Received: 30 August 2023 | Revised: 1 April 2024 | Accepted: 3 April 2024

DOI: 10.1111/cobi.14301

Conservation Biology

CONTRIBUTED PAPER

Quantifying the impacts of future shoreline modification on biodiversity in a case study of coastal Georgia, United States

Daniel J. Coleman[1] | **Rachel K. Gittman**[2] | **Craig E. Landry**[1] | **James E. Byers**[1] | **Clark R. Alexander**[1,3] | **G. Paul Coughlin**[1] | **C. Brock Woodson**[1]

[1]Institute of Resilient Infrastructure Systems, University of Georgia, Athens, Georgia, USA

[2]Department of Biology, East Carolina University, Greenville, North Carolina, USA

[3]Skidaway Institute of Oceanography, University of Georgia, Savannah, Georgia, USA

Correspondence
Daniel J. Coleman, University of Georgia, 211 ISTEM2, 302 East Campus Road, Athens, GA 30601, USA. Email: Daniel.Coleman@uga.edu

Article impact statement: Shoreline armoring leads to a decline in biodiversity, but adopting natural infrastructure can mitigate or reverse that decline.

Funding information
U.S. Army Corps of Engineers; Engineering With Nature Cooperative Ecosystem Studies Unit Agreement

Abstract

People often modify the shoreline to mitigate erosion and protect property from storm impacts. The 2 main approaches to modification are gray infrastructure (e.g., bulkheads and seawalls) and natural or green infrastructure (NI) (e.g., living shorelines). Gray infrastructure is still more often used for coastal protection than NI, despite having more detrimental effects on ecosystem parameters, such as biodiversity. We assessed the impact of gray infrastructure on biodiversity and whether the adoption of NI can mitigate its loss. We examined the literature to quantify the relationship of gray infrastructure and NI to biodiversity and developed a model with temporal geospatial data on ecosystem distribution and shoreline modification to project future shoreline modification for our study location, coastal Georgia (United States). We applied the literature-derived empirical relationships of infrastructure effects on biodiversity to the shoreline modification projections to predict change in biodiversity under different NI versus gray infrastructure scenarios. For our study area, which is dominated by marshes and use of gray infrastructure, when just under half of all new coastal infrastructure was to be NI, previous losses of biodiversity from gray infrastructure could be mitigated by 2100 (net change of biodiversity of +0.14%, 95% confidence interval −0.10% to +0.39%). As biodiversity continues to decline from human impacts, it is increasingly imperative to minimize negative impacts when possible. We therefore suggest policy and the permitting process be changed to promote the adoption of NI.

KEYWORDS
armoring, biodiversity, natural infrastructure, shoreline modification, wetlands

INTRODUCTION

Shoreline modification is the common practice of altering coastal ecosystems to augment particular services, such as the mitigation of coastal erosion and protection against storm surge and waves (Dugan et al., 2011; Kittinger & Ayers, 2010). Conventional approaches, most of which can be classified as gray infrastructure, are often focused on near-term property protection but often entail sacrifice of ecological services, such as wildlife habitat and water quality maintenance, associated with functioning natural systems (Arkema et al., 2013; Dugan et al. 2018). Gray infrastructure, also called *shoreline armoring* or *hardening*, utilizes common construction and engineering techniques and can take the form of bulkheads, seawalls, and revetments. These approaches often give little to no consideration to habitat quality or other natural services. In contrast, natural infrastructure (NI) (or green infrastructure) uses natural processes and ecosystem services, often in conjunction with some type of built components (i.e., hybrid approach), to meet engineering objectives (Institute for Resilient Infrastructure Systems, 2024; Smith et al., 2020). NI along coasts includes living shorelines, salt marshes, dunes, and reefs and can provide protections similar to gray infrastructure (Sutton-Grier et al., 2015).

There are numerous benefits to using NI instead of gray infrastructure. For example, NI can be cheaper and easier to maintain, adaptive to climate change, and less harmful to the environment (Gittman et al., 2014; Mitchell & Bilkovic, 2019; Narayan et al., 2016; Sicangco et al., 2021; Sutton-Grier et al., 2015). Additionally, NI often offers more cobenefits or services in addition to the primary purpose of hazard protection than gray infrastructure. For example, living shorelines designed to mitigate wave impacts have been shown to provide nursery habitat for commercially important fisheries (Gittman et al., 2016; Scyphers et al., 2011). Alternatively, utilizing gray infrastructure often leads to environmental degradation and loss of ecosystem services (Balouskus & Targett, 2016; Jaramillo et al., 2021; Peterson et al., 2019). For example, gray infrastructure can negatively affect species abundance and biodiversity (Sobocinski et al., 2010; Toft et al., 2021).

Although shoreline modification can alter a whole host of ecological functions, biodiversity is often used as a proxy for overall ecosystem functioning. Biodiversity is relatively easy to measure and is consistently reported in empirical studies, but it is in decline globally (Butchart et al., 2010; Habibullah et al., 2022; Hooper et al., 2012). There are numerous factors contributing to the global decline in biodiversity (Williams-Subiza & Epele, 2021; Jaureguiberry et al., 2022), including local (direct) human impacts, such as the construction of a dam (Jumani et al., 2018), and global (indirect) impacts, such as human-induced climate change (Trisos et al., 2020). Certain global impacts can also prompt additional local impacts. For example, rising sea levels and increased intensity and severity of coastal storms create a greater impetus for shoreline modification (McNamara & Keeler, 2013).

The way in which society addresses the pressure for shoreline modification may have significant impacts on coastal biodiversity (Gittman et al., 2016). If this pressure is met with environmentally unsustainable practices, such as widespread installation of gray infrastructure, this could result in greater declines in biodiversity and ecosystem function. In other words, biodiversity will likely continue to decline due to indirect human impacts (Trisos et al., 2020), which could then trigger greater direct impacts. If the demand for shoreline modification is met with more green and hybrid infrastructure, however, it could be possible to manage shoreline risks in a sustainable manner that avoids synergistic negative impacts on biodiversity (Feagin et al., 2021).

We analyzed the balance of gray and green shoreline infrastructure projects that could provide shoreline protection while minimizing potential negative impacts on coastal biodiversity. Furthermore, we explored the prospective magnitude of green infrastructure projects that could allow reversal of negative biodiversity impacts of shoreline modification that have already occurred. To these ends, we estimated the relationships among gray infrastructure, NI, and biodiversity by analyzing the results of previously published studies. We specifically focused on NI that could be considered hybrid structures (i.e., they contain both an anthropogenic component and a natural or nature-based feature). We used geospatial data produced from a temporal sequence of armored shoreline surveys (Alexander, 2010, 2016, unpublished data 2021) and US Fish & Wildlife Service's National Wetland Inventory (NWI) to develop a model to project future shoreline modification in coastal Georgia (United States). We applied the model to analyze various mixes of gray and green infrastructure to predict future changes in biodiversity under different infrastructure investment scenarios.

METHODS

Literature-derived estimates of infrastructure impact

To determine how different infrastructure types affect biodiversity, we built on an existing meta-analysis by Gittman et al. (2016). Because of the substantial interest in shoreline modification—particularly NI—since 2016 (Smith et al., 2020), we included additional studies up to 2023. To gather these additional studies, we conducted a systematic search of the Google Scholar with search terms for structure type (*seawall* OR *bulkhead* OR *riprap* OR *breakwater* OR *sill* OR "*natural infrastructure*" OR *NNBF* OR "*green infrastructure*"), response metric (*richness* OR *diversity* OR *biodiversity*), and shoreline modification indicators ("*shore* hard**" OR "*shore* armor**" OR "*shore* stabilization*" OR "*shore protection*" OR "*shore* modification*" OR "*coast* modification*"). We supplemented this search with a more targeted approach, such as searching the references of relevant studies. We were interested in NI designs that contained both an anthropogenic built component and a natural ecosystem component. As such, we excluded studies that represented pure environmental restoration and ecoengineered seawalls (i.e., seawalls with slight design alterations aimed at promoting colonization of fouling organisms but do not include natural habitat). As in Gittman et al. (2016), studies were added only if they included data for both a shoreline modification structure and a natural ecosystem. This paired approach prevents comparing shoreline modification structures studied by one research team with a particular methodology to a natural ecosystem studied by a different research team with a different methodology.

We were interested in biodiversity metrics and thus excluded other metrics (such as abundance) from our analyses. This resulted in 20 studies from the original meta-analysis plus 18 additional studies from the 2016–2023 search. Following Gittman et al. (2016), we extracted means, standard deviations, and sample sizes of ecological community metrics (such as richness and diversity) for each shoreline structure and natural ecosystem comparison. Within a study, values were pooled for shoreline structure and ecosystem classification and kept separate for habitat-use group: flora, benthic infauna, birds, epibiota, and nekton (subdivided into fishes and mobile macroinvertebrates where possible). From the 38 studies, we extracted 98 unique comparisons (e.g., species richness of birds from seawalls vs. sandy beaches from study 18 [Appendix S1]). These comparisons mainly included species counts (i.e., species richness) and biodiversity indices (e.g., Shannon–Wiener diversity). For all metrics, higher values indicated higher diversity (Appendix S1).

Conservation Biology

To predict the effect of shoreline modification, we created proportional response probability distribution functions. First, we calculated the proportional change in the biodiversity metric between the gray infrastructure site and the corresponding natural site as:

$$P_{\Delta C} = \frac{\text{structure} - \text{natural}}{\text{natural}}. \quad (1)$$

The comparisons were then grouped based on structure and natural ecosystem. The structure categories were armored (which included rip-rap revetments and seawalls or bulkheads) and NI (all of which were hybrid structures, such as living shorelines). For ecosystem categories, we divided the studies into biogenic coasts (e.g., mangroves and marshes), sandy beaches, and rocky shorelines. We then created probability density functions (PDFs) of the proportional changes for each pairing of the 2 structural categories (armored or NI) in each of the 3 environmental contexts (biogenic coast, sandy beach, rocky shore).

Present and future shoreline modification

We created a shoreline composed of standardized segments as the framework for our geospatial analyses. We first extracted the shoreline from the NWI (US Fish &Wildlife Service, 2023) by determining the boundary that separated marine and estuarine polygons from dry land polygons. Using ArcGIS, we automatically subdivided the shoreline into 50-m segments, which represented shoreline frontage for hypothetical waterfront properties. All shoreline segments were contiguous, and coastline sections that could not accommodate at least 3 segments (i.e., islands with a perimeter <150 m) were excluded. When subdividing the shoreline results in a segment <25 m, those segments were joined to an adjacent segment. Segments ≥25 m remained independent. This process resulted in 68,900 coastline segments, 98.7% of which were 50 m and all of which ranged from 25 to 75 m. We used artificial segments instead of property parcels to simplify our analyses, but future research could incorporate property and social data to enhance the behavioral and policy elements of the study (Peterson et al., 2019).

We defined key parameters for the shoreline segments with existing geospatial data sets. To quantify shoreline infrastructure, we used data mapped for the state of Georgia in 2006, 2012, and 2018 by Alexander (2010, 2016, unpublished data 2021) (Figure 1). We buffered the segments by 25 m inland and 25 m into the water to account for discrepancies in location between the NWI-derived shoreline and the shoreline implicit to the modification data sets. We then determined the length of modified shoreline that fell within the segment. For each shoreline segment, we determined whether it contained NI or gray infrastructure in 2006, 2012, or 2018; whether the neighboring shoreline segments contained infrastructure in 2006, 2012, or 2018; mean slope; and adjacent ecosystem. If a shoreline segment contained a modified shoreline with a length ≥25% of the segment length, it was coded as modified. Knowing shoreline segment modification status and the status of adjacent segments allowed modeling of neighboring parcel effects, which are a strong driver of shoreline modification patterns (Gittman et al., 2021; Peterson et al., 2019; Scyphers et al., 2015). To determine adjacent ecosystem, we associated the shoreline segments with the NWI code for the adjacent marine or estuarine NWI polygon. The NWI codes were grouped as marsh, beach, or other (predominantly open water or scrub–shrub land cover). There were no NWI polygons in the study region that represented rocky shorelines.

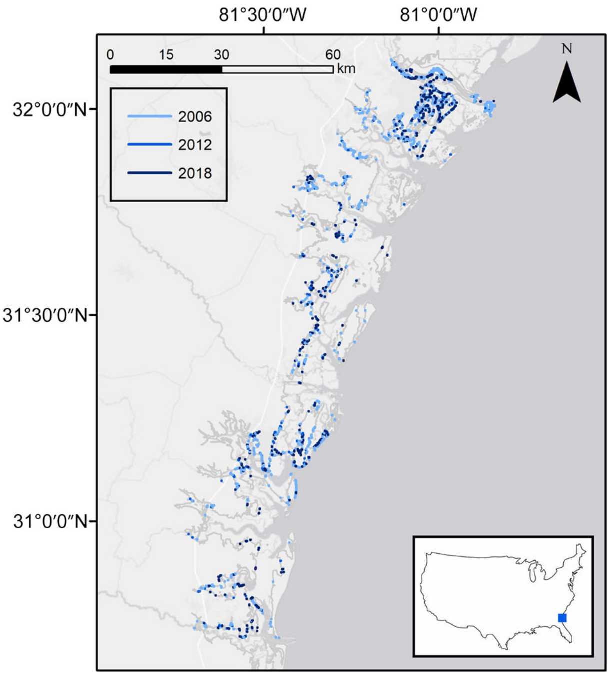

FIGURE 1 Study area and location of shoreline modification in a given year (inset, position of the Georgia coast in the United States).

To verify the importance of neighboring segment modification status, we developed linear regression models. Specifically, we included the presence of shoreline modification at a neighboring shoreline segment in the previous survey year as independent variables and whether a segment became modified from one time step to the next as the dependent variable. We developed one linear regression for the presence of new shoreline modification from 2006 to 2012 and one for 2012 to 2018. The models for 2006–2012 and 2012–2018 are very similar; both were statistically significant in their explanation of approximately one half of the variability (2012: $p < 0.001$, $R^2 = 0.51$; 2018: $p < 0.001$, $R^2 = 0.51$).

We created a probabilistic, decision-tree model to predict the location of shoreline modification in the future (Appendix S2). For each time step (6 years), we assigned a random number from the unit interval (0–1) to all unmodified shoreline segments that was then compared with the corresponding probability of modification. To derive these probabilities, we first classified all shoreline segments based on adjacent ecosystem type and presence of neighboring armoring in the previous survey. For each classification, we divided the number of shoreline segments that were modified in 2012 but not in 2006 by the total number of shoreline segments in that class in 2012. We repeated this for shoreline segments modified in 2018 but not 2012 and averaged the values for each time step. If the random number was below the probability of modification, the shoreline segment became modified in that time step. Once a shoreline segment became modified, it could not return to an unmodified state. We ran the model for 14 time steps (from 2018 to the year 2102) and repeated the model run 50 times to determine uncertainty and variability in model predictions. From the 50 runs, we calculated the mean proportion of the shoreline that was armored in each time step and the 95% confidence interval (mean and 2 SD).

Modeling biodiversity changes

To simulate the impact of shoreline modification on biodiversity, we applied the PDFs derived from the quantitative literature review to the model of future shoreline modification. When a shoreline segment was first modified, it was randomly assigned a proportional change value from a cumulative density function (CDF) derived from the PDF for either NI or armored of the corresponding ecosystem type. Specifically, shoreline segments in the marsh or other categories were assigned values from the biogenic CDFs, and shoreline segments in the beach category were assigned values from the beach CDFs. This was done such that the distribution of proportional change values for the modified shoreline segments would correspond to the PDFs derived from the quantitative literature review for a given ecosystem and infrastructure category. Because the impact of shoreline modification projects that were previously built was not measured, segments modified prior to 2019 were assigned proportional change values in the same manner.

We ran the model with different ratios of NI to gray infrastructure to determine whether biodiversity losses could be mitigated or reversed. We ran separate simulations for scenarios in which 100% of new modification was armoring or gray infrastructure and then varied the proportion of gray to NI as follows: 80:20; 60:40; 40:60; 20:80. The final simulation consisted of 100% NI. At the time of our study, there were only a few small-scale, living shorelines in coastal Georgia (Georgia Department of Natural Resources, 2013). They represented <0.1% of modified shoreline segments. As such, all shoreline modification projects from the 2006, 2012, and 2018 data sets were considered gray infrastructure for all scenarios. Variability between runs was produced via randomization in determining whether a segment became modified and had the corresponding (randomized) impact on biodiversity. Each scenario was run 50 times to determine consistency between runs, mean change in biodiversity for each time step, and the 95% confidence intervals. We also investigated the results of a more gradual adoption of NI. For this, we repeated our analyses with a proportion of NI to gray infrastructure that changed with each time step rather than a fixed ratio. Beginning with 0% NI and 100% gray infrastructure in 2018, each subsequent time step had x% more NI and x% less gray infrastructure, continuing until the year 2102 or 100% NI. We tested multiple values of x to determine which, if any, resulted in approximately 0% change in biodiversity by 2102.

RESULTS

Shoreline modification impacts in the literature

With the exception of biogenic coasts (NI n = 19, armored n = 27), there were considerably more data for armored coastlines than NI (beaches: NI n = 4, armored n = 21; rocky coasts: NI n = 5, armored n = 22) (Figure 2). For beaches and rocky coasts, the small sample sizes for NI obscured statistical comparisons with armored shorelines. Fortunately, having less resolved values for beaches and rocky coasts was a minor problem because beaches accounted for only 2.6% of shoreline segments in our study location and rocky coasts were absent entirely. For biogenic coasts, the impacts of NI and armoring were significantly different (2-sample Kolmogorov–Smirnov test, $p < 0.001$). Approximately two thirds (26 out of 38) of the studies were conducted in North America (most represented continent in the coastal NI literature [Smith et al., 2020]).

Armored coastlines tended to have lower biodiversity than their natural ecosystem counterparts. This pattern was consistent for beaches, biogenic coasts, and rocky coasts (Figure 2d–f). Armored shorelines exhibited biodiversity metrics that were on average 26% (SE 6) lower than unmodified beaches. Of the 21 comparisons between armored shorelines and beaches, only 3 showed a positive difference in biodiversity. Compared with unmodified biogenic coasts, armored coastlines had biodiversity metrics that were a mean of 18% (4) lower; 8 of 27 comparisons had higher biodiversity. Finally, armored shorelines had biodiversity metrics a mean of 11% (12) lower than unmodified rocky coasts; only 3 of 22 comparisons had higher biodiversity at the armored site.

The impact of NI on biodiversity was different from that of armored shorelines. For NI, biodiversity metrics were a mean of 37% (SE 13) higher than unmodified biogenic coasts; 14 of 19 comparisons had higher biodiversity measures (Figure 2a–c). There were only 4 comparisons between NI and beaches and 5 between NI and rocky coasts. For NI, biodiversity metrics were a mean of 6% (6) lower than unmodified beaches and 43% (8) lower than unmodified rocky coasts. One of the 4 comparisons between NI and beaches had higher biodiversity for NI, and there were no comparisons between NI and rocky coasts in which there was higher biodiversity for NI.

Present and future shoreline modification

Shoreline segments were more likely to become modified if there was an adjacent modified shoreline for all ecosystems. The probability for a marsh-adjacent shoreline segment with a modified neighbor to become modified was the greatest at 36.4%, followed by the other ecosystem category at 25.6%. The probability of a beach-adjacent shoreline segment with a modified neighbor was notably lower, at 7.59%. The

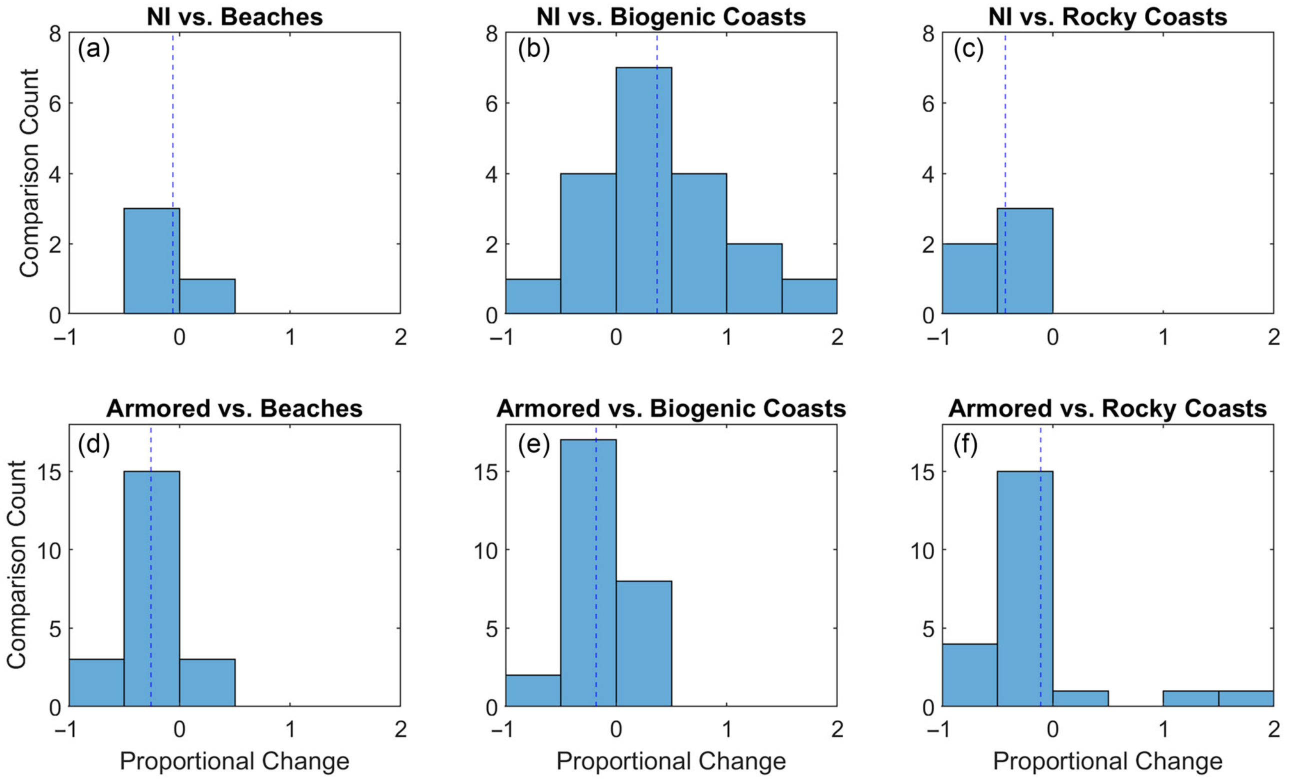

FIGURE 2 Probability distributions of comparisons between the proportional change in biodiversity metrics under (a–c) natural infrastructure (NI) and (d–f) armored shoreline modification versus a given natural ecosystem type (dashed line, mean proportional change of the distribution; 0 proportional change, biodiversity of the modified shoreline not different from the corresponding natural ecosystem; −1 proportional change, complete loss of biodiversity with the artificial structure; +1 proportional change, increase in biodiversity of 100%).

probability of modification without a modified neighbor was much lower; other ecosystems had the greatest probability at 1.11%, followed by marshes at 0.49%, and beaches at 0.20%. Despite there being far more shoreline segments without a modified neighboring shoreline (2012: 6131 with a modified neighbor compared with 62,769 without; 2018: 6951 relative to 61,949), the majority of new shoreline modification occurred at shoreline segments with a modified neighbor (2012: 1467 with a modified neighbor compared with 378 without; 2018: 1658 relative to 385).

The proportion of modified shoreline increased rapidly from 2006 to 2018 and continued to do so during the time frame of our model. In 2006, 5.9% of the total coastline of Georgia was composed of modified shoreline segments (Figure 3) (6.7% and 7.5% in 2012 and 2018, respectively). By 2102, the percentage was predicted to increase to 46.7% (95% confidence of 45.9–47.5%) (Figure 3). This increase in shoreline modification was based solely on observed trends from 2006 to 2018 and did not take into consideration how future change, such as sea-level rise, may alter the rate of shoreline modification.

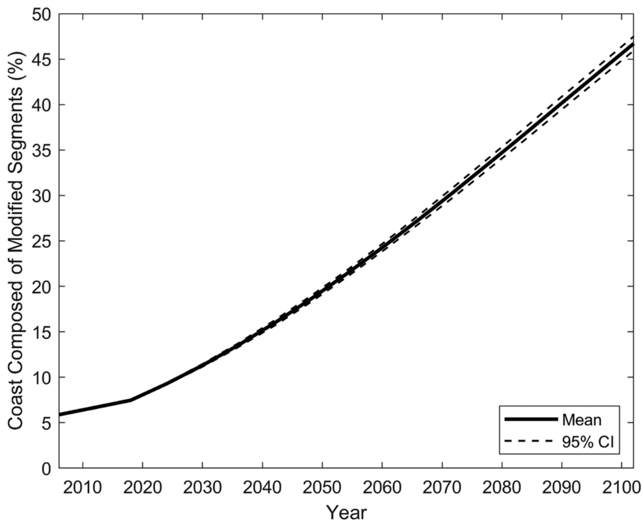

FIGURE 3 Percent of the length of Georgia coast predicted to be composed of modified shoreline segments. Rate of increase in the time frame is exponential.

Biodiversity changes

The overall impact on biodiversity from shoreline modification was heavily dependent on the percentage of new modification that is NI. The shoreline modification that occurred through 2018 was estimated to have changed biodiversity by −1.51% (95% confidence interval −1.47% to −1.55%) (Figure 4). Continuing the current trend of approximately 0% of new

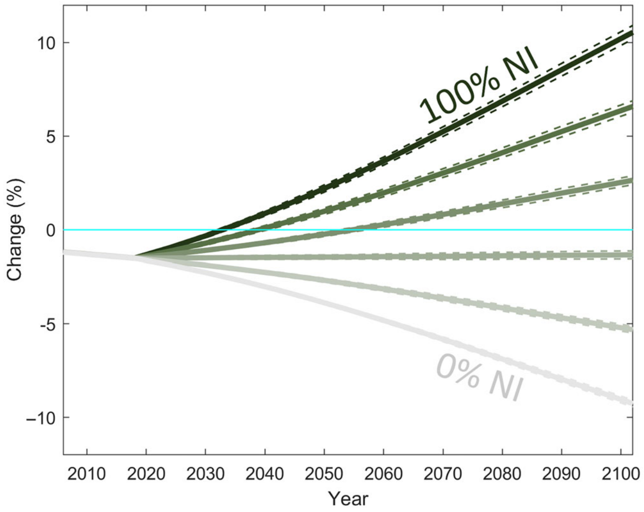

FIGURE 4 Percent change in biodiversity under increasing use of natural infrastructure (NI) shoreline modification (light gray line, continuance of the proportion of infrastructure type from 0% NI to 100% gray infrastructure [i.e., armoring]; from bottom to top, each line 20% increase in new NI; dashed lines, 95% confidence intervals; blue line, 0% change in biodiversity in unmodified coast). All scenarios have 0% NI and 100% gray infrastructure for 2006–2018 to represent historic conditions.

modification being NI by 2102, biodiversity was predicted to change by −9.26% (−9.07 to −9.44) from the premodification coastline (Figure 4). Alternatively, when all new modification was NI, biodiversity in 2102 was predicted as 10.5% (10.2–10.9%) greater than the premodification coastline (Figure 4). When approximately 47.5% of all new modification was NI, there was no estimated net impact on biodiversity from shoreline modification (+0.14% [−0.10% to +0.39%]) by 2102 (Appendix S3). For the gradual adoption of NI scenarios, when the proportion of NI increased by 6% every time step (i.e., new shoreline modification from 2018 to 2024 6% NI and 94% gray, from 2024 to 2030 12% NI and 88% gray, and so on until new shoreline modification from 2096 to 2102 was 84% NI and 16% gray), the net impact to biodiversity from shoreline modification was approximately zero by 2102 (Appendix S3).

DISCUSSION

The staggering loss of biodiversity is fueled by human activities, both indirect and direct (Ceballos et al., 2015). Indirect human impacts, such as warming temperatures and ocean acidification, are often global in scale (Trisos et al., 2020; Turley & Gattuso, 2012). These factors will likely continue to have some effect on biodiversity into the future despite potential changes in actions and policies (Trisos et al., 2020; Turley & Gattuso, 2012). Shoreline modification represents a direct human impact on coastal biodiversity, others of which include overfishing, flow modification, and environmental contamination (Williams-Subiza & Epele, 2021). The extent to which these factors affect biodiversity in the future is directly related to future actions and policies. Efforts made to mitigate direct human impacts can have immediate effects on biodiversity and are therefore especially pertinent (Jaureguiberry et al., 2022).

Our results indicated that losses in biodiversity caused by shoreline modification can be mitigated and even potentially reversed through the implementation of NI. Furthermore, mitigating biodiversity losses does not require a complete (100%) shift from gray infrastructure to NI. If just under one half of all new shoreline modification projects were NI, coastal Georgia could essentially reach net zero changes in biodiversity compared with an unmodified shoreline by 2102. Furthermore, adopting 6% more NI (and 6% less gray) each time step resulted in approximately net zero changes in biodiversity compared with an unmodified shoreline by 2102, but mandates that 84% of shoreline modification added in the final time step (2096–2102) be NI.

As with all models, there are inherent assumptions that simplify the complexity of real-world systems. Our work is based on the assumption that NI is suitable for all coastline segments in our study area. Some segments may be physically unsuitable for NI (Nunez et al., 2022), which could ultimately limit the implementation of NI. We also assumed that the changes in biodiversity are spatially and temporally independent at a given segment; the impact of modification at one segment does not affect biodiversity at another segment. Similarly, we determined the overall percent change in biodiversity to be the average of all segments. Interaction between segments, such as synergistic or compensatory processes, could result in biodiversity changes different from our predictions. We did not differentiate between species richness or biodiversity metrics, the latter of which can capture ecologically important factors, such as relative abundances and evenness. By measuring biodiversity change in percentage terms (Equation 1), we essentially limited the degree of negative change to 100%, but there was no limit to positive change. Because the PDFs were unbounded in the positive direction, our model may be biased toward the positive effects of infrastructure, especially NI because it has the most potential for a positive effect. We believe this is acceptable, however, because it mimics the possibilities of actual infrastructure projects. For instance, if a hypothetical ecosystem has 10 species, it is only possible to lose 10 species (limit of −100%), but it is possible to gain any number of species (no natural upper bound). Therefore, we assumed that there is an unlimited pool of biodiversity that can be added to the system. If there are a limited number of species capable of migrating to new NI projects because of extinction or extirpation, dispersal barriers, a depleted regional pool of biodiversity, and so forth, then the positive impact of NI predicted here may be unrealistic.

The adoption of NI is a powerful tool in mitigating biodiversity losses because many studies have found NI to have greater biodiversity than the natural counterpart. This is likely because most NI designs increase habitat heterogeneity by replacing a single natural ecosystem with multiple engineered ecosystems. For example, a common living shoreline design implemented at eroding salt marshes is that of a biomimic structure that enhances oyster recruitment directly in front of a restored portion of the marsh (Davenport et al., 2018; Guthrie et al., 2022). In this example, the living shoreline consists of 2 distinct coastal habitats: a hard substrate and a salt marsh. The influx of organisms that rely on hard substrates, such as oysters, thus increases the biodiversity associated with the living shoreline in comparison to the natural marsh. Increases in biodiversity associated with marshes could also be due to increases in the magnitude of habitat (a pure area effect). In either situation, biodiversity would be preserved and enhanced. This contrasts with shoreline armoring approaches, which often remove or entirely replace existing habitats (Currin et al., 2010). It is also important to note that not all NI has a beneficial impact. Nearly one half of comparisons of NI to natural habitats show a negative impact on biodiversity from NI (13 out of 28 comparisons), and these potential effects were accounted for in our simulations. Interestingly, this positive impact of NI is driven almost entirely by NI installed on biogenic coasts. We excluded rocky coasts from our study due to their absence in the Georgia coast. There are very few studies that compare NI with unmodified beaches, and they report a small negative impact, although not significantly different from zero. Furthermore, 97% of the shoreline segments are biogenic coast.

In addition to the numerous factors not included in our model that may limit the possible positive impact of NI, biodiversity does not always represent a positive change for the ecosystem. Invasive species are often the first organisms to inhabit new or disturbed environments (Byers, 2002), and we did not directly address the impacts of invasive species on changes in biodiversity. The presence of invasive species can increase biodiversity (Lean, 2021), but may hinder other ecosystem services (Coleman et al., 2022; Lean, 2021). Of the 38 studies that ultimately had a comparison included in our analyses, 7 (approximately 18%) explicitly documented the presence of an invasive species at a modified location. In some cases, invasive species can lead to higher biodiversity at gray infrastructure sites compared with their natural counterpart (Vaselli et al., 2008). Although the impact of shoreline modification on invasive species is beyond the scope of this work, it raises the question of whether or not net zero changes in biodiversity compared with an unmodified coast translate to a net zero change in other ecosystem services. For instance, it is unclear how NI may differ from natural systems with regard to habitat quality, the sizes of populations a given area can support, cultural value, or support for interconnected systems.

Our results are based off current available data for both the rate of new shoreline modification and the impact of structures on biodiversity. As sea level continues to rise and the risk of storms increases, there will likely be greater rates of shoreline modification (McNamara & Keeler, 2013). The difference in impact between gray infrastructure and NI becomes more meaningful as more of the shoreline becomes more intensely modified (see Figure 4). Furthermore, it is possible that the impact of converting a natural ecosystem to either gray infrastructure or NI may change. For example, improvements in NI design could further enhance their effect on biodiversity. Species that currently can inhabit a particular type of modification structure may not be able to persist if they are experiencing other stressors, such as warmer temperatures. Therefore, our estimates were based on observable trends and patterns and would benefit from updated information because these trends and patterns will change over time (through improved designs and continued climate change).

Given increasing concerns about environmental change, sustainability, and resilience, there is growing interest in promoting coastal NI across the globe. Ecological design principles can be integrated into marine infrastructure projects to promote cobenefits (Cameron & Blanuša, 2016; Dafforn et al., 2015; Dennis et al., 2018), but more research is needed to improve and test scientific understanding of various aspects of effectiveness of NI and to gain better insights into human perceptions and social acceptance (Evans et al., 2017, 2019; Morris et al., 2016, 2019; Smith et al., 2020). Research into best practices in coastal NI should also seek a holistic approach to benefits assessment across terrestrial, freshwater, and marine domains (Lowe et al., 2022) and should take care to avoid green washing of widespread coastal development (Firth et al., 2020).

Despite intrepid policy goals for climate change adaptation, coastal restoration, and biodiversity conservation in the United States, the European Union, Australia, and other parts of the world, there still exist significant barriers to adoption of coastal NI (Evans et al., 2019; Morris et al., 2019). Many people fail to recognize how marine ecosystems support ecological services and how environmental degradation due to shoreline armoring diminishes sustainability, resilience, and overall social welfare (Elwell et al., 2018; Evans et al., 2019; Lau et al., 2019; Scyphers et al., 2020). There are also legislative and regulatory barriers (Bellantuono, 2014; Dhakal & Chevalier, 2017; Morris et al., 2019). In many US states, including Georgia, it is more difficult to obtain a permit for NI relative to conventional hard armoring. Although research continues into the feasibility, design, and performance of NI in coastal Georgia (Bliss et al., 2014; Bugbee, 2020), many regulators and property owners have been concerned about unknown or unproven effects (Smith et al., 2020). Moderate investments in demonstration projects may be effective in ameliorating community misperceptions of efficacy and benefit of coastal NI (Evans et al., 2019; Morris et al., 2019; Sutton-Grier et al., 2015), and hybrid approaches can help address multiple objectives of coastal infrastructure (Evans et al., 2019; Morris et al., 2019; Waryszak et al., 2021). Focusing on regulatory variability in the US southeast, Jones and Pippin (2022) highlight potential policy levers that could assist in promoting wider adoption of NI, when appropriate. Their insights combined with more detailed analysis of human behavior in adoption of coastal protection strategies could provide a sound basis for future policy research and simulations.

The coastline of Georgia is characterized by expansive wetlands and limited but expanding human development (Gittman et al., 2015). We selected this region as a case study because of the availability of temporal data of shoreline modification, which is crucial for our model. Our findings most directly correspond to other lightly developed coasts dominated by wetlands. For more developed and modified regions, it is likely that biodiversity losses compared with an unmodified state have been greater. Such locations would therefore require either a higher percentage of NI or a longer time to mitigate biodiversity losses. Shoreline modification rate is also related to how fast biodiversity can change within the system. With faster rates of modification, more drastic changes in biodiversity can occur sooner. Our model predicted relatively rapid rates of modification for Georgia, with the 7.5% extent of armoring in 2018 approximately doubling to 16% by 2042 (Figure 3). In contrast, a study of the urban centers of New Zealand found approximately 35% of the shoreline was modified in 2018 and predicted an increase in armoring of 49–76% by 2043 (Floerl et al., 2021). This greater current amount of armoring and slower rate of change mean a much greater proportion of modification would need to be NI to fully mitigate biodiversity losses. Additionally, the positive impact of NI was driven by installations on biogenic coasts, which dominate Georgia. In regions where biogenic coasts compose only a small fraction of shoreline length, recovering biodiversity through the use of NI might not be possible at all based on currently available data. More research is needed to investigate the impact of NI on biodiversity in sandy and rocky coastlines as these habitats are currently underrepresented in the literature.

Our results offer coastal land managers potential tools to analyze and promote more sustainable shoreline modification choices. We used the simplest, effective model for predicting shoreline armoring to support reproducibility by local managers and practitioners. The model only requires multiple time steps of geospatial shoreline armoring data. Although additional time steps help account for variability, our method could be applied to any coastal setting with just 2 time steps of geospatial data. By applying the relationship between shoreline modification structure and biodiversity derived here, local practitioners can replicate our study for their chosen domain. The relationship between NI and biogenic coasts is robust, but the same cannot be said for NI versus beaches or rocky coasts. As such, our relationships are best applied to locations dominated by biogenic coasts, such as Georgia. Utilizing the tools presented here allows local regulatory and management authorities to calculate the proportion of infrastructure that could be focused on NI to potentially avoid or reverse biodiversity decline. We hope that such a tool will prove valuable in the implementation and possible revision of the shoreline modification permitting process.

Our literature review indicated that NI is less harmful to biodiversity than gray infrastructure, particularly at biogenic coastlines. As the rate of shoreline modification increases in the future, utilizing NI over gray infrastructure can mitigate loss in biodiversity. Based on current available data, if approximately 47.5% of new coastal infrastructure in Georgia is NI, biodiversity losses caused by past shoreline modification could be reversed by 2102. In locations with a greater percentage of existing shoreline modification, slower rates of modification, or a lower percentage of biogenic coasts, recovering previously lost biodiversity by 2102 would only be possible with a higher proportion of NI. Considering the lasting effects of global and indirect human impacts on biodiversity, it is crucial to minimize the impact of direct and local human impacts.

ACKNOWLEDGMENTS

This work was supported by the US Army Corps of Engineers Engineering With Nature Initiative through Cooperative Ecosystem Studies Unit Agreement. We thank M. Risse, J. S. Pippin, and K. Samples for discussions that helped formulate this project and for technical guidance.

ORCID

Daniel J. Coleman https://orcid.org/0000-0003-4243-9153
Rachel K. Gittman https://orcid.org/0000-0001-8376-8960

REFERENCES

Alexander, C. R. (2010). *GIS and field-based documentation of armored estuarine shorelines in Georgia*. Georgia Department of Natural Resources.

Alexander, C. R. (2016). *Geospatial characterization studies for advancing estuarine restoration and management in Georgia*. Georgia Department of Natural Resources.

Arkema, K. K., Guannel, G., Verutes, G., Wood, S. A., Guerry, A., Ruckelshaus, M., Kareiva, P., Lacayo, M., & Silver, J. M. (2013). Coastal habitats shield people and property from sea-level rise and storms. *Nature climate change*, *3*(10), 913–918.

Balouskus, R. G., & Targett, T. E. (2016). Fish and blue crab density along a riprap-sill-hardened shoreline: comparisons with Spartina marsh and riprap. *Transactions of the American Fisheries Society*, *145*(4), 766–773.

Bellantuono, G. (2014). The regulatory anticommons of green infrastructures. *European Journal of Law and Economics*, *37*, 325–354.

Bliss, T. H., Coleman, S., MacKinnon, J., Hurley, D., & Lambert, C. (2014). *Living shoreline – Little St. Simons Island Georgia*. NOAA. https://repository.library.noaa.gov/view/noaa/35145

Bugbee, K. O. (2020). *Strategies for increased functionality of living shorelines on the Georgia coast* (Doctoral dissertation). University of Georgia.

Butchart, S. H., Walpole, M., Collen, B., Van Strien, A., Scharlemann, J. P., Almond, R. E., Baillie, J. E., Bomhard, B., Brown, C., Bruno, J., Carpenter, K. E., Carr, G. M., Chanson, J., Chenery, A. M., Csirke, J., Davidson, N. C., Dentener, F., Foster, M., Galli, A., … Watson, R. (2010). Global biodiversity: Indicators of recent declines. *Science*, *328*(5982), 1164–1168.

Byers, J. E. (2002). Impact of non-indigenous species on natives enhanced by anthropogenic alteration of selection regimes. *Oikos*, *97*(3), 449–458.

Cameron, R. W., & Blanuša, T. (2016). Green infrastructure and ecosystem services–is the devil in the detail? *Annals of Botany*, *118*(3), 377–391.

Ceballos, G., Ehrlich, P. R., Barnosky, A. D., García, A., Pringle, R. M., & Palmer, T. M. (2015). Accelerated modern humaninduced species losses: Entering the sixth mass extinction. *Science Advances*, *1*(5), e1400253.

Coleman, D. J., Schuerch, M., Temmerman, S., Guntenspergen, G., Smith, C. G., & Kirwan, M. L. (2022). Reconciling models and measurements of marsh vulnerability to sea level rise. *Limnology and Oceanography Letters*, *7*(2), 140–149.

Currin, C. A., Chappell, W. S., & Deaton, A. (2010). Developing alternative shoreline armoring strategies: The living shoreline approach in North Carolina. In H. Shipman, M. N. Dethier, G. Gelfenbaum, K. L. Fresh, & R. S. Dinicola (Eds.), *2010: Puget Sound Shorelines and the Impacts of Armoring—Proceedings of a State of the Science Workshop, May 2009: U.S. Geological Survey Scientific Investigations Report 2010–5254* (pp. 91–102). U.S. Geological Survey.

Dafforn, K. A., Mayer-Pinto, M., Morris, R. L., & Waltham, N. J. (2015). Application of management tools to integrate ecological principles with the design of marine infrastructure. *Journal of Environmental Management*, *158*, 61–73.

Davenport, T. M., Seitz, R. D., Knick, K. E., & Jackson, N. (2018). Living shorelines support nearshore benthic communities in upper and lower Chesapeake Bay. *Estuaries and Coasts*, *41*, 197–206.

Dennis, H. D., Evans, A. J., Banner, A. J., & Moore, P. J. (2018). Reefcrete: Reducing the environmental footprint of concretes for eco-engineering marine structures. *Ecological Engineering*, *120*, 668–678.

Dhakal, K. P., & Chevalier, L. R. (2017). Managing urban stormwater for urban sustainability: Barriers and policy solutions for green infrastructure application. *Journal of Environmental Management*, *203*, 171–181.

Dugan, J. E., Airoldi, L., Chapman, M. G., Walker, S. J., & Schlacher, T. (2011). Estuarine and coastal structures: Environmental effects, a focus on shore and nearshore structures. In E. Wolanski & D. McLusky (Eds.), *Treatise on estuarine and coastal science* (pp. 17–41). Academic Press.

Dugan, J. E., Emery, K. A., Alber, M., Alexander, C., Byers, J. E., Gehman, A. M. M., McLenaghan, N., & Sojka, S. (2018). Generalizing ecological effects of shoreline armoring across soft sediment environments. *Estuaries and Coasts*, *41*, 180–196. https://doi.org/10.1007/s12237-017-0254-x

Elwell, T. L., Gelcich, S., Gaines, S. D., & López-Carr, D. (2018). Using people's perceptions of ecosystem services to guide modeling and management efforts. *Science of the Total Environment*, *637*, 1014–1025.

Evans, A. J., Garrod, B., Firth, L. B., Hawkins, S. J., Morris-Webb, E. S., Goudge, H., & Moore, P. J. (2017). Stakeholder priorities for multi-functional coastal defence developments and steps to effective implementation. *Marine Policy*, *75*, 143–155.

Evans, A. J., Firth, L. B., Hawkins, S. J., Hall, A. E., Ironside, J. E., Thompson, R. C., & Moore, P. J. (2019). From ocean sprawl to blue-green infrastructure–A UK perspective on an issue of global significance. *Environmental Science & Policy*, *91*, 60–69.

Feagin, R. A., Bridges, T. S., Bledsoe, B., Losos, E., Ferreira, S., Corwin, E., Lodder, Q., Beck, M. W., Reguero, B., Sutton-Grier, A., Figlus, J., Palmer, R., Nelson, D. R., Smith, C., Olander, L., Silliman, B., Pietersen, H., Costanza, R., Gittman, R. K., … Guidry, T. (2021). Infrastructure investment must incorporate Nature's lessons in a rapidly changing world. *One Earth*, *4*, 1361–1364.

Firth, L. B., Airoldi, L., Bulleri, F., Challinor, S., Chee, S. Y., Evans, A. J., Hanley, M. E., Knights, A. M., O'Shaughnessy, K., Thompson, R. C., & Hawkins, S. J. (2020). Greening of grey infrastructure should not be used as a Trojan horse to facilitate coastal development. *Journal of Applied Ecology*, *57*(9), 1762–1768.

Floerl, O., Atalah, J., Bugnot, A. B., Chandler, M., Dafforn, K. A., Floerl, L., Zaiko, A., & Major, R. (2021). A global model to forecast coastal hardening and mitigate associated socioecological risks. *Nature Sustainability*, *4*(12), 1060–1067.

Georgia Department of Natural Resources. (2013). *Living shorelines along the Georgia coast: A summary report of the first living shoreline projects in Georgia*. Coastal Resources Division.

Gittman, R. K., Fodrie, F. J., Popowich, A. M., Keller, D. A., Bruno, J. F., Currin, C. A., Peterson, C. H., & Piehler, M. F. (2015). Engineering away our natural defenses: An analysis of shoreline hardening in the US. *Frontiers in Ecology and the Environment*, *13*(6), 301–307.

Gittman, R. K., Peterson, C. H., Currin, C. A., Joel Fodrie, F., Piehler, M. F., & Bruno, J. F. (2016). Living shorelines can enhance the nursery role of threatened estuarine habitats. *Ecological Applications*, *26*, 249–263.

Gittman, R. K., Popowich, A. M., Bruno, J. F., & Peterson, C. H. (2014). Marshes with and without sills protect estuarine shorelines from erosion better than bulkheads during a Category 1 hurricane. *Ocean & Coastal Management*, *102*, 94–102.

Gittman, R. K., Scyphers, S. B., Baillie, C. J., Brodmerkel, A., Grabowski, J. H., Livernois, M., Poray, A. K., Smith, C. S., & Fodrie, F. J. (2021). Reversing a tyranny of cascading shoreline-protection decisions driving coastal habitat loss. *Conservation Science and Practice*, *3*, e490.

Guthrie, A. G., Bilkovic, D. M., Mitchell, M., Chambers, R., Thompson, J. S., & Isdell, R. E. (2022). Ecological equivalency of living shorelines and natural marshes for fish and crustacean communities. *Ecological Engineering*, *176*, 106511.

Habibullah, M. S., Din, B. H., Tan, S. H., & Zahid, H. (2022). Impact of climate change on biodiversity loss: Global evidence. *Environmental Science and Pollution Research*, *29*(1), 1073–1086.

Hooper, D. U., Adair, E. C., Cardinale, B. J., Byrnes, J. E., Hungate, B. A., Matulich, K. L., Gonzalez, A., Duffy, J. E., Gamfeldt, L., & O'Connor, M. I. (2012). A global synthesis reveals biodiversity loss as a major driver of ecosystem change. *Nature*, *486*(7401), 105–108.

Institute for Resilient Infrastructure Systems. (2024). *What is natural infrastructure?* https://iris.uga.edu/about-infrastructure/

Jaramillo, E., Dugan, J., Hubbard, D., Manzano, M., & Duarte, C. (2021). Ranking the ecological effects of coastal armoring on mobile macroinvertebrates across intertidal zones on sandy beaches. *Science of the Total Environment*, *755*, 142573.

Jaureguiberry, P., Titeux, N., Wiemers, M., Bowler, D. E., Coscieme, L., Golden, A. S., Guerra, C. A., Jacob, U., Takahashi, Y., Settele, J., Díaz, S., Molnár, Z., & Purvis, A. (2022). The direct drivers of recent global anthropogenic biodiversity loss. *Science Advances*, *8*(45), eabm9982.

Jones, S. C., & Pippin, J. S. (2022). Towards principles and policy levers for advancing living shorelines. *Journal of Environmental Management*, *311*, 114695.

Jumani, S., Rao, S., Kelkar, N., Machado, S., Krishnaswamy, J., & Vaidyanathan, S. (2018). Fish community responses to stream flow alterations and habitat modifications by small hydropower projects in the Western Ghats biodiversity hotspot, India. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *28*(4), 979–993.

Kittinger, J. N., & Ayers, A. L. (2010). Shoreline armoring, risk management, and coastal resilience under rising seas. *Coastal Management*, *38*(6), 634–653.

Lau, J. D., Hicks, C. C., Gurney, G. G., & Cinner, J. E. (2019). What matters to whom and why? Understanding the importance of coastal ecosystem services in developing coastal communities. *Ecosystem Services*, *35*, 219–230.

Lean, C. H. (2021). Invasive species increase biodiversity and, therefore, services: An argument of equivocations. *Conservation Science and Practice*, *3*(12), e553.

Lowe, E. C., Steven, R., Morris, R. L., Parris, K. M., Aguiar, A. C., Webb, C. E., Bugnot, A., Dafforn, K., Connolly, R., Mayer-Pinto, M., & Pinto, M. M. (2022). Supporting urban ecosystem services across terrestrial, marine and freshwater realms. *Science of the Total Environment*, *817*, 152689.

McNamara, D. E., & Keeler, A. (2013). A coupled physical and economic model of the response of coastal real estate to climate risk. *Nature Climate Change*, *3*, 559–562.

Mitchell, M., & Bilkovic, D. M. (2019). Embracing dynamic design for climate-resilient living shorelines. *Journal of Applied Ecology*, *56*, 1099–1105. https://doi.org/10.1111/1365-2664.13371

Morris, R. L., Deavin, G., Hemelryk Donald, S., & Coleman, R. A. (2016). Eco-engineering in urbanised coastal systems: Consideration of social values. *Ecological Management & Restoration*, *17*(1), 33–39.

Morris, R., Strain, E. M., Konlechner, T. M., Fest, B. J., Kennedy, D. M., Arndt, S. K., & Swearer, S. E. (2019). Developing a nature-based coastal defence strategy for Australia. *Australian Journal of Civil Engineering*, *17*(2), 167–176.

Narayan, S., Beck, M. W., Reguero, B. G., Losada, I. J., van Wesenbeeck, B., Pontee, N., Sanchirico, J. N., Ingram, J. C., Lange, G.-M., & Burks-Copes, K. A. (2016). The effectiveness, costs and coastal protection benefits of natural and nature-based defences. *PLoS ONE*, *11*, e0154735.

Nunez, K., Rudnicky, T., Mason, P., Tombleson, C., & Berman, M. (2022). A geospatial modeling approach to assess site suitability of living shorelines and emphasize best shoreline management practices. *Ecological Engineering*, *179*, 106617.

Peterson, N. E., Landry, C. E., Alexander, C. R., Samples, K., & Bledsoe, B. P. (2019). Socioeconomic and environmental predictors of estuarine shoreline hard armoring. *Scientific Reports*, *9*(1), 16288.

Scyphers, S. B., Beck, M. W., Furman, K. L., Haner, J., Keeler, A. G., Landry, C. E., O'Donnell, K., Webb, B., & Grabowski, J. H. (2020). Designing effective incentives for living shorelines as a habitat conservation strategy along residential coasts. *Conservation Letters*, *13*(5), e12744.

Scyphers, S. B., Picou, J. S., & Powers, S. P. (2015). Participatory conservation of coastal habitats: The importance of understanding homeowner decision making to mitigate cascading shoreline degradation. *Conservation Letters*, *8*, 41–49.

Scyphers, S. B., Powers, S. P., Heck, K. L., Jr., & Byron, D. (2011). Oyster reefs as natural breakwaters mitigate shoreline loss and facilitate fisheries. *PLoS ONE*, *6*(8), e22396.

Sicangco, C., Collini, R., Martin, S., Monti, A., & Sparks, E. (2021). *Cost-benefit analysis of a small-scale living shoreline project*. Program for Local Adaptation to Climate Effects: Sea Level Rise [PLACE: SLR]. https://placeslr.org/wp-content/uploads/2021/07/Living-Shoreline-Cost-Benefit-Analysis_withPub.pdf

Smith, C. S., Rudd, M. E., Gittman, R. K., Melvin, E. C., Patterson, V. S., Renzi, J. J., Wellman, E. H., & Silliman, B. R. (2020). Coming to terms with living shorelines: A scoping review of novel restoration strategies for shoreline protection. *Frontiers in Marine Science*, *7*, 434.

Sobocinski, K. L., Cordell, J. R., & Simenstad, C. A. (2010). Effects of shoreline modifications on supratidal macroinvertebrate fauna on Puget Sound, Washington beaches. *Estuaries and Coasts*, *33*, 699–711.

Sutton-Grier, A. E., Wowk, K., & Bamford, H. (2015). Future of our coasts: The potential for natural and hybrid infrastructure to enhance the resilience of our coastal communities, economies and ecosystems. *Environmental Science & Policy*, *51*, 137–148.

Toft, J. D., Dethier, M. N., Howe, E. R., Buckner, E. V., & Cordell, J. R. (2021). Effectiveness of living shorelines in the Salish Sea. *Ecological Engineering*, *167*, 106255.

Trisos, C. H., Merow, C., & Pigot, A. L. (2020). The projected timing of abrupt ecological disruption from climate change. *Nature*, *580*(7804), 496–501.

Turley, C., & Gattuso, J. P. (2012). Future biological and ecosystem impacts of ocean acidification and their socioeconomic-policy implications. *Current Opinion in Environmental Sustainability*, *4*(3), 278–286.

U. S. Fish and Wildlife Service. (2023). *National Wetlands Inventory website*. U.S. Department of the Interior, Fish and Wildlife Service.

Vaselli, S., Bulleri, F., & Benedetti-Cecchi, L. (2008). Hard coastal-defence structures as habitats for native and exotic rocky-bottom species. *Marine Environmental Research*, *66*(4), 395–403.

Waryszak, P., Gavoille, A., Whitt, A. A., Kelvin, J., & Macreadie, P. I. (2021). Combining gray and green infrastructure to improve coastal resilience: Lessons learnt from hybrid flood defenses. *Coastal Engineering Journal*, *63*(3), 335–350.

Williams-Subiza, E. A., & Epele, L. B. (2021). Drivers of biodiversity loss in freshwater environments: A bibliometric analysis of the recent literature. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *31*(9), 2469–2480.

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

How to cite this article: Coleman, D. J., Gittman, R. K., Landry, C. E., Byers, J. E., Alexander, C. R., Coughlin, G. P., & Woodson, C. B. (2024). Quantifying the impacts of future shoreline modification on biodiversity in a case study of coastal Georgia, United States. *Conservation Biology*, e14301. https://doi.org/10.1111/cobi.14301