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A Review of Habitat Impacts from Residential Docks and Recommended Best Management Practices with an Emphasis on the Northeastern United States

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Abstract
Small docks and floats are common in estuaries and coastal waters worldwide. These structures serve a role in coastal recreation by facilitating access to waterways. However, they can impact shoreline ecological function. While individual environmental impacts are generally minor, increasing dock proliferation and overlap with sensitive coastal resources can result in cumulative impacts that pose threats at the ecosystem level. Docks promote changes in habitat and aquatic communities through alteration of environmental conditions. Here, we review the potential environmental impacts of docks on estuarine and coastal flora and fauna and discuss best management practices (BMPs) to avoid or minimize such impacts with a focus on New England. We consider impacts in relation to the structural components of docks: the piles, decking, and floats. Impacts to salt marsh and submerged aquatic vegetation are a particular focus given the important ecosystem services these vegetated habitats provide and their vulnerability to dock-induced habitat alteration. Potential environmental impacts depend on structure size, design, and location, and can include both short-term (e.g., turbidity from pile installation) and long-term (e.g., salt marsh loss from chronic shading) effects. Such effects can be minimized through BMPs (e.g., construction outside sensitive time-of-year periods, designs to reduce shading). As BMPs tend to reduce rather than avoid environmental effects, cumulative impacts also need to be considered in the permitting process. We recommend that managers develop plans or bylaws that identify sensitive habitats where dock construction should be avoided as well as BMPs to make remaining dock proliferation less impactful.

Keywords Boardwalks · Coastal development · Cumulative impacts · Piers · Shading · Walkways

Introduction

A dock is a structure that extends into waterways to enhance access for a variety of activities. The term dock is similar to wharf or pier, with distinctions between the terms typically hinging on size, materials, or whether or not vessels are accommodated. In this paper, we focus on docks, and we use the word "dock" to describe small, residential-scale structures, which typically support recreational vessel berthing for small motor or sailing craft < 40 ft (12 m) and paddle craft such as kayaks and canoes. The major structural components of docks are pile-supported decks connected to shore and floating structures, or "floats" that are often attached to decks or shore with gangways (Fig. 1). An alternative to a pile-supported structure is a solid-fill structure or causeway. Habitat impacts are higher for a solid-fill structure, which displaces considerably more area than a pile-supported structure. Due to their habitat

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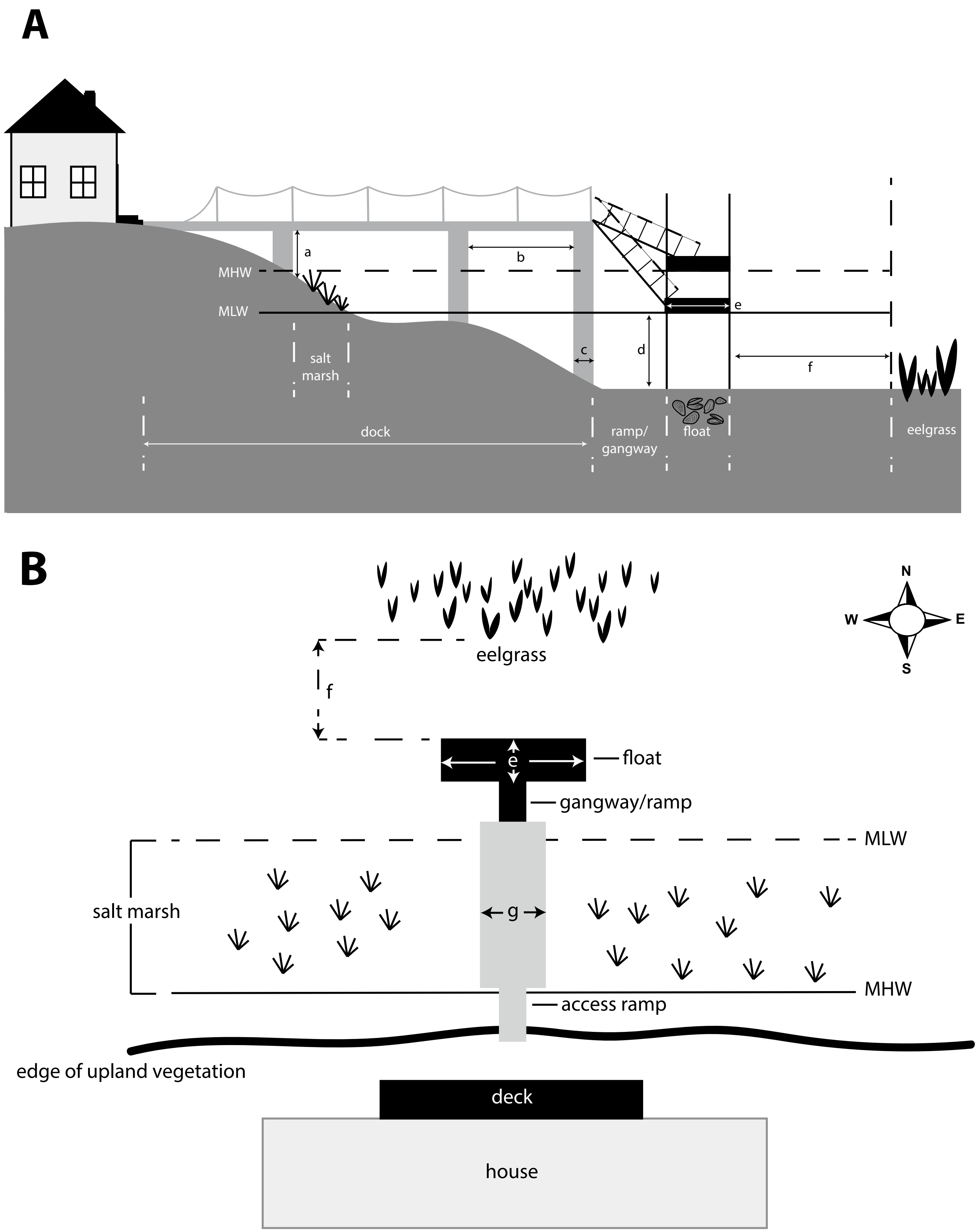
A
MHW
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salt
marsh
dock
ramp/
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eelgrass
B
N
W
E
S
eelgrass
f
e
float
gangway/ramp
MLW
salt marsh
g
MHW
access ramp
edge of upland vegetation
deck
house

◂**Fig. 1** Diagram of the components of a typical dock structure shown as (A) horizontal and (B) plan view: Pile-supported decking typically connects the bordering upland area to the adjacent shoreline. A ramp or gangway extends from the seaward end of the decking to a float located in the intertidal or subtidal zone. To minimize impacts to aquatic resources, (a) decking over salt marsh should have a height ≥ 1.5 decking width, (b) pile spacing should be maximized, (c) pile width minimized, (d) depth under float at mean low water (MLW) should be 2.5 ft (0.8 m) in shellfish habitat, (e) float surface area should be minimized, (f) the seaward edge of the float should be located at least 25 ft (7.6 m) from any bordering submerged aquatic vegetation (SAV), and (g) decking width minimized. Docks should be set at a north–south orientation when feasible to minimize shading to underlying vegetation

impacts and cost, solid-fill structures are rarely seen as new construction projects, so this review focuses solely on pile-supported docks.

Docks are common in estuaries worldwide (Gissy 1985; Kennish 2002, 2016; Kelty and Bliven 2003). Coastal development combined with a desire for water access has led to dock proliferation along the east coast of the United States (U.S.) in recent decades resulting in the construction of thousands of individual structures (Chinnis and Stidham 2001; Kelty and Bliven 2003; Seabrook 2012; Logan et al. 2018a). This proliferation has led to concerns among coastal resource managers about the cumulative environmental impacts of these structures (Buchsbaum 2001; Bliven 2003, 2005; Kelty and Bliven 2003). Due to their location along the shore, docks often conflict with coastal resource areas including salt marsh, seagrass, oyster reefs, and mudflats.

Many U.S. states have guidance documents that describe best management practices (BMPs) for docks to avoid or minimize impacts to coastal resources (Table 1). The BMPs (Table 1) are either required (e.g., ACOE Programmatic General Permits) or recommended (e.g., state guidelines) during dock design and construction. However, most available guidelines pre-date many recent studies of dock impacts (e.g., Eriander et al. 2017; Logan et al. 2018a, b). Here, we provide an updated summary of the existing literature regarding the environmental impacts of docks with a particular focus on marine and estuarine resources in the northeastern U.S. region of New England. Our review focuses on the three main structural components of docks: the piles, decking, and floats. We also consider impacts of associated boat use as well as approaches to address cumulative impacts. We evaluate the existing information to recommend BMPs that avoid and minimize dock impacts and limit cumulative impacts at the scale of a waterbody.

Piles

The only structural components of docks that always result in direct habitat impacts are the support piles. Support piles are often made of wood due to its low cost, ease of use, and resistance to corrosion, rust, and spalling (i.e., fragmenting) (Lebow et al. 2019). Cement, steel, and composites are also options as pile materials (Bright and Smith 2002). There are several primary impact-producing factors from piles: habitat displacement, ecosystem interactions with alternative materials, creation of habitat for invasive species, shading and circulation impacts to bordering habitats and marine resources, contamination of water and fouling organisms from the leachates from pressure-treated and creosote-treated wood, and construction-related impacts. Each of these impacts is described below.

Habitat Creation and Alteration

Piles displace existing habitat and create new habitat, so they represent a direct source of habitat alteration. There are three variables to consider how piles affect habitats: the piles themselves, the pile design (standard and bio-friendly designs), and the pile material. How each of these variables affects fish and invertebrate diversity and productivity is described below.

Piles

Habitat displacement can benefit some fish species while negatively impacting others (Grothues et al. 2016; Brandl et al. 2017). Piles benefit some fish species by providing structure and increased foraging habitat (e.g., Gallagher and Heppell 2010), but the addition of hard structures in a waterbody may also improve the habitat for invasive species (Vaselli et al. 2008; Carman et al. 2019). In New England, wooden piles provide habitat and refuge for a variety of small fish species, primarily cunner (*Tautogolabrus adspersus*), rock gunnel (*Pholis gunnellus*), and grubby (*Myoxocephalus aenaeus*) (Brandl et al. 2017). Fish abundance was higher in pile-fields and wrecks than open-water habitat in the Hudson River estuary due to the increased prevalence of structure-seeking species like the mummichog (*Fundulus heteroclitus*), bay anchovy (*Anchoa mitchilli*), naked goby (*Gobiosoma bosc*), American eel (*Anguilla rostrata*), and northern pipefish (*Syngnathus fuscus*) (Duffy-Anderson et al. 2003). In Australia, large mobile species were found to move between structures (e.g., marina piles) and open water while smaller species were only found in close proximity to these structures (Clynick et al. 2007). Piles can also disrupt school formation, and the added structure may facilitate ambush predation relative to open water environments (Grothues et al. 2016). Juvenile winter flounder (*Pseudopleuronectes americanus*) and tautog (*Tautoga onitis*) held in cages in open pile fields had similar growth rates to individuals held in open water (Able et al. 1999), but a sonar survey in the Hudson River found fewer small fishes in open pile habitats relative to open water during daytime (Grothues et al. 2016).

Table.1 Summary of best management practice guidelines and general permit conditions for dock design in coastal states in the continental U.S

	Maximum dock width	Minimum dock height	Decking	Minimum float depth	Materials	Orientation
Maine (Swan and Sowles 2008)	NA	1:1 H:W[a,b]	0.75 in (1.9 cm) spacing[a,b]	8-12 in (~20-30 cm) float stops[c]	No creosote	NA
New Hampshire (NHDES 2020)	6 ft (1.8 m)	1:1 H:W	0.75 in (1.9 cm) spacing	2 ft (0.6 m)	Non-toxic	N-S
Massachusetts (Bliven and Pearlman 2003)	NA	1:1 H:W[k] 4 ft (1.2 m)[b]	0.75 in (1.9 cm) spacing or alternative decking[b,k]	1.5 ft (0.5 m)[x] 2.5 ft (0.8 m)[x,y] 4 ft (1.2 m)[b]	No creosote	N-S[b,k]
Rhode Island (RICRMP 2007)	4 ft (1.2 m)	4 ft (1.2 m) or 1:1 H:W[a]	NA	1.5 ft (0.5 m)[x] No float[b]	No creosote	NA
Connecticut (CTDEEP 2015)	4 ft (1.2 m)[a,b]	5 ft (1.5 m)[n] > 1:1 H:W[a]	NA	No float[b]	NA	NA
New York (NYDEC 2013)	4 ft (1.2 m)	4 ft (1.2 m)[a] 5 ft (1.5 m)[b,o]	open grated[a]	2.5 ft (0.8 m)[z]	No creosote	N-S
New Jersey (NJDEP 2020)	4 ft (1.2 m)[b] 6 ft (1.8 m)[a,c] 8 ft (2.4 m)[d]	4 ft (1.2 m)	0.38 in (1.0 cm), 0.5 in (1.3 cm), 0.75 in (1.9 cm), 1 in (2.5 cm) or alternative decking	4 ft (1.2 m)	Non-polluting materials[y]	NA
Delaware (ACOE 2020)	3 ft (0.9 m)[a,b,c] 4 ft (1.2 m)[e,f] 5 ft (1.5 m)[g] 6 ft (1.8 m)[h,i] 8 ft (2.4 m)[j]	3 ft (0.9 m)[a,b,c] 4 ft (1.2 m)[e]	0.38 in (1.0 cm), 0.5 in (1.3 cm), 0.75 in (1.9 cm), 1 in (2.5 cm) or alternative decking	Avoid grounding	NA	NA
Maryland (ACOE 2011a)	3 ft (0.9 m)[a] 6 ft (1.8 m)[d]	3 ft (0.9 m)[a,q] 4 ft (1.2 m)[r]	NA	NA	NA	NA
VMRC (1999, 2005)	5 ft (1.5 m)[a]	3 ft (0.9 m) or width + 1 ft (0.3 m)[a]	NA	NA	NA	NA
North Carolina (NCDEQ 2021)	6 ft (1.8 m)	3 ft (0.9 m)[a]	NA	NA	NA	NA
South Carolina (SCDHEC 2008)	4 ft (1.2 m)	3 ft (0.9 m)[o]	NA	NA	NA	NA
Georgia (ACOE 2017)	6 ft (1.8 m)	6 ft (1.8 m)[s]	NA	2 ft (0.6 m)	NA	NA
Florida (Florida DEP 2020, ACOE and NMFS 2001)	4 ft (1.2 m)[b,k,l]	4 ft (1.2 m)[k] 5 ft (1.5 m)[b,o]	0.5 in (1.3 cm) spacing[b]	1 ft (0.3 m)[x]	NA	N-S[b]
Alabama (ACOE 2011b)	5 ft (1.5 m)[b,m]	1:1 H:W[b,m,o]	0.75 in (1.9 cm) spacing	NA	NA	NA
Mississippi (ACOE 2018)	6 ft (1.8 m)[b,m]	1:1 H:W[b,m,o]	NA	NA	NA	NA
Louisiana (K. Morgan, LA CZM Pers. Comm.)	NA	NA	1–2 in (2.5–5.0 cm) spacing[k]	NA	NA	NA
Texas (ACOE 2021)	4 ft (1.2 m)	5 ft (1.5 m)[t]	1 in (2.5 cm)[u]	NA	NA	NA
Washington (WADOE 2011)	NA	NA	Grating over at least 50% of surface area[v]	4–5 ft (1.2–1.5 m)[z]	Avoid wood treated with toxic compounds[A]	N-S
Oregon (ODFW 2016)	6 ft (1.8 m)	NA	NA	NA	Avoid pressure treated (PT) wood	NA

Table.1 (continued)

	Maximum dock width	Minimum dock height	Decking	Minimum float depth	Materials	Orientation
California (Metz 2019)	NA	NA	NA	NA	No creosote[B]; Avoid PT or use wrap (piles); if PT used, ACZA or ACQ recommended[C]	NA

[a]If over wetlands

[b]If over submerged aquatic vegetation (SAV)

[c]If over intertidal flats

[d]If over water

[e]If municipal or residential community structure over wetlands, SAV, or intertidal flats

[f]General requirement for piers

[g]General requirement for docks

[h]General requirement for docks with boatlifts

[i]General requirement for community, municipal, or commercial pier structure

[j]General requirement for community, municipal, or commercial dock structure

[k]If over salt marsh

[l]If over mangroves

[m]If over non-forested wetlands

[n]Height above substrate at mean high water (MHW) line for public access if stairs or other public access not provided

[o]Height above MHW

[p]If walkway over scrub-shrub wetlands

[q]Height above mean low water (MLW) if decking < 5 ft (1.5 m) width

[r]Height above MLW if 6 ft (1.8 m) width over open water to protect SAV

[s]Height above MHW if constructed over a tributary that can be bridged

[t]Height above MHW or OHW (ordinary high water) measured from top surface of decking if over special aquatic sites

[u]If over special aquatic sites

[v]If over nearshore or littoral area

[w]Measured at low tide in coastal wetland resource areas and ordinary high water or annual high water for inland wetland resource areas

[x]Depth at MLW

[y]If over shellfish habitat

[z]Depth at mean lower low water (MLLW)

[A]Applies to structures on or over state-owned aquatic lands

[B]Creosote not recommended in new structures but may be acceptable if adding small number of piles to existing structure

[C]If PT wood used, ammoniacal copper zinc arsenate (ACZA) recommended if no human or marine mammal contact anticipated, alkaline copper quaternary (ACQ) if such contact anticipated

The former study suggests that piles can provide suitable habitat and food source for many fish species, but the latter study suggests piles may not benefit overall fish production.

Piles displace existing habitat for infaunal invertebrates (e.g., clams, polychaete worms) but can increase habitat for a variety of epifaunal invertebrates (e.g., mussels, oysters, barnacles) by providing surface area for settlement and attachment. Piles in a Florida lagoon supported a variety of filter-feeding invertebrates that collectively provided approximately 30% of the filtration capacity of oyster reefs in the system (Layman et al. 2014). Fish species diversity and abundance around piles in marinas in Australia were correlated with the percent cover of pile epibiota in experimental manipulations (Clynick et al. 2007), and piles that promoted colonization by mussels, foliose algae, and other epibiota enhanced fish habitat in some cases. Red king crabs (*Paralithodes camtschaticus*) were more abundant on experimental pile structures in Alaska than adjacent seafloor, possibly due to habitat provided by colonizing hydroids (Stevens et al. 2004). Similarly, docks in the southeast U.S. provided forage from attached fouling organisms growing on piles for mangrove tree crabs (*Aratus pisonii*) (Cannizzo et al. 2018). In Australia, subtidal epibiota diversity and abundance increased on piles relative to open water, and unshaded piles had similar communities as natural rocky habitat (Connell and Glasby 1999).

Pile Design and Materials

Incorporation of creative designs and materials in pile installations can enhance habitat value (Dyson and Yocom 2015; O'Shaughnessy et al. 2020), but may also have unintended effects (e.g., spread of invasive species, trophic alterations) (Dafforn 2017; Malerba et al. 2019). For example, installation of more textured piles could promote colonization of epibiota resulting in increased water filtration (Layman et al. 2014) as well as invertebrate and fish diversity and abundance (Clynick et al. 2007; Perkol-Finkel and Sella 2015) depending on the colonizing species. Given documented use of piles and other hard structures by invasive species (e.g., Cordell et al. 2013), monitoring of invasive species colonization rates and longevity should accompany installation of these creative designs to better understand the biological responses to designs intended to promote colonization (Dyson and Yocom 2015). Some eco-engineering designs have demonstrated reductions or prevention of biofouling and invasive colonization (Paalvast et al. 2012; Perkol-Finkel and Sella 2015). Certain installation practices (e.g., controlling timing of installation, pre-seeding piles with appropriate native filter-feeders, grazers or predators) show promise in better controlling the communities that ultimately colonize piles (Dafforn 2017). Pile material affected habitat value for epibiota in studies in the U.S. and Europe (Layman et al. 2014; Paalvast et al. 2012; Perkol-Finkel and Sella 2015). In a Florida lagoon, filter-feeding communities varied greatly across pile types. Communities on concrete piles contributed 68% of total filtration capacity even though concrete piles were found on only 7% of sampled docks. Communities on wooden piles with pile wrap supported 10% of filtration even though wooden piles were most common, associated with 69% of sampled docks (Layman et al. 2014). Piles in New York outfitted with rough textured concrete designed to promote invertebrate colonization had greater species diversity than fiberglass piles (Perkol-Finkel and Sella 2015). In both studies, filter feeders settled more on the rougher texture of the concrete piles. In the Netherlands, piles outfitted with synthetic, free-hanging ropes increased colonization by native species (blue mussels (*Mytilus edulis*) below mean low water (MLW), seaweed above MLW line) but not invasives (Pacific oyster (*Crassostrea gigas*)) relative to standard piles (Paalvast et al. 2012). In Florida, dock piles were wrapped with oyster mats and had oyster bags suspended adjacent to piles in a pilot study examining potential means of creating a "living dock" (Weaver et al. 2018). Both dock augmentations allowed successful colonization of oysters and a variety of invertebrates (e.g., barnacles, sponges, mussels).

For fishes, it is unclear if pile material influences community composition. Pile material (concrete, wood, or polyvinyl chloride (PVC)) did not affect fish community composition around docks in Belize and Panama. However, sample sizes for each material were low ($n \leq 6$ for each type) and may have been inadequate to detect differences (Brandl et al. 2017). Fish communities should be affected indirectly since pile material affects epibionts (Layman et al. 2014; Perkol-Finkel and Sella 2015), and the fish community will likely respond to changes in the epibiota.

Invasive Species

Piles can also serve as habitat for invasive invertebrate species (Ruiz et al. 2009). For example, new piles represent unoccupied hard substrate, and such open spaces promote the colonization of invasive marine sessile invertebrates (Stachowicz et al. 1999). When piles are installed in areas with little or no natural hard substrate, invasive species associated with hard substrate may be provided with previously unavailable habitat. In North America, the greatest numbers of invasive species in coastal waters were observed in docks and marinas among all surveyed habitat types followed by rocky reefs (Ruiz et al. 2009). In Australia, piles supported approximately twice as many invasive species as rocky reefs (Glasby et al. 2007) and 10–80% greater percent cover (Dafforn et al. 2012). While colonization of piles and floats by invasive species is well documented globally (Lambert and Lambert 1998, 2003; Lambert 2002; Ruiz et al. 2009; Simkanin et al. 2012), conditions that promote or limit the spread of invasives to natural habitats are not well understood (Simkanin et al. 2012). If natural hard bottom habitat is proximal to piles, these structures may serve as a gateway for the spread of invasives to neighboring habitats (Simkanin et al. 2012). However, an experimental predator manipulation study found that the high abundance of predators in natural hard bottom habitat can inhibit colonization of those natural habitats and piles by invasive invertebrates (Dumont et al. 2011; Kimbro et al. 2013). Given that invasive colonization of piles and other artificial hard substrates is well documented, limiting the proliferation of piles reduces the potential for the spread of invasive species (Wasson et al. 2005).

Shading and Circulation

Piles can alter the surrounding environment through shading and alteration of water circulation. Piles cast shade and reduce the availability of light to the water column and benthic habitat. Piles can trap floating debris, which further affects light and habitat (Nightengale and Simenstad 2001). Structures placed in moving water can alter circulation (Ramos et al. 2016), which, depending on the conditions, can either cause scour and erosion or lead to increased deposition of sediments (Pentilla and Doty 1990; Kelty and Bliven 2003). Salt marsh vegetation biomass decreases under dock decking as pile spacing decreases (Logan et al. 2018a), possibly due to increased shading from piles and associated support structures (e.g., cross bracing, horizontal stringers) and/or shading by wrack trapped by piles. Salt marsh losses under a dock system that used a powered cart atop thin rails rather than traditional decking may have also been due to indirect pile impacts as this system required closer pile spacing (10 ft; ~ 3 m) relative to other construction methods with greater spacing of 12–20 ft (~ 3.7–6.1 m) (Alexander 2012).

Piles can trap materials that smother underlying salt marsh or submerged aquatic vegetation (SAV). While wrack naturally accumulates on marsh surfaces, piles magnify burial effects on marsh vegetation by preventing tidal action from moving such material and instead causing it to remain in place for prolonged periods. Pilings alter marsh wrack distribution in marshes, trapping wrack and keeping it in place for longer periods of time (Alexander 2008). Wrack burial impacts are greatest for large mats that remain in place for longer time intervals and at higher marsh elevations (Valiela and Rietsma 1995). *Spartina patens*, the dominant high marsh vegetation in U.S. east coast salt marshes, is more susceptible to mortality from burial (due to less frequent inundation and flushing) than the low marsh dominant, *S. alterniflora*, but both species die off when subjected to wrack burial for greater than two months (Bertness and Ellison 1987). In the absence of vegetation, these wrack-covered areas can erode and cease to be suitable marsh habitat due to lowered elevation and increased tidal inundation. Similarly, if scouring occurs in SAV the vegetation community can be eliminated or altered (Beal and Schmit 2000). Like areas of marsh loss, denuded subtidal areas around piles can fill in with shell debris and detritus, making revegetation unlikely due to habitat alteration and smothering (Beal and Schmit 2000).

Toxic Leachates

Wood piles are commonly treated with anti-microbial chemicals to lessen decay from insects and rotting and consequently increase pile lifespan (Kelty and Bliven 2003; Gorenier and Lebow 2006). Anti-microbial treatments include metal additives absorbed into the wood under pressure (pressure-treated wood) and creosote coatings. While rarely used in contemporary dock construction, creosote coatings are still used in commercial applications and a legacy pollution effect remains (Younie 2015). Both treatment types are known to leach contaminants into the water and surrounding sediments (Malins et al. 1985; Weis and Weis 1996). These effects are described in turn.

Pressure-Treated Wood

The main chemical additives in pressure-treated wood are chromated copper arsenate (CCA) and various copper-based alternatives that do not contain arsenic but have higher concentrations of copper than CCA (e.g., alkaline copper quaternary (ACQ) and copper azole (CA)) (Lebow et al. 2019). CCA is no longer produced for use in residential settings (i.e., inside a house) due to the toxicity of arsenic to humans (Gerstein and Zaccaria 2004), but it can be used for dock construction. Copper is toxic to marine organisms at very low concentrations (Ansari et al. 2004), so it is effective in anti-fouling compounds but has the potential to cause adverse environmental impacts (Weis and Weis 1992a, 1996). Pressure-treated wood is known to leach the metal additives over time in proportion to the original concentrations of metals (Hingston et al. 2001); therefore, the copper-based alternatives to CCA-treated wood leach higher concentrations of copper (Stook et al. 2005; Temiz et al. 2006; Dubey et al. 2007) than CCA. Metal leachate concentrations are also influenced by environmental conditions, and generally increase with lower pH, higher salinities, and higher temperatures (Hingston et al. 2001; Moghaddam and Mulligan 2008). Metal concentrations are higher in recently (≤ 3 years) constructed structures (Weis and Weis 2002), and leaching is most rapid in the first month following pile installation due to an initial release of a pulse of unfixed or poorly fixed preservatives followed by an exponential decline towards a lower leaching rate (Breslin and Adler-Ivanbrook 1998; Lebow et al. 2004). Leached metals are transferred to sediment, organisms, and waters surrounding the wood piles. Metal concentrations are higher in fine-grained sediments (i.e., silts and clays) and poorly flushed systems with reduced tidal exchange (Weis and Weis 1992a; Weis et al. 1993a, 1993b, 1998). For example, copper concentrations of oysters (*Crassostrea virginica*) growing on pressure-treated bulkheads in a canal with minimal flushing were twelve times higher than background levels while conspecifics on similar structures in open water were twice background levels (Weis and Weis 1992b). Typically, transfer of metals to surrounding sediments and sessile organisms is spatially constrained to areas within approximately 33 ft (10 m) of structures containing pressure-treated wood (Wendt et al. 1996; Weis et al. 1998; Weis and Weis 2002; Vasilas et al. 2011a), but the spatial range of impact can be wider under older structures (Weis and Weis 2002; Vasilas et al. 2011a), possibly due to metal transport after release through mineralization (Vasilas et al. 2011a).

Negative effects on aquatic biota subjected to metal leachates from pressure-treated wood in laboratory settings range from reduced fitness to mortality (Weis and Weis 2004). CCA-treated wood leachates slowed limb regeneration in fiddler crabs (*Uca pugilator*) (Weis et al. 1991,

1992) and induced mortality at higher concentrations in fiddler crabs and mummichog embryos (Weis et al. 1991). CCA leachates also indirectly induced mortality in mud snails (*Ilyanassa obsoleta*) consuming green algae exposed to leachates (Weis et al. 1991; Weis and Weis 1992b) and reduced sea urchin (*Arbacia punctulata*) fertilization (Weis et al. 1992b). ACQ-treated wood was more toxic than CCA as well as all other tested materials (e.g., untreated wood, steel, concrete) for zooplankton (*Daphnia magna*) and fishes (rainbow trout (*Oncorhynchus mykiss*) and threespine stickleback (*Gasterosteus aculeatus*)). Untreated wood was also toxic to brine shrimp (*Artemia franciscana*) and oyster (*C. gigas*) embryos in laboratory exposure (Libralato et al. 2007). Amphipods (*Ampelisca abdita*) exposed to wood pile leachates had reduced survival with untreated but not CCA-treated wood, possibly due to toxic effects of naturally leaching compounds (e.g., phenols) that might be extracted or chemically altered during the preservation process (Baldwin et al. 1996). As noted by Weis et al. (1991), these laboratory studies do not mimic the full range of ambient conditions (e.g., dilution from tidal flushing, avoidance for mobile fauna) and do not necessarily represent impacts to fauna in field conditions. For example, the proximity of CCA-treated docks in estuaries with moderate (~ 5–6.5 ft; 1.5–2.0 m) tidal ranges did not affect oyster growth, survival rates, and metal concentrations over a 6-week exposure period nor the short-term (< 1 week) survival rates of mud snails, mummichogs, juvenile red drum (*Sciaenops ocellatus*), or juvenile white shrimp (*Penaeus setiferus*) (Wendt et al. 1996).

The broader ecological effects of pressure-treated wood structures are less clear. Epibiota colonizing CCA-treated wood in New York estuaries had lower species diversity, biomass, and abundance relative to communities on untreated piles (Weis and Weis 1992a), and epibiota diversity decreased with increasing CCA concentrations in the Mediterranean Sea (Karayanni et al. 2010). For experimental wood panels submerged at different European coastal sites, epibiota diversity (Brown et al. 2003) and abundance (Brown and Eaton 2001) were greater on CCA-treated wood compared to untreated wood, possibly due to modifications to the wood surface and positive responses of dominant taxa, respectively. Metals could be transported greater distances away from docks via trophic transfer as organisms acquire elevated levels of metals indirectly through the consumption of contaminated prey (Weis and Weis 2004). However, this uptake by local producers and impacts on associated food webs are not well understood (Sanders 2008).

Creosote-Treated Wood

Creosote is another chemical that is added to wood under pressure to lessen decay from insects and rotting (Brooks 1995). Creosote-treated piles release polycyclic aromatic hydrocarbons (PAHs) into the surrounding environment resulting in negative impacts to nearby aquatic organisms (Malins et al. 1985; Duncan et al. 2017; West et al. 2019). Though creosote-treated piles have been banned in many regions, their use still persists in some locations (e.g., Metro Vancouver) primarily due to their cost-effectiveness and longevity (Younie 2015). Therefore, BMP guidelines put forth by U.S. state agencies frequently call for avoidance of creosote-treated piles or note the unlawfulness of their use (e.g., Swan and Sowles 2008; Table 1). Creosote-treated piles installed prior to being banned remain in place in some harbors. For example, 30,000 abandoned creosote-treated piles were recently mapped across San Francisco Bay (Werme et al. 2010). In Washington state, 16,000 creosote-treated piles remained after the removal of more than 25,000 (Robertson 2018).

The release of PAHs from creosote-treated piles varies with environmental conditions (Brooks 1995; Stratus Consulting Inc. 2006; Perkins 2009). Elevated concentrations of PAHs are mostly localized in close proximity to the piles (Duncan et al. 2017) and are highest in environments with poor circulation and low sediment oxygen concentrations due to limited dilution and reduced rates of microbial decomposition, respectively (Brooks 1995; Stratus Consulting Inc. 2006). The release of PAHs from creosote-treated piles can continue long after installation. For example, herring (*Clupea pallasii*) larvae collected near creosote-treated piles in place for > 100 years had elevated levels of PAHs (West et al. 2019). Thus, similar environmental conditions (e.g., low flow, fine sediments) that elevate heavy metal contamination from pressure-treated wood piles also increase PAH contamination from creosote-treated piles, but the release of leachates of creosote-treated piles continues over much longer time spans (decades to centuries).

Impacts of creosote on marine organisms have been documented in both laboratory (e.g., Vines et al. 2000) and field (e.g., West et al. 2019) settings (Cherr et al. 2017). More vulnerable taxa include benthic fishes and invertebrates, and egg and larval stages of fishes due to heightened exposure to sediment-bound PAHs and greater sensitivity to low PAH concentrations, respectively (Malins et al. 1985; Vines et al. 2000; Duncan et al. 2017). Creosote exposure can result in both diminished health in adults as well as mortality in larvae. For example, English sole (*Parophrys vetulus*), a benthic flatfish, had a high prevalence of hepatic lesions in habitat with creosote-contaminated sediment (Malins et al. 1985). In laboratory exposure, Pacific herring embryos that attached directly to creosote-treated wood failed to further develop a few days post-incubation (Vines et al. 2000).

Alternatives to Pressure- and Creosote-Treated Wood: Challenges and Trade-offs

Many U.S. states recommend avoiding creosote and pressure-treated wood (e.g., WADOE 2011, ODFW 2016; Table 1), but ultimately all pile materials have trade-offs in terms of cost, efficacy, availability, and environmental impacts that will need to be weighed for individual projects (Table 2). These trade-offs among materials often make identifying the least environmentally damaging construction materials complex (May et al. 2017). For example, external wraps that reduce chemical leaching can also reduce pile biological function by limiting colonization by filter feeders (Layman et al. 2014), and the risk of greater leachate release from pressure-treated piles without wrap may be outweighed by the reward of ecosystem services in an embayment with high flushing (i.e., lower sensitivity to leaching impacts) and populations of native filter-feeders (e.g., oysters). Using untreated local hardwoods from the U.S. (e.g., black locust (*Robinia pseudoacacia*), cedar (*Thuja* spp.), white oak (*Quercus alba*)) would alleviate metal leaching effects but such materials may be less readily available and have a shorter lifespan than pressure treated wood materials (Gorenier and Lebow 2006; Lebow et al. 2019). Even pressure-treated wood will ultimately still be susceptible to degradation by fungi and marine borers (Lopez-Anido et al. 2004). Amazonian hardwoods (e.g., Greenheart (*Chlorocardium rodiei*)) have high durability (Crossman and Simm 2004) and will also alleviate metal leaching effects but may compound broader environmental effects (e.g., deforestation and unsustainable harvest practices) (Treu et al. 2019).

Life cycle assessments (LCAs) offer a broader, more holistic evaluation in material selection. An LCA estimated that wood-plastic composite decking resulted in fourteen times greater greenhouse gas (GHG) emissions relative to ACQ-treated lumber (Bolin and Smith 2011) while plastic pile GHG emissions were about 2.5 times greater than CCA-treated piles (Bolin and Smith 2012) (Table 2; May et al. 2017). However, these life cycle comparisons were sensitive to assumptions of relative virgin vs. recycled plastic content and not necessarily representative of all available composite products (Platt et al. 2005; Bolin and Smith 2011). Estimates of GHG emissions from wood-plastic composites decreased by 28% when virgin plastic was substituted with 100% recycled plastic (Fuchigami et al. 2020). Steel and concrete GHG emissions also exceeded those for the life cycle of CCA-treated piles (Bolin and Smith 2012). The production of Portland cement, an essential component of concrete, is highly energy intensive and consequently has a large carbon footprint (Meyer 2009; Cooke et al. 2020). Therefore, each project should consider the surrounding environment and determine an appropriate approach on a case-by-case basis within a system level analysis. An online model developed by Oregon State University and the Western Wood Preservers Institute (WWPI) can be used as a tool to estimate the potential impact of different construction designs on local water column and sediment contaminant concentrations (Western Wood Preservers Institute 2018a). When pressure-treated wood is used, careful selection of source material that has been produced using BMPs to reduce leaching will aid in minimizing environmental impacts (Lebow et al. 2019).

Table.2 Summary of benefits and negative aspects of different potential construction materials for dock decking and pile supports

Material	Pros	Cons
CCA	•Commonly used •Can use local wood supply •Long lifespan •Low lifecycle greenhouse gas (GHG) emissions	•Release of arsenic •Release of chromium •Release of copper
ACQ	•Commonly used •Can use local wood supply •Long lifespan •Low lifecycle GHG emissions	•Release of higher copper concentrations than CCA
Creosote	•Long lifespan •Can use local wood supply	•Release of PAHs
Local (U.S.) hardwood (e.g., black locust, cedar, white oak)	•No leaching of metals or PAHs	•Less commonly available •Shorter lived than pressure-treated wood
Amazonian hardwood	•No leaching of metals or PAHs	•GHG emissions •Sustainability concerns for many sources
Chemically-modified wood	•No leaching of metals or PAHs •High durability	•Less commonly available
Concrete	•No leaching of metals or PAHs •Can promote colonization by filter-feeders	•May promote spread of invasive species •High lifecycle GHG emissions
Composite	•No leaching of metals or PAHs •Widely available as decking material	•High lifecycle GHG emissions

Several piling designs and materials have future potential as replacements of conventional pressure-treated wood, but currently require further development. Chemical modification of wood involves covalently bonding a chemical group to a wood component, resulting in a stable bond between the reagent and wood cell wall components (Mantanis 2017). Chemically modified wood, which has been shown to have greater durability than CCA-treated wood in marine trials (Westin et al. 2016), is a potential future alternative that avoids the release of toxic leachates (Mantanis 2017). Fiber reinforced composites (FRPs), which often use high percentages of recycled plastic, are also highly durable and corrosion resistant, but further assessment of long-term resilience is needed (Guades et al. 2010; Zyka and Mohajerani 2016) and the potential for plastic contamination needs to be studied. Finally, eco-engineering approaches show promise in enhancing the ecological function of support piles (e.g., Paalvast et al. 2012; Perkol-Finkel and Sella 2015; O'Shaughnessy et al. 2020), but still need to be further evaluated in terms of the potential spread of invasive species (Vaselli et al. 2008; Carman et al. 2019).

Construction Impacts

Pile installation via pile driving, jetting, or blasting can temporarily impact fishes and invertebrates through direct mortality as well as the introduction of turbidity and noise (Mulvihill et al. 1980; Iafrate et al. 2016). These latter impacts may adversely affect fish and shellfish eggs, larvae, and adults. Eggs and larvae of estuarine fishes exhibit some of the most sensitive responses to elevated turbidity. Impacts from suspended sediments include egg burial, hatching delay, reduced feeding success, behavioral changes, and mortality (Wilber and Clarke 2001; Berry et al. 2011). For shellfish, pumping rates and growth can be impacted (Loosanoff 1962; Wilber and Clarke 2001). In New England, winter flounder spawn in coastal embayments during winter months (Pereira et al. 1999), and their demersal eggs are also vulnerable to physical damage from pile installation and associated turbidity (Berry et al. 2011). Installation during the spawning season could negatively impact species like alewife (*Alosa pseudoharengus*) that spawn along pond shorelines (as reviewed in Mather et al. 2012) due to direct impacts to spawning substrates. Pile-driving in areas of contaminated sediment may negatively impact nearby fishes that are attracted to the increased suspension of benthic organisms as the water column and substrate surfaces are disturbed (Nightengale and Simenstad 2001). Noise from pile driving can trigger avoidance behavior in some fish species (Hawkins et al. 2014; Iafrate et al. 2016) and limit their use of habitat near the dock site during construction, potentially interfering with migratory pathways.

Noise and turbidity impacts can be reduced with the use of modified timing and installation procedures. Construction-related impacts can be minimized through time-of-year restrictions that avoid sensitive life history stages of species present at a dock installation site that are more vulnerable to turbidity or noise-related impacts of pile installation (Evans et al. 2015). Noise levels vary among piling installation methods with vibratory hammers producing lower noise vibrations than impact hammers (Denes et al. 2016). For any pile driving activity, potential impacts to mobile fauna can be reduced with a "soft-start" approach, which initiates the construction process with lower noise level activity (Robinson et al. 2007). This approach can potentially allow mobile organisms to leave the area before greater noise production occurs. Turbidity associated with pile-driving can be reduced by avoiding jetting (Gabr et al. 2004) and contained through the use of silt curtains.

In addition to direct mortality, turbidity, and noise, jetting or blasting methods for pile installation can cause scouring in the benthic habitat surrounding installed piles that can persist for years post-construction (Shafer and Robinson 2001) and can impact SAV (Beal and Schmit 2000). In Florida, pile installation using jetting produced a halo with a diameter up to 1.6 ft (0.5 m) around the piles, and the resulting bare area was found to deepen over time due to scouring (Beal and Schmit 2000). Bare areas around piles ranged from approximately 3.0–6.5 ft (0.9–2.0 m) in diameter in St. Andrew Bay, Florida, even though the age of the piles varied widely and thus some areas should have presumably revegetated. In many cases, bare areas from adjacent piles were observed to overlap and coalesce into continuous expanses of bare sediments causing fragmentation of the SAV meadow (Shafer and Lundin 1999; Shafer and Robinson 2001).

Installation of new as well as removal of existing piles can also have environmental impacts depending on the pile material. Complete removal of piles is generally preferable in restoring aquatic habitat relative to pile cutting. For creosote piles, cutting can release additional PAHs into the environment possibly by exposing previously unweathered wood material (West et al. 2019). To minimize additional PAH release, the Washington Department of Natural Resources (WADNR) recommends full removal as the preferred option and collection of any splintered wood, debris, and fragments (WADNR 2017). Similar debris collection and proper disposal practices are recommended in the removal of pressure-treated wood piles as well as during installation if any such debris is generated (Western Wood Preservers Institute 2018b). When cutting is the only option during pile removal, the WADNR recommends cutting occur at least 2 ft (0.6 m) below the mudline in intertidal and shallow (< -10 ft (3 m) mean lower low water (MLLW) subtidal waters and at least

1 ft (0.3 m) below the mudline in deeper water environments for creosote piles (WADNR 2017; Abercrombie 2018). The Oregon State Marine Board (OSMB) recommends a minimum 1 ft (0.3 m) below mudline cutoff depth for all pile materials and marine habitats. For creosote piles, the OSMB recommends the addition of clean fill to cover the remaining stump (OSMB 2012).

Regardless of timing, all pile construction activities can impact habitat beyond the footprint of the structures. Operation of the work vessel in shallow water can result in grounding or propeller scour, which could damage benthic resources including shellfish and SAV, and also cause turbidity (Sagerman et al. 2020). Anchored barges also limit light availability. These vessel impacts can occur in areas adjacent to the dock footprint as well as nearby areas encountered in transit to and from the work site. This is a particular concern for SAV and other shallow water resources that could be present in the general project vicinity.

Recommendations

- Avoid sensitive habitat when siting piles: When feasible, pile locations should be adjusted to avoid or minimize direct impacts to the most sensitive and ecologically valuable areas.
- Minimize area of direct impact: The total pile footprint should be reduced to the maximum extent practicable within the constraints of the structure's engineering requirements through the use of monopiles or by maximizing paired pile spacing and/or reducing pile diameter.
- Choose appropriate pile materials: Most appropriate pile materials will vary as a function of project site habitat characteristics and biological communities. Among pressure-treated wood alternatives, materials with lower concentrations of copper, such as CCA, are recommended over those with higher concentrations of copper, such as ACQ. Creosote treatment of untreated wood is not recommended under any conditions.
- When using pressure-treated piles, the amount of leaching can be reduced with project-specific BMPs relating to both product selection and installation including purchasing BMP-certified wood, minimizing on-site cutting or including proper containment, and adding external pile wrap.
- Guidance on when to use pressure-treated wood alternatives: Traditional, pressure-treated wood appears to provide a relatively low impact use for applications in locations with greater flushing and coarser, more sandy sediment, but consider wraps or non-leaching materials in poorly flushed systems.
- Consider cumulative effects: Highly developed embayments with low flushing represent areas of particular concern as they pose the potential to compound localized impacts.
- Adhere to time-of-year restrictions in the project sequencing: Construction-related impacts, such as turbidity or noise, can be minimized through careful coordination of timing to avoid sensitive life history stages of species present at a dock installation site.
- Use construction methods that minimize habitat disturbance: Vibratory hammer use is recommended with a slow-start approach for pile driving while jetting should be avoided. Silt curtains or other containment methods should be used in fine-grained sediment to contain turbidity.
- Construction vessel BMPs: For docks installed in shallow water environments, construction activities should be staged to only occur near high tide to reduce the risk of vessel grounding or propeller scouring.

Decking

The decking for a dock is elevated with pile supports and so does not cause any direct impacts. However, decking can cause a variety of indirect impacts to marine resources through leaching (Weis and Weis 1996) and shading (Burdick and Short 1999; Able and Duffy-Anderson 2005; Logan et al. 2018a). In particular, salt marsh and seagrasses and the variety of associated ecosystem services provided by these habitats (Barbier et al. 2011) can be negatively impacted by shading from dock decking (Burdick and Short 1999; Logan et al. 2018a, b).

Leaching

While some guidelines only recommend avoidance of pressure-treated wood for in-water structures (e.g., WADOE 2011), decking composed of pressure-treated wood also results in the leaching of heavy metals into the aquatic environment through weathering and rainfall events (Khan et al. 2006; Shibata et al. 2007; Lebow 2014). In general, leaching and rainfall rates are inversely related since water has less contact time with wood during heavier rain events (Lebow 2014). Leaching rates from decks are lower than from piles since piles have routine exposure to water (Lebow et al. 2004). Like pile impacts, leaching impacts from decking are higher in the first year after construction, in poorly flushed environments, and for animals and habitats in close proximity to the pressure-treated wood structures (Weis and Weis 1996, 2002, 2004).

Shading

The primary indirect impact imparted by decking is shading. Reduced light levels under decking adversely affect productivity of salt marsh and seagrasses and can alter fish behavior (Burdick and Short 1999; Munsch et al. 2017; Logan et al. 2018a, b). Decking that shades salt marsh vegetation can reduce underlying production by decreasing stem density and biomass (Kearney et al. 1983; Sanger et al. 2004; Alexander and Robinson 2006; Vasilas et al. 2011b; Logan et al. 2018a, b). Relative impacts vary among decking designs (Kearney et al. 1983; Colligan and Collins 1995; Sanger and Holland 2002; Alexander and Robinson 2004; Sanger et al. 2004; Vasilas et al. 2011b; Alexander 2012; Logan et al. 2018a, b). *Spartina* stem density under docks along the U.S. east coast was reduced by approximately 30 to 70% (Alexander and Robinson 2004, 2006; Sanger et al. 2004; Alexander 2012; Logan et al. 2018a) relative to unshaded habitat. Biomass effects were more variable across dock studies and ranged from a 63% reduction to a 23% increase, with the latter observed increase occurring for tall-form *S. alterniflora* (Alexander and Robinson 2006; Alexander 2012; Logan et al. 2018a).

Light penetration increased with decking height in field studies conducted in both the northeast and southeast U.S. (Alexander 2012; Logan et al. 2018a, b). Salt marsh vegetation under northeast U.S. docks also responded positively to increased decking height with clear increases in aboveground production as decking height increased towards 4 ft (1.2 m) and higher (Kearney et al. 1983; Logan et al. 2018a, b). Docks positioned closer to the ground frequently have large bare patches along the underlying marsh substrate due to heightened shading (Fig. 2). Decking set > 5 ft (1.5 m) above the marsh platform had the least impact on aboveground production in Massachusetts estuaries (Logan et al. 2018a, b). Current U.S. state guidelines include minimum dock height recommendations ranging from 3 ft (0.9 m) (NCDEQ 2021) to 6 ft (1.8 m) (ACOE 2017; Table 1). Several U.S. state BMP guidelines recommend a minimum dock H:W ratio of 1:1 (e.g., NHDES 2020; Table 1), but a controlled study found that docks with the 1:1 design had only approximately 50% of the *Spartina* biomass found in unshaded controls (Logan et al. 2018b). A 1.5:1 design (i.e., 6 ft (1.8 m) above the marsh platform for a 4 ft (1.2 m) wide dock) had significantly greater marsh production relative to 1:1 docks (Logan et al. 2018b).

Decking width and orientation effects on underlying salt marsh production varied across different regions of the U.S. east coast. Width was the most important predictor of relative marsh production under mid-Atlantic U.S. docks but was not a significant predictor for New England docks (Kearney et al. 1983; Colligan and Collins 1995; Logan et al. 2018a). Most U.S. state guidelines recommend a four ft (1.2 m) maximum dock width (Table 1), and most docks in Massachusetts constructed over salt marsh have a four ft (1.2 m) width (Logan et al. 2018a), so uniformity in dock width may have limited detection of width effects on marsh production in New England studies. Many U.S. state guidelines recommend a north–south orientation for docks constructed over vegetation (Table 1) as a strategy to decrease shading since light availability should be enhanced as the sun rises in the east and sets in the west. Light reduction from decking designs will likely vary with latitude since the elevation and angle of the sun are related to latitude. Consequently, designs that promote light penetration in field settings at a given latitude should be applicable to that geographic region as well as more southern latitudes, but may not impart the same benefits to docks constructed at more northerly latitudes. Similarly, designs that promote light penetration in U.S. estuaries in spring and summer seasons may not produce the same improvements during

Fig. 2 Typical salt marsh loss under docks constructed closely over the marsh platform. (A) Private dock in MA, USA and (B) experimental dock constructed at < 1:1 H:W ratio (Logan et al. 2018b)

fall or winter (Alexander 2012). Dock orientation affected light availability under northeast and southeast U.S. docks (Alexander 2012; Logan et al. 2018a). In New England, relative decreases in aboveground production declined as dock decking shifted towards a north–south orientation with the greatest marsh loss observed for docks set east–west (Logan et al. 2018a). However, dock orientation did not influence salt marsh production under docks in the mid-Atlantic (Vasilas et al. 2011b) or southeast U.S. regions (Sanger and Holland 2002; Alexander and Robinson 2004; Sanger et al. 2004).

Relative impacts to salt marsh varied among decking designs (Kearney et al. 1983; Colligan and Collins 1995; Sanger and Holland 2002; Alexander and Robinson 2004; Sanger et al. 2004; Vasilas et al. 2011b; Alexander 2012; Logan et al. 2018a, b). Several U.S. state BMP guidelines for dock construction recommend alternative decking to promote light penetration (Table 1). Decking type was not a significant predictor of salt marsh stem density under northeast U.S. docks (Logan et al. 2018a) while stem density reductions under southeast U.S. docks were higher for grated than traditional decking (Alexander 2012). Controlled studies of light penetration through alternative decking in the southeast U.S. demonstrated that during fall and winter when the sun's elevation was low, light was not able to penetrate the grated decking any more than traditional plank decking. During spring and summer, when the sun's elevation was higher, alternative decking did enhance light penetration, but only produced a small (< 10%) increase in photosynthetically active radiation (PAR) relative to traditional plank decking (Alexander 2012). For docks in the northeast U.S., where the sun's elevation is lower, total light availability did not significantly differ between decking types (Logan et al. 2018a).

SAV is also vulnerable to shading, and light obstruction from decking and floats can result in eelgrass mortality and sub-lethal impacts (Burdick and Short 1999; Beal and Schmit 2000; Eriander et al. 2017; Sagerman et al. 2020). Extent and magnitude of impacts vary greatly depending on seagrass species, orientation, and other environmental conditions (Burdick and Short 1999; Gladstone and Courtenay 2014; Eriander et al. 2017). A recent review by Sagerman et al. (2020) found that SAV coverage under decking was only 18% of nearby control areas, but effects varied across studies and species with a range of 9 to 36%. Even structures that cause partial shading can largely eliminate existing eelgrass with minimal chance for regrowth (Pentilla and Doty 1990). Using experimental docks, Shafer and Robinson (2001) found Florida seagrass shoot densities to be 52 and 58% of control densities under docks placed at 5 ft (1.5 m) and 4 ft (1.2 m) above mean high water (MHW), respectively. In Australia, Gladstone and Courtenay (2014) found that *Zostera muelleri* subsp. *capricorni* is reduced by 75% under docks compared to controls.

Eutrophication effects (e.g., decreases in dissolved oxygen and water clarity, increases in sediment sulfides) impart additional stresses on eelgrass that can compound decking shading impacts (Goodman et al. 1995). Shading can also alter light availability towards levels more favorable for eelgrass competitors such as macroalgae (Markager and Sand-Jensen 1992). Once algae becomes established, it can outcompete eelgrass for space and light (Short et al. 1995; Hauxwell et al. 2003), as well as lead to hypoxia and eutrophication upon decomposition (Han and Liu 2014). In Massachusetts, this displacement is particularly concerning in inner-harbor areas subject to eutrophication. Benthic communities in Waquoit Bay, Massachusetts shifted from eelgrass to macroalgae-dominated due to deteriorating water quality, leading to shading impacts to the remaining eelgrass, decreases in dissolved oxygen, and increases in sulfide in the sediments, which are toxic to eelgrass (Hauxwell et al. 2003).

Similar to studies of docks constructed over salt marsh in New England (Kearney et al. 1983; Logan et al. 2018a, b), Burdick and Short (1999) and Short et al. (2009) found dock height to be the most important variable determining underlying light availability and by association, eelgrass bed quality. Shoot density of *Halodule wrightii* beneath docks in Perdido Bay, Alabama was reduced by 40 to 50% at light levels of 16 to 19% surface irradiance (SI) (Shafer 1999), with no seagrass found under docks with less than 14% SI. Mesocosm studies found 30% SI would support 50% of the normal eelgrass production (Short et al. 1995) and a significant reduction in photosynthetic response after 40 days of shading at ≤ 34% surface irradiance (Ochieng et al. 2010). Burdick and Short (1999) used mesocosm information to model dock heights that would achieve adequate surface irradiance (30%) for eelgrass growth around and beneath the structure in a microtidal lagoon in southern Massachusetts. They determined a 3 ft wide (1 m) dock requires a minimum height of approximately 10 ft (3 m) off the seafloor.

Dock height above the water surface has been shown to be positively correlated with underlying eelgrass coverage (Eriander et al. 2017). Short et al. (2009) created a "Dock Eelgrass Calculator" (DEC) that includes dock height above mean sea level (MSL), orientation and width variables that users can manipulate to estimate how impacts to eelgrass vary with different designs. For a 4 ft wide (1.2 m) dock typical of Massachusetts, the DEC estimates a north-oriented dock would need to be approximately 5.7 ft (1.7 m) above MSL to avoid impacts to underlying bed quality. For an east-facing dock, the required height to avoid impacts would increase to approximately 9.7 ft (3 m). Seagrass presence under docks in Florida was positively correlated with height based on a sampled height range of 0 to 5.6 ft (1.7 m) above MHW (based on Florida Department of Environmental Protection survey described in Beal and Schmit 2000). Experimental docks set at heights of 3 ft (0.9 m) and 4.9 ft (1.5 m) above MHW over seagrass beds in Florida also caused a decline in seagrass (percent cover and shoot density) although only shoot density effects scaled with dock height (Beal and Schmit 2000). Shafer and Robinson (2001) found that seagrass could persist under docks in Florida that were elevated 4 to 5 ft (1.2–1.5 m) above mean sea level and used fiberglass grid for the entire dock. Shading effects may be less where larger tidal ranges exist, due to the greater separation between the decking base and the water surface during low tides and potential for more light infiltration, but to our knowledge there are no studies that confirm this theory.

Orientation and width influence shading impacts on SAV (Burdick and Short 1999). East–west oriented docks cause all-day shading resulting in either poor quality beds or the complete loss of vegetation (Burdick and Short 1999). According to the Burdick and Short (1999) and Short et al. (2009) models, a dock constructed in an east–west orientation would require approximately twice the height to allow a given light penetration relative to a north–south oriented dock in the same location. Light levels are greater under the centers of narrow docks than under wide docks (Burdick and Short 1999). In Australia, seagrass (*Zostera muelleri* subsp. *Capricorni*) bed biomass was only approximately 25% of unshaded controls but the losses under docks in Australia were independent of dock orientation, possibly due to the narrow width (3–3.6 ft (0.9–1.1 m)) of sampled docks (Gladstone and Courtenay 2014). Tropic and subtropic seagrass species may also respond to dock shading differently than temperate seagrasses.

Applications of alternative decking materials have had variable success in limiting impacts to underlying SAV. Docks with grated decking in Australian coastal waters caused seagrass loss, although bed loss was less than that caused by traditional wooden decking (Gladstone and Courtenay 2014). Similarly, the installation of prisms along docks was found to have only minor benefits to eelgrass growth, unless used in great quantity (Blanton et al. 2002). The same study investigated several other products (e.g., SunTunnels, halide lights), and while some improved light conditions, they did not completely eliminate impacts. Beal et al. (1999) tested fiberglass and alternating wood-and-fiberglass decking and found only minor differences in irradiance. Landry et al. (2008) found that grated docks improved light irradiance under the structure relative to traditional float designs, but *Halophila johnsonii* densities were still reduced relative to unshaded reference areas. Gayaldo et al. (2001) tested the use of reflective alumino-metallic film attached to the structure's underside and piles, and found an increase in reflected light under the structure with light levels elevated from 1–3 to 9–11%, which allowed eelgrass to recolonize beneath the structure.

While impacts of small docks on vegetation have been well studied, studies of dock shading impacts on fish and invertebrate communities to date are mostly limited to large structures (i.e., piers, marinas, and bridges) and results may not scale to smaller private structures (Able et al. 2013). For example, most studies of piers have found negative shading effects on fish and invertebrates (Able and Duffy-Anderson 2005) while a study of small docks in the southeast U.S. showed that decking shading provided a beneficial thermal refuge to mangrove tree crabs (Cannizzo et al. 2018). Previous studies of large commercial piers have shown a variety of impacts including alteration of fish migratory behavior, habitat use, community composition, and growth (Munsch et al. 2017). Light levels under large urban piers decrease towards the pier interior (Able et al. 2013; Munsch et al. 2014), and shading appears to be the main driver of negative fish responses (Munsch et al. 2017). The large municipal and commercial piers included in previous studies all had widths > 60 m (200 ft) with a maximum width of 837 ft (255 m) for Pier 40 in the Hudson River estuary (Able and Duffy-Anderson 2005; Munsch et al. 2014). Bridges assessed for benthic invertebrate impacts were > 29.5 ft (9 m) wide (Struck et al. 2004). By contrast, the width of private docks in Massachusetts is typically < 5 ft (1.5 m) (Bliven and Pearlman 2003; Logan et al. 2018a) but situated closer to the water surface than larger piers (Munsch et al. 2014) and bridges (Struck et al. 2004).

Shading from decking can impact the communities that colonize underlying piles and nearby benthic habitat. For example, the percent cover of serpulid polychaetes, sponges, and solitary ascidians was often greater on piles in Australian marinas than rocks while spirorbid polychaetes, foliose and filamentous algae were more prevalent on rocks (Glasby 1999a). The differences between piles and rocks were attributed to shading, rather than the pile structure itself, as unshaded piles supported similar communities as rocks while shaded piles had significantly different species composition (Glasby 1999b). Therefore, shading from overlying dock decking may counteract any positive habitat contributions of the support piles (See review by Munsch et al. 2017). Shading from piers and other urban infrastructure in Brazil affected rocky intertidal community composition by reducing macroalgal biomass and cover as well as grazer size (Pardal-Souza et al. 2017). Relatedly, shading from decking combined with increased filter-feeder colonization of piles could potentially result in dock structures becoming carbon dioxide sources and energy sinks due to the resulting imbalance of primary producers and consumers (Malerba et al. 2019). Relationships between total productivity and increased filter-feeder colonization of shaded piles are complex, so indirect effects of net carbon uptake and carbon dioxide generation are not entirely clear (Riascos et al. 2020) but nonetheless represent a further example of potential unintended effects of shading. Additional research on fish and invertebrate communities associated with docks and associated environmental conditions would be useful in furthering our understanding of scaling effects from what is currently known for larger commercial piers.

Recommendations

- Avoid salt marsh and SAV whenever feasible: Dock construction should be avoided in areas containing salt marsh or SAV given the ecosystem services provided by these vegetation types and their susceptibility to shading.
- Focus on dock height, width, and orientation over salt marsh and SAV: When avoidance is not feasible, a minimum 1.5:1 H:W ratio is recommended for docks constructed over salt marsh and a 10 ft (3 m) height above the seafloor (or 5.7 ft (1.7 m) above MSL) for docks over SAV. Orientation should be within 10° of north–south to increase light penetration.
- Maintain light transmission even in unvegetated habitats: A 1:1 H:W ratio is recommended for docks constructed over non-vegetated intertidal and subtidal habitats to reduce shading impacts.
- Use alternative materials in poorly flushed areas: In areas with low flushing, the decking component should use alternative materials such as composite, fiberglass, or local hardwood.
- Use construction BMPs to minimize pressure-treated wood impacts: As with pile installation, on-site cutting should be avoided or proper containment used to prevent sawdust and shavings from entering the aquatic environment when using pressure-treated wood.

Floats

Floats are located at the seaward or channelward extent of a dock structure (Fig. 1) and can serve as a platform for swimming or fishing as well as a location for vessel tie-up. Floats are typically secured with piles or chain and anchor structures. Some dock structures do not include floats, but instead have pile-supported decking that terminates over the water. In some areas, floats are installed on a seasonal basis, and in other areas they are present year-round. Seasonal floats only impact the resource area for part of the year, but can have additional negative impacts if stored in sensitive resource areas (e.g., salt marsh). Often, floats rise and fall freely with the tide, secured in place either with piles or anchor systems.

Data on float impacts on benthic habitats are limited relative to other dock structural effects, but preliminary research has shown impacts do occur (e.g., Alexander et al. 2006). Depending on water depth and underlying marine

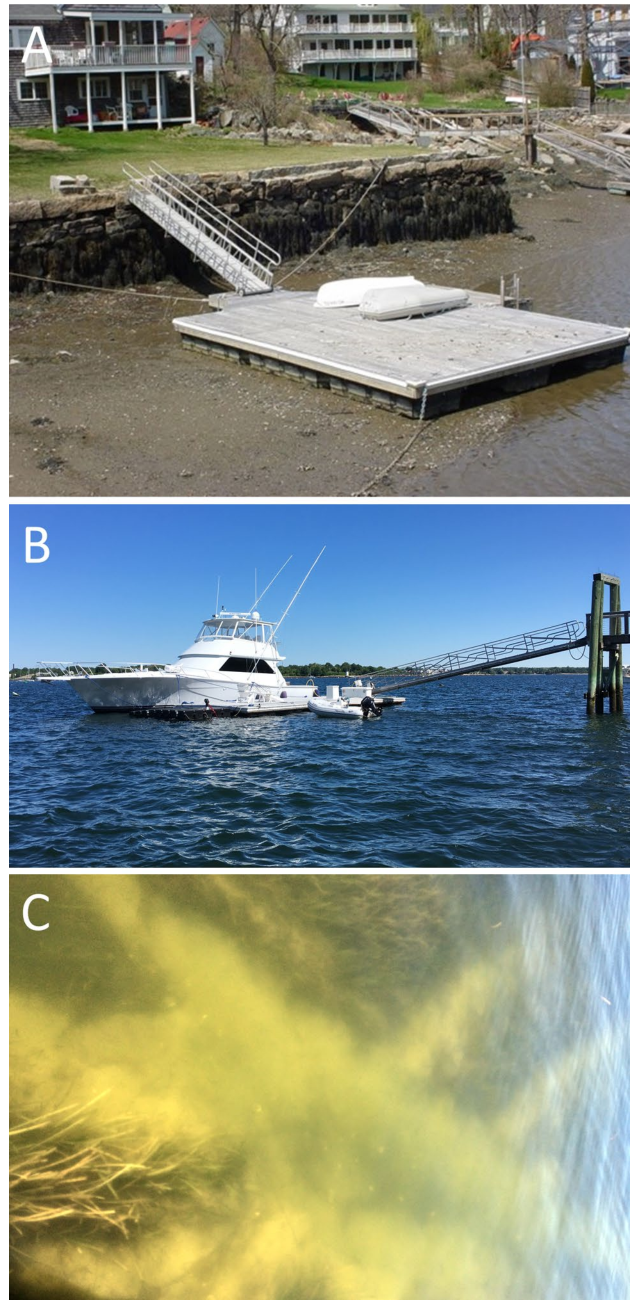

Fig. 3 Examples of potential float impacts on marine resources including (A) a float grounding on a mudflat at low tide, (B) increased shading impacts from a large vessel tied to a float, and (C) propeller scouring of eelgrass in shallow water that can occur when vessels transit to and from dock floats. All images taken in MA, USA

resources, dock floats can have both direct and indirect impacts. Direct impacts can occur when floats rest on the substrate during part of the tidal cycle (Fig. 3; Kelty and Bliven 2003; Alexander et al. 2006) while indirect impacts are mainly associated with shading (Burdick and Short 1999). Grounded floats can lead to crushing, smothering, burial, damage to vegetation, and disturbance to substrates (Nightengale and Simenstad 2001). As floats rise and fall with the tide they can lift sediments (i.e., "pumping") and impact water clarity (Kelty and Bliven 2003). Float mooring systems common with New England docks also impact the surrounding seafloor since the anchors and mooring lines come in contact with the seafloor. The placement of a float over shellfish habitat can lead to direct habitat displacement by piles or anchors. U.S. state guidelines generally recommend float water depths be a minimum of 1 to 4 ft (0.3 to 1.2 m) at MLW to minimize habitat impacts depending on the resources present (Table 1).

Floats installed in intertidal habitat can directly impact benthic habitat by grounding at low tide or indirectly impact nearby salt marsh through erosion or circulation alteration. In addition to the direct conversion of mudflat to artificial structure, there is often a "dead zone" of soft sediment surrounding float piles or footings (MacFarlane 1996). In such areas, impacts can be reduced through alternative means of water access including the use of a dinghy to reach a vessel moored in subtidal habitat, use of a nearby public boat launch, or creation of a shared, community dock. When alternatives are not feasible, pile-supported float stops are sometimes used. Float stop systems incorporate brackets near the bottom of the support piles to stop the float from falling below a certain height at low tide, suspending the float at that height until the next incoming tide. The area of direct impact can also be reduced through the installation of legs at the corners or skids running along the float edges.

Benthic Impacts

Assessments of benthic habitat under and adjacent to floats in Georgia identified impacts to sediment grain size and by association carbon and nitrogen content. Biological impacts included reductions in benthic algal production, macrofaunal numbers, and biomass (Alexander et al. 2006). The floats included in the Alexander et al. (2006) study all rested on the substrate at low tide and porosity profiles provided evidence that the floats were compressing the upper < 1 in (1–2 cm) surface sediment layer of the seafloor (Alexander et al. 2006). Many of the detected impacts also occurred under the float sections that did not contact the substrate. The authors attributed these impacts to flow acceleration and removal of finer, higher organic content sediment for the substrate areas under the overall float structure and adjacent to the sub-floats that rested on the substrate. Alexander et al. (2006) also found sediment grain size alterations downstream of the float, which suggests that float impacts may have a larger spatial footprint in systems with higher tidal velocities.

Shading

In addition to the smothering and crushing impacts of floats, SAV can be adversely impacted by shading, which has been documented in both U.S. (Fresh et al. 1995, 2006; Burdick and Short 1999) and European (Eriander et al. 2017) waters. As with decking, the terminal float, if positioned over existing or historic eelgrass areas, can reduce, limit, or entirely prevent eelgrass from vegetating the area directly under the float as well as a halo area, which can be as much as twice the size of the float (Fresh et al. 1995; Smith and Mezich 1999). Overall, shading from floats has greater impacts on SAV than pile-supported decking (Burdick and Short 1999; Eriander et al. 2017). The float impacts are likely a result of a combination of shading and direct impact, since floats may directly come in contact with eelgrass. Unlike pile-supported decking, floats are directly on the water surface and so impart a greater shading impact. Burdick and Short (1999) found that no eelgrass could be supported under floats where tidal range was < 3.3 ft (1 m). Impacts were greater under floats compared to fixed-height docks, and eelgrass was completely absent under three of the four floats that they examined. Fresh et al. (1995) observed that mean eelgrass density beneath floats in Puget Sound, Washington was 24% of the mean undisturbed control density, and eelgrass density one structure width away from the dock was 60% of the control. In Sweden, eelgrass was completely absent under all sampled floats while habitats under and adjacent to pile-supported docks had an average reduction in coverage of approximately 70% (Eriander et al. 2017). Shading impacts are not constrained to the area directly under floats, but instead can extend as far as 20 to 26 ft (6–8 m) from the float edge (Eriander et al. 2017). This broader area of shading impact can be a result of side-shading from the float itself as well as additional shading created by vessels tied up to the float (Fig. 3B; Eriander et al. 2017).

Several studies have investigated the effectiveness of alternative construction materials to enhance light penetration under floats with most studies documenting continued seagrass loss. Steinmetz et al. (2004) found that the use of glass prisms as part of the float did not prevent loss of underlying seagrass in Florida. Fresh et al. (2006) used both steel and fiberglass grated decking over *Z. marina,* and found that floats with as much as 50% grated space still impacted eelgrass in Puget Sound and were of no ecological benefit compared to traditional planked floats. Shafer and Robinson (2001) studied alternative materials such as acrylic, lexan, aluminum grating, and fiberglass grating, and found significant declines in turtle grass

(*Thalassia testudinum*) in Florida in both spring and summer in nearly all treatments. Only the fiberglass grating showed promise for continued turtle grass growth. Vulnerability to shade stress varies among SAV species (Collier et al. 2016; Nelson 2017) and light transmission through grating varies with latitude so design benefits may also depend on the species present and geographic location of a given dock site.

Anchor System Impacts

The anchor system that secures the float to the seafloor can also result in direct impacts, whether concrete block anchors with chain, or helical anchors and flexible rodes commonly known as conservation moorings. While common in New England, anchor systems are not typically used in the southeast U.S. and so present more of a regional problem (C. Alexander, Pers. Comm). A study investigating the efficacy of conservation mooring systems in five Massachusetts harbors found that these systems can minimize impacts to eelgrass when they are entirely floating above the sediment, which requires proper installation and regular maintenance (MA DMF 2019). In some cases, once the eelgrass is uprooted, eelgrass detritus can linger and prohibit new growth (MA DMF Pers. Obs.; Walker et al. 1989). Impacts to the rhizome system can lead to reduced habitat and destabilized sediments. Furthermore, moorings in some embayments are prone to heavy fouling by mussels, tunicates, and other organisms that weigh the mooring down causing additional scour. If conservation mooring rodes are left in the water to over-winter, unlike chain dropped to the bottom in one place, the floating rode will partially sink to the bottom, resulting in a larger scour scar in the surrounding eelgrass (MA DMF 2019). While these systems can result in lower impacts to the seafloor and SAV than traditional block and chain systems with proper design, maintenance, and adequate water depth, in practice they often still cause damage to eelgrass meadows (MA DMF 2019) and should not be viewed as a mitigating measure to permit float construction in eelgrass beds. Further study is also required to assess their efficacy in minimizing impacts to seagrass when used as float anchors.

Invasive Species

Like piles, floats can serve as habitat for invasive species (Lambert and Lambert 1998, 2003; Lambert 2002; Arenas et al. 2006; Ashton et al. 2006; Minchin 2007) by providing colonizing substrate. Floats can potentially facilitate the further spread of invasives to natural hard substrate if present in the surrounding waterbody. Potential invasive species interactions are described in greater detail in the "Invasive Species" sub-section of the "Pile Design and Materials" section and not repeated here as impacts from floats are similar.

Boating Impacts

Boats are commonly moored to dock floats, and boating activity associated with floats can also negatively affect the benthic environment and SAV in particular (Orth et al. 2006; Sagerman et al. 2020) if the float is located in shallow water. SAV can be indirectly impacted by boats tied up to floats through shading (Eriander et al. 2017). Floats in Sweden had a distinct border between low and full eelgrass coverage that was 7 to 8 m (~ 23 to 26 ft) from the float edge. This extended area of shading impact outside of the float footprint was likely influenced by additional shading from associated vessels (Eriander et al. 2017). Boating activity around floats can also result in direct impacts. Propeller dredging occurs when a boat's propeller or water jets suspend sediments. This may result in the direct loss of vegetation or loss through burial by sediments (Fig. 3; Burdick and Short 1999; Sagerman et al. 2020). Propeller scarring of SAV has been well documented in shallow water habitats worldwide (e.g., Gonzalez-Liboy 1979; Loflin 1995; Burdick and Short 1999; Martin et al. 2008; West 2011; Hallac et al. 2012). Propeller scarring in Florida Bay was most prevalent in shallow waters ≤ 2 m (6.6 ft) (Sargent et al. 1995; Hallac et al. 2012). The disturbed area may become unsuitable for seagrass growth due to changes in depth or substrate type and become permanently void of growth or take many years to heal. For example, eelgrass scars from mussel dragging in Maine, USA were estimated to require from 6 to 20 or more years to repopulate depending on growth conditions (Neckles et al. 2005). Other potential adverse effects of propeller scarring include destabilization of sediments as well as creation of habitat for algae and invasive species to take hold. Scar areas also support lower macrofaunal abundance and diversity relative to SAV (Uhrin and Holmquist 2003). In addition to SAV impacts, turbidity generated by propellers can disrupt or even bury newly settled shellfish (MacFarlane 1996) and finfish eggs (Whitfield and Becker 2014).

Recommendations

- Minimize float size: Float size should be the minimum length and width necessary to berth the associated vessel. Minimizing the float footprint will reduce the scale of potential impacts (e.g., shading, turbidity, benthic scour).
- Place floats in the deepest portion of the site: Floats should be placed in the deepest available water and in areas where the structure will not ground at low tides. We support the continued use of existing guidelines rec-

ommended for the protection of shellfish habitat (2.5 ft (0.8 m) at MLW) and all other locations (1.5 ft (0.5 m)) to protect other benthic fauna in Massachusetts as these buffers will prevent float grounding and propeller scour from smaller vessels. These depths should be measured from the substrate to the lowest structural float component.

- Use alternative designs or means of water access when the float cannot reach subtidal habitat: For locations where float placement in subtidal habitat is not feasible, alternatives to individual dock and float systems should first be explored. In the absence of viable means of alternative water access, we recommend pile-supported float stops to keep the entire float from contacting the seafloor at low tide. If piles and float stops are not feasible, legs or skids should be used to minimize the area of direct float impact.
- Avoid SAV: Given the vulnerability of SAV to direct and indirect impacts from float installation, MA DMF recommends avoidance as the best strategy for floats and any associated anchors or piles. Terminal floats should be located at least 7.5 m (~25 ft) from the edge of existing or historically mapped eelgrass.
- Maintain buffer from salt marsh: Maintain at least 7.5 m (25 ft) buffer from the edge of the nearest salt marsh to avoid circulation and/or erosion impacts.
- Seasonal floats should be stored outside of aquatic resource areas: For cases where floats are removed seasonally, storage locations should avoid sensitive resource areas. Floats should be stored above the high tide line and outside of any salt marsh or other sensitive aquatic habitats.

Cumulative Impacts

While individual small docks may have only limited impacts on any given aquatic resource, dense build-out can collectively result in greater overall fragmentation, alteration, and loss of habitat that should be considered in coastal planning (MacFarlane 1996; Peterson and Lowe 2009). For example, in Massachusetts, > 2500 docks occupied over 6 ha of salt marsh (Logan et al. 2018a). Docks occupied > 28 ha and 60 ha of coastal habitat, in Georgia and South Carolina, respectively (Sanger et al. 2004; Alexander and Robinson 2006). In addition to causing direct habitat loss, docks and other artificial structures can disrupt ecological connectivity by creating barriers to the movement of certain species, providing new structure that facilitates the movement of other species, and altering trophic connectivity (Bishop et al. 2017). While cumulative impacts are important to quantify and account for in coastal management, quantifying broader impacts remains a challenge due to the current lack of consensus on an approach.

System-level assessments and planning may be achieved by considering individual dock applications in the context of relative ecosystem-level impacts and quantitative comparisons of benefits and negative impacts. MacFarlane et al. (2000) described an index to identify areas of greater resource sensitivity in Pleasant Bay, Massachusetts, which could then be avoided during construction activities. The sensitivity index used a variety of abiotic, biological, and anthropogenic factors to assign a ranking to a given region of the study system of 0, 0.5, or 1, for low, moderate, and high resource sensitivity, respectively. Sites with total scores > 5.5 were considered highly sensitive and consequently unsuitable for dock installation (MacFarlane et al. 2000). This framework was used by the towns surrounding Pleasant Bay to identify regions within Pleasant Bay to restrict new dock construction (Pleasant Bay Resource Management Alliance 2018). Needles et al. (2015) described a similar matrix approach that provides managers with a decision support tool to quantify tradeoffs of a variety of management actions. The matrix considered cultural, climate regulation, biological services (e.g., food), storm protection, water quality, and biodiversity services by assigning positive, negative, or neutral effects of each considered management action on these services. The Programmatic General Permit (PGP83) of the U.S. Army Corps of Engineers for private docks in Georgia included a cumulative impact assessment based on annual dock build-out and estimates of the percent of different coastal habitat types being impacted (e.g., marsh, tidal flats). The environmental impact assessment estimated a variety of direct and indirect impacts (e.g., shading, wrack accumulation, sediment re-suspension, float grounding) and found that the impacted area was < 1% of the state's total marsh habitat, although cumulative impacts were not assessed at the level of individual systems (ACOE 2012; King and Blair 2012).

Best management practices including the siting, timing, installation methods, materials, and designs employed in the construction of docks can minimize environmental impacts of individual dock structures, but cumulative impacts still should be accounted for in the permitting process. Most BMPs will function to minimize rather than completely avoid environmental impacts and so adoption of these practices without consideration of the cumulative impacts with dock proliferation may instill a false sense of environmental preservation. For example, dock shading and loss of underlying marsh production can be minimized by BMPs that increase the dock's height to width (H:W) ratio, but even docks with optimized H:W under practical constraints will induce some level of shading and marsh loss. Docks set at a 1.5:1 H:W ratio and oriented N-S reduced shading

Table.3 Best management practice (BMP) recommendations for docks in New England estuarine and coastal waters

Component	Recommendation
Piles	
Design – general	•Minimize footprint (diameter and number) •Maximize pile spacing
Design – over sensitive habitats (salt marsh, SAV)	•Avoid where feasible
Installation	•Work from upland where feasible •Use shallow-draft barge that avoids grounding •Use vibratory rather than impact driving where feasible •Avoid jetting •Employ "soft start" to reduce turbidity and noise impacts when using impact driving •Use silt-curtains in areas with fine-grained sediment •Avoid sensitive life history periods (e.g., fish spawning)
Materials	•Use CCA or local hardwood instead of ACQ or other high-copper content treated wood •Avoid cutting over resource area to reduce leachate release to aquatic environment
Decking	
Design – general	•Minimize width (≤ 4 ft (1.2 m)) •Maximize H:W ratio (≥ 1:1) •Orient within 10° of North •Avoid add-ons that increase shading (e.g., bump-outs, gazebos)
Design – over sensitive habitats (salt marsh, SAV)	•Avoid where feasible by micro-siting walkway in area of property outside sensitive habitats •Follow General Design BMPs with the following modifications: oSalt Marsh -Maximize H:W ratio (≥1.5:1) oSAV -Minimize width (≤ 3 ft (0.9 m)) -Maximize height above seafloor (> 10 ft (3 m))
Material	•Use composite or local hardwood
Floats	
Design – general	•Place in deepest available water to avoid direct and indirect impacts during low tide •Maintain minimum 1.5 ft (0.5 m) depth under float at mean low water (MLW) •Avoid intertidal habitat where feasible •If only intertidal habitat available, use pile-supported float stops to avoid or legs/skids to minimize contact with seafloor •Orient within 10° of North •Minimize float area to size required for intended use
Design – over sensitive habitats (salt marsh, SAV, shellfish)	•Follow General Design BMPs with following modifications: •Maintain minimum 25 ft (7.5 m) separation from any bordering salt marsh or SAV •Avoid placing float anchors in SAV •Maintain minimum 2.5 ft (0.8 m) depth under float at mean low water (MLW) in shellfish habitat
Installation	•Avoid grounding of work barges •Remove seasonally and store outside resource areas
Cumulative impacts	
Community planning	•Quantify cumulative impacts at ecosystem-level to place individual dock permitting in proper context •Develop by-laws and tools for minimizing cumulative impacts to sensitive resources (e.g., salt marsh, SAV) and cultural resources (e.g., shellfishing flats) •Require mitigation for unavoidable impacts
Alternative access approaches	
	•Public boat ramps •Community docks •Moorings •Boat lifts

impacts relative to lower H:W designs, but still caused an approximate 30% decline in low marsh production (Logan et al. 2018b). When viewed in the context of a single dock, such impacts may be considered minor and consequently permittable, but dense dock build-out even following BMPs (Table 3) could cause collective, cumulative impacts to salt marsh and other estuarine resources.

In some U.S. states, towns can implement BMP conditions comprehensively through the development of local bylaws that can be more specific and conservative than state laws (Patterson 2003; Goetsch 2011; Massachusetts Association of Conservation Commissions 2016). In Massachusetts, many towns have taken this approach to maximize protection of eelgrass and other marine resources (Goetsch 2011; Massachusetts Association of Conservation Commissions 2019). In Massachusetts, the Waterways Regulations includes language (310 CMR 9.38(2)(b)) that prohibits state permitting of docks in areas that have been identified by town-level management plans as being unsuitable due to competing uses (e.g., shellfishing) or presence of sensitive resources (e.g., SAV). The Massachusetts Association of Conservation Commissions created a template to help guide towns in developing such bylaws (Massachusetts Association of Conservation Commissions 2016). A bylaw approach would allow a town to broadly implement BMPs to all future dock projects, which serves the added benefit of streamlining the permitting process by clearly defining required designs and practices. In Massachusetts, individual town bylaws include specific conditions relating to dock construction methods (e.g., prohibition of jetting in pile installation), location relative to sensitive marine resources (e.g., floats need to be seaward of salt marsh and eelgrass), and size (e.g., 4 ft (1.2 m) maximum width) (Town of Bourne 2000; Town of Hingham 2014; Town of Yarmouth 2016).

Recommendations

- Map and quantify important resource areas at a system level: Improved characterization of existing resources and abiotic conditions of regions within individual estuaries can guide BMPs and restriction decisions at a subembayment scale.
- Develop sensitivity indexes to guide the use of BMPs vs. moratoriums: Sensitivity indexes should be developed for individual estuaries to identify areas where dock installation could occur with limited environmental impact with appropriate BMPs as well as areas where restrictions are more appropriate due to the presence of more vulnerable resources (e.g., SAV) and/or important cultural or recreational resources (e.g., productive shellfish flats).
- Consider BMP guidance in the context of anticipated dock build-out: A system-wide approach to management may identify areas where dock build-out can be accomplished with limited resource impacts. If restrictions are not feasible for a given town or estuary due to state or federal permitting standards, individual dock BMPs should be scaled towards the most conservative parameters to account for cumulative impacts. Decisions on BMP guidance for individual dock projects can best be made in the context of anticipated cumulative impacts from continued build-out of similar structures in a given system.
- Consider mitigation strategies for unavoidable resource impacts: When avoidance is not feasible, cumulative impacts of dock build-out within an estuary can also be addressed through mitigation approaches. For cases where BMPs are employed but unavoidable impacts still occur, mitigation can be required to avoid net loss of habitat within a system.

Data Gaps and Next Steps

Additional data are still needed to better understand dock impacts (e.g., float grounding), possible designs to minimize impacts (e.g., effective light-transmitting decking), and strategies to incorporate cumulative impacts. Data assessing potential impacts of grounding floats on epi- and infaunal communities are lacking as is an assessment of how such impacts may be avoided or minimized across different minimum depth thresholds. While alternative designs intended to reduce shading have largely been unsuccessful to date, some designs have shown promise and further research into where those designs work as well as new designs and materials is warranted. For example, further development and testing of different grated decking designs as well as light reflective pile material or reflective material under decking or floats is warranted to potentially reduce shading and associated habitat alteration. Further development of pile materials and designs that promote colonization by native species is also warranted as such an approach could allow dock structures to promote ecosystem services (e.g., biomass of filter feeders). Relatedly, further study of the relationship between submerged dock structures (piles and floats) and invasive species colonization is warranted as piles should ideally be designed and situated in a manner that does not promote the spread of invasives (Chapman et al. 2018). Continued development of ecological engineering designs (See review in O'Shaughnessy et al. 2020) and similar field trials are needed to identify appropriate designs for different habitat types and geographical regions. Minimization of dock proliferation combined with selection of designs and materials that limit negative impacts and where possible provide habitat enhancements are key management strategies for the preservation of coastal and estuarine ecosystems.

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