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

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Introduction

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 projected 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 have not been a primary concern in the design and materials used (Chapman and Underwood 2011); rather, the major consideration has been 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 funding and design consideration 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 be 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 and 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 1. 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 2: 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 2 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 3. 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 4. 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 mostly 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 EU 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). Currently, 36% of the core NY-NJ Harbor estuary is comprised of hardened shoreline, more than 80% of the area's tidal wetlands have been filled in the past century, and the harbor's historic oyster reefs are functionally extinct (Bain et al. 2007, NOAA 2003). The percent of hardened shoreline is as high as 87% in more urban reaches, such as the Upper Bay of the estuary - Bayonne New Jersey, Staten Island's north shore, industrial portion of Brooklyn (U.S. Army Corps of Engineers and others, 2009). 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). These are outlined in Table 1. 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 communities 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 5. 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 killie (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 killie trap
* Rare species, infrequently captured

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*
	killifish, eastern		*diaphanus*
Cyprinodontidae - killfishes	banded	*Fundulus*	*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*

Table 5 cont.

FAMLY	**COMMON NAME**	**GENUS**	**SPECIES**
	spot (Lafayette)	*Leiostomus*	*xanthurus+*
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 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 enhancements of urban shorelines 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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