitats available on some other natural hard shorelines (Davis et al. 2002). Pister (2009) surveyed intertidal quadrats to show that in southern California the overall community structure did not significantly differ between riprap and naturally rocky habitats, but that mobile species occurred in greater diversity on natural shores (see also references in Table 1 of Pister 2009). In Puget Sound, Washington, enclosure nets and snorkel surveys were used to show that the largest effects on nearshore fish communities occur where shoreline modifications, such as using riprap for stabilization, extend from the supratidal into the subtidal zone (Toft et al. 2007). An assessment of benthic communities along a variety of shorelines in the Lynnhaven River System within Chesapeake Bay (natural marsh, oyster shell reef, rip-rap and bulkhead) showed variable effects of shoreline type on specific taxa within benthic communities (Lawless and Seitz 2014). Overall density of benthic fauna was, however, clearly highest in assemblages associated with oyster reef habitats; these densities were twice those of assemblages within natural marsh habitats and bulkhead habitats were associated with the lowest benthic faunal densities. Although this study examined infaunal sediment assemblages, it is an important recent examination of the ecological effects of shoreline modification on benthic physical and faunal characteristics that can help to identify potential mechanisms or differences across shorelines.

Besides those materials occurring directly on engineered shorelines, other manmade structures that are used as novel habitats by aquatic biota include pontoons and pier pilings (Connell and Glasby 1999, Glasby 1999, Connell 2000, Connell 2001). Provision of novel habitats by some of these structures may facilitate the establishment of nonindigenous species (Glasby et al. 2007). Floating pontoons in marinas on the south coast of England provide habitat for barnacles, limpets, and numerous colonial ascidian and bryozoan species whose larvae may be dispersed by watercraft (Arenas et al. 2006). In the Hudson River estuary (between the shorelines of Hoboken, NJ and West Village, NY), passive traps were used to show that juvenile fishes preferentially inhabit wooden pile fields and open water habitats over those under large piers (Able et al. 1998). Specifically, only 14 of the 25 fish species found during the study occurred under piers (Able et al. 1998). Despite anthropogenic disturbances, some habitats in the lower Hudson River still appear to act as a nursery area for some fish species (Able et al. 1998).

Restoration of habitat value to engineered shorelines require strong collaboration between ecologists, engineers and managers of environmental assets (Bulleri and Chapman 2010, Browne and Chapman 2011). In Sydney Harbor, the creation of novel intertidal pool habitat by omitting some blocks within a sandstone seawall resulted in increased diversity of sessile invertebrates, mobile animals and foliose algae, particularly high along shorelines owing to species from lower shore levels being able to colonize the pools (Chapman and Blockley 2009). A caveat was that the community structure in the pools provided in seawalls did not mimic that in natural rockpools (Chapman and Blockley 2009). In the same harbor, addition of novel intertidal microhabitats (i.e., holes and crevices) to the seawalls resulted in short-term increases in mobile invertebrates (Browne and Chapman 2011). However, small holes eventually were colonized and covered by sessile invertebrates, such as mussels, precluding their use by mobile invertebrates. The use of concrete flower pots attached to seawalls to mimic intertidal rock-pools provided habitat for different communities than those occurring on adjacent seawall, with 25 species in pots not found on walls (Browne and Chapman 2011). The additional taxonomic diversity supported by pots was greatest for mobile animals (118% more species), followed by algae (50% more species) and sessile animals (39% more species) (Browne and Chapman 2011). There is the potential for future engineered shorelines to include design elements targeted at benefitting specific species, as was demonstrated for seahorses in Sydney Harbour (Hellyer et al. 2011).

Metrics used in assessment of relative habitat value of estuarine shorelines

It should be noted that none of methods or response variables used in the assessments of relative habitat value of engineered shorelines and other manmade structures mentioned in the previous subsection were designed for inclusion in a readily repeatable protocol, rather they were chosen to answer the specific aims of each stand-alone study. As for methodologies in collection of biotic samples, there are a plethora of response variables used in ecological assessments (Downes et al. 2002, Borja and Dauer 2008). Metrics selected for ecological assessments must be scientifically valid and responsive to the ecosystem attribute being measured (e.g. relative habitat value of hardened shorelines), measurable with low error, easy to interpret, and should be cost-effective to enable widespread application if they are to be used in a protocol (Gibson et al. 2000). An ecologically parsimonious approach dictates assessing the utility of existing metrics, prior to developing new ones (Borja and Dauer 2008). Common univariate response variables used in ecological assessments are taxon diversity, diversity indices (e.g. Shannon-Wiener), productivity (e.g. abundance or biomass of different taxonomic groups), relative proportions of different taxonomic groups, proportions of native and invasive species, and indicator species (Downes et al. 2002, Borja and Dauer 2008). Multivariate ordination (e.g. multidimensional scaling) is commonly used to determine whether the relative similarity of communities between sites corresponds with the predictor variable/s being tested (Warwick and Clarke, 1991). To compare whole assemblages between plates, as well as within sites and regions, permutational multivariate analysis of variance (PERMANOVA) is a very useful tool that calculates the probability that the similarities and differences between communities, calculated from a similarity matrix such as that generated by the Bray–Curtis function, are likely to be a result of random chance, or determined by the predicator variables and factors applied in the model(Anderson 2001, McArdle and Anderson 2001) There is also evidence that examination of functional traits of macroinvertebrates (e.g. proportional representation of animals from different feeding guilds and functional traits) can be valuable in assessments of aquatic communities, including those inhabiting estuaries (Bremner et al. 2003, Diaz et al. 2004, Hewitt et al. 2008). Because of varying sensitivities of the different response variables, several of them should be used concurrently in ecological assessments, providing greater certainty of the data interpretation than reliance on any single measure (Gibson et al. 2000). Multimetric approaches incorporate standardized information from a suite of variables into a single index of relative ecological condition (e.g. Pinto et al. 2008, Stoddard et al. 2008). Although useful for simplifying the results of bioassessments, important information is obscured by multimetric approaches (Borja and Dauer 2008) and these do not allow assessment of the correlation between each individual metric and habitat structure that may prove useful in informing future shoreline management.

Indicator species are those organisms whose presence or absence reliably indicates some critically favorable or unfavorable attribute/s of an ecosystem (Gibson et al. 2000, Carignan and Villard 2001, Whitfield and Elliot 2002, Bilkovic et al. 2005). For example, if an organism known to be particularly sensitive to a pollutant is found to be highly abundant at a site, high water quality is inferred using the indicator species approach. Conversely, dominance by pollution tolerant organisms implies that a site is highly polluted (Gibson et al. 2000). Reliable indicator taxa can be an important and cost-effective tool in ecological assessment. However, they must be used with caution as the presence of ubiquitous pollution tolerant organisms is not necessarily associated with ecosystem degradation, unless they numerically dominate more sensitive taxa. The use of the presence of sensitive species that are consistently absent from all degraded sites as an indicator of favorable environmental conditions is less subjective. Greatest confidence in indicator species occurs where the abundances of sensitive and robust species changed in opposite directions across gradients of environmental degradation (Gibson et al. 2000). It is unclear whether any species is likely to be consistently sensitive to the design and materials used in shoreline stabilization. Studies showing particular groups (e.g. limpets, Moreira et al. 2006) sensitive to shoreline habitat structure… In a study of effects of port construction on epibenthos inhabiting hard substrate in a Spanish estuary, several species were identified as potentially indicative of sedimentation but this was not directly to construction (Saiz-Salinas and Urkiaga-Alberdi 1999). Owing to the relatively low taxon richness of epibenthos in coastal marine waters, which is further diminished by broad-scale effects of urbanization, these communities may not be adequately sensitive to enable detection of local-scale habitat effects (Gibson et al., 2000). Gibson et al. (2000) also suggest that if low richness precludes the utility of epibenthos community structure in habitat assessments, the use of indicator species may be more appropriate.

Determining the suite of indicators likely to provide useful information for determination of shoreline habitat quality in estuaries is likely to be challenging. Given that estuarine shorelines are naturally highly variable in their physicochemical characteristic and urban shorelines are further disturbed by multiple anthropogenic stressors, the biota on theses shorelines may be adapted to stress and demonstrate limited response to changes in local habitat structure (i.e., 'The Estuarine Quality Paradox', Elliot and Quintino 2007). Consideration of the role of habitat modification heterogeneity in structuring estuarine assemblages is therefore crucial in developing indices with which to assess habitat quality in these regions (e.g. Hewitt et al. 2008). The central principle of ecological assessments is the valid comparison of a disturbed site to biological criterion, which is typically based on evaluation of the community in relatively undisturbed 'reference' sites in similar ecosystems (Gibson et al. 2000). This is not possible in highly urbanized regions where significant modification has occurred at all sites within the region. Further, it is recognized that many shorelines will continue to be disturbed and in these regions it is not feasible to restore ecosystems to pristine conditions. Rather, ecological restorations are performed to maximize natural function by providing enhanced local structural habitat complexity and more natural elements, within the constraints of disturbances operating at broad spatial scales in urbanized estuaries. It is desirable to assessment the relative habitat value of traditional shoreline stabilization techniques (e.g. seawall, bulkhead) against 'ecologically enhanced' shoreline stabilization techniques. To this end, a standardized protocol is required to compare the communities on 'ecologically enhanced' shorelines to those on traditional shoreline structures (which will act as 'controls'). A protocol should be useful for providing ecologically relevant and scientifically defensible information in assessing the relative successes of different ecologically enhanced shoreline projects. In complex natural ecosystems, it is difficult to accurately predict the effects of both human disturbances and restoration actions. It may take considerable time after restoration actions for the intended benefits to manifest and unforeseen side effects of restoration action may occur (Kelaher et al. 2003). For future restoration initiatives it will be feasible to apply a Before–After Control–Impact monitoring design (Downes et al. 2002) using the protocol. This will provide information on the timescales of recovery, if any occur.

In the near-term, the goals of the project are to develop and pilot a standard protocol for assessing the habitat value of ecologically enhanced shorelines in the urban core of the NY–NJ Harbor Estuary. Longer-term, the protocol development should lead to further development of a means for standardized monitoring of various techniques and developing case studies over time; inform decision-making by regulatory agencies, engineers, and land-owners surrounding emerging and existing stabilization techniques employed in urban coastal areas. The development and use of a standard protocol will also be useful for building a database of information about ecosystem responses to shoreline restoration strategies, which can be used to inform future management.

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