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Research article

Multi-site salt marsh restoration can recover key natural functions despite long-term structural deviations and site-level differences

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ABSTRACT

Restoration studies often focus on short-term structural recovery, while achieving consistent, long-term functional outcomes across diverse sites remains a challenge, limiting scalability. We evaluated the long-term structural and functional effectiveness (average match to reference natural sites) and outcome consistency (dispersion around the average) of two salt marsh restoration methods in the Venice Lagoon: RC which favours tidal creek formation at the low shore, and RB, which does not. Sampling was conducted at mid and low shore levels in restored marshes over 10 years old.

At the mid shore, neither method fully replicated the natural sediment characteristics, and restored sites often showed greater site-level differences than natural marshes. Nevertheless, average vegetation structure and function, including biomass and macrofaunal diversity, were comparable to natural sites. Soil carbon was also similar between restored and natural sites, although this likely reflects the carbon content of dredged sediments rather than new atmospheric CO_2 sequestration. In contrast, at the low shore, neither method supported the native cordgrass *Spartina maritima,* despite sediment characteristics more closely resembling natural conditions. Only the non-native *S. anglica* colonised the low shore at RC sites, exhibiting biotic and functional attributes similar to those of the widespread *S. anglica* stands at natural sites, while RB sites remained unvegetated.

These findings suggest that restoration methods promoting natural foreshore dynamics, such as RC, more effectively support vegetation establishment and replicate low-shore functions, even if these functions are delivered by a non-native species. We hypothesize that restoring a network of spatially diffuse sites may improve ecological outcomes by better replicating key natural structural and functional conditions, despite persistent structural deviations and elevated site-level differences.

1. Introduction

Large-scale ecosystem restoration is crucial for recovering lost or degraded functions and services, as well as for promoting resilience to intensifying global changes (Abelson et al., 2020; Suding, 2011). Restoration science is rapidly advancing, driven in part by global initiatives like the United Nations Decade of Restoration (UNEP, 2021) and policy frameworks, such as the EU Nature Restoration Law (EU, 2024). Long-term monitoring is essential for evaluating the effectiveness of different approaches and identifying the most cost-effective ones (Suding, 2011).

Restoration projects vary in their objectives and methodologies, ranging from preventive management aimed at reducing stress to full habitat reconstruction (Airoldi et al., 2021). These efforts often rely on the (re)establishment of prominent habitat-forming species, such as trees, corals, and seagrasses. The underlying assumption is that restoring these foundation species will also restore their associated functions and services (Bulleri et al., 2018). However, restored sites may not always

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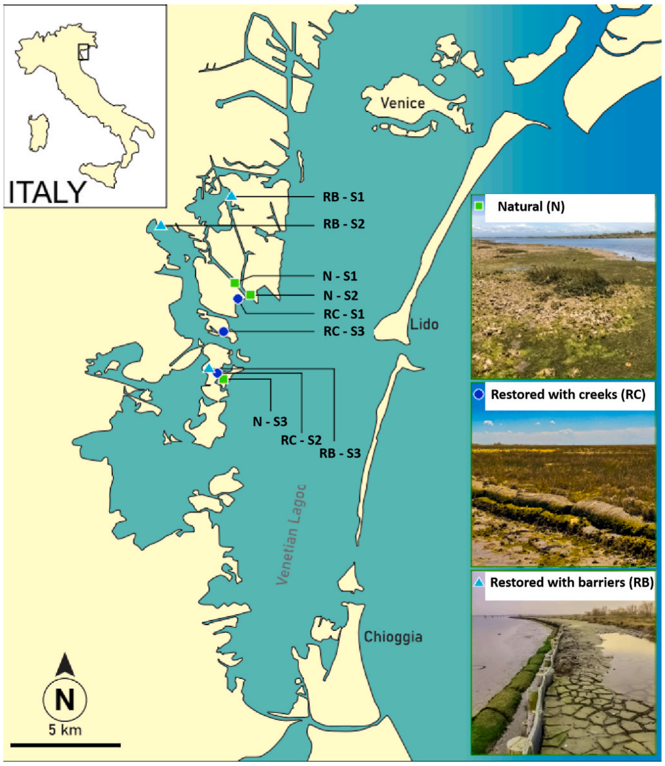

Fig. 1. Location of Natural (N), Restored with creeks (RC) and Restored with barriers (RB) salt marshes in the Venice Lagoon, Italy, with representative pictures.

progress towards the desired or anticipated state (e.g., Pilon et al., 2023). Additionally, native habitat recovery is increasingly challenged by rapid climate change, species invasions, and land use changes, as traits that optimize species' future survival may shift (e.g., increasing droughts favouring drought-resistant species; Laughlin et al., 2017). Consequently, restoration increasingly focuses on restoring ecosystem functions, which underpin services (Kollmann et al., 2016), even if the resulting ecosystem exhibit some degree of "novelty" (*sensu* Hobbs et al., 2009). In this context, even if restored sites do not fully replicate the structure or taxonomic composition of reference natural sites, restoration projects can still deliver intended benefits, particularly when the species recovered are functionally similar to the original targets (Brudvig et al., 2017; Isdell et al., 2021; Laughlin et al., 2017a). Yet, most studies focus on short-term structural recovery, while long-term data on functional attributes remain limited, leaving uncertainties about the sustained functionality of restored ecosystems.

Another important yet under explored factor that could influence restoration success is the variability in outcomes depending on how, where, and when restoration is conducted (Brudvig et al., 2017; Fraschetti et al., 2021). Currently, most restoration projects assess success by comparing average metrics of target conditions (e.g. species abundance) to those of reference sites or desired states (Brudvig et al., 2017). However, if restoration outcomes are variable, multiple restored sites may on average be comparable to reference habitats, despite they individually do not match target goals. Additionally, some sites may achieve restoration goals even if the overall intervention is deemed unsuccessful (Brudvig et al., 2017). Therefore, the dispersion of restoration outcomes around the mean provides a meaningful measure of consistency, that is how closely outcomes are clustered regardless of whether the restoration target is achieved. Such consistency in turn affects scalability, as consistent outcomes build confidence in the ability to reliably predict and replicate results at new sites, thereby supporting broader implementation. A recent meta-analysis of terrestrial systems highlighted that restored sites tend to exhibit consistently higher variability compared to reference sites (Atkinson et al., 2022), but little is known about the underlying causes. Far less is known about the replicability of restoration in coastal and marine environments, where restoration science and practice are less developed and outcomes remain uncertain (Bayraktarov et al., 2016). Potential sources of variation include the context of the restoration, the approach or implementation method used, or the outcomes considered (Fraschetti et al., 2021).

Salt marshes are key targets for marine ecosystem restoration (Moraes et al., 2022; Wang et al., 2022). Marsh vegetation provides multiple key ecosystem services, including erosion control and coastal protection, nutrient and contaminant removal, and carbon sequestration, but human activities and global changes have caused over 50 % habitat loss (Airoldi and Beck, 2007; Gedan et al., 2008). This is prompting investments in salt marsh restoration and protection (Bayraktarov et al., 2016; Wang et al., 2022). Marsh restoration has employed various approaches, such as de-embanking old pasture fields to restore tidal regimes and creating elevated intertidal areas using dredged sediment (Billah et al., 2022). However, as with other ecosystems, measuring restoration success in salt marshes has largely focused on average, short-term, structural-compositional recovery (e.g., plant species diversity and composition; Billah et al., 2022; Mossman et al., 2012; Wolters et al., 2005), with scarce long-term data on functional attributes or on the variability of outcomes across multiple restored sites (Morris et al., 2022).

We quantified the long-term effectiveness (average match to reference natural sites) and consistency (dispersion around the average) of restoration outcomes across different sites and restoration methods in the Venice Lagoon (Italy). Here, over 11 km^2 of marshes have been restored since 1985 using sediments dredged from navigation channel maintenance and containing barrier structures (Scarton et al., 2013). We compared structural (sediment characteristics, vegetation coverage and above- and below-ground biomass) and functional (soil carbon, chlorophyll *a* in sediments, and supported macrobenthos diversity) outcomes at replicated restored marshes over 10 years old across two restoration methods: RC which favours the formation of tidal creeks and mudflats at the salt marsh edge due to partial barrier removal over time, and RB which does not due to continuous edge barriers. Our goals were to determine which method better matches natural marsh conditions and yields more predictable outcomes. We hypothesised that RC marshes will align more closely with natural conditions because of their greater tidal flat dynamism, while RB marshes will show greater outcome consistency due to their more uniform edge design. We also hypothesised that the variation among multiple restored sites would be greater at the low than at the mid shore, reflecting the different marsh edge designs, and more variation in structural than functional parameters, as different vegetation types can support similar functions.

2. Materials and methods

2.1. Study area

The Venice Lagoon (Fig. 1), the largest Mediterranean lagoon (550 km^2, average depth 1.2 m) is renowned for its natural and cultural heritage. It is a microtidal system (average tidal range 60 cm, up to 120 cm during spring tides), characterized by strong physical and chemical gradients (Solidoro et al., 2010).

Humans have largely shaped the lagoon, including the diversion of rivers in the 16th and 17th centuries to prevent sediment infilling of navigation channels (Solidoro et al., 2010). The long-term effects of sediment starvation, salinity modifications, urban sprawl, land subsidence, and sea level rise have caused a loss of salt marshes from 170 km^2 to 37 km^2 (Tommasini et al., 2019). To restore marsh ecosystems, over 80 tidal islands were created since 1985 using sediments dredged from shipping canals (Scarton et al., 2013). The sediments varied in grain size, from silt to sand, and were contained using rows of logs (~30 cm in

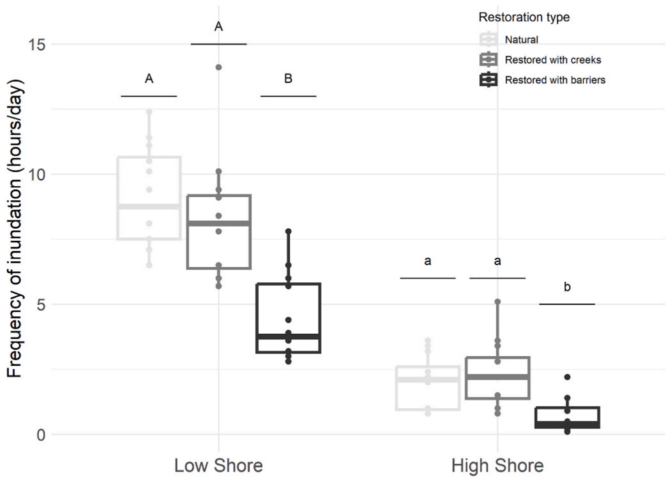

Fig. 2. Inundation frequency (hours) at the low and mid shores of Natural (N), Restored with creeks (RC) and Restored with barriers (RB) marshes. Data are means ± 1 SE (4 plots for each of 3 sites, n = 12).

diameter) and/or 2–3 rows of rock gabions, often combined with geotextiles. Portions of these containing structures were subsequently removed from some islands to promote more natural tidal flat evolution and the formation of tidal channels (Scarton et al., 2013). The exact timing of marsh construction and partial barrier removal is not publicly documented. Using google maps imagery we estimated that all restored marshes were at least 10 years old (which are considered stable marshes according to Tagliapietra et al., 2018), with most marshes likely between 15 and 25 years old. Barrier removals appeared to have occurred several years before our sampling, allowing sufficient time for tidal creek development.

We considered three types of salt marshes (Fig. 1).

1. Natural (N): marshes without visible containment structures, either due to a lack of recent human interventions or because of natural evolution following earlier restoration.
2. Restored with creeks (RC): marshes with partial removal of the containment barriers (log poles, gabions, or both) along the low shore to promote tidal creeks and mudflats at the marsh edge.
3. Restored with barriers B (RB) marshes with intact containment, resulting in abrupt margins at the low shore.

We identified three marshes with comparable hydrological settings for each category within a 3 × 10 km area. At the mid shore (located further inland far from the influence of the edge structures) all the three typologies of marshes were densely covered by mixed vegetation, primarily *Limonium narbonense* Mill., *Puccinellia palustris* (Host) Parl, and *Salicornia fruticosa* (L.) L. At the low shore (located near the marsh edge) the habitat was more heterogeneous both within and across sites (see session 3. Results). The prevailing habitat was unvegetated mudflats, interspersed at some sites with small patches (about 0,5-2 m diameter) of either the native cordgrass *Spartina maritima* (Curtis) or the non-native *S. anglica* (Hubb) (syn.: *Sporobolus maritimus* and *S. anglicus*, Peterson et al., 2014; here Bortolus et al., 2019). *S. anglica* has largely established in the Venice Lagoon (Wong et al., 2018), replacing the native *S. maritima*, highly vulnerable to climatic and other global changes (Strain et al., 2017; Wong et al., 2015).

2.2. Sampling

Between May and July 2021, we sampled three replicated marshes for each category: N, RC and RB. Each marsh was sampled at both mid and low shore levels, using replicated quadrats (0.5 × 0.5 m), which were randomly interspersed within each shore level, and placed at least 10 m apart. At the mid shore, all quadrats were located within dense vegetation, and all parameters (structural and functional) were estimated in 4 replicated quadrats per marsh. At the low shore our original intention was to stratify sampling according to the two vegetation types, *S. maritima* and *S. anglica*. However, *S. maritima* was found only at N sites, *S. anglica* only at N and RC sites, and no *Spartina* species were found at RB marshes (see session 3. Results). To provide the most balanced comparison possible across N, RC, and RB marshes under these constraints, we sampled the physical structural parameters from 4 replicate unvegetated quadrats per marsh, while the biotic and functional parameters were sampled from 4 replicate *S. anglica* vegetated quadrats for N and RC sites only.

We recognize that there is some inconsistency in the literature regarding how certain variables, particularly standing biomass, are categorized, which can understandably lead to differing interpretations. Some authors (e.g., Loreau et al., 2001; Bayraktarov et al., 2020) consider standing biomass as a functional variable, as it reflects ecosystem productivity. However, others classify it as structural variable (e.g., Dixon et al., 1994; Xu et al., 2021), emphasizing its status as a static measure at a given point in time. In our study, we adopted the latter interpretation and treated standing biomass, vegetation cover, and canopy height as structural variables. In contrast, we considered soil carbon stock, chlorophyll-a (Chl-a), and macrofaunal diversity as proxies for ecosystem function. These metrics reflect underlying ecological processes such as carbon accumulation, primary productivity, and habitat provisioning, consistent with previous studies (Bayraktarov et al., 2020; Eppley et al., 1985; Friedland et al., 2012; Schmidt et al., 2011; Lal, 2004).

In each quadrat, surface elevation (marsh height) was measured using real-time kinematics (RTK) GPS (Glonass GNSS) and referenced to the Italian Ordinance Datum (IOD, Genoa, 1942). Marsh height was used to calculate the daily frequency of inundation of the low and mid shore at each site, based on average tidal levels from 2020 to 2022 from the Piazza Vigo tidal gauge station of Chioggia (https://www.venezia.isprambiente.it) referenced to IOD. For each plot, we calculated the average daily time (in hours) that the tidal level exceeded the plot height over a three-year period.

In each vegetated quadrat we recorded cover (%) and canopy height, measured using a Styrofoam "drop disc" (~80 g, diameter: 30 cm) and a calibrated stick as a ruler (Stewart et al., 2001). We then clipped all the aboveground biomass (AGB) and collected three sediment cores (20 cm in depth and 9 cm diameter) for analyses of belowground biomass (BGB), macrobenthos and physicochemical parameters. Surface sediments (top 3 cm) were collected using a sterile, modified syringe, wrapped into an aluminium foil wrap, packed into a cool icebox and stored at −20 °C in the laboratory for chlorophyll *a* analyses.

2.3. Laboratory analyses

AGB and BGB samples were washed on a 0.5 mm sieve to remove all sand and silt, oven-dried at 105 °C until constant weight, and weighed to the nearest 0.001 g. Macrobenthos samples were washed on a 500 μm screen. The retained organisms were preserved in 80 % alcohol for later sorting, counting and identification to the lowest practicable taxonomic level under a dissecting microscope. Macrobenthos diversity was computed using the Shannon-Wiener diversity (H) index.

Sediment cores were air dried in the laboratory and passed through a 500 μm sieve to discard gravels, plant materials and mollusc shell. Sediment salinity (as electrical conductivity) and pH were measured in a 1:5 (w/v) soil-distilled water extract, using a multiparametric probe (Hanna HI9829) and a pH meter (CRISON, Basic 20) respectively. Bulk density (BD) was obtained as:

$$\mathrm{BD}\ (\mathrm{g\ cm}-3) = \frac{\mathit{Oven\ dried\ sediment}\ (g)\ (\mathit{dried\ at}\ 105\ ^{\circ}C)}{\mathit{Volume\ of\ sediment}\ (cm^{3})}$$

Sediment sub-samples were ignited in a muffle furnace for 4 h at

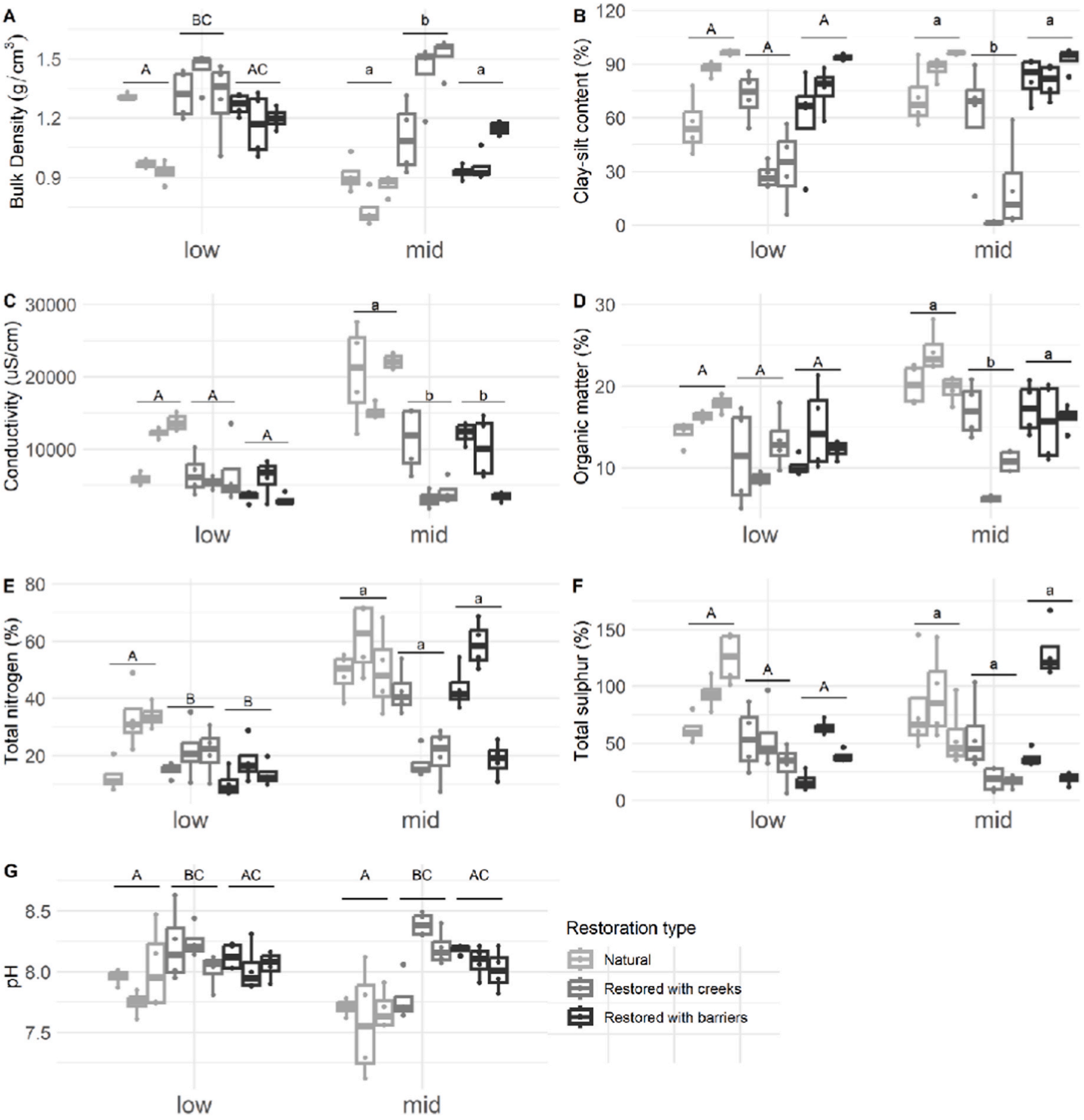

Fig. 3. Bulk density (gcm^{-3}; A), % clay-silt (B), conductivity (i.e. salinity uS/cm; C), %OM (D), total nitrogen (%; E), total sulphur (%; F) and sediment pH (G) at each of 3 sites (same colour bars) at the low and mid shores of Natural (N), Restored with creeks (RC) and Restored with barriers (RB) marshes ($n = 4$). Letters indicate significant differences among restoration types ($\alpha = 0.05$; Tukey's test).

550 °C to determine the organic matter content (% OM), calculated as the difference between the initial weight (Wi) and final weight (Wf) after combustion: $(Wi - Wf)/Wi$. Organic carbon (%C) was determined using the equation $\%C = 0.37 \times (OM) - (0.0024) \times OM$ (Puppin et al., 2023). Sediment carbon stocks (SCS) of top 20 cm was calculated following Howard et al. (2014) and expressed as Mg ha^{-1} of C:

$$\text{Sediment carbon stocks (SCS)} = \text{BD} \times (\%\text{C} / 100) \times 20$$

The combusted sediments were sieved (<63 μm) to quantify the mud content expressed as % clay-silt content. Chlorophyll *a* (Chl *a*) in the frozen sediments was extracted using 10 mL of 90 % acetone, quantified spectrophotometrically and expressed as micrograms per gram (μg/g). Total nitrogen, TN and total sulphur, TS contents (expressed ass % dry weight of the sediment) were determined using a CHNOS Elemental Analyzer.

2.4. Statistical analyses

We tested if sediment characteristics (bulk density, salinity, sediment pH, % mud fraction, and % OM, %TN and %TS) in restored marshes matched those in natural marshes and how this varied by shore level using linear mixed-effect models with the *lmer* function (lme4 package; Bates et al., 2015), with restoration (3 levels: N, RC, and RB) and shore level (two levels: low and mid shore) as fixed factors. Replicated marshes nested within restoration type were included as a random intercept. When the restoration type was statistically significant, we ran Tukey post-hoc comparisons using the *emmeans* package (Lenth, 2024).

We also tested if the dominant vegetation in restored marshes matched that in natural marshes in terms of structural (% cover, canopy height, AGB and BGB) and functional (soil carbon stock, Chl *a,* H' diversity of macrobenthos) characteristics. Because of the unbalanced distribution of the vegetated habitats, we ran separate analyses for the mid and the low shores. For the mid shore, we used mixed-effect models

Table 1
Linear mixed-effect models of the effects of restoration (Natural N, Restored with creeks RC and Restored with barriers RB) and shore level (mid and low shore) on: BD = bulk density; CD = electric conductivity (i.e., salinity); OM = organic matter.

Response variable	Predictor	SS	MS	Num. def.	Den. d.f.	F-value	P-value	mR^2	cR^2
BD									
	Restoration	0.321	0.160	2	6	9.699	0.013	0.57	0.74
	Shore	0.385	0.385	1	60	23.301	<0.001		
	Restoration:Shore	0.186	0.93	2	60	5.618	0.006		
Mud %									
	Restoration	2490.5	1245.2	2	6	5.379	0.046	0.46	0.79
	Shore	18.33	18.3	1	60	0.079	0.779		
	Restoration:Shore	2229.3	1114.6	2	60	4.815	0.011		
CD									
	Restoration	$254*10^6$	$127*10^6$	2	6	11.04	0.01	0.60	0.72
	Shore	$342*10^6$	$342*10^6$	1	60	28.67	<0.001		
	Restoration:Shore	$226*10^6$	$113*10^6$	2	60	9.80	<0.001		
pH									
	Restoration	0.676	0.338	2	6	7.73	0.022	0.36	0.47
	Shore	0.139	0.139	1	60	3.17	0.080		
	Restoration:Shore	0.249	0.125	2	60	2.85	0.065		
OM									
	Restoration	179.92	89.96	2	6	8.67	0.017	0.49	0.62
	Shore	176.55	176.55	1	60	17.02	<0.001		
	Restoration:Shore	78.29	39.14	2	60	3.77	0.029		
Total Nitrogen									
	Restoration	0.070	0.035	2	6	3.11	0.118	0.50	0.68
	Shore	0.722	0.722	1	60	64.67	<0.001		
	Restoration:Shore	0.146	0.073	2	60	6.53	0.003		
Total Sulphur									
	Restoration	1.51	0.756	2	6	2.81	0.138	0.29	0.61
	Shore	0.46	0.460	1	60	1.71	0.196		
	Restoration:Shore	1.99	0.996	2	60	3.70	0.031		

All models include site nested within Restoration as random factor. CD and Total sulphur were square-root and log transformed, respectively. mR^2 = marginal R^2, cR^2 = conditional R.^{2}.

with restoration (three levels: N, RC, and RB) as the main effect and site nested within restoration as a random effect. When restoration was statistically significant, we ran a Tukey HSD post hoc comparison. For the low shore, comparisons focused on *S. anglica* vegetation only, restricting the analysis to RC vs N sites. No comparison was possible for *S. maritima*.

Sediment salinity, BGB (mid shore), and Chl *a* were square-root transformed and canopy height (mid shore), AGB (low shore, *S. anglica*), and Chl *a* (low shore) were log transformed to meet normality assumption. For macrobenthos diversity (H′), bulk density, and mud content, normality assumptions were not met even after transformation. In these cases, we used PERMANOVA which requires no assumptions about normality (Anderson, 2017). Since results were consistent between the PERMANOVAs and the mixed-effect models, we reported the latter for ease and consistency of interpretation.

To evaluate the consistency of different restoration methods, we quantified the variation of each structural and functional property in RC and RB and compared it to the natural levels of variability in N. For each marsh type, we calculated the coefficient of variation (CV = mean/standard deviation) and the natural logarithm of the CV ratio (lnCVR): the CV of each structural or functional property of restored sites (RC or RB) was the numerator, while the average CV of N sites was the denominator, serving as reference natural variability. LnCVR is similar to the log response ratio (Atkinson et al., 2022), commonly used to quantify an experimental effect relative to the control (Harris et al., 2014), and values above or below 0 indicate higher or lower variability, respectively, in restored marshes compared to natural marshes. We then computed the 95 % confidence intervals for each parameter to investigate if LnCVRs differed from 0 (i.e., from the reference natural marshes). All statistical analyses were performed in R version 4.2.1 (R Core Team, 2022). Figures were created using the *ggplot2* package (Wickham, 2016).

3. Results

3.1. Effectiveness of restoration

RC marshes experienced similar inundation frequencies to N marshes in both the low and mid shores, while RB marshes, located at higher elevations within the tidal range, experienced less frequent inundation compared to both N and RC (Fig. 2).

Sediment characteristics varied across restoration types (Fig. 3). These differences were most pronounced in RC than RB and at the mid than the low shore, as evident from the Restoration type and Shore level interactions (Table 1; Tables S1 and S2). At the mid shore, RC exhibited the highest bulk density (BD) and the lowest mud (% of clay and silt) and organic matter (OM) contents. Sediment pH was also higher in RC compared to N but not to RB. Sediment salinity was consistently lower in RC and RB, which had similar values, than in N. Total nitrogen and sulphur levels were comparable among N, RC and RB. At the low shore zone, both RC and RB had lower nitrogen levels compared to N, and RC showed higher bulk density and pH than N. All other parameters were similar across N, RC and RB.

There were also differences in the biotic characteristics of the restored sites, particularly at the low shore. Here neither restoration method supported the native *S. maritima,* which was found only in N sites. RC sites were vegetated by the non-native *S. anglica*, which was widespread at both N and RC. In contrast, RB sites were virtually devoid of cordgrass vegetation.

When we compared the structural and functional parameters of the vegetated habitats separately for the mid and low shores, we found that at the low shore zone, where only N and RC could be compared, the structural (Fig. 4) and functional (Fig. 5) parameters associated with *S. anglica* were similar between RC and N (Table 2; Supplementary materials, Table S3). At the mid shore zone, where a full comparison was possible between N, RC and RB, only RB had lower belowground biomass (BGB; Table 2; Table S4) than N, while all other parameters were comparable between RC, RB and N.

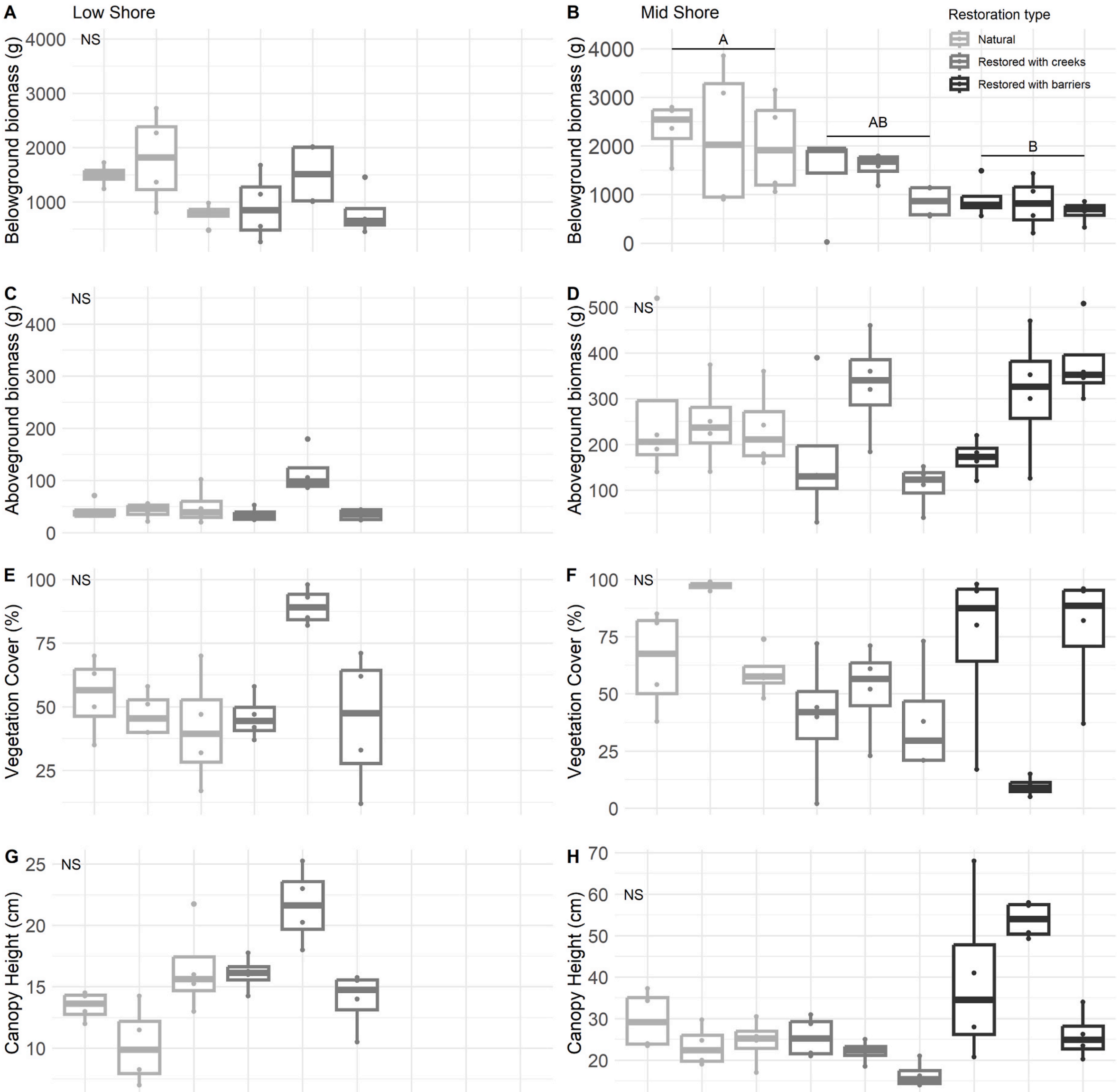

Fig. 4. Below ground biomass (BGB, g m-2; A and B), above ground biomass (AGB, g m-2; C and D), vegetation cover (%) (E and F), and canopy height (cm; G and H) at the low and mid shores of Natural (N), Restored with creeks (RC) and Restored with barriers (RB) (n = 4). Letters indicate significant differences among restoration types (α = 0.05; Tukey's test).

3.2. *Consistency of restoration*

Significant variability was observed among replicated marshes across restoration types for multiple structural and functional parameters (Figs. 3–5), as shown by the differences between marginal and conditional RB of the mixed-effect models (Tables 1 and 2). In RC, this spatial variability often exceeded the natural range (Fig. 6, Supplementary materials Table S5). In RC, sediment salinity and clay-silt content were consistently more variable than natural in both low and mid shores. Additionally, BD, OM, sulphur content and BGB were more variable than natural at the low shore, and Chl *a* and vegetation cover were more variable than natural at the mid shore. However, some parameters, such as sediment pH at the mid shore, and canopy height, AGB, and macrobenthos diversity at the low shore, showed less variability than natural.

In RB, the trends differed between the low and mid shores. At the low shore, RB marshes were consistently devoid of cordgrass vegetation, yet the physical parameters — such as BD, salinity, OM, and total nitrogen—were generally more variable than natural. In contrast, at the mid shore, RB sites displayed less variability than natural for BD, pH, total sulphur, and AGB, but greater variability in vegetation cover and Chl *a*.

4. Discussion

In the Venice Lagoon, after more than 10 years, neither restoration method fully replicated the sediment and biotic characteristics of natural marshes or supported the native cordgrass *S. maritima,* with site-level differences often exceeding natural levels. Nonetheless, at the mid shore the restored vegetation generally aligned with most natural biotic structural and functional parameters. At the low shore, only RC sites, which prompted more natural tidal foreshore dynamic, supported the establishment of vegetation, albeit represented by the non-native *S. anglica*, exhibiting biotic and functional attributes comparable to those of the widespread *S. anglica* meadows at natural sites, while RB sites remained unvegetated.

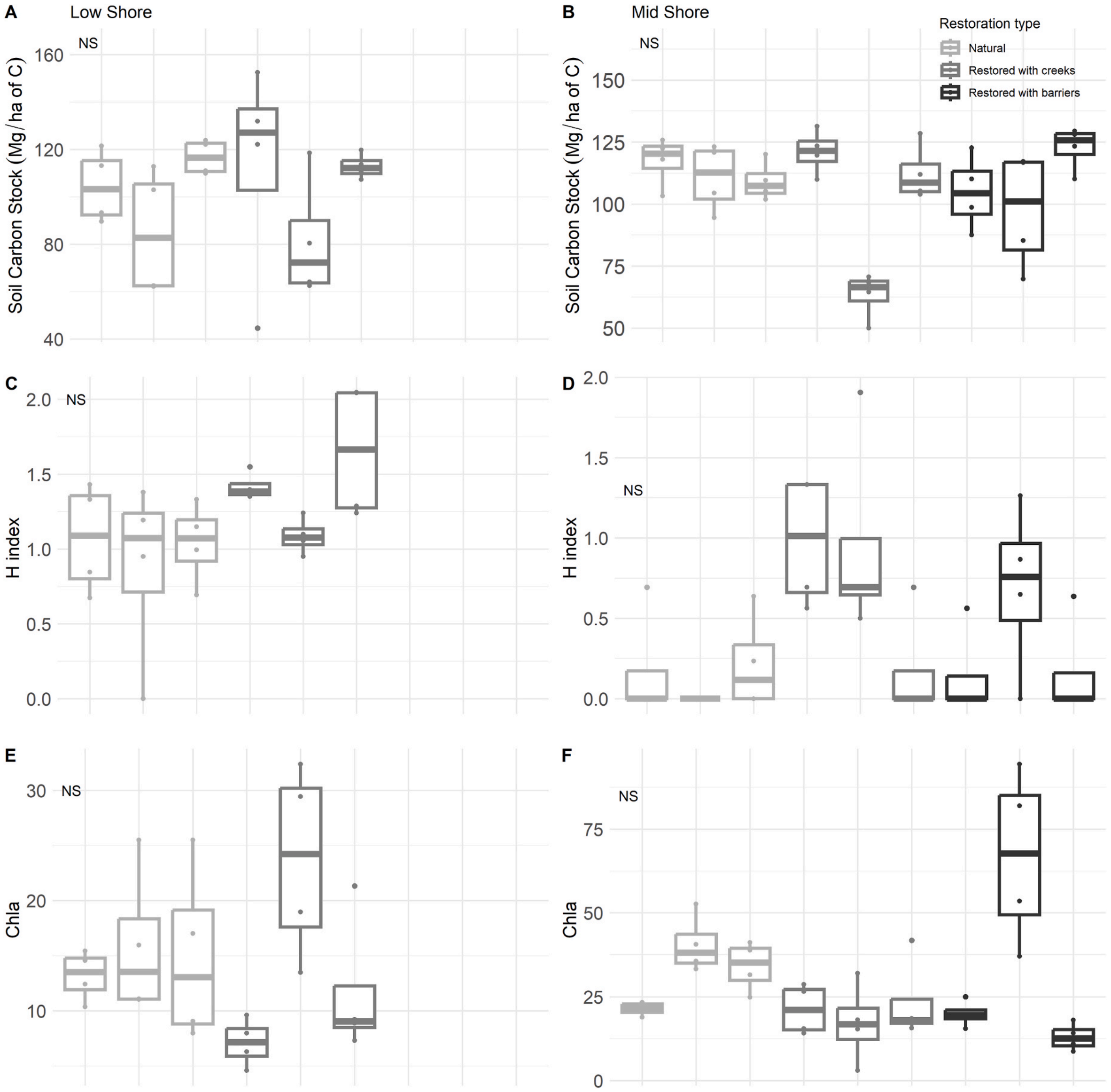

Fig. 5. Sediment carbon stocks (SCS, Mg C ha^{-1}; A and B), Shannon diversity index (H') of the macrobenthos (C and D) and Chlorophyll *a* (Chl *a*, μg/g dry weight; E and F) at the low and mid shores of Natural (N), Restored with creeks (RC) and Restored with barriers (RB) marshes (n = 4).

4.1. Structural and functional effectiveness

Dredged sediments are widely employed to create artificial salt marshes, which can cause long-lasting differences in sediment characteristics between restored and natural marshes (Atkinson, 2003; Isdell et al., 2021), as also observed in our study particularly at the mid shore. Contrary to initial predictions, RC marshes exhibited the greatest deviations from natural conditions, despite their more natural positioning within the tidal range. Both types of restored marshes showed lower sediment salinity and nitrogen levels compared to N marshes at the mid and low shores, respectively. Additionally, RC marshes showed higher bulk density and pH at both shore levels, along with lower mud and organic matter content at the mid shore. These discrepancies and the greater variability among restored sites likely stem from variable sediment sources and differing containment margin designs affecting sediment dynamics.

The sediments used for marsh restoration were dredged from the nearby navigation channels to maintain ship traffic (Scarton et al., 2013). Depending on where these sediments were dredged, their grain size can vary highly, from mud to sand up to 97 % (D'Alpaos et al., 2010; Scarton et al., 2013). Coarser sediments enhance water drainage, oxygenation, and redox status, promoting salt flushing and higher decomposition rates of litter and organic matter (Abbott et al., 2019). Thus, initial differences in sand content possibly explain the greater among-site variability in sediment characteristics of RC and RB compared to N, along with other differences among restoration types, such as higher BD and lower mud and organic content in RC and lower salinity in both RC and RB. Furthermore, design differences may have also contributed to shape sediment properties. Removing barrier sections in RC marshes reinstated connection with tidal movement and sediment erosion/deposition dynamics. Thus, in RC tidal influence may have increased sand erosion or allowed for greater deposition of sediments, increasing site level variability in sediment grain size.

Despite the notable differences in sediment characteristics, RC and RB marshes displayed vegetation structure and functions like those of comparable natural vegetated habitats. The only consistent difference was the lower belowground biomass of mid shore vegetation in RC and RB marshes. Lower BGB was likely due to higher soil salinity at natural sites, in line with observations of *S. anglica* in Wales (UK) and *S. alterniflorus* in the USA, where elevated salinity prompted greater

Table 2
Linear mixed-effect models of the effects of restoration (Restored with creeks (RC) and Restored with barriers (RB) at the mid shore and N vs RC at the low shore) on: BGB = belowground biomass; AGB = aboveground biomass; VC= Vegetation cover; height = canopy height; SCS = sediment carbon stocks; diversity = macrobenthos Shannon diversity index (H'); Chl *a* = Chlorophyll *a*.

Response variable	Shore level	Predictor	SS	MS	Num. d.f.	Den. d.f.	F-value	p-value	mR^2	cR^2
BGB										
	Low	Restoration	165090	165090	1	4	0.56	0.495	0.04	0.35
	Mid	Restoration	2033.7	1016.9	2	33	9.79	<0.001	0.36	0.36
AGB										
	Low	Restoration	0.045	0.045	1	4	0.23	0.654	0.02	0.51
	Mid	Restoration	16680	8340.2	2	6	0.64	0.557	0.06	0.32
Vegetation cover										
	Low	Restoration	182.66	182.66	1	4	0.65	0.466	0.06	0.53
	Mid	Restoration	1185	592.51	2	6	1.14	0.382	0.13	0.56
Canopy height										
	Low	Restoration	12.487	12.487	1	4	1.71	0.261	0.17	0.69
	Mid	Restoration	0.452	0.226	2	6	3.60	0.094	0.32	0.63
SCS										
	Low	Restoration	12.722	12.722	1	4	0.01	0.907	0.001	0.59
	Mid	Restoration	751.59	375.79	2	6	0.66	0.549	0.10	0.67
Macrobenthos H'										
	Low	Restoration	0.665	0.665	1	4	4.92	0.091	0.21	0.28
	Mid	Restoration	0.716	0.358	2	6	2.343	0.177	0.20	0.45
Chl *a*										
	Low	Restoration	0.033	0.033	1	4	0.20	0.674	0.02	0.47
	Mid	Restoration	1.078	0.539	2	6	0.55	0.603	0.09	0.69

All models include site nested within Restoration as random factor. At the mid shore BGB and Chl a were square-root transformed and canopy height was log transformed. At the low shore, AGB and Chl *a* were log transformed. mR^2 = marginal R^2, cR^2 = conditional R^2.

investment in belowground structures (Alldred et al., 2017; De Battisti et al., 2020). Artificial marshes can develop similar vegetation cover and biomass to natural marshes within just five years despite significant differences in tidal level and sediment characteristics (Abbott et al., 2019). Our study corroborates this pattern in the micro-tidal environment of the Venice Lagoon.

Interest in salt marsh restoration and protection is high due to their potential for carbon storage (Duarte et al., 2013). The sediment carbon stocks (SCS) of restored marshes were comparable to those of natural marshes and aligned with broader natural estimates for the Venice area (Puppin et al., 2023), Australia (Macreadie et al., 2017), and global averages (Duarte et al., 2013). This is noteworthy because marshes restored with dredged sediments often exhibit lower SCS than natural sites, even after >30 years (Abbott et al., 2019), while marshes restored by managed realignment may require over 100 years (Burden et al., 2019). In our study, however, restored marshes reached SCS levels comparable to both natural and global benchmarks just over 10 years after restoration. This apparent rapid recovery is striking but cannot be accounted for by the relatively low carbon accumulation rates typically observed in the area (on average 0.85 Mg ha-1 of C per year; Puppin et al., 2023). Rather, the high SCS values likely reflect the carbon already present in the dredged sediments at the time of deposition. As such, while restored marshes may appear to provide substantial carbon storage, this represents a redistribution of existing carbon within the system rather than newly sequestered atmospheric CO_2. This distinction is essential when interpreting carbon stocks as indicators of functional recovery and in evaluating the climate mitigation potential of restoration strategies involving dredged materials.

The sediment macrofauna diversity (*H'*) was similar, on average, between restored and natural marshes, consistent with studies showing that macrobenthos typically require 5–25 years to fully recover (Ferguson and Rakocinski, 2008). However, restored sites showed considerable variability, with some developing richer or poorer communities than natural ones. This suggests that sediment characteristics may have influenced macrobenthos recovery at specific sites, but overall, restoration was successful. This supports the idea that restoration success should be evaluated at multiple scales: while local conditions may hinder recovery at individual sites, large-scale restorations encompassing diverse conditions can achieve overall success. This underscores the importance of considering scale and context in restoration (Fraschetti et al., 2021).

Edge design had a significant impact on the distribution of low shore vegetation. Neither method supported the native *S. maritima* cordgrass, but RC sites supported stands of *S. anglica*, with structural and functional characteristics similar to *S. anglica* stands at natural sites. Salt marshes are highly dynamic systems, with periods of range extension and contraction. The use of barriers to contain dredged sediments fails to account for this dynamism, potentially disrupting natural connectivity between the front and the backshore and limiting tidal flux, as well as the dispersal of plant seeds (Bouma et al., 2014). These effects could be particularly detrimental to the native *S. maritima*, which colonises new areas through seeds (Castillo et al., 2010). In contrast, *S. anglica* has a greater capacity for colonisation through both seeds and vegetative dispersion (Holmer et al., 2002), which may explain its growth in RC sites, where the partial removal of barriers restored some tidal flux.

The absence of *S. maritima* at the restored sites signals that restoration interventions may not fully reestablish native vegetation communities, and can, inadvertently promote non-native species such as *S. anglica*. The presence and spread of non-native species (neobiota) is a challenge for nature conservation and ecosystem restoration, which increasingly incorporate some degree of "novelty" (sensu Hobbs et al., 2009). Ecological restoration often involves interventions, such as soil disturbance, that create open niches and bare ground. Without careful consideration of landscape-level propagule pressure, seedbank composition, or ecological filters, these disturbed restored sites may inadvertently facilitate opportunistic non-native species. In our case, the restored low-shore marshes were colonised by *S. anglica*, a widespread non-indigenous species in the region that has likely occupied niches left vacant by the decline of the native *S. maritima* (Wong et al., 2018). While this colonisation contributed to vegetation cover and sediment stability, two important indicators of functional recovery, it also challenges assumptions about what defines restoration success.

Future research should explore the ecological consequences and trade-offs of this replacement, such as such as impacts on biodiversity, ecosystem functioning, and long-term habitat stability. It is also relevant to develop anticipatory eco-engineering approaches that account for local species' biology and promote native species establishment (Airoldi et al., 2021; Morris et al., 2019). Restoration methods that enhance the

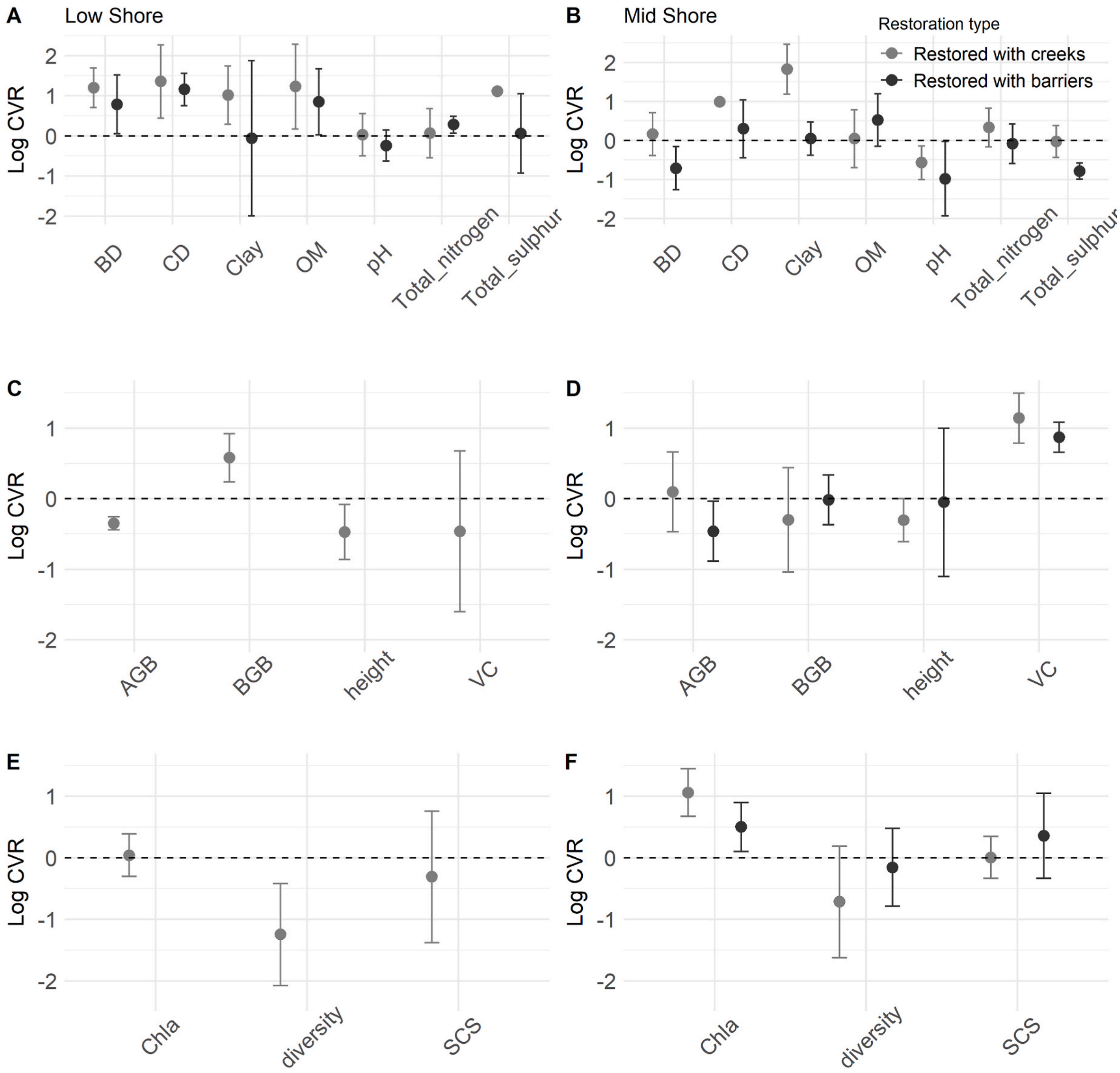

Fig. 6. Natural logarithm of the coefficient of variation ratio (lnCVR) for: sediment variables (CD = conductivity i.e., salinity); BD = bulk density, clay = % clay-silt fraction; OM = organic matter; A and B), vegetation structural variables (AGB = aboveground biomass; BGB = belowground biomass; height = canopy height, VC= Vegetation cover; C and D) and functional variables (Chl *a* = Chlorophyll *a*; diversity = macrobenthos Shannon diversity index (*H'*); SCS = sediment carbon stocks; E and F) at the low and mid shores of Natural (N), Restored with creeks (RC) and Restored with barriers (RB) (n = 4). Bars indicate confidence intervals at 0.95.

natural foreshore dynamics more than traditional complete or partial barriers, may better support cordgrass dispersal. A promising example in the northern Venice Lagoon is the use of spatially diffuse, permeable brushwood modules to reduce boat waves, prevent sediment erosion and stabilise the marsh edge (Barausse et al., 2015). Multi-ecosystem approaches, such as restoring oyster habitat in front of a marsh (Smith et al., 2025; Wellman et al., 2022), could be even more effective, providing essential protection from erosion to stabilise the edge margins while also facilitating cordgrass seed recruitment (Bouma et al., 2014). This area of research is advancing and warrants further exploration of novel eco-engineering solutions that maximise both coastal protection and conservation priorities (Schoonees et al., 2019; Marin-Diaz et al., 2021).

4.2. *Consistency of restoration*

Restoration projects are typically judged on their capacity to restore, on average, target metrics that resemble those of reference sites. Yet, this goal is not always met, as outcomes can vary depending on site-specific conditions such as wave exposure, local stressors, or greater exposure to climatic extremes like rain or drought (e.g., Brudvig et al., 2017; Brudvig and Catano, 2021). As in terrestrial systems (Atkinson et al., 2022), outcomes differed widely among restored sites. Multiple RC and RB sites exhibited as much or more variation than natural sites, even in the relatively uniform RB design, with limited consistency at the site level. Interestingly, abiotic structural outcomes were generally less consistent than biotic structural and functional ones. This contrasts with the hierarchy of predictability proposed by other authors (Brudvig et al., 2017; Brudvig and Catano, 2021), ranking abiotic conditions as the most predictable, followed by species richness and taxonomic composition. We suggest that ecosystem functions may be more consistently restored than other structural characteristics when sufficient functional redundancy is present, as different habitat and community structure can still maintain similar levels of functioning. Similar conclusions were reached by a long-term restoration study in a ponderosa pine-bunchgrass ecosystem in northern Arizona (Laughlin et al., 2017b). Confirming the generality of this finding across a variety of long-term restoration projects would assist practitioners in setting realistic success targets for different outcomes, ensuring the reinstatement of critical ecosystem services even when natural conditions cannot be fully restored.

Our findings challenge the prevailing preference in restoration policy and planning for contiguous large habitat areas over multiple smaller sites (Armsworth et al., 2018), a preference often based on the assumption that larger restoration size inherently leads to greater ecological success (Atkinson et al., 2022; Fahrig et al., 2022). While our study did not directly compare large versus small restorations, our results suggest that multiple sites, when assessed collectively, may more reliably approximate average natural functional conditions. This hypothesis is consistent with observations from other wetland systems (Vélez-Martín et al., 2018) and is especially relevant given the high differences observed across both restored and reference sites. A recent global meta-analysis of salt marsh restorations further supports this hypothesis, finding no consistent relationship between restoration size and the ability to replicate natural conditions (De Lauretis et al., 2025).

These results have clear implications for adaptive management and monitoring strategies. Rather than focusing solely on site-level outcomes, restoration planning could benefit from a landscape-level perspective that emphasizes the functional complementarity of multiple sites. To enhance practical relevance, future research should explore how site selection can optimize ecological representativeness and how connectivity—functional or structural—can be maintained or enhanced across fragmented restored areas. For example, selecting sites that span a range of environmental gradients, fill ecological or functional gaps in the landscape, or lie within potential dispersal corridors could improve overall restoration outcomes (Suding, 2011; Wolters et al., 2008). Similarly, functional connectivity may be enhanced by locating some of the restored sites near existing natural habitats, aligning with species movement pathways, or using stepping-stone configurations to support ecological flows. Such insights could support policy frameworks that recognize dispersed restoration configurations as viable and potentially more resilient alternatives to single-site approaches.

5. Conclusions

Our results highlight the challenges with replicating natural conditions, especially concerning native species recovery, and the high inconsistency of salt marsh restoration outcomes, emphasizing the need for a more nuanced approach to restoration planning. While no restoration method fully replicated natural marsh conditions in the Venice Lagoon, approaches that supported natural foreshore dynamics, such as RC, more closely resembled natural conditions, by supporting vegetation establishment at the low shore, albeit non-native. Moreover, although the high variability in restoration outcomes made site-specific predictions challenging, multiple restored sites, on average, approximated key ecological functions. We hypothesize that restoring multiple, dispersed sites may be a more effective strategy for scaling up restoration efforts, enhancing habitat quality and functionality, even if full ecological recovery remains elusive.

CRediT authorship contribution statement

Laura Airoldi: Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization. **Md Masum Billah:** Writing – original draft, Methodology, Investigation, Formal analysis, Conceptualization. **Marco Bonato:** Writing – review & editing, Investigation. **Federica Bortolami:** Writing – review & editing, Investigation. **Folco Giomi:** Writing – review & editing, Investigation. **Anjali Gopakumar:** Writing – review & editing, Methodology, Investigation, Conceptualization. **Davide De Battisti:** Writing – review & editing, Writing – original draft, Visualization, Formal analysis, Data curation.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Laura Airoldi reports financial support was provided by EU project CASCADE (INTERREG V-A Italy-Croatia CBC Programme ID 10255941. Davide De Battisti reports financial support was provided by Italian Ministry of Research (PON, grant 19-G-12540-1). Laura Airoldi reports financial support was provided by National Biodiversity Future Center (CN00000033, Italian MUR under the National Recovery and Resilience Plan, Mission 4 Component 2 Investment 1.4. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2025.126921.

Data availability

Data have been made available on the University of Padova public repository system at https://researchdata.cab.unipd.it/id/eprint/1607.

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