Estuaries and Coasts (2018) 41:661–675
DOI 10.1007/s12237-017-0323-1

MANAGEMENT APPLICATIONS

Effects of Docks on Salt Marsh Vegetation: an Evaluation of Ecological Impacts and the Efficacy of Current Design Standards

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Received: 28 April 2017 / Revised: 18 August 2017 / Accepted: 10 September 2017 / Published online: 25 September 2017

Abstract Private docks are common in estuaries worldwide. Docks in Massachusetts (northeast USA) cumulatively overlie ~ 6 ha of salt marsh. Although regulations are designed to minimize dock impacts to salt marsh vegetation, few data exist to support the efficacy of these policies. To quantify impacts associated with different dock designs, we compared vegetation characteristics and light levels under docks with different heights, widths, orientations, decking types and spacing, pile spacing, and ages relative to adjacent control areas across the Massachusetts coastline ($n = 212$). We then evaluated proportional changes in stem density and biomass of the dominant vegetation (*Spartina alterniflora* and *Spartina patens*) in relation to dock and environmental (marsh zone and nitrogen loading) characteristics. Relative to adjacent, undeveloped habitat, *Spartina* spp. under docks had ~ 40% stem density, 60% stem biomass, greater stem height and nitrogen content, and a higher proportion of *S. alterniflora*. Light availability was greater under taller docks and docks set at a north-south orientation but did not differ between decking types. Dock height best predicted vegetation loss, but orientation, pile spacing, decking type, age, and marsh zone also affected marsh production. We combined our proportional biomass and stem elemental composition estimates to calculate a statewide annual loss of ~ 2200 kg dry weight of *Spartina* biomass (367 kg per ha of dock coverage). Managers can reduce impacts through design modifications that maximize dock height (> 150 cm) and pile spacing while maintaining a north-south orientation, but dock proliferation must also be addressed to limit cumulative impacts.

Keywords Aboveground biomass · Cumulative impacts · Light attenuation · Shading · *Spartina* · Stem density

Communicated by Nathan Waltham

Electronic supplementary material The online version of this article (https://doi.org/10.1007/s12237-017-0323-1) contains supplementary material, which is available to authorized users.

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Introduction

Salt marshes provide a variety of ecosystem services including carbon sequestration (Chmura et al. 2003; Duarte et al. 2005), fish habitat and energy sources (Boesch and Turner 1984; Deegan and Garritt 1997; Deegan et al. 2000), and erosion control (Koch et al. 2009; Barbier et al. 2011; Shepard et al. 2011; Ysebaert et al. 2011). Anthropogenic impacts threaten these functions (Kennish 2001; Gedan et al. 2011), particularly in the context of cumulative impacts (Peterson and Lowe 2009; Needles et al. 2015). Sea level rise and interactions with other stressors like eutrophication and coastal development pose threats to salt marsh systems in New England (Gedan et al. 2011) and throughout the USA (Kennish 2001). Coastal development can directly displace or fragment existing salt marsh and also prevent shoreward migration of salt marsh vegetation in response to sea level rise (National Research Council 2007).

Regulatory agencies have developed a variety of permitting conditions and best management practice guidelines aimed at reducing impacts to salt marsh through modifications to dock designs. Conditions relate to dock height and width (e.g., Bliven and Pearlman 2003; New Hampshire Department of Environmental Services 2009; Connecticut Department of Energy and Environmental Protection 2015; COMAR

2016), deck spacing (e.g., Bliven and Pearlman 2003; New Hampshire Department of Environmental Services 2009), decking type (e.g., The Savannah District U.S. Army Corps of Engineers 2012), and orientation (e.g., Bliven and Pearlman 2003). While such guidelines and permit conditions are designed to reduce shading and consequently limit vegetation loss, design-specific data are often lacking or do not support the presumed environmental benefits (e.g., Vasilas et al. 2011; Alexander 2012).

The reduction in vegetation stem density and biomass associated with private docks constructed over salt marsh (Kearney et al. 1983; Sanger et al. 2004; Alexander and Robinson 2006; Vasilas et al. 2011) can lead to a reduction in ecosystem services. Observed reductions in marsh stem density under dock structures (Kearney et al. 1983; Sanger et al. 2004; Alexander and Robinson 2006; Vasilas et al. 2011) can reduce the effectiveness of marsh vegetation in preventing erosion and storm damage (Gleason et al. 1979). Shading-induced reduction of aboveground biomass (Alexander 2012) translates to a direct loss of detrital inputs to estuarine food webs and can lead to diminished benthic invertebrate diversity and abundance (Struck et al. 2004). Loss of belowground biomass would potentially shift these areas of salt marsh under docks from carbon sinks to sources (Macreadie et al. 2013). Coastal eutrophication, another stressor negatively impacting salt marsh ecosystem services (Deegan et al. 2012), could interact with shading stress to compound impacts to vegetation. While individual impacts may be relatively minor, docks are common along the US east coast (Kelty and Bliven 2003; Patterson 2003a; Patterson 2003b) and pose potential cumulative ecosystem-level impacts (Peterson and Lowe 2009; Needles et al. 2015). In South Carolina, an estimated 60 ha of salt marsh was lost through a decade of dock proliferation (Sanger et al. 2004).

We quantified vegetation characteristics under and adjacent to docks in Massachusetts, related aboveground marsh production to structural design and environmental variables, and estimated cumulative impacts. We present results in the context of potential management implications.

Methods

We sampled docks in Massachusetts (n = 212) from July to September 2014 (Table 1). We identified potential dock sampling sites by reviewing Google Earth v. 6.1 imagery and identifying existing structures constructed over salt marsh. We distributed our sampling effort to cover all regions of the mainland Massachusetts coastline containing both salt marsh and dock structures. We only sampled docks that were constructed over a continuous undisturbed (i.e., lacking docks or other shoreline structures) band of salt marsh vegetation that extended ≥ 5 m laterally on at least one of the two dock sides as assessed in the field. Field sampling was limited to docks accessible by foot or kayak.

For each sampling site, we measured a suite of dock characteristics and collected vegetation samples under and adjacent to the dock. We performed sampling at the section of the dock centered midway between the upper and lower bounds of the marsh zone overlaid by the dock. We further centered the sampling locations midway between adjacent support pilings within this sampling area. If the dock traversed both high and low marsh zones, we sampled both zones with sampling location centered midway within each respective zone (n = 9 docks). At each sampling location, we measured geolocation of the centered sampling location on the dock (latitude, longitude), dock width (cm), spacing between decking planks (cm), decking type (traditional planks or alternative grated design), spacing between support pilings (cm), and dock height measured from the marsh platform surface to the base of the support stringer (cm). We estimated dock orientation and age post-field sampling using the ruler and time slider tools, respectively, in Google Earth v. 6.1. For the age estimate, we assigned dock age to the oldest available historical image year for which the dock was present. We collected eight 1/16 m^2 clip plot samples under each dock with individual quadrat locations based on a randomized grid system that equally represented the entire area under the dock. For each sample, we clipped all stems originating within the quadrat area at the marsh surface. Following the same sampling methods, we collected eight clip plot samples from an adjacent control site located five meters perpendicular to the dock. The five meter separation was based on methods used in previous dock shading studies (Alexander and Robinson 2004; Sanger et al. 2004; Alexander 2012) and designed to avoid dock shading effects while remaining close enough to the dock site to properly characterize the marsh region where the dock was located. If both sides of the dock contained salt marsh vegetation, we determined the control sampling side based on a coin toss. We transferred all clip plot samples to frozen storage at - 18 °C within 24 h of sampling.

For vegetation characteristics we measured community composition as well as stem density, dry biomass, elemental composition, height, and nitrogen stable isotope ($\delta^{15}N$) values. We approximated elevation and nitrogen loading based on marsh zone (high and low) and stem $\delta^{15}N$, respectively. For each quadrat sample, we first thawed, rinsed, and separated live and dead stems. We further separated live stems by species and performed species-specific stem counts. For cases where a dock and control both contained *Spartina*, we measured the five tallest stems from each quadrat sample of the numerically dominant species (*Spartina alterniflora* (n = 171 sites) or *Spartina patens* (n = 35 sites)) using an electronic measuring board (± 0.1 mm). We then dried all live stems at 70 °C for ≥ 48 h and weighed (± 0.01 g) dried stems separately by species. For samples containing *Spartina*, we

Table 1 Characteristics of docks sampled in Massachusetts

Decking	Number	Height (cm)	Width (cm)	Height/width (*H*/*W*)	Orientation (°)	Deck Spacing (cm)	Age (years)	Piling spacing (m)	Latitude (°)	Longitude (°)
All										
Planking	197	108 ± 38	117 ± 23	0.9 ± 0.3	169 ± 98	1.9 ± 0.8	16 ± 6	3.2 ± 0.9	41.8 ± 0.3	− 70.7 ± 0.3
Grated	24	108 ± 47	116 ± 20	0.9 ± 0.4	175 ± 97	1.7 ± 0.7	9 ± 7	3.8 ± 1.1	41.7 ± 0.3	− 70.5 ± 0.1
Both	221	108 ± 39	117 ± 23	0.9 ± 0.3	170 ± 97	1.9 ± 0.8	15 ± 6	3.3 ± 1.0	41.8 ± 0.3	− 70.7 ± 0.2
High marsh										
Planking	71	100 ± 35	115 ± 23	0.9 ± 0.3	150 ± 96	2.0 ± 0.8	15 ± 5	3.1 ± 0.8	41.9 ± 0.4	− 70.7 ± 0.2
Grated	9	96 ± 44	114 ± 21	0.8 ± 0.3	195 ± 99	2.1 ± 0.8	12 ± 7	4.2 ± 0.9	41.8 ± 0.5	− 70.6 ± 0.1
Both	80	99 ± 36	115 ± 22	0.9 ± 0.3	155 ± 96	2.0 ± 0.8	15 ± 5	3.2 ± 0.9	41.9 ± 0.4	− 70.7 ± 0.2
Low marsh										
Planking	126	113 ± 39	118 ± 24	1.0 ± 0.4	180 ± 97	1.9 ± 0.8	16 ± 6	3.3 ± 1.0	41.8 ± 0.3	− 70.6 ± 0.3
Grated	15	115 ± 48	116 ± 20	1.0 ± 0.4	163 ± 98	1.4 ± 0.5	7 ± 6	3.6 ± 1.2	41.7 ± 0.2	− 70.5 ± 0.1
Both	141	113 ± 39	118 ± 23	1.0 ± 0.4	178 ± 97	1.8 ± 0.8	15 ± 6	3.3 ± 1.0	41.8 ± 0.3	− 70.6 ± 0.3

Values are mean ± standard deviation. For the combined high and low marsh dataset, nine sites had docks over both zones and characteristics include measurements from sampling locations from both zones

homogenized the dominant *Spartina* species (*S. alterniflora* or *S. patens*) using a blender and archived a subsample of the homogenate. We later weighed and packed approximately 6–7 mg of each subsample (± 0.001 mg) with sufficient material (*S. alterniflora* $n = 167$; *S. patens* $n = 33$) into a tin capsule. Packed samples were analyzed for percent carbon and nitrogen (control and dock samples) and nitrogen stable isotope ratios ($\delta^{15}N$; $^{15}N/^{14}N$) (control samples only) at the Viking Environmental Stable Isotope Lab (VESIL) at Salem State University. All isotope samples were calibrated using USGS40 and USGS41 standards and are reported as per mil relative to atmospheric N_2. Reference gas stability on the elemental analyzer/isotope ratio mass spectrometer (EA/IRMS) system was 0.02‰. For elemental analysis, values were corrected using a sulfanilamide standard, and for each daily run, factors ranged between 0.95 and 1.05. The mean standard deviations of duplicate samples were 0.2% (N, $n = 41$), 0.1% (C, $n = 41$), and 0.1‰ ($\delta^{15}N$, $n = 21$).

In July–September 2015, we installed HOBO light loggers (model UA-002-64; Onset Computer Corporation) under and adjacent (five meters perpendicular) to a subset of the docks sampled in 2014 ($n = 31$) with different height, orientation, and decking characteristics. HOBO loggers record total light in lux, which is significantly correlated with photosynthetically active radiation (PAR) (Long et al. 2012; Medeiros et al. 2013). We mounted the light loggers on wooden stakes set 0.3 m above the marsh platform and collected six full days of data at each dock site. We converted dock light data to percent control values.

To assess cumulative dock impacts, we visually identified all docks overlying salt marsh in Massachusetts using the most recent Google Earth imagery available for a given region. Since some structures are seasonal, we viewed multiple years of imagery for sites with the most recent images taken during winter months. We also viewed multiple years of recent imagery in cases where the most recent image contained shadows or attributes that obscured our view of dock structures. We calculated the area of each dock directly overlying salt marsh using the ruler tool in Google Earth. To estimate measurement error, a second analyst randomly selected 50 docks from the full list of identified docks over salt marsh and measured dock areas using the same methods applied by the primary analyst. We combined our total dock area estimates with our field estimates of *Spartina* dry biomass and elemental composition to generate estimates of total marsh biomass, carbon, and nitrogen loss due to dock effects across the state of Massachusetts.

To assess dock impacts in relation to statewide salt marsh habitat, we quantified the total area of salt marsh in Massachusetts using the most recent Massachusetts Department of Environmental Protection (DEP) salt marsh habitat layer (Massachusetts Office of Geographic Information 2009). The DEP assessment, completed in 2009 using photography from 1990, 1991, 1992, 1993, 1999, and 2000, used orthophotography interpreted at a scale of 1:12,000 to map wetlands. We selected all polygons coded "salt marsh" and "barrier-beach salt marsh" and summed the area of all selected polygons to generate an estimate of 18,378.1 ha of total marsh area after accounting for an estimated loss of 2.11 ha of salt marsh between 2009 and 2014 (Massachusetts Office of Geographic Information 2014).

Statistical Analysis

We compared stem density, dry weight, and elemental composition between dock and control samples using a Wilcoxon signed-rank test. We performed stem density and dry weight analyses for the entire dataset and separately for the high and low marshes. We made high and low marsh classifications based on the dominant plant species in terms of stem density for each site's control sample. We classified a site as low marsh if control samples had *S. alterniflora* as the dominant species and/or co-dominance of *S. alterniflora* and *Salicornia* spp. We assigned high marsh classification to sites with dominance and/or co-dominance of *S. patens*, *Distichlis spicata*, and *Juncus gerardii*. We performed the analysis for pooled *S. alterniflora* and *S. patens* and also performed separate analyses for each species individually for subsets of the dock dataset for which each species was present in the dock and/or control sample. For stem density and biomass, we calculated percent control values for each dataset both as pooled values and as median ± median absolute deviation (MAD) for individual dock site values. We also calculated coefficient of variation (CV) among the eight individual clip plots for each dock and control sample to compare patchiness between groups.

We compared maximum stem heights between dock and control samples for *S. alterniflora* and *S. patens* using a mixed model ANOVA in the "nlme" package (Pinheiro et al. 2017) in R (R Core Team 2016). We included treatment (dock or control) as a fixed effect and site as a random effect to account for multiple samples per site. To assess treatment significance, we used *F* tests with the "ANOVA" function in R.

We compared vegetation community composition for dock and control sites using analysis of similarities (ANOSIM), multivariate analysis of variance using distance matrices (adonis), and the similarity percentage technique (SIMPER) in the "vegan" package (Oksanen et al. 2016) in R. ANOSIM and adonis test whether sites within categories are more similar than sites in different categories. The resulting R statistic provides a correlation coefficient where values close to zero reflect minimal correlation between groups while values closer to one or negative one represent strong correlation. For these analyses, we first converted stem counts for each species in a given site to proportions. We then used these proportions to calculate a Bray-Curtis (BC) dissimilarity index using the following equation:

$$d_{jk} = \frac{\sum_{i=1}^{n} \left| x_{ij} - x_{ik} \right|}{\sum_{i=1}^{n} \left(x_{ij} + x_{ik} \right)}$$

where d is the Bray-Curtis dissimilarity distance, j and k are two sample sites, x is the proportion of stems of species type i in a given site sample, and n is the total number of site samples. This metric converts all community composition comparisons between sites to values ranging from 0 to 1 such that sites with no species overlap will have a value of 0, sites with complete overlap will have values of 1, and sites with partial overlap will have values between 0 and 1. We calculated Bray-Curtis metrics separately to also generate estimates based only on comparisons within individual dock-control pairs. Using this Bray-Curtis matrix, we compared community composition between dock and control sites with ANOSIM and adonis. We then determined the relative contribution of each species to observed differences between dock and control sites using SIMPER. We performed these analyses for the entire dataset and also separately for the high and low marsh zones.

We assessed the effects of a suite of dock characteristic and environmental variable co-variates on relative stem counts ($Stem\ Count\ Proportion = \frac{Dock\ Stem\ Count}{Dock\ Stem\ Count + Control\ Stem\ Count}$) and dry weights ($Stem\ Weight\ Proportion = \frac{Dock\ Stem\ Weight}{Dock\ Stem\ Weight + Control\ Stem\ Weight}$) using generalized additive models (GAMs) with the "mgcv" package (Wood 2006) in R. For this analysis, we included sampling locations for which *S. alterniflora* and/or *S. patens* were present in both dock and control samples ($n = 205$ sites; $n = 214$ samples). We first performed a logit transformation on proportion data and performed GAM analyses on the logit-transformed proportions using a Gaussian family of errors with an identity link function. We performed diagnostics on the residuals and did not detect any deviations from normality using the global model with the "gam.check" function in the "mgcv" package in R. We analyzed the full dataset and pooled all live stem data for the dominant species (*S. alterniflora* and *S. patens*). We excluded remaining rare species from GAM analyses as our main questions related to dock impacts on *Spartina* production. Covariates included both dock (height, width, decking spacing, piling spacing, decking type, age, and reciprocal orientation) and environmental (marsh zone, stem $\delta^{15}N$) variables. Based on a positive linear relationship between *S. alterniflora* and *S. patens* stem $\delta^{15}N$ and percent of nitrogen derived from wastewater previously reported in New England estuaries (McClelland et al. 1997; McClelland and Valiela 1998; Martinetto et al. 2006; Wigand et al. 2007), we used control stem $\delta^{15}N$ as a proxy for site eutrophication. *S. patens* stem $\delta^{15}N$ was lower than *S. alterniflora*, consistent with previous findings (e.g., Deegan and Garritt 1997; Wainright et al. 2000; Wigand et al. 2007), so we normalized all site $\delta^{15}N$ values based on *S. patens* with a 0.9‰ increase. This normalization reflects the median difference that we observed between species for sites where we analyzed stem $\delta^{15}N$ from both marsh zones ($n = 6$).

We analyzed all possible models using the "MuMIn" package (Bartoń 2016) in R with estuarine site included in all candidate models as a random effect to account for clustered sampling within individual estuaries. While this approach of evaluating all possible combinations of co-variates has been criticized (Burnham and Anderson 2002), we chose it in the absence of a priori information upon which to develop a smaller set of candidate models. The dock and environmental co-variates included in our models were all based on our original hypotheses regarding potential interactions among dock shading, dock design, and environmental conditions. We ranked models based on Akaike's Information Criterion values corrected for small sample sizes (AIC_c). We also calculated the sum of Akaike weights of models and predictor variables for the subset of models with $\Delta AIC_c < 4$, as all models within this ΔAIC_c range have some support as best models (Burnham et al. 2011). In both cases, Akaike weights range from 0 to 1. Models with higher weights have more support as best models. Variables found in a higher percentage of models will have higher Akaike weight values with variables found in all or none of the considered models having Akaike weight values of one and zero, respectively (Symonds and Moussalli 2011). For the "best" model in each analysis (i.e., $\Delta AIC_c = 0$), we also calculated percent deviance explained and adjusted R^2 to provide an absolute measure of model fits and explanatory power (Galipaud et al. 2014).

Prior to analysis, we examined correlations among our continuous co-variates based on Pearson correlation coefficients calculated using the "cor" function in R. Dock H/W was strongly correlated with height ($r = 0.78$) while height and width had a low correlation ($r = 0.39$), so we included height and width as separate co-variates and omitted H/W. Correlations among remaining continuous predictor variables were negligible ($r < 0.30$; Online Resource 1). We compared our continuous and categorical predictors (decking type and marsh zone) using Mann-Whitney U tests using the "coin" package (Hothorn et al. 2008) in R with Holm adjustments for multiple comparisons due to observed departures from normality among groups. Age and longitude significantly differed ($P < 0.05$) between docks with grating and traditional planking while dock longitude significantly differed between high and low marsh zones ($P < 0.05$). Remaining comparisons were not significant ($P > 0.05$).

We assessed the relationship between light penetration and dock height in relation to dock orientation and decking type following a nested model approach. We used beta regression with the "betareg" function within the "betareg" package (Cribari-Neto and Zeileis 2010) in R with a logit link. Beta regression shares properties with conventional linear models but is able to model variables constrained between 0 and 1 (Cribari-Neto and Zeileis 2010). The simplest model assumed all parameters were the same across dock orientations (north-south ($n = 41$) vs. east-west ($n = 14$)) and decking types (traditional wooden planks ($n = 44$) vs. grated decking ($n = 11$)) ($n = 55$ docks total), the intermediate model assumed orientation-specific parameters, and the full model assumed both orientation and decking-specific parameters. We compared models using a likelihood ratio test with the "lrtest" function in the "lmtest" package (Zeileis and Hothorn 2002) in R. The north-south traditional decking dataset included both private dock data ($n = 12$) and light data from an experimental dock array installed in a Massachusetts estuary ($n = 24$) (Logan et al. 2017).

For cumulative impacts to *Spartina* biomass, carbon, and nitrogen pools, we generated mean, standard deviation, and 95% confidence interval estimates that incorporated propagated error using the "propagate" package (Spiess 2014) in R. This package uses first-/second-order Taylor approximation and Monte Carlo simulation to calculate uncertainty propagation. For biomass change estimates, we used the following equation:

$$\begin{aligned}\Delta Biomass = & (Total\ Dock\ Area \times Biomass_{Control}) \\ & -(Total\ Dock\ Area \times Biomass_{Dock})\end{aligned} \quad (1)$$

where total dock area is the estimated mean of all dock surfaces constructed over salt marsh (m^2), $Biomass_{Control}$ is the mean dry *Spartina* biomass (kg/m^2) among all control sites, and $Biomass_{Dock}$ is the mean dry *Spartina* biomass (kg/m^2) among all dock sites. To estimate changes in carbon and nitrogen pools, we added percent carbon and nitrogen estimates to biomass estimates described in Eq. 1. We included mean and standard deviation for all values using "stat" within the "propagate" package in R.

Results

General Dock and Marsh Characteristics

Docks sampled in our study ($n = 212$) had a variety of design attributes (Table 1). Most docks ($n = 188$) contained plank decking while a smaller sample ($n = 24$) contained grated decking. Average dock age was 15 years. Plank decking docks were older ($\bar{x} = 16\ years$) than docks with grated decking ($\bar{x} = 9\ years$). Dock orientation, pile spacing, and height were highly variable with orientations ranging from 5° to 360°, pile spacing ranging from 124 to 780 cm, and height from 21 to 222 cm. Dock width and spacing between decking boards ranged from 58 to 183 and 0.1 to 4.0 cm, respectively (Table 1).

S. alterniflora and *S. patens* were the dominant species in the low and high marsh regions, respectively (Table 2). These two species accounted for more than 85% of the live stems observed at dock and control sites across the entire survey. *D. spicata* was a secondary species observed mainly in the high marsh (Table 2).

Dock Effects on Light Transmission

Docks reduced light availability relative to controls with the extent of shading varying among dock designs (Fig. 1). Light availability under docks significantly increased with height

Table 2 Mean ± SD proportion of all species observed in dock and control plots based on live stem density

Species	All data		High marsh		Low marsh	
	Control	Dock	Control	Dock	Control	Dock
Smooth cordgrass (*Spartina alterniflora*)	0.60 ± 0.42	0.69 ± 0.41	0.10 ± 0.15	0.39 ± 0.43	0.88 ± 0.19	0.85 ± 0.28
Salt meadow cordgrass (*Spartina patens*)	0.29 ± 0.37	0.18 ± 0.32	0.70 ± 0.29	0.39 ± 0.39	0.06 ± 0.12	0.06 ± 0.19
Spike grass (*Distichlis spicata*)	0.06 ± 0.13	0.07 ± 0.16	0.12 ± 0.18	0.16 ± 0.25	0.03 ± 0.08	0.02 ± 0.06
Common glasswort (*Salicornia* spp.)	0.02 ± 0.09	0.04 ± 0.14	0.02 ± 0.07	0.03 ± 0.11	0.02 ± 0.10	0.06 ± 0.15
Black grass (*Juncus gerardii*)	0.02 ± 0.11	0.02 ± 0.08	0.05 ± 0.17	0.03 ± 0.11	0.00 ± 0.00	0.01 ± 0.06
Sea blite (*Suaeda maritima*)	0.01 ± 0.07	0.00 ± 0.00	0.01 ± 0.07	0.00 ± 0.00	0.01 ± 0.06	0.00 ± 0.00
Sea lavender (*Limonium nashii*)	0.00 ± 0.01	0.00 ± 0.01	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.01	0.00 ± 0.01
Salt marsh bulrush (*Scirpus robustus*)	0.00 ± 0.01	0.01 ± 0.07	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.01	0.01 ± 0.09
Marsh orach (*Atriplex patula*)	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00

(P < 0.001 for all comparisons), and increases varied in relation to dock orientation but not decking type (Fig. 1). Model fit of relationships between light availability and dock height improved with the inclusion of orientation ($\lambda^2(1)$ = 18.84, P < 0.001) but not decking type ($\lambda^2(1)$ = 0.31, P = 0.577). Across heights, total light under docks averaged < 10% of the unshaded control light levels for docks with heights < 50 cm (n = 3) while tallest docks (> 150 cm) averaged > 60% of controls (n = 14).

General Dock Effects on Marsh Vegetation

Spartina vegetation under docks was more patchy than unaltered salt marsh habitat. The average coefficient of variation (CV) among sampling sites was 68.2 and 48.3% for dock and control stem density, respectively. Docks and controls had CV values of 76.2 and 51.1%, respectively, for biomass.

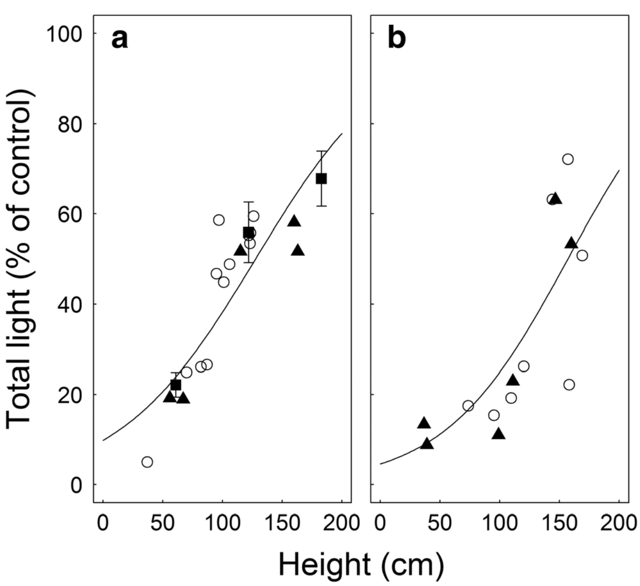

Fig. 1 Total light (% of control) in relation to dock height (cm) for docks oriented **a** north-south and **b** east-west. Data include private docks with traditional plank decking (open circles) and grated decking (solid triangles) as well as experimental docks with traditional plank decking oriented north-south (solid rectangles; mean ± SD of eight docks). Solid lines are best fit lines based on beta regression

Docks had a significant negative effect for combined *S. alterniflora* and *S. patens* stem density (P < 0.001) and standing live biomass (P < 0.001; Online Resource 2). These same patterns held for both high and low marsh samples (P < 0.001 for all comparisons). Pooled stem density and biomass reductions were consistent across zones with dock sites having approximately 40 and 60% of control values for the respective metrics (Table 3).

Maximum stem length was greater under docks for both *S. alterniflora* ($F_{1,10,572}$ = 353.26, P < 0.001) and *S. patens* ($F_{1,2497}$ = 292.31, P < 0.001). Stem length (mean ± 1 standard deviation) was 48.9 ± 13.8 cm (dock 1220 stems) and 43.4 ± 10.0 cm (control 1314 stems) for *S. patens*. For *S. alterniflora*, dock and control stem lengths were 52.6 ± 26.0 cm (4645 stems) and 47.2 ± 23.4 cm (6100 stems), respectively.

Docks had a significant effect on stem elemental composition (Online Resource 2). Dock effects were positive for nitrogen content for both *S. alterniflora* (P < 0.001) and *S. patens* (P < 0.001). Dock effects were negative for carbon content (P < 0.001 for both species) and C/N ratios (P < 0.001 for both species; Online Resource 2).

Dock Effect on High and Low Marsh Zones

Dock effects were also significant for the two *Spartina* species when examined separately, but effects varied by marsh zone (Online Resource 2; Table 3). Docks had significant negative effects for the two *Spartina* species when both marsh zones were combined (P < 0.001 for all comparisons). Relative losses varied between marsh zones for both species with higher percentage reductions for *S. alterniflora* in the low marsh and *S. patens* in the high marsh. For *S. alterniflora*, significant reductions occurred in the low marsh (P < 0.001 for stem density and biomass) but not the high marsh (stem density $P = 0.132$; stem biomass $P = 0.570$). Patterns in relative stem density and biomass were opposite for *S. patens*, with significant negative effects for both metrics in the high marsh zone ($P < 0.001$ for stem density and biomass). In the low marsh, *S. patens* stem density was not significantly affected ($P = 0.054$), and while significant ($P = 0.042$; Online Resource 2), the effect size for stem biomass ($r = 0.20$) was small (Cohen 1988).

Community composition significantly differed between dock and control sites (Table 4). Differences were mainly due to the relative proportions of *S. alterniflora* and *S. patens* observed in each site. *S. alterniflora* was more prevalent in dock sites while *S. patens* made up a higher proportion of control plot live stems. While we observed significant differences in community composition for the entire dataset as well as the two marsh zone subsets in the ANOSIM analysis, correlation coefficients for the entire dataset and low marsh were relatively low. Adonis results were only significant for the full and high marsh datasets. In the low marsh, the proportion of *Salicornia* spp. did increase slightly under docks (6%) relative to controls (2%; Table 2), accounting for approximately 16% of variation, but the most pronounced differences occurred in the high marsh. *S. alterniflora* made up 40% of dock live stems and only 10% of control site stems. *S. patens* in the high marsh instead accounted for about 70% of the stem density at control sites and less than 40% at dock sites. These differences were greater for docks with grated decking, under which *S. alterniflora* was the dominant high marsh species (66%) while *S. patens* only accounted for 18% of live stems (Table 5). In the high marsh, *D. spicata* also showed a small increase under docks (16%) relative to controls (12%; Table 2) and accounted for approximately 17% of the difference in community composition between dock and control samples. Median Bray-Curtis dissimilarity index values for the three datasets ranged from 0.10 (low marsh) to 0.52 (high marsh) with the complete dataset having a median Bray-Curtis value of 0.15 (Table 4).

Effects of Dock and Environmental Characteristics on Aboveground Production

Dock height was included in best candidate models for GAM analyses of both *Spartina* biomass and stem density while other dock characteristics differed in importance among analyses (Figs. 2 and 3; Online Resources 3 and 4). Height was part of all top candidate models (i.e., $\Delta AIC_c < 4$; Online Resources 3 and 4) with predicted *Spartina* proportional biomass and stem density both increasing with dock height (Figs. 2 and 3; Online Resources 3 and 4). Dock age was part of the best model for stem density while the best biomass model instead included decking type. Marsh biomass and stem density both decreased with increasing dock age while biomass was elevated for docks with grated decking relative to traditional plank decking. Age significantly differed between docks with grated and traditional plank decking ($P < 0.05$), and a candidate model including age in place of decking type had a similar AIC_c value ($\Delta AIC_c < 1$; Online Resource 3). Dock reciprocal orientation and pile spacing were both part of the best models of proportional marsh biomass with relative proportions increasing towards a north-south orientation and with greater pile separation. Deck spacing and dock width were absent from all best models and were not part of most top models (Akaike weights < 0.4 for all analyses).

For environmental co-variates, marsh zone was part of the best model for stem density while stem $\delta^{15}N$ was not included in either best model. Proportional stem density varied between marsh zones with higher values in the low marsh. While stem $\delta^{15}N$ was not included in either best model, a stem density model including stem $\delta^{15}N$ in place of dock age had a similar AIC_c value ($\Delta AIC_c < 2$; Online Resource 4), and included a positive linear association between proportional stem density and $\delta^{15}N$. The explanatory power of our GAMs was relatively low with best models having adjusted R^2 of only approximately 0.18 and percent deviance explained of approximately 20%.

Cumulative Impacts

We estimated a total of 18,376.1 ha of salt marsh habitat within the entire state of Massachusetts, with a total of 2673 docks overlying approximately 6.00 ± 0.06 ha of this area (< 0.1% of total area). This current dock build-out results in an average annual loss of *Spartina* aboveground biomass of 2204 ± 3797 kg although the confidence interval for this estimate is wide and includes zero (95% confidence interval − 5229 to 9629 kg). This biomass reduction is equivalent to 367 kg per ha of dock area. Based on this biomass loss estimate, annual average losses of aboveground carbon and nitrogen pools are 933 ± 1534 kg (− 2076 to 3957 kg) and 16 ± 62 kg (− 106 to 140 kg), respectively.

Discussion

Dock Impacts

Consistent with other studies along the US east coast (Kearney et al. 1983; Colligan and Collins 1995; Alexander and Robinson 2004; Sanger et al. 2004; Alexander and Robinson 2006; Vasilas et al. 2011; Alexander 2012), we found that docks in Massachusetts altered a variety of salt marsh vegetation characteristics. Docks reduced available light causing effects to underlying vegetation ranging from individual stem to community scales. *Spartina* vegetation under docks in Massachusetts had lower stem density and biomass, lower stem carbon content, higher stem nitrogen content, taller

Table 3 Percentage of control stem density and biomass of *Spartina alterniflora* and *S. patens*

Species	Number	Stem density		Stem biomass	
		Pooled (%)	Median ± MAD	Pooled (%)	Median ± MAD
All					
Spartina spp.	215	40.8	36.6 ± 20.2	62.3	60.7 ± 24.9
S. alterniflora	181	40.7	43.8 ± 23.5	67.3	69.1 ± 28.3
S. patens	65	50.7	36.8 ± 28.4	50.9	35.5 ± 27.4
High marsh					
Spartina spp.	75	39.6	32.2 ± 20.8	59.5	57.6 ± 28.8
S. alterniflora	42	68.4	77.3 ± 49.8	92.8	120.6 ± 62.8
S. patens	51	45.3	36.8 ± 25.4	47.4	32.4 ± 21.9
Low marsh					
Spartina spp.	140	42.6	40.4 ± 19.9	63.5	62.7 ± 22.0
S. alterniflora	139	36.7	36.0 ± 17.7	63.4	63.2 ± 22.4
S. patens	15	204.7	36.5 ± 29.7	189.4	59.0 ± 52.9

Values are pooled percentages as well as median percentages ± median absolute deviation (MAD). Sample sizes are the number of dock sites where species were present in both dock and control samples. For the combined high and low marsh datasets, nine sites had docks over both zones and percentage estimates include samples from both zones

stems, and a higher proportion of *S. alterniflora* relative to unshaded habitat.

Individual plant level responses were consistent with light limitation. Similar changes in nitrogen content have occurred in *Spartina* (Logan et al. 2017) and seagrass in reduced light conditions (van Lent et al. 1995; Grice et al. 1996; Moore and Wetzel 2000; Peralta et al. 2002) and may be related to reduced growth or disruption of carbon production under low light levels. Elemental composition measurements could provide a rapid assessment tool to evaluate relative shade stress in future studies of structural impacts on vegetation. Increased *Spartina* stem height under docks was consistent with other New England dock studies (Logan et al. 2017; Kearney et al. 1983; Colligan and Collins 1995) and an etiolation response

Table 4 Analysis of similarities (ANOSIM), similarity percentage technique (SIMPER), and multivariate analysis of variance using distance matrices (adonis) results for vegetation community composition analyses of dock and control sites

		All data	High marsh	Low marsh
Bray-Curtis	Median Bray-Curtis index	0.15	0.52	0.11
ANOSIM	R statistic	0.013	0.162	0.006
	P value	0.010*	0.001*	0.033*
SIMPER	*S. alterniflora*	0.44	0.33	0.46
	S. patens	0.34	0.41	0.22
adonis	R^2	0.013	0.124	0.003
	P value	0.009*	0.001*	0.411
	F	5.753	22.41	0.914

*Significant result ($P < 0.05$)

by which plants put more energy towards vertical growth to escape light limitation. In the Mid-Atlantic and southeast USA, researchers noted visual observations in the field of taller *S. alterniflora* stems under docks (Sanger and Holland 2002; Vasilas et al. 2011). Direct measurements showed both no height effect (Alexander and Robinson 2004) and a significant height increase under docks (Alexander and Robinson 2006).

While relative loss of aboveground production was consistent between marsh zones, community-level effects contributed more to high marsh vegetation changes. Docks disrupt the natural competitive hierarchy in the high marsh as the competitive dominant, *S. patens* (Bertness and Ellison 1987; Bertness 1991), was partly displaced by *S. alterniflora* and *D. spicata* in our study and under Mid-Atlantic docks (Vasilas et al. 2011). *Spartina* stem density also declined with dock age, which is consistent with a shift towards lower density *S. alterniflora* stems. As shading causes a reduction of stem density under docks over time, the marsh platform may gradually erode resulting in a lower elevation and more frequent tidal inundation, conditions less suitable for *S. patens* survival. *D. spicata*, which displaced *S. patens* in our study and under both southern New England and Mid-Atlantic docks (Kearney et al. 1983; Vasilas et al. 2011), is able to colonize disturbed, bare patches with vegetative runners (Bertness and Ellison 1987) and could expand to open areas under docks where *S. patens* is thinned out by shading or other abiotic stressors. The slight proportional increase of *Salicornia* spp. under docks in the low marsh in our study is consistent with the same pattern of opportunistic expansion to bare patches created by dock shading since *Salicornia* species

Table 5 Mean ± SD proportion of *Spartina alterniflora* and *S. patens* observed in dock sites with grated and traditional plank decking and respective control plots based on live stem density

Species	Grated decking			Traditional decking		
	n	Control	Dock	*n*	Control	Dock
All						
S. alterniflora		0.59 ± 0.38	0.76 ± 0.38		0.60 ± 0.42	0.68 ± 0.41
S. patens	24	0.29 ± 0.36	0.09 ± 0.22	197	0.29 ± 0.37	0.19 ± 0.33
High marsh						
S. alterniflora		0.16 ± 0.12	0.66 ± 0.43		0.10 ± 0.15	0.36 ± 0.42
S. patens	9	0.71 ± 0.21	0.18 ± 0.33	71	0.70 ± 0.30	0.41 ± 0.39
Low marsh						
S. alterniflora		0.85 ± 0.21	0.83 ± 0.35		0.89 ± 0.19	0.86 ± 0.27
S. patens	15	0.04 ± 0.08	0.03 ± 0.09	126	0.06 ± 0.12	0.06 ± 0.20

are also able to rapidly populate open areas (Bertness and Ellison 1987). While the high marsh is structured mainly through competitive interactions, the low marsh in New England estuaries is dominated by *S. alterniflora* due to its ability to tolerate the physical conditions associated with the more frequent tidal inundations in this zone (Bertness and Ellison 1987). This likely explains the relative consistency in community composition for dock and unaltered sites in our low marsh samples and may also explain the lack of reported species shifts in other studies based in *S. alterniflora*-dominated marshes (e.g., Alexander and Robinson 2004; Sanger et al. 2004).

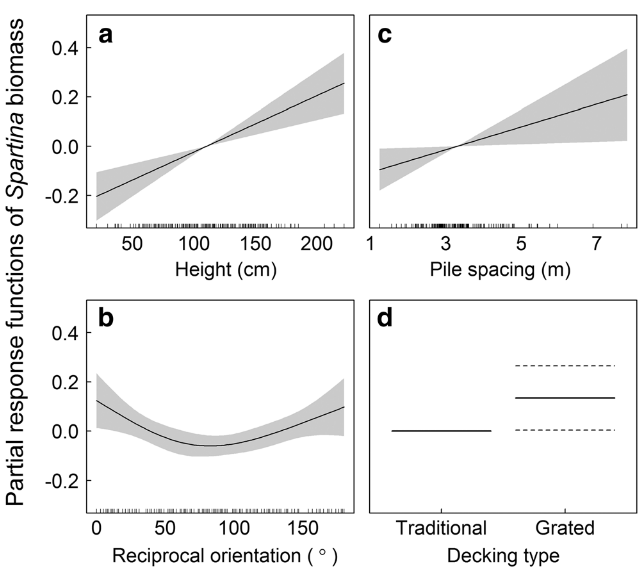

Fig. 2 Partial response plots for the best candidate model of *Spartina* proportional biomass. The best model included **a** dock height (cm), **b** dock reciprocal orientation (°), **c** pile spacing (m), and **d** decking type (traditional planking or grated). Shaded areas (**a**–**c**) and dashed lines (**d**) represent 95% confidence intervals, and the tick marks on the *x*-axis are rug plots showing sampling density

Spartina vegetation under docks in Massachusetts had lower stem density and aboveground biomass than unaltered surrounding marsh habitat. Relative to unshaded control sites, marsh under docks in the southeast USA had reductions in *S. alterniflora* stem density of approximately 30 to 70% (Alexander and Robinson 2004; Sanger et al. 2004; Alexander and Robinson 2006; Alexander 2012). Our Massachusetts docks caused a similar (~ 60%) reduction, suggesting a general consistency across latitudes. *S. alterniflora* biomass alteration under southeast US docks varied among sites with different decking or *Spartina* growth forms and ranged from a 63% reduction to a moderate (23%) increase (Alexander and Robinson 2006; Alexander 2012). Our dock dataset, which consisted mostly of docks with traditional plank decking (Table 1), showed a lower percent biomass reduction relative to stem density impacts similar to the reduction (22%) reported for southeast US docks with traditional decking (Alexander 2012). Our estimated loss of *Spartina* aboveground biomass per unit area due to dock impacts (367 kg dry weight/ha) was lower than estimates that we derived from data collected in Georgia for tall-form *S. alterniflora* (1320 kg dry weight/ha) but similar to estimates for short-form *S. alterniflora* (450 kg dry weight/ha) from the same region (Alexander and Robinson 2006).

Dock Characteristic Effects on Aboveground Production

Light availability and relative *Spartina* production both increased with dock height. While our field light measurements are supported by previous measurements under experimental docks set at varying heights in Massachusetts (Logan et al. 2017) and Georgia (Alexander 2012), the relationship between height and relative vegetation loss is less consistent across studies. Height was the main dock characteristic affecting relative stem density reductions under southern New England docks (Kearney et al. 1983) and significantly affected

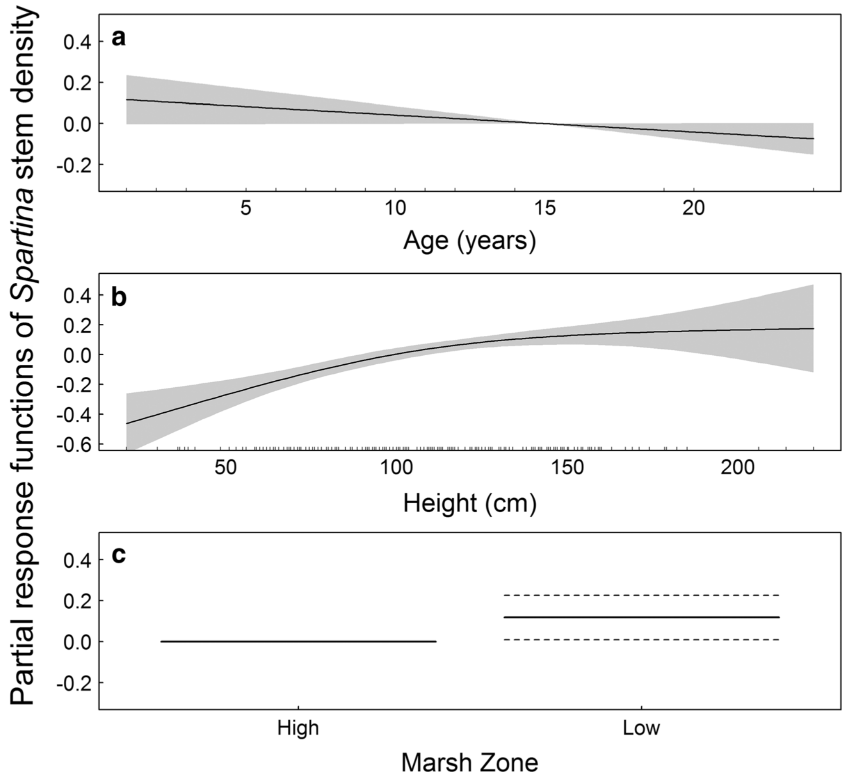

Fig. 3 Partial response plots for the best candidate model of *Spartina* proportional stem density. The best model included **a** dock age (years), **b** dock height (cm), and **c** marsh zone (high or low). Shaded areas (**a**, **b**) and dashed lines (**c**) represent 95% confidence intervals, and the tick marks on the *x*-axis are rug plots showing sampling density

stem density and biomass in experimental docks in Massachusetts (Logan et al. 2017). In the Mid-Atlantic USA, height had no significant effect on vegetation density for private docks (Vasilas et al. 2011), but *H*/*W* was positively correlated with underlying light levels as well as vegetation stem density and aboveground biomass for bridges (Struck et al. 2004; Broome et al. 2005). In the southeast USA, dock *H*/*W* and stem density reductions were only weakly correlated ($r^2 = 0.17$), and the relationship for height alone was not significant (Alexander and Robinson 2004).

Docks oriented north-south had greater light availability and underlying *Spartina* stem biomass than east-west docks. Orientation was not significantly related to relative vegetation losses under docks in the Mid-Atlantic (Vasilas et al. 2011) or southeast USA (Sanger and Holland 2002; Alexander and Robinson 2004; Sanger et al. 2004), despite demonstrated changes in light availability across orientations in the latter region (Alexander 2012). For New England docks, salt marsh vegetation stem height was significantly correlated with dock orientation (Colligan and Collins 1995), consistent with differential shade stress (Logan et al. 2017; Kearney et al. 1983; Colligan and Collins 1995), while orientation was a significant predictor of underlying eelgrass (*Zostera marina*) bed quality (Burdick and Short 1999).

Docks with grated decking had higher relative vegetation biomass than docks with traditional decking, but light availability did not vary between decking types. While biomass was higher under grated decking in our study, docks with grated decking in the southeast USA had biomass reductions of 55–63% compared with a 22% decrease and 1% increase under traditional decking (Alexander 2012). Decking type was not part of our best model for stem density, but stem density reductions under docks in the southeast USA were actually higher for grated (55–58%) relative to traditional decking (31–44%) (Alexander 2012). Detailed analyses of PAR levels under docks with the two decking types in the southeast USA detected only a minimal (< 10%) increase under grated decking during the growing season (Alexander 2012). The biomass effect that we observed coincides with higher proportions of *S. alterniflora* under grated decking relative to traditional decking. This differential species shift is surprising given the lack of a detectable difference in light penetration between decking types, but grated decking may be altering abiotic conditions in other ways that further favor *S. alterniflora* expansion into the high marsh. Our sample size of docks with grated decking was small ($n = 24$), and observed patterns may simply be spurious results. Decking type and age were correlated in our dataset, so the relative impacts of these two variables are difficult to decipher. Since the docks that we sampled with grated decking were younger overall than the traditional decking sample, the docks with grated decking may not have reached an equivalent extent of biomass loss at the time that we sampled them.

Pile spacing was part of the best models predicting relative biomass losses, which is consistent with compounded effects of piling and associated support structures on nearby vegetation. We centered our sampling equidistant between adjacent support piles to standardize methodology across docks. Since we detected a pile spacing effect, even samples collected furthest from the piles in docks with close pile spacing experienced greater biomass loss than docks with wider pile spacing. While we did not collect light level data in relation to pile spacing, increased shading caused by bordering piles and associated cross bracing is a likely cause of observed vegetation declines for docks with close pile spacing. These support structures could also negatively impact bordering marsh biomass by trapping wrack, which can in turn smother underlying vegetation (Bertness and Ellison 1987).

Dock width and decking spacing were not part of our best models, but these attributes were relatively uniform among the docks in our survey and may be influential at further extremes. Width was not a significant predictor of marsh vegetation loss under other New England docks (Kearney et al. 1983; Colligan and Collins 1995) but did affect marsh and eelgrass under docks in the Mid-Atlantic and New England regions, respectively (Burdick and Short 1999; Vasilas et al. 2011). Deck spacing was not significantly related to vegetation loss for docks in the Mid-Atlantic USA (Vasilas et al. 2011) or southern New England (Kearney et al. 1983), but spacing was also fairly constrained for docks sampled in these studies (Kearney et al. 1983; Vasilas et al. 2011). We would expect positive and negative vegetation impacts with increases in deck spacing and width, respectively, beyond the ranges represented by docks in our study.

Environmental Effects on Aboveground Production

Stem $\delta^{15}N$ was part of a top stem density model providing support for an interaction between nitrogen loading and relative dock shading effects. Ambient *S. alterniflora* stem densities decrease with increased nitrogen loading for sites in our study (Logan Unpublish Data) and an experimental fertilization study site in northern Massachusetts (Johnson et al. 2016). Our GAM analysis instead showed a positive trend for relative *Spartina* stem density under docks with increased nitrogen loading. This positive response is likely due to the diminished ambient production in the more eutrophic sites rather than an enhancement of below-dock production but nonetheless suggests that shading and nitrogen loading are not compounding stressors. While the positive linear relationship between percent wastewater contributions to nitrogen loading and *Spartina* $\delta^{15}N$ is well established (McClelland et al. 1997; McClelland and Valiela 1998; Martinetto et al. 2006; Wigand et al. 2007), nitrification and denitrification within estuaries can also influence baseline $\delta^{15}N$ values (Cifuentes et al. 1989; Horrigan et al. 1990). These and other factors may have also contributed to the variability in *Spartina* $\delta^{15}N$ in our dataset.

Model Limitations

Our models of proportional *Spartina* biomass and stem density under docks only explained approximately 20% of the deviance, so other factors not included in our models are also influencing *Spartina* abundance under docks. The information-theoretic approach that we applied in our model selection process provides a mechanism for identifying the best considered models but does not ensure that the best models are robust predictors (Galipaud et al. 2014). Shifts in the relative proportions of *S. alterniflora* and *S. patens* under dock and control plots created additional variability in our pooled *Spartina* estimates of dock impacts since the two species differ in stem density and biomass. Vegetation distribution under docks was less dense than control plots, and our randomized sampling across the full dock width may not have always sufficiently characterized overall biomass and density under the dock footprint. While we included a comprehensive suite of dock characteristics in our GAMs reflective of permitting agency regulations and guidelines, our characterization of environmental conditions was limited to marsh zone as a proxy for elevation and stem $\delta^{15}N$ as a eutrophication proxy. Ambient vegetation stem density and biomass vary as a function of a wide variety of abiotic factors that were not directly accounted for in our models such as soil aeration and salinity (Mendelssohn and Morris 2000). Abiotic conditions may interact with dock shading to amplify or reduce changes in biomass and stem density under dock structures. If much of the unexplained deviance in our models is a product of such abiotic interactions rather than our vegetation sampling protocol, general dock design guidelines may not impart the expected benefits to marsh production in all locations.

Cumulative Impacts

While individual docks only impact a small area of salt marsh habitat, cumulative impacts at the town, embayment, or state level can impart broader loss of marsh production and associated ecosystem services. Estimates of cumulative impacts from dock proliferation were higher in the southeastern USA than in New England. In South Carolina, researchers estimated a cumulative dock coverage of 60 ha over a decadal scale (Sanger et al. 2004) while in Georgia, the cumulative estimates for all docks constructed on or after 1974 was approximately 28 ha (Alexander and Robinson 2006). This 28 ha area caused an estimated loss of 10,000 to 17,000 kg of carbon per year due to associated reductions in *Spartina* aboveground production (Alexander and Robinson 2006). South Carolina and Georgia historically have the highest area of *Spartina*-dominated salt marsh along the US east coast accounting for > 65% of all *Spartina* habitat (Teal 1986). Massachusetts instead accounts for < 1% of east coast salt marsh (Teal 1986), so the higher estimates of cumulative marsh loss from dock proliferation in southeastern US states are consistent with the relative differences in total marsh habitat between regions. At the state level, Massachusetts docks only impact a small (< 0.1) percentage of existing salt marsh habitat similar to the relative impact reported in South Carolina (Sanger et al. 2004), but impacts to individual systems with high dock build out and limited salt marsh habitat will be greater.

Our estimates of cumulative *Spartina* loss have a high degree of uncertainty due to spatial and temporal variability in salt marsh production under different abiotic conditions (Mendelssohn and Morris 2000) and variability in proportional marsh biomass under docks with different designs. We collected all of our data during a single growing season and consequently were unable to account for the high degree of annual variability in salt marsh production. Estimates of annual aboveground production of *S. alterniflora* at an estuary in northern Massachusetts varied widely from 343 to 1324 g per m^2 (Morris et al. 2013). We also estimated ambient production from a single reference site at each dock location and so did not account for variation across marsh zones. In a southern New England estuary, *S. alterniflora* aboveground biomass production estimates varied from approximately 100 to 1100 g per m^2 across different elevations (Culbertson et al. 2008). While our cumulative impact estimates have a high degree of uncertainty, impacts to marsh production that we observed across individual dock structures can translate to a broader reduction in salt marsh ecosystem services when considered at the scale of individual estuaries or coastal regions. Decreases in marsh stem density and biomass limit *Spartina*'s capacity for erosion control, carbon sequestration, and detrital production. The cumulative reduction in aboveground biomass production has the potential to impact higher trophic levels through associated reductions in benthic invertebrate prey (Struck et al. 2004) and diminished nekton production (Kneib 2003; Alexander and Robinson 2006).

Management Implications

For docks constructed over salt marsh in the northeast USA, permitting guidelines and regulations based on dock height or *H*/*W* can reduce shading and associated vegetation loss. In Massachusetts, the recommended 1:1 *H*/*W* guideline (Bliven and Pearlman 2003) reduces vegetation loss, but additional reductions could be achieved with a more conservative (1.5:1) guideline (Logan et al. 2017). Our top model showed a linear increase in proportional biomass with increasing height suggesting continued increases in proportional biomass beyond the 1:1 *H*/*W* threshold given that most sampled docks had widths of approximately 120 cm. Stem density had a more asymptotic relationship with a maximum at a height of approximately 150 cm although our sample size of docks decreased with height beyond this value. Controlled experimental manipulations of dock height also showed a relationship with proportional biomass and stem density (Logan et al. 2017) with docks set at a 180-cm height (1.5:1 *H*/*W*) showing an overall reduction in vegetation loss relative to 120 cm height docks set at the 1:1 *H*/*W* threshold (Logan et al. 2017).

Managers can also reduce individual dock impacts to salt marsh vegetation by requiring maximal pile spacing and a north-south orientation. Engineering requirements for structural integrity will constrain the former while property boundaries will constrain the latter. While dock pathways from the upland area to a bordering waterway may not be modifiable at many property sites, managers can partly address orientation effects by modifying other design characteristics. For docks in New England, Burdick and Short (1999) found that in order to meet light requirements for eelgrass, docks oriented east-west needed to double their height relative to equivalent docks constructed in a north-south orientation. Docks in sites that require an east-west orientation over salt marsh to reach bordering waterways would warrant more conservative height regulations (e.g., ≥ 1.5:1 *H*/*W* or 180 cm) (Logan et al. 2017) given the shading effect of this orientation.

Our results combined with similar findings from docks in the southeast USA do not provide support for the use of alternative decking as the principle means of reducing vegetation loss. Docks with grated decking experience vegetation loss similar to or greater than losses under traditional decking, and light levels under grated decking are still sensitive to height and orientation design parameters. While height and *H*/*W*-based guidelines are best supported by our results, our models had relatively low explanatory power, and high vegetation losses may still occur even when dock design best management practices are implemented.

While design modifications can reduce dock impacts per unit area by reducing shading effects, continued dock proliferation will result in broader marsh losses as even design best management practices tend to impart some degree of marsh loss. Marsh losses associated with indirect effects like dock shading are not typically accounted for in mitigation and habitat restoration requirements in the permitting process. We identified > 2500 docks constructed over an estimated six hectares of salt marsh in the state of Massachusetts. By reducing underlying stem density and biomass, this structural proliferation diminishes a variety of ecosystem services associated with salt marsh including erosion control, habitat, production, and carbon sequestration (Barbier et al. 2011).

Managers can control dock impacts to salt marsh through a combination of broader system-level regulation of dock build-out (total area of impact) and dock design best management practices (impacts per unit area). Cumulative impacts can be approached through system, regional, or statewide assessment of impacts and regulations set at the corresponding scale of impact (MacFarlane et al. 2000; Needles et al. 2015). Dock impacts per unit area can be reduced through regulations with height conditions (> 150 cm), requirements to maximize pile spacing within engineering constraints, and, where feasible, installation in a north-south orientation. Grated decking designs assessed in our survey did not improve light penetration or overall vegetation production and are not adequate substitutes for height-based regulations. Docks in Australia with aluminum mesh decking had reduced seagrass impacts relative to traditional wooden docks (Gladstone and Courtenay 2014), providing some support for benefits of alternative decking designs and/or regional variability in design efficacy. Dock orientation is often pre-determined by site conditions, and height or H/W ($\geq$ 1.5:1) (Logan et al. 2017) best management practices may not always be practical. Given these constraints, coastal engineers should continue to explore creative new decking designs to develop structures that increase light availability.

Acknowledgements We would like to thank Holly Williams, Alina Arnheim, Drew Collins, David Behringer, Christian Petitpas, Tay Evans, Katelyn Ostrikis, Jillian Carr, and Wesley Dukes for their assistance with field sampling. Vincent Manfredi provided equipment for grass measurements. Tara Rajaniemi (University of Massachusetts—Dartmouth) provided lab facilities for sample preparation. Brad Hubeny and the staff at the Salem State Viking Environmental Stable Isotope Lab performed isotope and elemental analyses.

Funding Information This work was funded by a grant from the Massachusetts Bays National Estuary Program (MassBays).

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