



Review

Identifying the consequences of ocean sprawl for sedimentary habitats



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ABSTRACT

Extensive development and construction in marine and coastal systems is driving a phenomenon known as “ocean sprawl”. Ocean sprawl removes or transforms marine habitats through the addition of artificial structures and some of the most significant impacts are occurring in sedimentary environments. Marine sediments have substantial social, ecological, and economic value, as they are rich in biodiversity, crucial to fisheries productivity, and major sites of nutrient transformation. Yet the impact of ocean sprawl on sedimentary environments has largely been ignored. Here we review current knowledge of the impacts to sedimentary ecosystems arising from artificial structures.

Artificial structures alter the composition and abundance of a wide variety of sediment-dependent taxa, including microbes, invertebrates, and benthic-feeding fishes. The effects vary by structure design and configuration, as well as the physical, chemical, and biological characteristics of the environment in which structures are placed. The mechanisms driving effects from artificial structures include placement loss, habitat degradation, modification of sound and light conditions, hydrodynamic changes, organic enrichment and material fluxes, contamination, and altered biotic interactions. Most studies have inferred mechanism based on descriptive work, comparing biological and physical processes at various distances from structures. Further experimental studies are needed to identify the relative importance of multiple mechanisms and to demonstrate causal relationships. Additionally, past studies have focused on impacts at a relatively small scale, and independently of other development that is occurring. There is need to quantify large-scale and cumulative effects on sedimentary ecosystems as artificial structures proliferate. We highlight the importance for comprehensive monitoring using robust survey designs and outline research strategies needed to understand, value, and protect marine sedimentary ecosystems in the face of a rapidly changing environment.

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Contents

1. Introduction	32
2. Impacts of artificial structures on sedimentary habitats	33
2.1. Placement loss, habitat degradation, and related effects	33
2.2. Changes to the sensory environment	33
2.3. Hydrodynamic effects	36

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2.3.1.	Large-scale hydrodynamic effects	37
2.3.2.	Intermediate-scale hydrodynamic effects	37
2.3.3.	Small-scale hydrodynamic effects	37
2.4.	Organic enrichment and material fluxes	38
2.5.	Contaminants	39
2.6.	Biotic effects	39
3.	Factors influencing the direction and magnitude of impacts	40
4.	Approaches employed in past studies	41
5.	Research gaps and future directions	41
5.1.	Monitoring	41
5.2.	Future research directions	42
6.	Conclusions	42
	Acknowledgements	43
	References	43

1. Introduction

The intensifying development of urban foreshores, coastlines, and offshore areas is driving a phenomenon commonly referred to as “ocean sprawl” (Duarte et al., 2012). Artificial structures are added to estuarine, coastal, and marine systems to protect shorelines from erosion (Dugan et al., 2011; Nordstrom, 2014), and to support marine aquaculture (Giles, 2008; McKindsey et al., 2011; Simenstad and Fresh, 1995), renewable energy generation (Bailey et al., 2014; Gill, 2005; Langhamer, 2010; Miller et al., 2013; Petersen and Malm, 2006),

natural resource extraction (Kingston, 1992; Peterson et al., 1996; Wilson and Heath, 2001), and recreational and commercial activities (Connell and Glasby, 1999; Connell, 2000). Artificial structures therefore take a variety of forms (Fig. 1), varying in size, from small objects such as ‘crab-tiles’ (Sheehan et al., 2008) to large, artificial islands (Cavalcante et al., 2011). Collectively, these structures are causing extensive modification of marine and coastal ecosystems and the important ecosystem services they support (Bulleri and Chapman, 2010; Dugan et al., 2011). While these structures are added to both hard and soft bottom habitats (Bulleri, 2005), most research has focused on the



Fig. 1. Examples of artificial structures in sedimentary environments. From left to right: groyne^a, pier^a, revetment^a, dumped appliance (toilet)^b, tire reef^b, and overwater causeway^b. Photo credit: ^a E. Strain, ^b E.C. Heery.

extent to which they modify and mimic natural hard substrates, with impacts on sedimentary ecosystems, by comparison, little studied.

In marine, coastal, and estuarine environments (hereafter, collectively referred to as marine), sediment is one of the most abundant ecosystems, spanning intertidal habitats, such as sandy beaches and tidal flats, to the deep sea floor (Masselink et al., 2014; Paris et al., 2011). Consequently, in many parts of the world, the distribution of marine sediments overlaps substantially with ocean sprawl, resulting in habitat loss and modification of the diverse communities and ecological functions that sedimentary habitats underpin (Snelgrove et al., 2014; Bishop et al., 2017-in this issue). Microbial, meiofaunal, macrofaunal, and macrophytic assemblages (e.g. saltmarsh, mangrove, seagrass) live on and within the sediments (Adam, 1990; Coull, 1988; Lugo and Snedaker, 1974; Orth et al., 1984; Paerl and Pinckney, 1996; Snelgrove, 1998) and provide prey resources for fishes, shorebirds, and large vertebrates, such as gray whales (*Eschrichtius robustus*), dugongs (*Dugong dugon*), and green sea turtles (*Chelonia mydas*) (Carruthers et al., 2002; Gray and Elliott, 2009; Lopez and Levinton, 1987; Weitkamp et al., 1992). These assemblages also underpin many ecosystem services of fundamental importance to humanity, including fisheries productivity, biogeochemical cycling, remediation of contaminants, and shoreline stabilization (Bolam et al., 2002; Snelgrove, 1997, 1999; Snelgrove et al., 2014; Weslawski et al., 2004).

Understanding the impacts of ocean sprawl on the structure and important functions of sedimentary ecosystems is necessary for the development of management strategies aimed at conserving biodiversity and ecosystem services (Gray, 2002; Snelgrove, 1999). Here, we review current knowledge of the impacts of artificial structures on marine sedimentary ecosystems (including the organisms living in or in close association with benthic sediments), as well as the limitations in current knowledge. Our objective is not to impose value judgements on these impacts, but rather to summarize the known literature. Sediment is a broad term that refers to a diverse range of loose materials that are derived from a parent source (i.e. bedrock, shells, plant and animal matter) ranging from megaliths (i.e. <1075 km diameter) through to the finest muds (Blair and McPherson, 1999; Wentworth, 1922). In this paper we focus on sediments that are entrained and transported under normal wave conditions (i.e. muds, <0.06 μ , to cobbles <256 mm diameter), thereby excluding larger cobbles and coarse boulders, which are typically entrained only under storm or tsunami conditions (Paris et al., 2011). We refer to all material greater than 2 mm diameter (i.e. gravels to cobbles) as coarse sediment and all material finer than 2 mm (i.e. sand, silt and clay) as fine sediment. We focus primarily on the effects of artificial structures on un-vegetated sediments, as these habitats have been particularly underrepresented in the literature to date, but utilize examples from vegetated sediments where relevant and informative for sedimentary ecosystems more broadly. Our review includes discussion of how artificial structures may affect sedimentary ecosystem functioning and, hence, the provision of ecosystem services. We make the case that the proliferation of artificial structures is of vital concern for sedimentary ecosystems and highlight knowledge gaps and future research that will be needed in order to protect ecosystem services provided by marine sedimentary habitats in the face of ocean and climate change.

2. Impacts of artificial structures on sedimentary habitats

Artificial structures modify soft sediment habitats directly, through displacement of flora and fauna by their foundations, and indirectly, by altering key physical, chemical, and biotic parameters that influence sediments beyond the footprint of the structure (Table 1, Fig. 2). Sedimentary organisms may respond to these direct and indirect effects at the population, community, or ecosystem level. In Sections 2.1 through 2.6, we summarize the direct and indirect effects of artificial structures on sediments, as well as the potential consequences with respect to sedimentary ecosystem functions.

2.1. Placement loss, habitat degradation, and related effects

Construction of artificial structures on top of surface sediments is arguably the most obvious and destructive direct impact because it reduces the area of habitat available to resident organisms, which are concentrated close to the seawater-sediment interface (Hines and Comtois, 1985). The habitat that is eliminated by the footprint of artificial structures is known as 'placement loss' (Dugan et al., 2008; Griggs, 2005). In the case of structures that are deployed in clusters over large areas, such as offshore wind farms, placement loss may be particularly extensive (Wilson and Elliott, 2009). In the UK alone, 1200–8600 km² of sedimentary habitat is expected to be lost to offshore wind farm development by 2020 (Byrne and Houlby, 2003; Wilson et al., 2010).

In addition to directly impacting the organisms living in sediments via placement loss, many structures have secondary effects on mobile and migratory species, particularly when artificial structures result in the loss or modification of large areas of habitat. For instance, in eliminating upper intertidal and supratidal beach habitat that supports invertebrate communities of beach hoppers and ghost crabs, seawalls placed in the low or mid intertidal zone negatively affect the foraging and roosting behavior of shorebirds and seabirds (Dugan et al., 2008). Similarly, swing mooring buoys, which uproot seagrass through chain drag on bottom sediments, and overwater structures such as piers and pontoons, which reduce macrophyte abundance through shading (Section 2.2), influence the invertebrate and finfish species that utilize vegetated sediments for food and shelter (Collins et al., 2010; Walker et al., 1989).

During the construction of artificial structures, placement loss is often coupled with a series of other physical and chemical changes. For example, the construction and maintenance of marine infrastructure often requires dredging of large amounts of sediment. Increased concentrations of suspended sediments during the dredging process can damage the gills and eyes of fish and prevent filter feeding by invertebrates (Knott et al., 2009). Moreover, estimates of up to 2300 m³ of sediment loss per turbine have been linked to dredging during wind farm construction (Lozano-Minguez et al., 2011). Dredging not only removes sediments but also resident fauna (Jones and Candy, 1981; Thrush and Dayton, 2002) and flora (Iannuzzi et al., 1996), and recovery of affected benthic communities can, in some instances, take as long as 2 to 4 years (van Dalssen et al., 2000). Dredge spoil that is dumped offshore or deposited intertidally to nourish eroding beaches or create artificial wetlands can have substantial and lasting impacts on the benthic communities, arising through smothering and alteration of sediment properties (Bishop, 2005; Bishop et al., 2006; Manning et al., 2014).

Artificial structures can also act as a physical barrier or deterrent to the movement of organisms across sedimentary seascapes. For instance, breakwaters and seawalls can inhibit the movement of sea turtles and terrapins from the sea to the supratidal area of sandy beaches where they lay eggs (Bouchard et al., 1998). Seawalls can also limit the tidal migration of sandy beach invertebrates up and down the shore to feed and avoid desiccation stress (Dugan et al., 2011). In the subtidal, the arrangement of structures, such as underwater turbines, jetties, pilings, and bulkheads may also create barriers to the movements and migrations of sediment-feeding organisms depending on their density and spatial arrangement in the seascape (Bulleri and Chapman, 2015; Dadswell and Rulifson, 1994; Gill, 2005). Effects of artificial structures on connectivity are reviewed by Bishop et al. (2017-in this issue), and therefore are not discussed here in detail.

2.2. Changes to the sensory environment

Artificial structures alter the sensory environment for sedimentary organisms in by modifying light and noise levels. Some structures produce light pollution (Davies et al., 2014), which may impact sedimentary organisms (Navarro-Barranco and Hughes, 2015). Conversely, many structures cast shadows on sedimentary habitats. The light level in the

Table 1
Documented effects from artificial structures, their scale, and their potential biotic effects as presented by the authors referenced.

Effect type	Structure type	Abiotic change	Scale	Potential biotic impacts	References
Placement loss	Bulkheads & seawalls	Elimination of upper intertidal	Area of structure	Reduced abundance of upper intertidal invertebrates and their predators	Dugan et al. (2008)
Physical barrier	Pilings	Act as a physical barrier inhibiting movement of large mobile species	10s of meters	Reduction in nesting of sea turtles	Bouchard et al. (1998)
Magnetic and electric currents	Cables	Magnetic fields surrounding cables	<1 m to 10s of meters	Apparently limited effect on invertebrates	Andrulewicz et al. (2003), Bochert and Zettler (2004), Petersen and Malm (2006)
	Wind farms	Magnetic and electric currents from cables	10s of meters	Possible effects on elasmobranchs and fish that are sensitive to magnetic fields	Petersen and Malm (2006)
Noise	Wind farms	Sound from rotors	Kilometers	Potential impact on cetaceans and other organisms; may deter some fish	Wahlberg and Westerberg (2005), Petersen and Malm (2006)
	Aquaculture	Shading from aquaculture structures	Variable	Lower abundance of macrophytes	Deslous-Paoli et al. (1998), McKindsey et al. (2011)
Light	Marinas	Increased turbidity/reduced light levels	Area of structure	Reduced sediment microalgal production	Iannuzzi et al. (1996), Rivero et al. (2013)
	Piers & docks	Shading from overwater structures	Meters to 10s of meters	Negative impacts on primary producers, poor feeding conditions and suboptimal foraging by juvenile fish, avoidance by mobile consumers, concentration of consumer populations in adjacent areas, altered assemblage structure	Burdick and Short (1999), Duffy-Anderson and Able (1999, 2001), Toft et al. (2007), Munsch et al. (2014), Ono and Simenstad (2014)
Interruption of current patterns	Artificial reefs	Changes in microtopography & ripple marks	<15 m	May impact variation in macrofauna and meiofauna composition	Sun et al. (1993), Barros et al. (2004)
	Breakwaters & groynes	Interruption of longshore currents & redistribution of sediments	100s of meters to kilometers	Elimination of downdrift depositional habitats	Duane (1976), Komar (1998), Cuadrado et al. (2005), Bostic et al. (2015)
	Bulkheads & seawalls	Modification of water circulation	100s of meters	Accumulation of eggs and larvae in between bulkheads	Jackson et al. (2015)
	Artificial reefs	Increased erosion & scour (updrift side)/Coarsening of sediment	<10 m	Greater variability in infaunal community composition	Davis et al. (1982), Ambrose and Anderson (1990), Barros et al. (2001)
Increases in flow	Breakwaters & groynes (exposed side)	Increased erosion & scour/Coarsening of sediments	<10 m	Shift towards larger macrofauna	Bertasi et al. (2007), Munari et al. (2011)
		Increased erosion & scour/Coarsening of sediments	<3 m	Lower density of meiofauna	Weis et al. (1993), Spalding and Jackson (2001)
	Bulkheads & seawalls	Narrowing of intertidal habitat/Increased steepness	Meters	Reduced nesting by sea turtles and colonization by swash-riding mollusks; Altered population dynamics for benthic organisms such as ghost crabs; Reduced burrowing habitat for benthic organisms; Reduced aquatic vegetation	Pilkey and Wright (1988), Hall and Pilkey (1991), Peterson et al. (2000), Brown and McLachlan (2002), Bozek and Burdick (2005), Toft et al. (2007), Lucrezi et al. (2010), Dugan et al. (2011), Rizkalla and Savage (2010), Heatherington and Bishop (2012), Morley et al. (2012)
Decreases in flow	Aquaculture	Accumulation of fine sediment	Meters to 10s of meters	Organic enrichment (see below)	McKindsey et al. (2011)
	Artificial reefs	Accumulation of fine sediment	<10 m	Organic enrichment (see below)	Ambrose and Anderson (1990), Fabi et al. (2002), Martin et al. (2005), Wilding (2006), Zalmon et al. (2012), Machado et al. (2013), Wilding (2014)
	Breakwaters & groynes (protected side)	Accumulation of fine sediment/longer residence time following storms	Meters	Organic enrichment (see below), Smaller macrofauna, shift in zonation patterns with depth; Lower abundance of benthic invertebrates in some locations	Martin et al. (2005), Zanuttigh et al. (2005), Bertasi et al. (2007), Munari et al. (2011)
	Marinas	Accumulation of fine sediment/longer residence time/slight increases in temperature and pH	Footprint	Change in infaunal community structure	Floerl and Inglis (2003), Balas and Inan (2010), Rivero et al. (2013)
	Aquaculture	Organic enrichment/hypoxic & sulfidic sediments	10s of meters	Decreased abundance of larger infauna and altered vertical biomass profiles in sediments; may alter meiofaunal community composition; Potential limitations in system-wide carrying capacity	Weston (1990), Deslous-Paoli et al. (1998), Wildish et al. (2001), Duarte et al. (2003), Holmer et al. (2005), Giles (2008), Cranford et al. (2009), Wilding (2012)
Organic enrichment	Artificial reefs	Organic enrichment/Site-specific reduction in sedimentary oxygen	<2 m	May alter meiofaunal community composition	Fricke et al. (1986), Danovaro et al. (2002), Wilding (2014)
	Breakwaters & groynes	Increased organic content	Meters	Changes in infaunal community structure	Bertasi et al. (2007)
	Oil and gas platforms	Organic enrichment/Lower sedimentary oxygen	<100 m	Increased abundance of deposit feeding polychaetes and nematodes	Kennicutt et al. (1996), Montagna and Harper (1996)
	Wind farms	Increased ammonia, detrital	10s of	Increased resource availability for infaunal	Maar et al. (2009)

Table 1 (continued)

Effect type	Structure type	Abiotic change	Scale	Potential biotic impacts	References
Material fluxes		material, and fecal pellets in down current sediments	meters	deposit feeders	
	Artificial reefs	Influx of shell fragments from encrusting invertebrate colonizing structure	<15 m	Unknown	Davis et al. (1982), Barros et al. (2001), Machado et al. (2013)
	Bulkheads & seawalls	Decrease in beach wrack and mangrove leaf litter	Meters	Reduction in terrestrial insects, which decreases prey resources for fish, as well as in beach invertebrates, such as amphipods and oligochaete worms, which may have cascading effects	Dugan et al. (2008), Sobocinski et al. (2010), Heatherington and Bishop (2012), Heerhartz et al. (2014, 2015, 2016)
	Oil and gas platforms	Influx of shell fragments	<500 m	Unknown	Kennicutt et al. (1996)
	Artificial reefs	Increase in zinc, benzothiazoles, and polycyclic aromatic hydrocarbons (tire reefs), metals (coal ash reefs)	Unknown	Elevated contaminants in invertebrate tissues and potential toxicity	Collins et al. (1995, 2002), Wik and Dave (2009)
Contaminants	Bulkheads & seawalls	Increase in copper chromated arsenate (CCA) bulkheads constructed with treated wood	<5 m	Unknown	Weis et al. (1993)
	Marinas	Contamination from vessel anti-fouling (AF) paints, CCA, metal biocides	Area of structure to 10 m	Increased contaminants in tissues of macroalgae and invertebrates, stress-induced changes in biotic interactions	McGee et al. (1995), Schiff et al. (2004), Singh and Turner (2009), Johnston et al. (2011), Rivero et al. (2013), Neira et al. (2014), Sim et al. (2015)
	Oil and gas platforms	Contamination and discharge from drilling	100s of meters to kilometers	Altered macrofaunal community composition	Kingston (1992), Olsgard and Gray (1995), Montagna and Harper (1996), Peterson et al. (1996)
	Pilings	Increase in CCA	<10 m	May reduce richness and diversity of infauna	Weis et al. (1993), Weis and Weis (1996), Hingston et al. (2001)
Bottom-up limitation	Wind farms	Depletion of plankton resources by sessile invertebrates colonizing platforms	<10 m	Lower infaunal biomass and altered infaunal community structure	Maar et al. (2009)
Increased predation	Artificial reefs	Increased predation from reef-associated predators	Meters to 10s of meters	Unknown	Davis et al. (1982), Nelson et al. (1988), Frazer et al. (1991), Posey et al. (1992)
	Pilings	Attracts consumers (surfperch, crabs)	Meters	Unknown	Toft et al. (2007)
	Wind farms	Increase of consumers associated with platforms/Increased physical disturbance from foraging activity	<10 m	Unknown	Maar et al. (2009)
Change in disturbance from humans	Crab tiles	Increased disturbance from trampling	Variable	Depletion of meiofauna	Sheehan et al. (2010a)
	Artificial reefs	Decreased disturbance from reduction of bottom trawling	Variable	Unknown	Cheung et al. (2009)

area directly underneath artificial structures and in the nearby vicinity can be several orders of magnitude less than that in adjacent open water (Burdick and Short, 1999; Deslous-Paoli et al., 1998). Overwater structures have been found to lower the growth rates and percent cover of macrophytes (Deslous-Paoli et al., 1998), including habitat-forming species such as seagrasses (Burdick and Short, 1999). Artificial structures are also likely to negatively affect growth of the microphytobenthos (MPB) (Pagliosa et al., 2012; Struck et al., 2004). MPB include microalgae and cyanobacteria that stabilize sediments (McIntyre, 1969; Underwood and Paterson, 2003), serve as an important food resource for several invertebrate grazers (De Jonge and Van Beuselom, 1992; Herman et al., 2000; Simith et al., 2017-in this issue), and fix nitrogen (Piehler et al., 1998, 2010). Impacts on these organisms due to shading effects from artificial structures are therefore likely to significantly affect some of the functioning properties of sedimentary systems in the photic zone. In the intertidal, shadows cast by artificial structures can lower temperatures and reduce desiccation stress, which may alter the growth rate and success of intertidal invertebrates and algae (Blockley and Chapman, 2006; Guichard et al., 2001). Low-light areas under subtidal piers, jetties and wharves also reduce abundances of and feeding activity by fish that rely on visual cues to forage for prey in sedimentary environments, including juvenile salmonids

(Munsch et al., 2014; Ono and Simenstad, 2014; Toft et al., 2007) and juvenile winter flounder, *Pseudopleuronectes americanus* (Duffy-Anderson and Able, 1999, 2001). These mobile consumers may also become concentrated in areas adjacent to artificial structures due to avoidance behavior that is driven by shadows (Munsch et al., 2014), and this may have secondary effects on sedimentary prey populations.

Construction, operation, and decommissioning of artificial structures, particularly those associated with offshore energy resources, can also significantly change the acoustic environment (Bailey et al., 2010; Nedwell et al., 2003, 2007). For example, the decibels of sound produced by pile driving is almost double that of background levels 100 m away from construction sites and can be detected above background noise up to 70 km away from the source (Bailey et al., 2010). To date, studies on the effect of structure-associated noise have primarily focused on marine mammals (Bailey et al., 2010; Koschinski et al., 2003; Tougaard et al., 2009) and have extended to few other taxa (Nedelec et al., 2014). We know that exposure to anthropogenic noise caused by boat traffic and seismic surveys can reduce successful development and early survival of marine invertebrates (de Soto et al., 2013; Nedelec et al., 2014). Noise can also have physiological and behavioral effects on marine invertebrates (Regnault and Lagardere, 1983; Wale et al., 2013a, 2013b). For

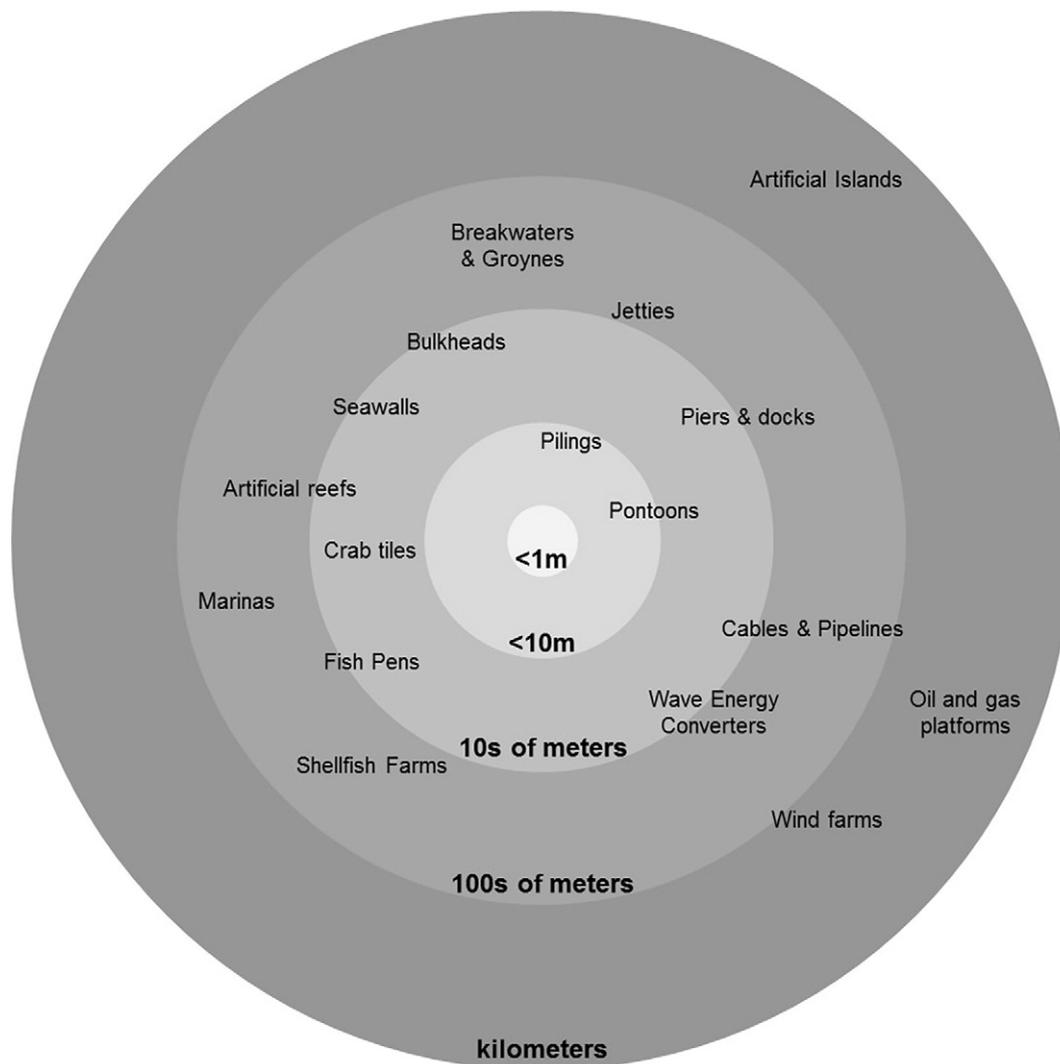


Fig. 2. Spatial scale of effects of different types of artificial structures.

example, ship noise can negatively affect foraging and antipredator behavior in the shore crab, *Carcinus maenas* (Wale et al., 2013b). It is likely, therefore, that organisms living in association with the sediments are affected by the noise produced during the life-cycle of marine infrastructures (construction, operation and decommissioning). Much of the research on anthropogenic noise related to artificial structures has focused on pile driving when constructing offshore infrastructure. Pile driving is regarded as one of the most extreme noises associated with artificial structures, therefore representing the worst-case scenario when assessing impacts (Madsen et al., 2006). Research needs to be extended to other potentially important noise sources, including near-shore renewable energy development and artificial structures associated with recreational boating, which correlate with boat traffic (Widmer and Underwood, 2004).

Another change to the sensory environment of sediments results from cables that connect the mainland with offshore infrastructure and generate electromagnetic (EM) fields. Many marine species are EM-sensitive, including cetaceans, turtles, certain groups of fish, and some crustaceans and mollusks (Gill et al., 2014). The nudibranch *Tritonia diomedea*, for instance, uses earth's magnetic field to navigate shallow sedimentary environments in the northeast Pacific (Lohmann and Willows, 1987; Willows, 1999; Wyeth and Willows, 2006). The intensity of EM fields emitted by submarine cables is potentially sufficient to interfere with such behaviors (Bochert and Zettler, 2004), however,

direct evidence of EM-related impacts from cables on the navigation and movement of sedimentary marine organisms is limited (Gill, 2005; Tricas and Gill, 2011). Andrulowicz et al. (2003) found no consistent change in the macrozoobenthos of sandy substrata from before and after the installation of a submarine cable system between Sweden and Poland, despite a strongly altered magnetic field. Similarly, no significant change in the behavior of the Atlantic halibut (*Hippoglossus hippoglossus*), Dungeness crab (*Metacarcinus magister*), or the American lobster (*Homarus americanus*) – species closely associated with sedimentary habitat (Woodruff et al., 2012) – were observed following exposure to a magnetic field in the laboratory. The mechanism by which EM fields might impact marine organisms remains under investigation. For instance, laboratory experiments have shown that EM fields can induce the expression of heat shock proteins (HSPs) 70 and 90 in immunocytes of the mussel *Mytilus galloprovincialis* that attaches to hard substrates (Malagoli et al., 2004). Similar effects on sediment-dwelling invertebrates may be expected, however, further study is needed.

2.3. Hydrodynamic effects

Artificial structures change the speed and direction of water movement. This results in a number of hydrodynamic effects at large, intermediate, and small spatial scales.

2.3.1. Large-scale hydrodynamic effects

At large spatial scales (hundreds of meters to kilometers), artificial structures can cause extensive modifications to water circulation patterns and sediment transport mechanisms (Bostic et al., 2015; Cavalcante et al., 2016; Cuadrado et al., 2005; Thomalla and Vincent, 2003; Zyserman et al., 2005). For instance, groynes, breakwaters, seawalls, and artificial reefs alter and restrict sediment dynamics by interrupting both longshore and tidal transport (Cuadrado et al., 2005; Pilkey and Wright, 1988). Sediment volume increases on the up drift side of these structures and decreases in down drift areas (Duane, 1976; Komar, 1998), which can reduce the extent of adjacent wetlands (Bostic et al., 2015) and beaches (Thomalla and Vincent, 2003).

Interrupted currents also affect gamete and larval transport (see Bishop et al., 2017-in this issue). Eggs of horseshoe crabs (*Limulus polyphemus*) tend to accumulate in shoreline discontinuities, such as along jetties and in the enclaves between bulkheads (Jackson et al., 2015). This, in turn, can increase aggregations of foraging shorebirds (Botton et al., 1994). While the concentration of predators can have greater ecological or evolutionary implications (i.e. - increasing intraspecific or interspecific competition among shorebirds, altering the timing and synchronicity of reproduction among prey), research on such effects has been limited.

It is important to note that these large-scale hydrodynamic effects from artificial structures do not happen in isolation. They are usually furthered by other human activities in the marine environment, such as dredging for navigation and/or mineral extraction, the damming of rivers, and construction of flood and shoreline defenses, which restrict fluvial and terrestrial sediment delivery to oceans (Dethier et al., 2016; Milliman and Farnsworth, 2013). Such modifications result in less sediment being available to counteract the negative hydrodynamic effects from artificial structures on sediment supply and dynamics (French, 2001). At the same time, modifications of rivers associated with agriculture and other development have increased supply of silts and total suspended solids (TSS) to marine systems. Both types of modifications additionally change the composition of sediment, which may alter the quality and suitability of sedimentary habitats (Section 2.3.3). Comparing changes in estuarine sediment communities from before to after the undamming of rivers, which is presently occurring the US (Gelfenbaum et al., 2015), may be a fruitful means of exploring these interactions. Directly testing the interaction effect between artificial structures and human modifications, such as beach nourishment, can also bring valuable insights for potential mitigating strategies (Colosio et al., 2007).

2.3.2. Intermediate-scale hydrodynamic effects

At large to moderate spatial scales (tens of meters), the interruption of circulation patterns by artificial structures changes the residence time of water. Wave energy and flow decrease in areas that are enclosed by recreational boating marinas (Balas and Inan, 2010; Floerl and Inglis, 2003; Rivero et al., 2013) and by networks of breakwaters and groynes (Zanuttigh et al., 2005). This increases water retention and the residence time of suspended particles, particularly following storm events (Zanuttigh et al., 2005). Longer residence times may influence the larval dispersal of infaunal species and affect recruitment to the benthos by inhibiting passive transport (Sim et al., 2015). Longer residence times also coincide with increases in turbidity, temperature, and pH (Munari, 2013; Rivero et al., 2013). While pH and temperature are known to impact larval and post-settlement survival of infauna (Talmage and Gobler, 2011) and infaunal assemblage structure (Hale et al., 2011), the extent of increases in these two factors that is attributable solely to artificial structures may be of little consequence biologically (Rivero et al., 2013). Increased turbidity, however, may have significant implications for infaunal communities, with particularly negative potential effects on suspension feeding bivalves (Bricelj et al., 1984; Ellis et al., 2002).

Interrupted circulation patterns also lead to changes in the bathymetric profile of sedimentary habitats. The seafloor becomes shallower over time in areas where artificial structures have reduced flow and increased sediment accumulation, such as on the landward sides of breakwaters (Scyphers et al., 2011). The shallower areas between breakwaters and the shore are known to support distinct assemblages of fish (Scyphers et al., 2011) and infauna (Bertasi et al., 2007; Martin et al., 2005; Munari et al., 2011). Depth-related zonation patterns of infauna also differ on the landward sides of breakwaters, with deeper-water species inhabiting shallower depths than in sediments where breakwaters are absent (Bertasi et al., 2007). However, these trends may primarily be the result of other small-scale hydrodynamic-related processes, such as changes in granularity (Section 2.3.3) or organic enrichment (see Section 2.4). The relative importance of these multiple, often co-occurring mechanisms, has yet to be evaluated directly in the field.

Seawalls and bulkheads reflect waves and thus tend to increase wave energy, scouring, and erosion of sediment (Pilkey and Wright, 1988). The extent and rate of erosion depends on local hydrodynamic conditions as well as sediment supply, and may not be evident within the first few years of seawall construction (Jaramillo et al., 2002). Sediment erosion may directly affect soft-sediment communities by causing concomitant erosion of small organisms such as meiofauna (Spalding and Jackson, 2001). It may indirectly affect sedimentary communities by reducing habitat availability for resident and dependent taxa (Brown and McLachlan, 2002; Rizkalla and Savage, 2010), by altering shoreline profile (Dugan et al., 2011), and by modifying key attributes of the abiotic environment such as sediment grain size (Section 2.3.3). Armored beaches tend to be steeper than unarmored beaches (Morley et al., 2012). This can limit the growth of macrophytes and negatively affect mobile organisms, which rely on them for food and nursery habitat (Morley et al., 2012; Peterson et al., 2000). Beach steepening is likely to increase as sea-level rise accelerates (Hansom, 2001), and may augment the impact of shoreline structures depending on local conditions (Kraus and McDougal, 1996). Additionally, intertidal habitats that are armored with seawalls are often narrower than unarmored shorelines (Bernatchez and Fraser, 2012; Fletcher et al., 1997; Hall and Pilkey, 1991; Heatherington and Bishop, 2012; Pilkey and Wright, 1988) and in many instances organisms are unable to compensate for lost habitat by increasing in density (Lucrezi et al., 2010; Schlacher et al., 2016).

Furthermore, modified currents and wave action can cause changes in intermediate- to small-scale sediment habitat features, such as scour holes and ripple patterns (Barros et al., 2004; Kambekar and Deo, 2003; Uijttewaai, 2005). Such features form as waves and currents move across surface sediments and reconfigure the distribution of individual grains (Blondeaux and Vittori, 2016). An interruption in waves and currents can therefore lead to modified topographical features and changes in structural complexity at intermediate spatial scales. In a habitat already at the low end of the complexity spectrum, this may have profound effects on the diversity of species (Byers and Grabowski, 2014), particularly communities of meiofauna (Sun et al., 1993). Ripple patterns in sediments have been shown to vary depending on distance from hard structures and coincide with distinct macrofaunal communities (Barros et al., 2004). More work is needed, however, to improve our understanding of the effects of altered topography from artificial structures on sediment community structure (Barros et al., 2004; Davis et al., 1982).

2.3.3. Small-scale hydrodynamic effects

At small spatial scales (centimeters to meters), artificial structures impact soft sediment assemblages via several flow-related mechanisms. Altered current-flow can impact sedimentary organisms directly. For instance, waves rebounding from seawalls and bulkheads might influence the feeding behavior of filter feeders at small scales by altering the dimensions of feeding apparatus and reducing the conditions that are suitable for feeding (see Li and Denny,

2004; Marchinko and Palmer, 2003 for examples from other wave-exposed settings). Reflected waves may also impact the morphology of sedimentary species, as stunting of growth forms has been observed in response to other causes of wave action (La Nafe et al., 2012; Norton-Griffiths, 1967).

Hydrodynamic changes also impact sedimentary organisms indirectly by altering other key physical variables at relatively small spatial scales. Modified patterns of flow cause considerable changes in granularity, or grain size composition, of surrounding sediments. Finer sediments accumulate where flow is reduced, such as on the landward sides of breakwaters (Zanuttigh et al., 2005) and artificial reefs (Fricke et al., 1986), and in areas where flow is impeded by structures for aquaculture (Guiral et al., 1996; McKindsey et al., 2011) and recreational boating (Rivero et al., 2013). Conversely, sediments become coarser where there is higher flow or wave energy increases scour, such as at the base of seawalls (Bozek and Burdick, 2005), on the down current sides of wind turbines (Maar et al., 2009), or surrounding anchor blocks associated with aquaculture structures (Guichard et al., 2001). As a general rule, the finer the sediment, the shallower the oxic layer, as finer grains have less interstitial space for water and air passage (Byers and Grabowski, 2014). Altered granularity may also therefore have consequences for primary production and the remineralization of organic matter in sedimentary systems as suggested by recent microbial studies (Sun et al., 2013).

Granularity is known to influence benthic communities (Snelgrove and Butman, 1994) and has been highlighted in many studies as a probable mechanism by which artificial structures alter soft sediment community composition (Ambrose and Anderson, 1990; Barros et al., 2001; Fricke et al., 1986). Infaunal assemblage structure tends to covary with grain size in the sediments surrounding breakwaters, for instance (Bertasi et al., 2007; Martin et al., 2005). In some cases, modified sediments support a higher density and abundance of deposit-feeding burrowers (Munari, 2013). Bioturbation from burrowers is a key process that influences sediment oxygenation and nutrient cycling (e.g. Lohrer et al., 2004; Norling et al., 2007; Olsgard et al., 2008). Changes in bioturbation may therefore impact nutrient and oxygen fluxes (Norling et al., 2007; Solan et al., 2004; Thrush et al., 2006). In other instances, the addition of breakwaters has been shown to increase the abundance of suspension feeding bivalves on sandy beaches (Bertasi et al., 2007). In separate experiments the removal of suspension feeders was found to cause an increase in microphyte standing stocks as well as an increase in $\text{NH}_4\text{-N}$ efflux in the light (Thrush et al., 2006). The authors also found that the removal of suspension feeders led to greater changes than the removal of deposit feeders (Thrush et al., 2006). Structural changes to sedimentary communities caused by artificial structures may therefore have important indirect consequences on functional properties.

The hydrodynamic changes that result from the introduction of artificial structures tend to covary with a number of other chemical and biotic parameters as well, each of which has additional implications for sedimentary ecosystems (Table 1). These and other interrelated effects are discussed below (Sections 2.4 and 2.5).

2.4. Organic enrichment and material fluxes

The accumulation of fine sediments tends to coincide with organic enrichment, which is known to impact soft sediment communities (Pearson and Rosenberg, 1978). In low flow settings, such as those surrounding some artificial structures (Al-Bouraei, 2013), sediment organic content is generally high (Snelgrove and Butman, 1994). Artificial structures may further enhance organic matter inputs to adjacent sediments by supporting flora and fauna that contribute dead tissue and organic waste to sediment (Airoidi et al., 2010; Cranford et al., 2009; Giles, 2008; Holmer et al., 2005; Kennicutt et al., 1996; McKindsey et al., 2011; Montagna and Harper, 1996; Wildish et al.,

2001). For example, Maar et al. (2009) found increased ammonia, detrital material, and fecal pellets down drift from blue mussel populations attached to offshore wind turbines. The mussels did, however, reduce the availability of phytoplankton and certain zooplankton species, which are food resources for infaunal suspension feeders (Maar et al., 2009). Similarly, the production of phytodetritus by natural rocky reefs can influence the organic content of soft sediments (Agnew and Taylor, 1986; Riggs et al., 1998), with flow-on effects to infaunal recruitment (Renaud et al., 1999), community composition, and trophic dynamics in some instances extending well beyond the immediate vicinity of the source (Bishop et al., 2010).

Conversely, where artificial structures enhance flow, reduce primary and secondary productivity, and/or serve as barriers to transport of allochthonous organic matter, they may reduce sediment organic content. Wrack accumulations are often less on armored than unarmored intertidal shorelines in part due to reduced wrack retention and likely also due to reduced wrack supply (Heatherington and Bishop, 2012; Heerhartz et al., 2014; Sobocinski et al., 2010). In some instances, seawalls may reduce organic matter retention by accelerating decomposition rates and/or decreasing organic matter residence times (Harris et al., 2014). In other instances, the reduced retention of wrack may be due to loss of the high intertidal and supratidal habitat, at which material accumulates on unarmored shorelines (Dugan et al., 2008; Heatherington and Bishop, 2012). The reclamation of land adjacent to seawalls can reduce terrestrial sources of leaf litter (Higgins et al., 2005) and the constraint by coastal armoring of intertidal habitat for primary producers, such as mangroves, may reduce autochthonous litter supply (Heatherington and Bishop, 2012). The net effect is reduced food and habitat for invertebrates, and consequently altered invertebrate communities (Dugan et al., 2008; Heerhartz and Toft, 2015; Heerhartz et al., 2014, 2016).

The paradigm is that the abundance of suspension feeders declines and the abundance of deposit feeders increases with sediment organic content (Pearson and Rosenberg, 1978). Yet, while several authors have found differences in soft sediment community structure that coincide with organic content (Ambrose and Anderson, 1990; Barros et al., 2001; Danovaro et al., 2002; Zalmon et al., 2014), they do not appear to follow a consistent pattern. Increased organic content in sediments surrounding artificial structures can reduce oxygen concentrations, leading in some cases to sediment hypoxia (Danovaro et al., 2002; Wilding, 2014). Hypoxia events can potentially alter net primary and secondary production and reduce the diversity and abundance of species in sedimentary habitats (Diaz and Rosenberg, 2008). However, hypoxia probably arises only when structures are added to already oxygen-deficient sediments (Wilding, 2014). In sufficiently oxygenated sediments, organic enrichment surrounding artificial structures may instead dampen seasonal variability in nutrient availability that would occur if the structures were absent. For instance, Machado et al. (2013) found that sediments surrounding subtidal artificial reef balls did not exhibit the same seasonal variation in reactive phosphorus, total nitrogen, or organic carbon as a control site without reef balls (Machado et al., 2013).

In addition to influencing organic matter inputs to sediments, artificial structures may also influence inputs of calcareous material. Sessile invertebrates on hard structures generate large amounts of shell material that fall to marine sediments when the organisms die, are damaged, or become dislodged (Ambrose and Anderson, 1990; Barros et al., 2001; Machado et al., 2013). This influx of shell material alters sediment granularity such that it becomes coarser immediately surrounding artificial structures (Barros et al., 2001). Presumably such habitat modification could alter sedimentary communities, by reducing the foraging efficiency of some sediment-feeding predators, and by impeding burial of some infaunal taxa (Gutiérrez et al., 2003). However, to our knowledge, no studies have tested whether such mechanisms are responsible for differences in infaunal community structure immediately surrounding artificial structures.

2.5. Contaminants

Artificial structures can also modify sedimentary communities by directly and indirectly altering their exposure to contaminants. The effects of contaminated sediments on aquatic communities have been extensively reviewed for a range of environmental conditions (Burton and Johnston, 2010) and here we focus solely on the role of artificial structures in modifying contaminant concentrations and exposure. Artificial structures may directly influence contaminants as a consequence of the materials from which they are constructed. They may indirectly influence contaminants by altering properties of the sediment that affect their affinity to bind contaminants, by influencing water retention, and as a consequence of the activities that they support.

The materials from which artificial structures are constructed and the biocidal coatings applied to them can have large influences on contaminant loads. In recent decades, there has also been growing concern about toxic leachate from car tires (Collins et al., 1995, 2002; Day et al., 1993; Degaffe and Turner, 2011; Wik and Dave, 2009), which have been used to construct artificial reefs (Collins et al., 2002; Fabi et al., 2011; Thierry, 1988), breakwaters, and other coastal defense installations (Collins et al., 1995, 2002). Tires leach zinc and polycyclic aromatic compounds (Collins et al., 1995, 2002; Degaffe and Turner, 2011). Zinc, in particular, likely penetrates adjacent sediments (Degaffe and Turner, 2011), bioaccumulates in invertebrates (Amiard et al., 2007; Hanna et al., 2013), and increases mortality of sedimentary organisms (Hanna et al., 2013). Artificial reefs have also been constructed from coal and oil ash (Collins and Jensen, 1995; Collins et al., 1992, 1994; Nelson et al., 1994; Vose and Nelson, 1998), which contain large amounts of heavy metals that can increase invertebrate mortality (Hamilton et al., 1993) if leachates are not contained via a stabilization process (Breslin and Roethel, 1995; Collins and Jensen, 1995; Pickering, 1996; Shieh and Duedall, 1994). Similarly, the treatment of wooden pilings in marinas and jetties with copper chromated arsenate (CCA) has been found to be a significant source of copper contamination (Hingston et al., 2001; Weis and Weis, 1996; Weis et al., 1993). When metal biocides are released into waterways their ions can bind to bottom sediments (Di Franco et al., 2011; Singh and Turner, 2009), and subsequently dissociate to enter the porewater and overlying water as free metal ions (Simpson et al., 2004). Contamination of sediments in turn influences sediment community structure and diversity (Neira et al., 2014; Rivero et al., 2013; Sim et al., 2015; Wilkie et al., 2010).

Changes in flow and granularity caused by artificial structures can also influence contamination indirectly. Increased deposition of fine sediments, for instance, has been linked to increased contamination due to the greater affinity and capacity of fine sediments to bind contaminants (Burton and Johnston, 2010; Simpson et al., 2013). Recreational marinas generally experience water retention and reduced flushing because they are built in low energy environments or surrounded by breakwaters. This has consequences for water quality and contaminant retention (Johnston et al., 2011; McGee et al., 1995; Schiff et al., 2004). Because vessel anti-fouling (AF) paints and the cleaning of pontoons and jetties are major contaminant sources (Srinivasan and Swain, 2007), marinas are hot spots of metal contamination in coastal and estuarine systems (Dafforn et al., 2011; Rivero et al., 2013; Schiff et al., 2007; Turner, 2010; Warnken et al., 2004).

The activities that artificial structures support are also a major source of contamination. For example, the >7000 oils and gas platforms installed around the world (Gray et al., 1990; Wilson and Heath, 2001) pollute the marine environment through accidental spillage, discharge of drill cuttings, and discharge of production water (Kingston, 1992). Studies investigating the impacts of offshore oil and gas drilling have found impacts to benthic sediment communities extending up to 500 kilometers from the rig or platform (Gray et al., 1990; Kingston, 1992). Opportunistic species may proliferate under moderate levels of pollution, but at high levels even opportunists are unable to persist (Gray et al., 1990; Kingston, 1992; Olsgard and Gray, 1995) More

broadly, responses of invertebrate communities to contaminants from oil and gas platforms can involve reduced cellular viability, considerable changes in abundance and reductions in diversity indices such as evenness (Edge et al., 2016; Gray et al., 1990; Johnston and Roberts, 2009; Kingston, 1992; Olsgard and Gray, 1995; Sandrini-Neto et al., 2016). The severity of decreases appears to depend on the frequency with which sedimentary communities are exposed to contaminants (Sandrini-Neto et al., 2016). While many infaunal organisms recover relatively quickly after contamination events, such as oil spills (Bolam et al., 2002; Sandrini-Neto and Lana, 2014; Sandrini-Neto et al., 2016), others are highly sensitive to oil contamination (e.g. *Bulla strita*, *Tellina versicolor*) (Sandrini-Neto et al., 2016).

2.6. Biotic effects

An additional mechanism by which artificial structures may affect sedimentary ecosystems is by modifying biotic interactions. Artificial structures can modify predator-prey interactions by altering predator abundance, prey abundance, or encounter rates (Caine, 1987; Davis et al., 1982; Kneib, 1991; Firth et al., 2017-in this issue). They may also modify positive interactions among species, such as facilitation, by altering the abundance of habitat forming species and eco-engineers. For instance, by aggregating green shore crabs (*Carcinus maenas*) for commercial harvest (Sheehan et al., 2008), foraging shorebirds (Sheehan et al., 2012), and mobile epifauna (Sheehan et al., 2010a), crab tiles may enhance predation in their vicinity. Reductions in the abundance of habitat-forming macrophytes as a result of shading (Section 2.2) or steepening of habitat profiles (Section 2.3) can affect the composition and abundance of infaunal taxa they facilitate (Eckman, 1983; Fonseca and Fisher, 1986; Ward et al., 1984).

Subtidal artificial reefs attract a variety of predatory fish (Brotto et al., 2006; Wilhelmsson et al., 2006), which move into surrounding areas (Henderson et al., 2014), feed on sedimentary organisms (Kurz, 1995; Lindquist et al., 1994), cause physical disturbances to sediments (Hall et al., 1991; Thrush et al., 1991; VanBlaricom, 1982), and introduce additional nutrients by excreting waste (Cheung et al., 2010). Off-reef foraging distance can vary, but the greatest foraging activity tends to occur within 10 m (Frazer et al., 1991; Nelson et al., 1988; Posey et al., 1992). Posey and Ambrose (1994) emphasized the importance of increased predation on infauna from consumers associated with natural reefs, but noted that these dynamics may differ on artificial structures (Posey and Ambrose, 1994). Such “halo” effects have been much discussed in the literature, but few studies have employed the experimental designs necessary to establish causal linkages between predation and the structure of sedimentary communities (but see Hill et al. (2013) for a small-scale experimental study). Gradients in reef-associated predation frequently coincide with gradients in hydrodynamic factors that may also affect infaunal composition (Galván et al., 2008; Jones et al., 1991; Langlois et al., 2005), raising the possibility of confounding variables.

Conversely, feeding by predators such as gray whales (Weitkamp et al., 1992) may be reduced in sedimentary habitats where artificial structures block their movement or foraging activities. Similarly, fishing activities by humans, and specifically bottom trawling, may be reduced in some cases by the introduction of artificial reefs, thus having positive effects on the abundance of some taxa and on species richness (Cheung et al., 2009; Liu et al., 2011; Munoz-Perez et al., 2000). Since bottom trawling can affect a variety of physical, chemical, and biotic processes (Thrush and Dayton, 2002), artificial structures placed in heavily trawled areas may indirectly affect sediment dynamics, lower sediment nutrient levels (Ambrose and Anderson, 1990), and facilitate the remineralization of organic matter, bioturbation, and bioirrigation of sediments (Cheung et al., 2009). This has not been empirically tested in the field and is potentially applicable only in trawled, subtidal sedimentary environments. Artificial structures that are used to attract harvested species, such as ‘crab-tiles’ in the *Carcinus maenas* fishery in the

United Kingdom, tend to increase the use and trampling of soft sediment habitats by humans, which impacts infaunal communities (Sheehan et al., 2010b).

Artificial structures may also affect demographic patterns and behavioral traits of soft sediment predators and infauna. For instance, Henderson et al. (2014) found that summer flounder (*Paralichthys dentatus*) were larger near artificial reefs than in more distant soft sediments. They suggested this pattern arose at least in part due to behavior, as fish associated with artificial structures tended to be more territorial, resulting in the competitive exclusion of smaller individuals (Henderson et al., 2014). Similarly, Long et al. (2011) found that fish feeding on juvenile blue crabs (*Callinectes sapidus*) in Chesapeake Bay were larger in size and had higher foraging rates along armored shorelines than near natural *Spartina* marshes where artificial structures were absent. Demographic responses to artificial structures are also plausible among infaunal populations. Dahlgren et al. (1999) found more larger-bodied and fewer small-bodied macrofauna near natural reefs. Differential demographic responses among fish and infauna would be expected if different size classes were affected in distinct ways by the altered physical, chemical, and biotic conditions surrounding structures. Such responses likely vary considerably over space and time (Langlois et al., 2006), but may be an important consideration for monitoring efforts and future research.

Lastly, in some instances, artificial structures appear to facilitate bioengineering species in surrounding sediments, but in areas with previously abundant bioengineers, they may have negative impacts. In transects extending perpendicularly from subtidal artificial structures in Southern California, Ambrose and Anderson (1990) found the tube-dwelling Onuphid polychaete *Diopatra ornata* was only present immediately adjacent to structures. *Diopatra* spp. was also documented to occur in high densities surrounding oil platforms in the same region (Davis et al., 1982). Similarly, Heery and Sebens (unpublished data) have observed higher densities of the tube-dwelling Chaetopterid polychaete *Spiochaetopterus costarum* immediately adjacent to artificial structures in Puget Sound, Washington. Ambrose and Anderson (1990) suggest that enhanced densities of polychaetes like *D. ornata* in the vicinity of structures may stabilize sediment, provide refuge habitat, or serve in some other facilitative capacity that ultimately increases infaunal diversity and abundance. However, they did not evaluate this hypothesis directly in their study.

3. Factors influencing the direction and magnitude of impacts

The way in which artificial structures modify sedimentary communities depends on their design and spatial configuration, the characteristics of the abiotic and biotic environment in which they are placed, and the scale of the impact, including area affected and duration (Airoldi et al., 2005; Martin et al., 2005). Unfortunately, many scientific-based assessments often neglect these complex interactions and scaling issues, which limits our current capability to predict the impacts of future developments (Loke et al., 2015). Studies to date have found tremendous variation in the patterns and trends they have observed in sedimentary habitats where artificial structures have been placed. This is likely due at least in part to inherent variation in the direction and magnitude of impacts from artificial structures, both over space and time, and across multiple spatial scales. For each of the effects documented above, there are a number of factors that likely influence variation in observed patterns in the field and are worth considering when seeking to identify generalizable trends.

Placement loss (Griggs, 2005; Section 2.1), by definition, increases with the aerial extent of foundations constructed in sedimentary habitat, and may be especially large in coastal areas where construction of structures is accompanied by backfill to reclaim land. In coastal environments, losses can be amplified by passive erosion, which results from structures inhibiting natural cycles of shoreline retreat (Griggs, 2005). The extent of such passive erosion can depend on the tidal elevation

at which a defense structure is built, as well as whether a shoreline is presently in an accretive or erosive state (Archetti and Romagnoli, 2011; Lin and Wu, 2014). Active erosion of sediment adjacent to structures, through wave reflection, scouring, and 'end effects' (Griggs, 2005) can also affect the magnitude of habitat loss. The effects are greatest where sand input is low and wave energy high (Lin and Wu, 2014; Miles et al., 2001). They are also dependent on the extent to which structures are designed to absorb versus reflect wave energy (e.g. hollow seabed versus solid concrete seawall designs) (Hettiarachchi and Mirihagalla, 1998; das Neves et al., 2015; Zanuttigh et al., 2005).

Impacts of artificial structures on sediment communities also vary spatially according to the extent to which they modify the abiotic and biotic conditions and local processes that control soft-sediment community assembly (Airoldi et al., 2005; Martin et al., 2005). The position (i.e. onshore vs offshore), orientation (i.e. perpendicular or parallel to shorelines), permeability (solid versus rock wall), dimensions and spacing of structures are all factors that could influence the extent to which structures intercept longshore drift, tidal and other currents, which in turn shape sedimentary communities by determining sediment, larval and resource (e.g. wrack and organic matter) transport and deposition (Martin et al., 2005; Bishop et al., 2017-in this issue). For example, Shyue and Yang (2002) found that the area of scour surrounding subtidal artificial reefs was heavily influenced by the structure's height, although differences in ambient flow between locations were also important (Shyue and Yang, 2002). The placement of seawalls with respect to tidal height and local wave energy are important factors determining the extent of scour and sediment coarsening in intertidal environments (Weigel, 2002). As another example, the impacts of oil rigs on adjacent sediment communities could be mitigated at deeper waters because of higher environmental stability and greater potential of dilution and dispersion of pollutants (Burns et al., 1999; Ellis et al., 1996). Terlizzi et al. (2008), however, reported an opposite trend, possibly because platforms at deeper sites are taller, therefore leaching greater amounts of contaminants or providing more surface area for growth of fouling invertebrates which slough off to influence sedimentary communities (Goddard and Love, 2010; Love et al., 1999; Terlizzi et al., 2008).

The spatial arrangement and isolation of artificial structures could affect sedimentary environments both directly, by affecting patterns of sediment deposition, and indirectly, by affecting the capability of artificial reefs to attract grazing and predatory fish communities. For example, on a Brazilian artificial reef, the proximity of reef balls to one another influenced their effect on organic and fine sediment inputs to adjacent habitat (Zalmon et al., 2014), with inputs greatest at a larger spacing. Overall, the large-scale effects of multiple structures (such as offshore structures) may differ from their local effects. For example, parks of offshore wind farms can act as a partial blockage of the overall current field: the blocked water volume is forced around the park, which leads to a decrease in the flow inside the park and an increase in flow velocities on the sides of the park (Airoldi et al., 2016). These blockages depend on the distance between piles (typically 600 to 1200 m), the diameter of the piles (6–10 m), the overall number of wind turbines in the park and the lay-out of the farm.

The sediment grain size and hydrodynamic regime can also determine the extension and severity of some of the impacts. For example, the effects of crab-tiles used to attract crabs for harvest depend on the grain size of the sediments where they are placed (Sheehan et al., 2010a). Similarly, the impacts from the sediment spills due to dredging for foundation and cable trenches of offshore activities will primarily be of local nature in low-current environments, while in high-current environments far-field impacts of lower intensity will prevail, due to advection and dilution (Airoldi et al., 2016). Further, the impacts of structures on sediments may vary spatially according to the processes occurring at the time of their construction, for example fouling community colonization (Underwood and Anderson, 1994) which in turn determines resource subsidies to adjacent sedimentary habitats (Airoldi et al., 2010; Goddard and Love, 2010; Love et al., 1999).

The effect of structures on sediment communities may also be expected to vary according to the diversity and identity of soft sediment communities at disturbed sites (Martin et al., 2005). For example, the diverse communities of dissipative beaches are more susceptible to the effect of structures than the more depauperate assemblages of exposed sandy beaches (Martin et al., 2005). Oil and gas rigs or artificial reefs that exclude fishing vessels may have large positive effects on biodiversity by removing or alleviating dredge or trawling disturbance to ecosystem engineers such as clams, tube worms, or seagrasses (González-Correa et al., 2005; Pearce et al., 2014). Conversely, if artificial structures have a negative effect on ecosystem engineers (e.g. Lemasson et al., 2017-in this issue; Teagle et al., 2017-in this issue).

Not only do the effects of structures vary spatially according to their abiotic and biotic context, but they may also vary temporally. Effects of artificial structures on sediment communities may strengthen or weaken with time since their construction. For example, because the development of fouling communities on structures takes time (Underwood and Anderson, 1994), indirect effects on sediment communities resulting from sloughing of algae or shell (Airoldi et al., 2010; Goddard and Love, 2010; Love et al., 1999) or fouling communities depositing feces (Maar et al., 2009), may increase with time since construction. Conversely, pulse impacts associated with the construction phase, such as those resulting from turbidity plumes or construction noise deterring benthic predators (Slabbekoorn et al., 2010) may weaken over time (Jaramillo, 2012). The effect of structures on sediment communities may also vary temporally according to natural variation in the strength of the abiotic and biotic processes they disrupt. For example, artificial reefs in Brazil reduce current velocities predominantly during months of high flow from the Paraíba do Sul River (Machado et al., 2013) and, conceivably, enhancement of predator foraging patterns around artificial structures may vary seasonally according to the biology of species.

4. Approaches employed in past studies

The effects of artificial structures on soft sediment ecosystems can currently be considered based on three types of information. Firstly, inferential studies that examine the response of soft sediment organisms to environmental changes associated with artificial structures (e.g. shading, modification of sediment grain size and so forth) provide proximal insights, but are primarily helpful for generating hypotheses. Secondly, surveys that examine how environmental variables and sediment communities vary spatially in the areas with and without artificial structures can lend further proximal information, but are limited by inherent spatial variation in sedimentary ecosystems and confounding variables. Lastly, Before/After and Control/Impact (BACI) designs test for causal effects of structure construction on sedimentary systems.

BACI designs are recognized as a robust approach for documenting environmental impacts (Hilborn and Walters, 1981; Underwood, 1994), as they test for causality (Underwood and Peterson, 1988). If effectively implemented, these designs provide the advantage of controlling for temporal changes that are confounded with the introduction of an artificial structure, as well as site-specific differences that are unrelated to structure introduction. Reference sites used in such designs must be selected carefully to ensure they are sufficiently similar to those where an artificial structure will be introduced without being in range of the structure's effects (Stewart-Oaten et al., 1986). In order to detect changes, data collection in BACI-type studies must also continue over a period of time that coincides with the temporal scale of the effects being measured (Stewart-Oaten et al., 1986). These limitations commonly make BACI-designs unfeasible, and the approach has been used only rarely as a means of characterizing the effects of artificial structures on soft sediment ecosystems (Jaramillo et al., 2002).

Most studies have instead sought to characterize patterns of spatial variation in soft sediment communities that correlate with the presence of or the distance from an existing artificial structure (Ambrose and

Anderson, 1990; Barros et al., 2001; Davis et al., 1982). Such studies have many limitations. Sites where artificial structures are constructed are also usually non-randomly selected, so it is likely that there are pre-existing differences between sites with and without structures that are unrelated to the construction or presence of the structures themselves. Even within a single site, it is difficult to discern patterns associated with artificial structures in surrounding sediments due to the inherent patchiness of soft sediment communities over time and space (Morrisey et al., 1992a, 1992b). Observationally derived differences in community structure do not therefore demonstrate causation, nor do they allow for conclusions regarding the mechanisms that are behind observed differences.

Many of the observational studies we reviewed emphasized specific physical, chemical, or biotic factors as the potential mechanism driving community and species distribution patterns observed in the field. However, there remains a strong need for research that tests the importance of multiple mechanistic processes associated with artificial structures on soft sediment ecosystem response. In addition, most studies focused on relatively local impacts of artificial structures on immediately surrounding soft sediments. Few studies have considered the cumulative impacts of multiple structures on sediments at larger spatial scales. There may be non-linear effects of adding more artificial structures to a seascape, such as a tipping point beyond which there is no longer sufficient sedimentary substrate to support particular groups of organisms, or beyond which the environment is no longer fit for habitation. Studies on the cumulative impacts of structures at the landscape scale are urgently needed as marine urbanization accelerates (Dafforn et al., 2015; Johnston et al., 2015; Bishop et al., this special issue).

5. Research gaps and future directions

As artificial structures extend across an increasingly large proportion of sedimentary seascapes (Airoldi and Beck, 2007), it is important that we improve our understanding of impacts on sedimentary ecosystem structure and function so that we can manage ocean sprawl in more ecologically sustainable ways. This will require developing and implementing rigorous monitoring programs, expanding academic research to encompass a wider breadth of testable hypotheses relating to artificial structure introduction, and improving the methodology in scientific studies so that the hypotheses in question are addressed more effectively (Dafforn et al., 2015).

5.1. Monitoring

Artificial structures can affect the ability of habitats and species to deliver ecosystem services that have societal benefits (Atkins et al., 2011). Regulatory frameworks can help to ensure that the style and scale of artificial structures are sustainable and do not risk the provision of ecosystem services (Mee et al., 2008). Such frameworks are only currently in place in certain areas of the world (e.g. EU Habitats Directive). Monitoring allows for regulatory bodies, where active, to evaluate the changes in assemblages or communities as a result of an intervention, such as building a seawall (Hiscock, 1998). Details about the techniques used to obtain monitoring data are not discussed here, as there are many other excellent sources (Kingsford and Battershill, 2000; McIntyre and Eleftheriou, 2005). However, several important considerations are worth emphasizing in relation to the design of monitoring studies.

'Before' and 'after' samples are essential in order to detect any modifications in the natural patterns in assemblages as a result of introducing an artificial structure. Environmental consequences of an intervention are actually variations in space and time of ecological processes which control the structure of species assemblages (Green, 1979; Underwood, 1992). Impacts can therefore be detected as changes in the absolute or relative abundances of taxa, changes in the variance of these abundance metrics, or changes in measured ecological processes

(Underwood, 1992). These changes need to be separated from natural variation through time (at a variety of scales) at the sites sampled (Underwood, 1992). In estuarine systems particularly, samples taken at the same site a few months, or even a week apart can differ significantly (Glasby, 1997; Morrissey et al., 1992a).

It is also necessary to compare potentially impacted sites with control/reference sites not subject to the impact (Stewart-Oaten et al., 1986). Any difference between a single reference site and the potentially impacted site may not be due to the impact because assemblages are naturally variable in space. To overcome confounding due to this natural variation, it is desirable to have replicated reference and impacted locations (Underwood, 1989). In most cases, however, only one impacted site exists. In such cases, patterns in the biota of the potentially impacted site are compared with the average of replicated reference sites to adequately detect the impact. This can be done using asymmetrical ANOVA in Beyond-BACI designs (Underwood, 1992). Information about the spatial scale of the impact is also necessary to understand and detect impacts (Bishop et al., 2002). Spatially nested designs can enable impacts to be assessed at multiple spatial scales.

Additionally, in many calls from management agencies for scientific information, there are requests for baseline monitoring in the belief that such monitoring can inform the design of subsequent monitoring efforts (Field et al., 2007). This can only be true in two sets of circumstances. The first is that the baseline sampling design is exactly the same as the subsequent monitoring as this can enable the direct comparison of previous and subsequent data to allow a test of the time x treatment interaction (Stewart-Oaten et al., 1986; Underwood, 1992). The second is where precision estimates and analysis outputs can be used to inform subsequent sample designs. In an example from marine conservation, Coleman et al. (2013) used pilot or baseline sample data to estimate the number of samples needed to retain the null hypothesis of no impact with confidence (Coleman et al., 2013); this was the number of samples used for subsequent monitoring. Only by explicitly connecting the baseline data with the analytical frameworks necessary to test the hypotheses of effects can we move beyond the limitations that exist in some monitoring data of the past (Burt, 1994) to generate reliable data on the effects of artificial structures on sedimentary assemblages.

Finally, monitoring of the impacts related to artificial structures are often considered on a case by case basis and have ignored the potential cumulative impact on sedimentary habitats (Halpern et al., 2008). Future impact assessment of artificial structures on sedimentary habitats and assemblages would be more appropriate if multiple development “impacts” were monitored as part of an integrated study, with predetermined comparable metrics, that are able to contextualize measured effects at ecosystem-relevant scales.

5.2. Future research directions

There remain many unanswered questions as artificial structures rapidly proliferate in sedimentary environments. Sedimentary ecosystems are dynamic, complex, and influenced by processes and feedbacks that remain poorly understood, and the introduction of artificial structures may cause complex patterns that are difficult to identify in the field, particularly when sampling regimes are temporally and spatially limited. Given the profusion of uncertainties surrounding sedimentary ecosystem dynamics in general, improving our understanding of the effects of artificial structures will require strategic and careful selection of research objectives.

We suggest several areas of study that would be particularly helpful for advancing current knowledge. Much of our understanding of the mechanisms by which structures modify sediment communities is inferential. There is therefore need for more studies that evaluate mechanism directly. This is particularly important if we hope to design structures in such a way that they have minimal impacts on sediment communities and in some instances provide benefits (i.e. ecoengineering, Loke et al., 2017-in this issue). Additionally, most studies to date have

quantified changes in the abundance or richness of macroinvertebrates, and future studies are needed to examine the effects on key biological parameters, such as reproduction and growth, as well as key ecological processes, such as trophic transfer. Past studies have primarily focused on small scale effects, and there is great need for studies that improve our understanding of impacts across large spatial scales (Dethier and Schoch, 2005; Thrush et al., 1994), including alterations to connectivity (Bishop et al., 2017-in this issue) and regional-scale cumulative changes (Duarte et al., 2003) as artificial structures proliferate across an increasingly large proportion of sedimentary habitats. Along these same lines, work that characterizes the current spatial extent of artificial structures and the scale of their effects on sedimentary ecosystems would represent a valuable contribution.

Certain taxonomic groups within sedimentary ecosystems have also been poorly represented in research to date. In particular, microbes in soft sediments likely have a central role in the functioning of ecosystems as they form the basal elements of food webs, affect sediment chemistry, and restrict nutrient availability (Gadd and Griffiths, 1977). Although there is no direct evidence of impacts of artificial structures on these communities at present, one study has shown that biofilms in natural habitats significantly differ from those on artificial structures (seawalls; Tan et al., 2015). Much work is needed to evaluate whether ocean sprawl affects the functionality of sediments via their effects on the microbiota associated with artificial structures.

Lastly, there is tremendous need for work that clarifies the link between ecosystem structure and function in sedimentary environments. Marine sediment ecosystems provide various important services, such as mediating global carbon, nitrogen and sulphur cycles, influencing water clarity, burying, transporting and metabolizing pollutants and stabilizing and transporting sediments (Snelgrove, 1997). These services are dependent on the ecological functions of the species comprising sedimentary communities, as well as the abiotic environment (Bulleri and Chapman, 2015; Johnston and Mayer-Pinto, 2015; Lenihan and Micheli, 2001; Lohrer et al., 2004). Present knowledge gaps preclude any comprehensive or quantitative evaluation of the sedimentary ecosystem functions that are most impacted by artificial structures. Throughout this paper, we have presented hypotheses linking observed effects from structures with potential implications for ecosystem function. Such hypotheses need to be tested directly and rigorously, with direct measurement of functional properties, to be useful in any further capacity. Ultimately, it is knowledge of this link between ecosystem structure and function, and the subsequent connection between functioning and ecosystem services that will allow us to understand the effects of artificial structure proliferation on human populations and societies more broadly.

6. Conclusions

Most research to date on sediment responses to artificial structures has highlighted local patterns associated with specific structure types (Ambrose and Anderson, 1990; Barros et al., 2001; Davis et al., 1982; Maar et al., 2009; Martin et al., 2005). This review compiled findings across structures, regions, and temporal and spatial scales to create a synthesis of the current knowledge about how ocean sprawl impacts on soft sediment ecosystems. The primary ways that artificial structures modify soft sediments, directly and indirectly, include placement loss, an altered sensory environment, hydrodynamic changes, organic enrichment, toxic contamination, and changes to species interactions and community dynamics. These changes have significant consequences for the diversity and structure of soft sediment communities, affecting, in turn, ecosystem functioning and services provided to humans. However, to date, empirical studies on the effect of structures on ecosystem functioning have been lacking. Relationships between biodiversity and ecosystem functioning in sedimentary environments are complex (Loreau et al., 2001; Naeem et al., 2009; Schmitz et al., 2015), and in order to accurately predict the effects of disturbances on

functions and services, direct measures of functioning are necessary (Johnston et al., 2015). Moreover, little is known about the mechanisms driving these impacts or their scale. Consequently, at this point it is only possible to hypothesize the large-scale functional consequences that may arise from structural changes in the assemblages caused by artificial structures and the mechanisms behind them. This knowledge can only be achieved through rigorous monitoring programs based on explicit experimental structures alongside more studies that address the issue of cumulative impacts from multiple structures and assess the collective impacts of ocean sprawl, rather than just considering structures individually. Reviews such as this one and Bishop et al. (2017-in this issue) will be complemented and progressed by the collection of more primary data from studies that incorporate neglected measures of ecosystem functioning and large-scale impacts. This knowledge will guide the design and management of ocean sprawl. With the predicted increase of construction in the ocean, there is a pressing need for this information to inform solutions-based research that can mitigate the impacts on soft sediments and protect this crucial habitat.

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