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Assessing habitat enhancement by living shoreline restoration: exploring potential caveats of nekton community metrics

Effects of shoreline restoration on nekton

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Abstract

Living Shorelines (LS) are a nature-based restoration technique that aims to stabilize shorelines while enhancing multiple ecosystem services. In particular, LS are frequently promoted as beneficial for enhancing essential coastal habitats for fish and mobile crustaceans (nekton). In this study, we aimed to assess the effects of LS on nekton habitat across eight LS restoration sites in coastal Alabama, USA, by applying widely-used community metrics. Nekton abundance, species richness, evenness, and diversity tended to be higher in LS and adjacent unrestored control sites than along nearby hardened shorelines. Community metrics showed no clear effect of LS compared to their adjacent controls, with substantial among-site variation masking any restoration effect. While this may suggest ecological equivalence between restored and control sites, there are other possible interpretations; that a lack of difference reflects sampling the same populations at nearby control and restored sites, that differences do exist but sampling effort was insufficient to detect them, or that community metrics may be too insensitive for comparing the effects of various restoration approaches on nekton habitat quality. Further exploration of individual-based metrics such as growth and condition of key species is warranted, as these may be more sensitive for assessing restoration outcomes and guiding future project designs.

Keywords: ecological equivalency, essential fish habitat, fisheries, habitat loss, nursery function, salt marsh ecology.

Implications for practice

- The interpretation of similarity between restoration and control sites in community metrics depends on the nature of the control sites.
- Similarity between degraded control and restored sites would suggest a lack of nekton response to restoration, while similarity between restored and healthy "target" reference sites would indicate success in achieving ecological equivalence.
- A finding of no difference between control and restored sites does not necessarily indicate ecological equivalence; other interpretations should be considered.
- Additional metrics such as growth and condition of key species may enhance evaluations of nekton responses to restoration.

Introduction

Coastal erosion threatens human infrastructure, as well as natural coastal ecosystems which provide many valuable services such as carbon sequestration, improved water quality, shoreline protection, and the provision of habitat for a diversity of species (Gilby et al. 2021). Shoreline erosion rates are likely to increase with climate change (Mentaschi et al. 2018), and the traditional response to protect coastal infrastructure is shoreline hardening. However, hardening has negative consequences for coastal ecosystem services (Balouskus & Targett 2016; Gittman, et al. 2016b; Kornis et al. 2018). Living shorelines (LS) are an increasingly popular nature-based solution that aim to stabilize shorelines while promoting the recovery of natural habitats and the functions they provide (Smith et al. 2020). While significant progress has been made in improving LS designs, the evaluation of LS effectiveness in restoring functional coastal habitats lags behind the rapid increase in the scale and number of LS projects (Bilkovic et al. 2016). This hampers our ability to guide future restoration designs to maximize ecological outcomes.

One of the stated objectives of many coastal restoration projects is the enhancement of habitats that support ecologically, socially, and economically important fish and mobile crustaceans (nekton). In addition to functioning as essential nursery habitats for many fisheries species (Lefcheck et al. 2019), complex habitats in coastal seascapes also provide rich foraging grounds for many species at various life stages (Sheaves et al. 2015). The quality and availability of these habitats can be a critical bottleneck in nekton life cycles (Fodrie et al. 2009; Sundblad et al. 2014). Consequently, it is reasonable to expect that LS implementation should enhance habitat values for a diversity of nekton species by providing additional structured habitats, stabilizing marsh edges, and enhancing prey resources (Colombano et al. 2021; Currin 2019; Polk et al. 2022).

Most assessments of nekton habitat enhancement in response to restoration use community-based metrics such as multivariate community composition, species richness, abundance, and diversity (Gittman et al. 2016a, Peterson et al. 2016; Guthrie et al. 2022). These approaches are logical, since we expect shoreline habitat enhancement to benefit multiple species, that abundance tends to be a reliable indicator of habitat quality (Lefcheck et al. 2019), and that diversity frequently reflects ecosystem functioning and resilience (Tilman et al. 2014; Troast et al. 2022). Some studies have reported higher abundances of nekton from restored sites than nearby controls (Scyphers et al. 2011; Gittman et al. 2016a), while others find no differences in community composition, diversity, or abundance between LS and control sites (Guthrie et al. 2022). The nature of the control sites in each study needs to be considered to interpret these findings. When a control site is a nearby unrestored and degraded shoreline (e.g. Scyphers et al. 2011), then successful restoration should lead to better habitat quality, reflected in higher community metrics at the restored sites. In contrast, if the control site is a high-quality natural reference marsh that the restoration is aiming to mimic, then similarity in metrics between control and restored sites would indicate success (e.g. Troast et al. 2022).

The importance of coastal seascapes for supporting many species, and the ongoing threats confronting these systems, warrant further investigations into the responses of nekton communities to restoration (Guthrie et al. 2022). In this study, we aimed to evaluate the response of nekton communities to shoreline restoration at 8 LS projects in coastal Alabama. We use seine data from 8 restoration sites, 8 nearby unrestored control sites, and 4 hardened shoreline sites to represent the traditional alternate shoreline defense strategy (Figure 1). Because our control sites represent nearby unrestored and unprotected marsh shorelines experiencing varying levels of erosion, we predicted that community metrics should be highest, indicating higher habitat quality at LS restored sites that provide additional structured habitat and restore the marsh edge, and lowest at hardened sites, which we expect to provide reduced habitat quality.

Methods

Study sites

We focused on 8 of the largest, publicly-funded Living Shoreline (LS) restoration projects in coastal Alabama (Figure 1). Each restoration project was implemented largely independently of each other at various times between 2009 and 2020 (Table S1). While BACI designs are the most robust approach to quantify nekton responses to restoration (e.g. Gittman et al. 2016a, Troast et al. 2022), few restoration projects provide funding for rigorous pre-construction monitoring despite its widely recognized value (Waltham et al. 2020). Our sites did not have pre-construction nekton monitoring, so instead we employed space-for-time substitution by sampling 8 control sites, which were un-restored marsh shorelines nearby to each restored site. These control sites are experiencing various levels of shoreline erosion with steep or escarped edges, and as such, should represent what the restored shorelines would look like if restoration had not been undertaken. With this design, we would interpret higher community metrics at restored sites to indicate habitat enhancement by restoration. We also sampled 4 riprap hardened shorelines intermixed among our LS sites, to represent the typical alternative shoreline erosion mitigation option, i.e. shoreline hardening. Control (and hardened) sites were chosen paired to LS sites if possible, but additional un-paired sites were included, as our intent was to represent the variability of site conditions across the study region rather than focus on pairwise comparisons at each location. For analyses, samples from each treatment were grouped as a single site if they were collected on immediately adjacent contiguous shorelines (PaP, CI, AP, ST), and treated as independent sites if they were not, for a total of 15 sites as reflected by the site names in Figure 1, and the panel groupings in Figure 5. Although this design somewhat confounds site and treatment, the control and hardened treatments were always the nearest such shorelines to our restoration sites, and our design allows for the detection of treatment effects on nekton communities.

Alabama is in the central northern Gulf of Mexico (GoM), with diurnal tides ranging around 0.8 m. Mobile Bay receives the second highest discharge of freshwater into the GoM (Stumpf et al. 1993), with freshwater pulses creating dynamic environmental conditions. Fringing marshes are dominated by *Spartina alterniflora* and *Juncus romerianus*, with *Phragmites australis*, *Distichlis spicata*, and *Spartina patens* along some eroding, scarped shores. Seagrass beds of *Halodule wrightii* and/or *Ruppia maritima* occur in parts of Portersville Bay. Our 8 LS sites include breakwaters ranging in length from 0.35 to 3.5 km, constructed of a variety of materials including, loose oyster shell, Reef Balls, Reef Blocks, precast concrete Wave Attenuation Devices, and detached stone riprap (Table S1). Hardened sites comprise riprap revetments along the shore to prevent erosion, ranging from 0.5 to 2 km. Further Living Shoreline site details are provided in Supplement S1.

Sampling procedures

Samples were collected using a 15.2 m center-bag seine with 6.4 mm stretched mesh and a 1.83 m deep bag. Sampling was standardized so the seine was pulled over a 10 m distance at a width of 10 m between seine poles, ensuring that a consistent bottom area of 100 m^2 was sampled. Seines were pulled parallel to the shoreline and as close to the shoreline as possible while ensuring the shoreward end was in water at least 20 cm deep. Between 4 and 12 replicate seine hauls were collected at each site (total = 105 seine hauls), based on the length of the LS breakwaters at each. Each seine replicate was located at least 75 m apart, usually >100 m, to ensure independence. Sampling was conducted between 7th of June and 27th of July, 2022. All organisms were returned to the laboratory where they were sorted, identified to the lowest practical taxonomic level, and enumerated. All fish sampling followed protocols of the

Institutional Animal Care and Use Committee (IACUC) of the University of South Alabama, USA, under protocol #1903663-2.

Data analysis

To compare nekton abundance, species richness, diversity and evenness among sites and treatment types (hardened, control, restored), we ran non-parametric Kruskal-Wallis tests with each seine haul being treated as independent replicates. Non-parametric tests were conducted because our data failed to meet the assumptions of two-way mixed-effects ANOVAs (Supplementary material, Tables S2 and S3). Pairwise comparisons between factor levels were conducted using the *post-hoc* Dunn's test for all univariate metrics, with Bonferroni correction to account for increased probability of type 1 errors.

To compare nekton community structure among treatment types (hardened, control, restored), two-way Permutational Analysis of Variance (PERMANOVAs) with a crossed design were executed for the top 95% most frequently encountered taxa. Separate PERMANOVAs were performed on log-transformed abundance and presence-absence (binary) data, utilizing Bray-Curtis and Jaccard distance resemblance matrices, respectively, with each replicate seine haul treated as an independent replicate. If significant differences were detected, Similarity Percentage Analysis (SIMPER) was executed to determine which taxa contributed the most to differences between relevant factors (Clarke 1993). Significant differences in community structure among treatments and sites were visualized using 3-dimensional non-metric multidimensional scaling (nMDS) ordination. All multivariate and ecological diversity analyses were conducted in the R package "vegan" (Dixon 2003), with PERMANOVA pairwise comparisons performed using the R package "pairwiseAdonis" (Martinez, 2020). To predict the total number of species that are likely to inhabit each site and compare it to observed species richness, species accumulation curves with extrapolation for each individual site and treatment type were produced using the R package "iNEXT" (Hsieh et al. 2016).

Results

A total of 8545 individuals in 58 taxa were sampled from 105 replicate seine hauls. The penaeids *Litopenaeus setiferus* (n = 3121) and *Farfantepeaneus aztecus* (n = 762), silver perch *Bairdiella chrysoura* (n = 247), pinfish *Lagodon rhomboides* (n = 245), anchovy *Anchoa mitchilli* (n = 178), and hardhead catfish *Arius felis* (n = 172) were dominant, representing about 55% by number of all nekton collected (Table S4). Abundance and species richness, but not diversity and evenness, were significantly higher in restored/control than in hardened treatments, while no differences were found between restored and controls (Figure 2, Table S5, Table S6). The Kruskal-Wallis test also revealed significant among-site variation in all univariate metrics (Table S5). Pairwise comparisons reveal that site effects were primarily due to sites that included only a hardened treatment differing from others, with 11 of 13 significant pairs including hardened-only sites (LP, SB, SA-H) (Table S7).

The PERMANOVAs for log-transformed abundances and presence-absence showed significant effects of both site and treatment on nekton community composition (Figure 3, Table S8). However, Levene's test showed that between-factor multivariate dispersions were heterogenous ($p < 0.001$), so PERMANOVA results should be interpreted with caution. The nMDS ordinations (Figure 3) exhibited moderate-high stress values, demonstrating that the position of individual replicates were considerably distorted when reduced to the 3-dimensional space. The treatment effect in the community composition was driven by hardened sites differing from controls and LS, while no differences were detected between controls and LS (Table S9, Figure 3). The site effect detected by the PERMANOVAs is only due to some weak, inconsistent site groupings that can be visualized when multivariate community composition data is coded by site (Figure 4), where some sites tend to cluster, but the majority of replicate samples for most sites tended to overlap throughout the ordination space. This indicates that the nekton community composition of most replicates from one site is as likely to be similar to replicates from distant sites than to other replicates from the same site due to a high degree of variability. Pairwise PERMANOVA comparisons between sites showed again that differences were driven by sites that included only hardened shorelines, with 16 of the 20 significant comparisons for log abundance, and all 11 for presence-absence including hardened-only sites (Tables S10 and S11).

SIMPER analysis show that most dissimilarities between hardened and LS/Control sites are driven by LS/Control having higher abundances of penaeid shrimp (*Farfantepenaeus aztecus* and *Litopenaeus setiferus*), hardhead catfish (*Arius felis*), silver perch (*Bairdiella chrysoura*), and pinfish (*Lagodon rhomboides*). Despite showing similar averaged abundances between treatments, anchovies (*Anchoa mitchlli*) drive dissimilarities due to high variability between replicate seine samples (Table 1). Species accumulation curves for some sites did not show an asymptote, meaning that observed species richness was often less than the predicted total number of species in those cases (Figure 5).

Discussion

Definitions of restoration success can vary according to desired or expected outcomes of each project (Baggett et al. 2015). The LS investigated in this study all include breakwaters that aim to protect eroding shorelines from wave action, an issue that threatens coastal habitat quality and is likely to increase with climate change (Paprotny et al. 2021). The breakwaters also provide structurally complex habitat, and hard substrate favorable to the settlement of ecosystem engineers such as oysters (Morris et al. 2019). Successful LS breakwaters also stabilize sediments (Luom et al. 2021; Morris et al. 2019; Wellman et al. 2022), and provide favorable ground for the establishment of a more diverse assemblage of prey types (Bilkovic et al. 2016; Bilkovic & Mitchell 2017; Wellman et al. 2022). In addition, the stabilization and recovery of marsh shorelines from steep eroding edges (such as the ones at our control sites) to more gently-sloped aggrading shorelines will allow more opportunities for nekton to access the flooded marsh surface and the benefits they gain from doing so (Minello et al. 2012; Baker et al. 2013; Plumlee et al. 2020). Therefore, the choice of the type of control used as comparisons to restored sites in studies utilizing space for time substitution is particularly important when interpreting findings. Specifically, studies utilizing natural, "target" controls consisting of shorelines relatively unaffected by erosion would aim for restored sites to become "equivalent" or similar to controls (e.g. Troast et al. 2022). Contrastingly, studies that chose degraded controls should expect restored sites to differ from controls (e.g. Scyphers et al. 2011). In our study system where LS projects were implemented to restore eroding and degraded marsh shorelines, and the control sites chosen were adjacent eroding shorelines, we expect that successful restoration should result in habitat enhancement for nekton at restored sites compared to controls.

Our data showed that hardened shorelines differ from both control and restored sites, usually showing decreased habitat quality according to our community metrics of abundance, richness, evenness, and diversity. Although our design limits our ability to distinguish among-site differences from treatment effects, the spatial patterns of variation in our various metrics showed the riprap hardened sites, particularly LP and SB, had lower values than control and restored sites, and no consistent patterns between control and restored regardless of treatment or geographic proximity. These findings are in agreement with much previous work showing degraded habitat values of hardened shorelines (Munsch et al. 2017; Legaspi et al. 2023). However, we found no evidence that LS have consistently higher abundance, richness, evenness or diversity than unprotected control sites. Instead, we see considerable among-site variation, which is typical of coastal nekton communities (Sheaves et al. 2012). The lack of clear differences between restored and control sites could be interpreted in a number of ways, each of which will be discussed further below: (1) LS may still be ecologically equivalent to unprotected controls in terms of fish habitat values; (2) the lack of difference, particularly at sites where LS and controls were paired and adjacent, could indicate we were actually sampling the same populations among treatments; (3) the LS sites sampled provide enhanced fish habitat, but our sampling effort was too low to detect these effects; or (4), even with higher levels of sampling effort, the community metrics commonly employed to evaluate restoration success are too insensitive to detect the more subtle effects of restoration for highly variable estuarine nekton communities.

The similarity in community metrics between restored and control sites in our study has multiple potential explanations. First, restored and eroding control sites may still provide equivalent ecological values for nekton. Several recent studies that examined restoration responses of marsh edge nekton communities showed no difference in community metrics between restored and adjacent control shorelines, concluding that the restored sites were ecologically equivalent nekton habitats to the control marshes (Guthrie et al. 2022; Isdell et al. 2021; Troast et al. 2022). If the objectives of the restoration projects were to restore degraded shorelines to have similar ecological values as nearby, healthy marsh-lined shores, then such findings could reasonably be interpreted as success. However, in projects such as ours that aim to improve ecological condition of restored sites over that of adjacent unprotected controls, a finding of equivalence would not indicate successful restoration, at least with respect to nekton habitat.

Second, the lack of difference in community metrics may be at least partially driven by our samples at control and LS sites actually sampling the same populations of nekton. It is well established in coastal ecology that nekton utilize coastal systems as a spatially connected and interdependent mosaic of different habitats (Sheaves 2009). In some of our smaller paired control/LS sites, the shoreline configuration meant that the only available seining locations at unprotected and unhardened (control) shorelines included locations close by to seining locations at the restored site, as close at 150 m at Alabama Port. In such cases, the lack of difference in community metrics may be partially driven by our samples at control and LS sites actually sampling the same populations making use of the paired sites as part of the interconnected mosaic of habitats. This would depend on the exact spatial scales at which individuals and populations use these mosaics, which is unresolved for many species, particularly for early juvenile life stages (Nagelkerken et al. 2015). Most of our control samples were located much farther from restored sites, and it seems unlikely that the proximity of control and LS sites alone could account for the lack of difference in our nekton community metrics. This does however highlight that while immediately adjacent control sites may be the most appropriate for monitoring metrics such as shoreline erosion rates and vegetation structure, control or reference sites for identifying fish community responses should be located far enough from the restoration sites to ensure they are not utilized by the same individuals and populations.

Absence of evidence is not necessarily evidence of absence (Altman & Bland 1995). In this case, this and previous studies that found no difference in community metrics between restored and controls do not necessarily indicate that LS are ecologically equivalent to the control marshes as habitat for nekton; instead, we may fail to detect differences. Such inability to detect effects of restoration can be related to the inherent stochasticity exhibited by nekton communities, which often drives most variation in community structure. For example, recruitment of fish into estuarine nurseries is influenced by random variation in environmental drivers, and recruitment can exert considerable influence on future abundance and community structure (Pierre et al. 2018). Stochastic variation in nekton community sampling can mask spatial and temporal patterns in community structure (Connolly et al. 2005; Grossman et al. 1982; Syms & Jones 2000). One potential remedy for sampling stochastic communities is higher levels of sampling effort to allow researchers to better approximate the real distribution of relevant metrics, such as community composition, abundance, and species richness (Connolly et al. 2005). Some studies with greater sampling effort than the present study have reported positive effects of LS restorations on fish communities. For example, Gittman et al. (2016) recorded higher CPUE of fish and crustaceans in fyke net samples from LS sites than adjacent controls, although their seine net sampling found no differences. Similarly, Scyphers et al. (2011) sampling LS sites in Alabama, including the same Alabama Port site sampled in the present study, found higher CPUE of larger mobile fishes from gill nets and decapod crustaceans from seines adjacent to LS breakwaters than along unrestored control shorelines, while again detecting no differences in the assemblages of fish collected in seine samples. The effort employed in this study is similar to the seasonal sampling effort of other studies (Guthrie et al. 2022; Scyphers et al. 2011). It is possible that with greater replication and a longer-term dataset, the community metrics examined here may reveal differences between the LS and control sites sampled in our study. However, other studies with larger sampling effort often still find no differences (e.g. Gittmann et al. 2016a), or detect differences only in some metrics. For instance, Guthrie et al. (2022) found higher nekton biomass at LS sites, driven by a few common species, but detected no differences in abundance. Given the inherent variability of coastal nekton communities (Sheaves et al. 2012), one of the alternative explanations for finding no differences is that community metrics may be too insensitive to detect subtle effects of restoration on nekton within a reasonable level of certainty.

A limitation of our study is that it used only one gear type, seine nets, to quantify nekton communities. While seine nets are effective at sampling nekton from shallow open waters, they cannot sample the flooded marsh surface and would therefore under-represent some members of the community (Peterson & Turner 1994). The use of additional gears like fyke or gill nets may help resolve more subtle differences between restored and control sites (Scyphers et al. 2011; Gittman et al. 2016a). However, whether or not higher levels of replication or using multiple gears might reveal more subtle effects of restoration, the practical reality is that few restoration monitoring programs provide the resources needed for more extensive fish monitoring (Guthrie et al. 2022). Even well-funded programs will often be focused on the more immediate and pressing ecosystem responses to restoration, such as shoreline erosion rates and responses of foundational habitat-forming species such as marsh grasses, seagrass, and oysters. So while enhancing nekton habitats is one of the more widely stated goals of living shoreline restoration projects, demonstrating success in this effort is one of the more challenging tasks. Our highlighting of the ambiguities in previous findings is not a criticism of previous work, but rather a reflection of the real challenges in quantitatively representing dynamic coastal nekton communities.

In addition to evaluating the success of individual restoration projects, monitoring and research efforts may also seek to compare the relative success of different restoration strategies at enhancing ecosystem condition, to guide the design of future restoration efforts and maximize beneficial outcomes (Gittman et al. 2016, Guthrie et al. 2022). If the inherent variability in nekton communities or logistical constraints on sampling effort results in ambiguous interpretations from community metrics, then consideration of additional, potentially more sensitive metrics is warranted. A key benefit that nekton gain from occupying complex coastal habitats is enhanced growth due to increased prey availability and shifts in growth-mortality trade-offs (Dahlgren & Eggleston 2000; Le Pape & Bonhommeau 2015; Lefcheck et al. 2019). Therefore, rather than using community wide analysis with many species, the growth rates of key indicator species that we expect should be enhanced by restoration may be more effective indicators of habitat quality (Murase & Iguchi 2020; Wilson et al. 2019; Woodland et al. 2012). Growth can be assessed by a great variety of methods such as modal/size progression analysis (de Barros et al. 2022; Zhou et al. 2022), otolith micro-increments (Fox et al. 2003), field-caging experiments (Baker & Minello 2010), and tagging (Francis 1988). While some of these approaches require substantial field sampling efforts, the analysis of otolith increments to estimate growth rates of widespread and ecologically important fish species can be conducted with much less field effort, and the technique has already been validated for key species in the SE USA such as spot (Siegfried & Weinstein 1989) and pinfish (Harter & Heck 2006). Nekton may also respond to enhanced habitat quality through increased energy storage (Amara et al. 2007; Maceda-Veiga et al. 2014; Suthers 1998). Such responses can be measured by condition metrics including morphometric methods (Froese 2006), or the measurement of energy density (Wedge et al. 2015). Condition metrics have been shown to respond to habitat quality, pollution, exploitation and even parasitism on marine organisms (Blackwell et al. 2000; de Barros et al. 2020; de Barros et al. 2021; Maceda-Veiga et al. 2014).

Although individual-based growth and condition metrics rely on the identification of suitable indicator species, they have proven to be sensitive to habitat quality and are worthy of additional investigation as metrics of restoration success (Amara et al. 2006; Gilliers et al. 2006). In this context, community metrics can be a useful tool to help identify potential key species that might be more sensitive to subtle habitat changes (e.g. SIMPER analysis, Table 1, but also see Souza and Vianna, 2022). There is a growing urgency to scale-up coastal restoration efforts due to the immense value of ecosystem services provided by coastal systems (Waltham et al. 2020). Scaling-up comes with increased costs, so to maximize the benefits of investments in coastal restoration, it is critical that future restoration efforts learn from and build on a robust evaluation of the relative successes of different restoration strategies. Enhancing fish habitats is one of the more widely stated goals of nature-based restoration approaches, yet it is one of the more challenging outcomes to quantify. We suggest that, in cases where community metrics provide ambiguous or uncertain interpretations, additional metrics such as growth and condition of ecologically important species be further investigated and added to the available toolkit to assess the effects of restoration on nekton.

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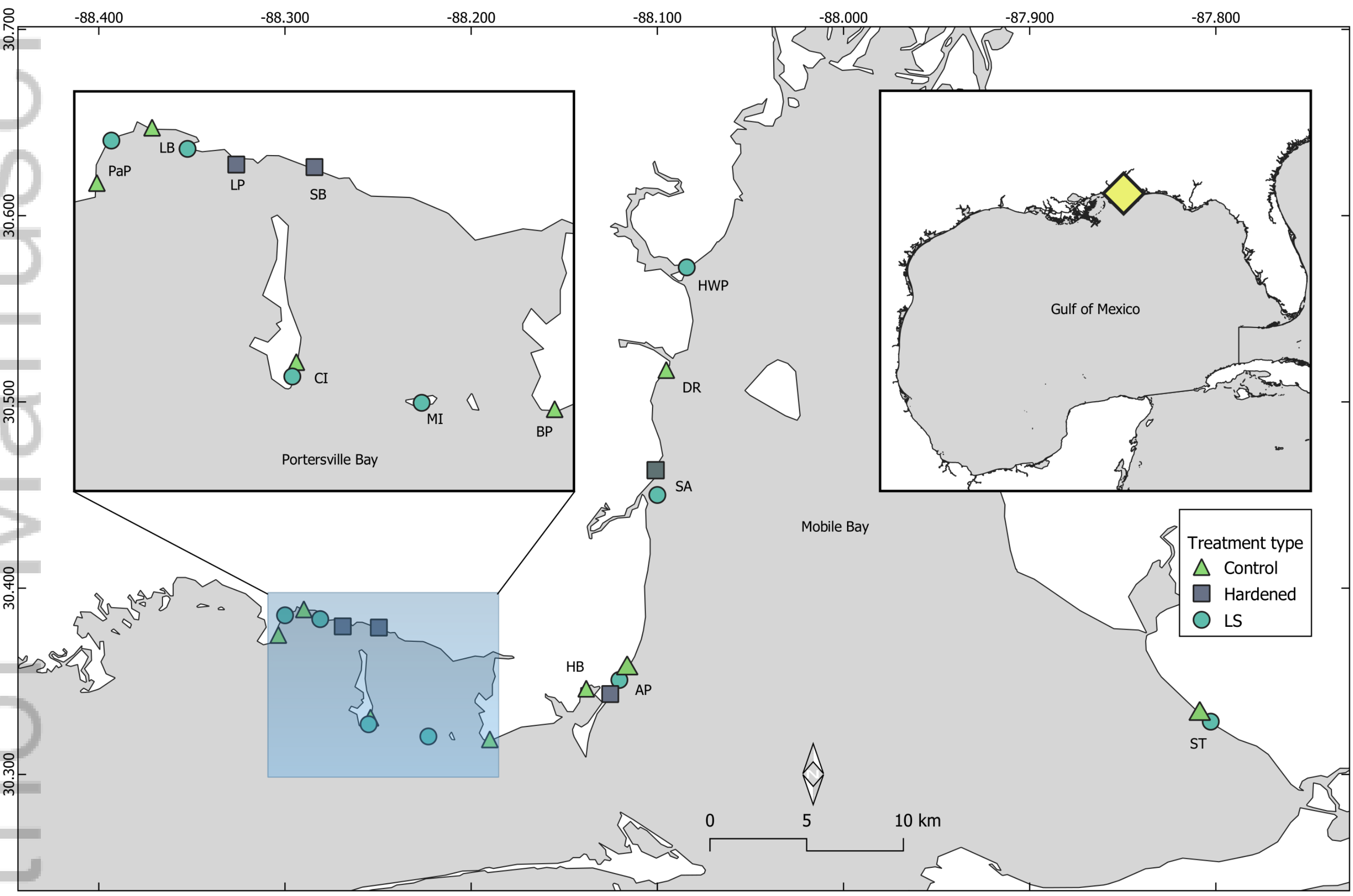
Table 1. Species that drive the differences between Control vs Hardened and LS vs Hardened sites indicated by SIMPER analysis. "Average" refers to the mean relative contributions of individual species to the overall dissimilarity between treatment types, and S.D. refers to the standard deviation.

	Control vs Hardened		**LS vs hardened**		**Average CPUE (ind/haul)**		
Species	**Average**	**S.D.**	**Average**	**S.D.**	**Hardened**	**LS**	**Control**
Litopenaeus setiferus	0.134	0.097	0.13	0.093	4.94	60	46.75
Farfantepaneus aztecus	0.075	0.058	0.105	0.069	0.777	13.61	11.95
Anchoa mitchilli	0.053	0.05	0.047	0.046	1.83	1.71	2.28
Arius felis	0.049	0.03	0.037	0.03	0.055	1.62	2.37
Bairdiella chrysoura	0.044	0.031	0.029	0.02	0.33	2.65	3.02
Lagodon rhomboides	0.036	0.025	0.034	0.027	0.11	6.68	2.15



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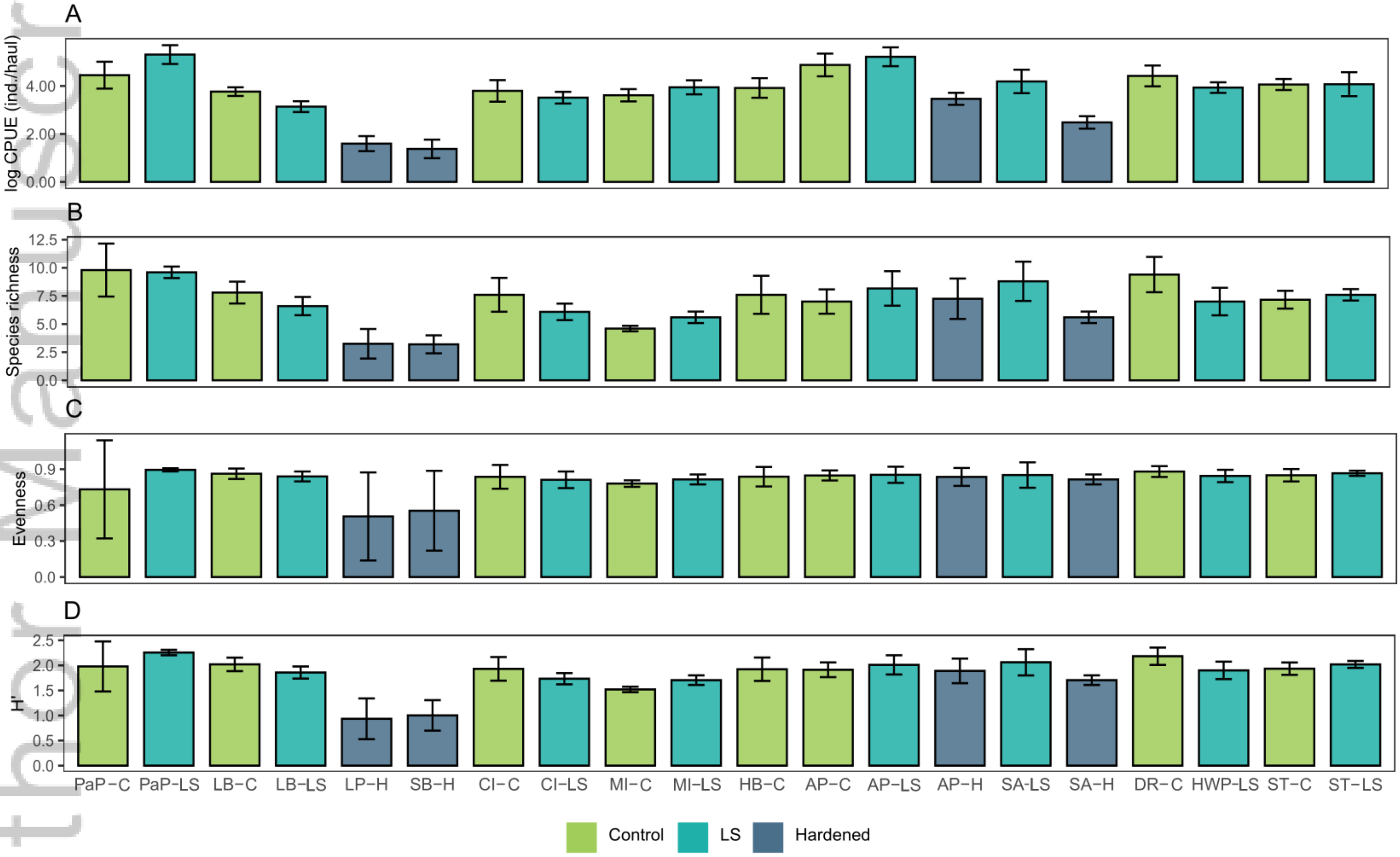

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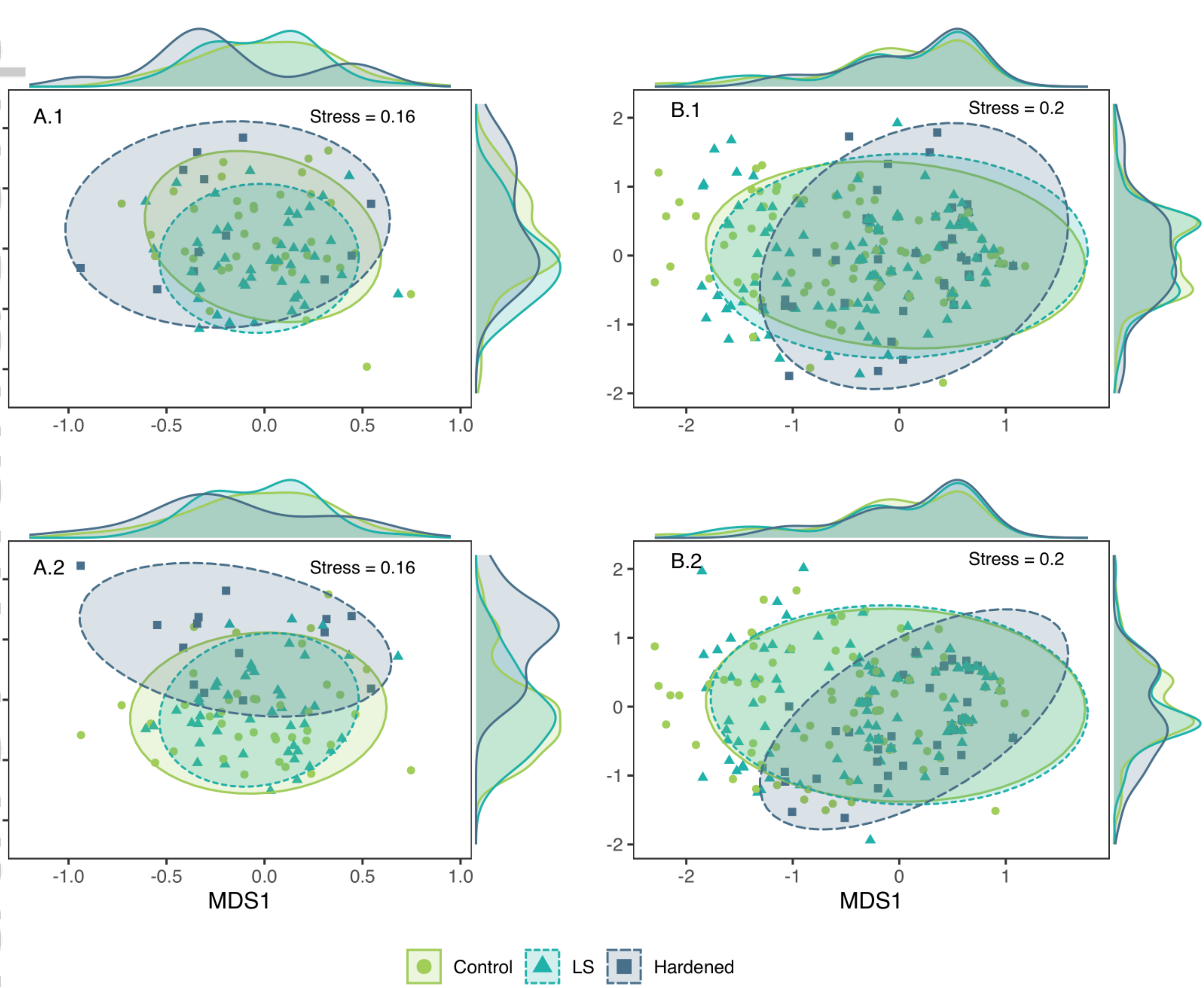

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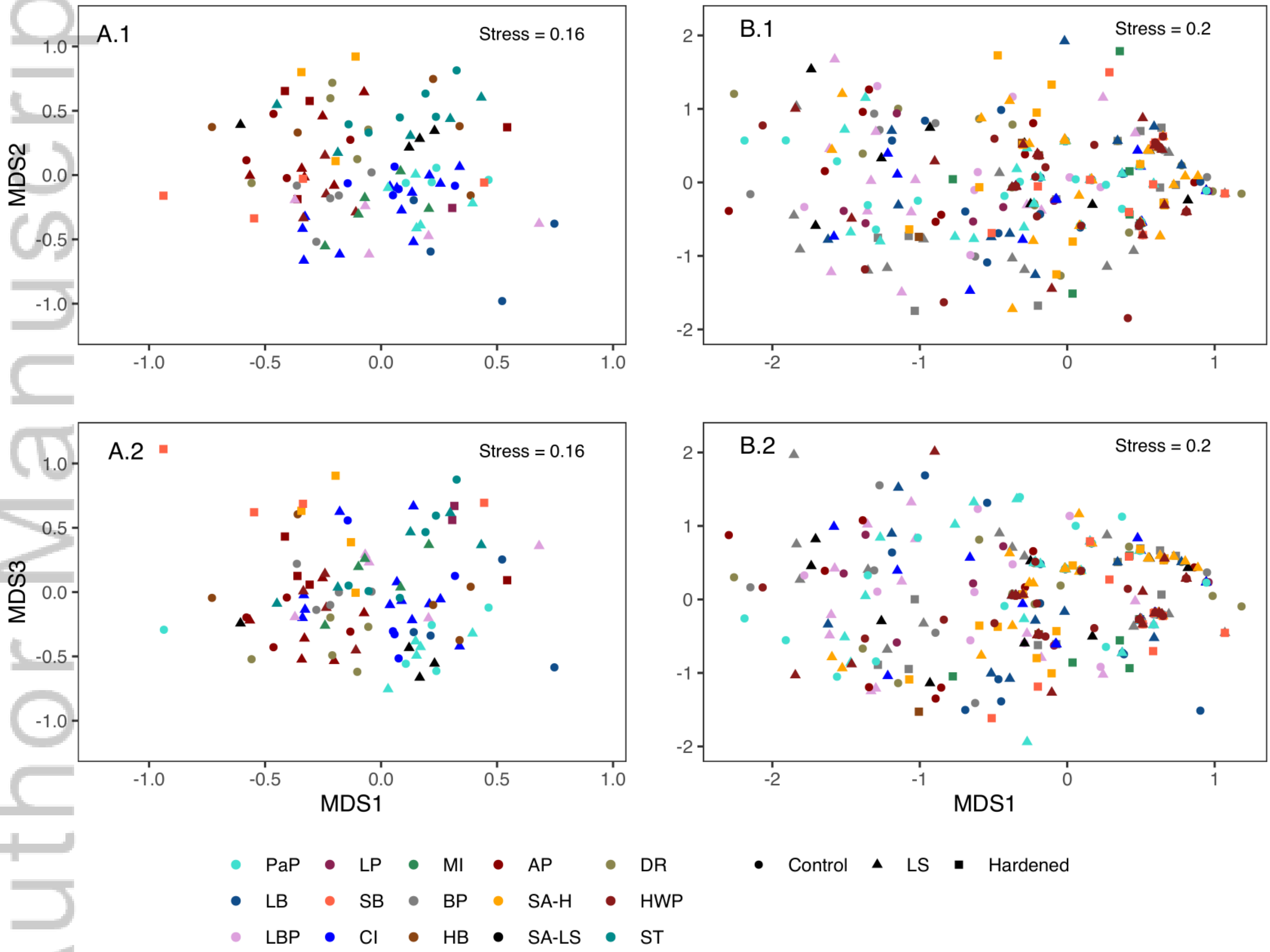

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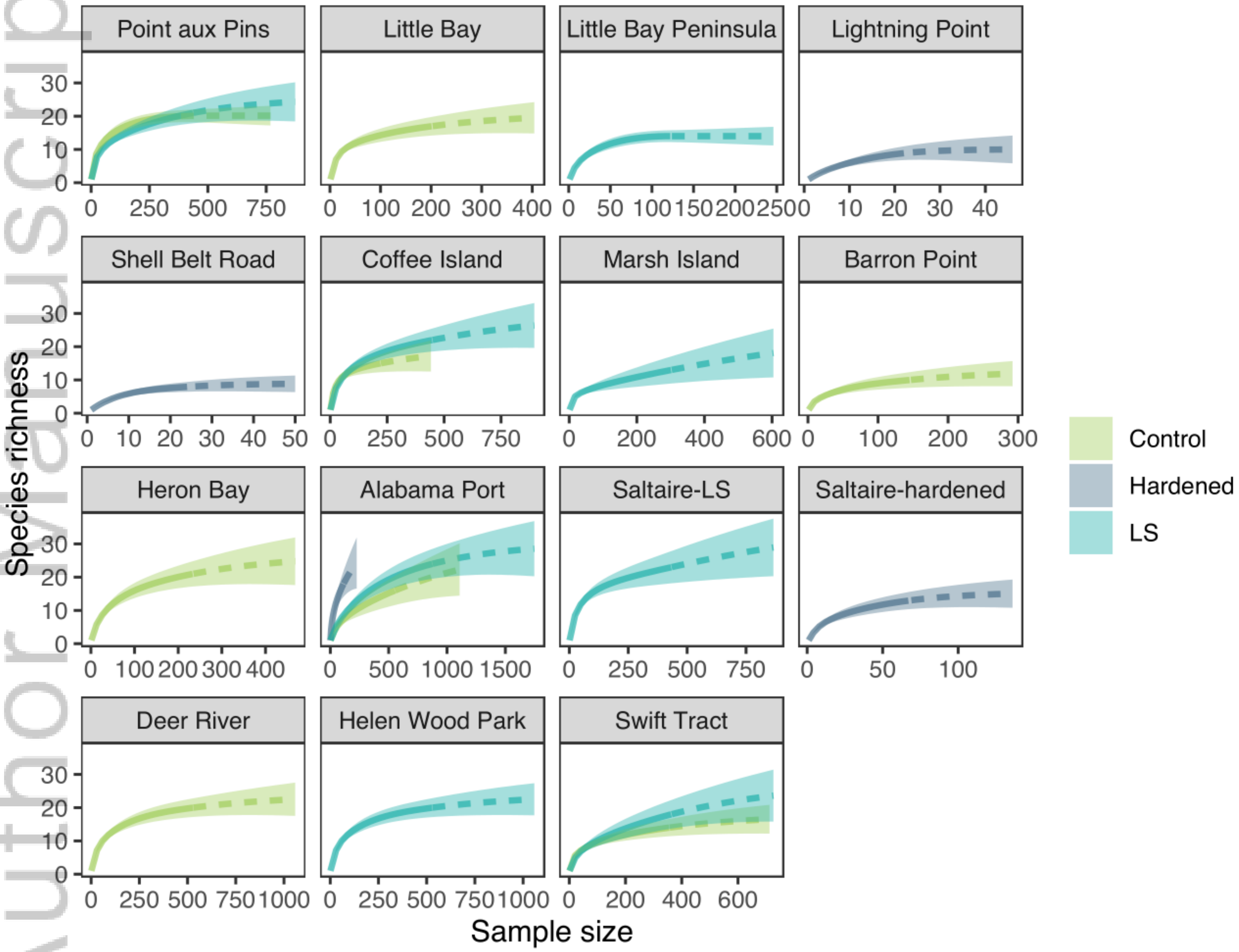

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