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Rapid multiple method assessment reveals the effects of tidal restriction and restoration on fish and crustacean communities in a California estuary

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ABSTRACT

Saltmarshes are one of the most valuable ecosystems on the planet, providing shoreline protection, carbon storage, and nursery habitat, but face significant pressure from human activity and sea level rise. Saltmarsh restoration strategies aim to recover ecological function and provide resilience to rising seas, yet their effects on mobile creek fauna (fish and decapod crustaceans) remain understudied on the North American Pacific coast. Researchers frequently seek rapid, cost-effective tools for studying these communities as they are effective indicators of marsh condition. This study assessed the ecological success of saltmarsh restoration and provides insight for future monitoring efforts. Assemblages were compared across one tidally restricted marsh, one sediment-addition restored marsh and two natural reference marshes in Elkhorn Slough, California using three different, but complementary methods over a two-month rapid assessment. In total, we detected 4833 individuals from 14 species. The restricted marsh supported the lowest abundance, smallest topsmelt (*Atherinops affinis*), and fewer and smaller transient species, indicating reduced nursery function. In contrast, the restored marsh closely resembled the adjacent natural marsh in community composition and ecological attributes present, such as trophic roles and residency patterns, and supported the highest native richness. Natural sites also differed from one another, highlighting the need for multiple reference sites in restoration assessments. Minnow traps provided consistent species detection and replication, seines captured the highest numbers of transient fish, and underwater videos detected large, elusive species. Results demonstrate that mobile creek fauna can rapidly re-establish after sediment-addition and that combining rapid sampling methods enhances understanding of ecological functions.

1. Introduction

Saltmarshes are vital ecosystems found around the world in coastal areas with low wave energy and fine sediment (Carter, 1988). Despite making up 1% of all land area, they provide 20% of all value derived from ecosystem services (Zu Ermgassen et al., 2021a). Some of these ecosystem services include protecting coastlines from flooding and erosion (Shepard et al., 2011), sequestering millions of tons of carbon each year (Barbier et al., 2011), nutrient cycling (Millennium Ecosystem Assessment, 2005), and supporting coastal food webs by acting as nursery grounds (Minello et al., 2003; Raposa and Talley, 2012; Zu Ermgassen et al., 2021b). However, a variety of anthropogenic pressures threaten saltmarshes resulting in extensive losses over the past century, including sea-level rise and tidal restriction. Human-caused climate change is expected to cause sea levels to rise as much as 1.01 m by 2100 (IPCC, 2023), threatening to drown saltmarshes that are unable to retreat landward (Zu Ermgassen et al., 2021a). A second major mechanism responsible for the degradation of many saltmarshes is tidal restriction via diking, which causes habitat fragmentation, impairs water quality and reduces animal diversity and abundance (Roman et al., 1984; Burdick et al., 1996; Raposa and Roman, 2001; Ritter et al., 2008; Cronin-O'Reilly et al., 2025).

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Several saltmarsh restoration projects have sought to reverse the negative impacts of tidal restriction by removing tidal barriers and restoring hydrological flow (Raposa and Talley, 2012; Prado et al., 2017; Broome et al., 2019). Restoring natural tidal flow, through removal of barriers and subsequent restoration of creek morphology and submerged aquatic vegetation (Zhu et al., 2026), has been shown to recover habitat function (Madon, 2008) and allow economically important species to increase in abundance and richness (Brockmeyer Jr. et al., 1996).A second major type of saltmarsh restoration is sediment addition which involves strategically depositing sediment (typically dredged sediment from nearby channels) onto low-lying marshes (Croft et al., 2006; VanZomeren et al., 2018; Raposa et al., 2022). Sediment addition restoration seeks to replace marsh elevation that was lost due to subsidence (Croft et al., 2006) and promote marsh resilience to sea level rise (VanZomeren et al., 2018; Raposa et al., 2022).

Monitoring is essential for evaluating saltmarsh restoration success and requires a multifaceted approach examining a variety of abiotic (physical and chemical attributes of water and soil) and biotic factors (plant and animal function) (Raposa and Talley, 2012). Monitoring efforts commonly focus on factors such as soil chemistry, vegetation and hydrology (Neckles et al., 2002; Zhao et al., 2016; Cadier et al., 2020; Billah et al., 2022). Whilst important, these metrics do not represent restoration impacts across multiple trophic levels (Whitfield and Elliott, 2002; Rochlin et al., 2012; Mahoney et al., 2021). Marsh restoration and subsequent monitoring often emphasize “bottom-up” factors, such as vegetation distribution, but “top-down” factors like predator abundance can play important roles in marsh structure and function (Silliman and Bertness, 2002).

Mobile creek faunal communities (fish and decapod crustaceans) can serve as excellent indicators of saltmarsh function (Whitfield and Elliott, 2002; Raposa et al., 2003; Rochlin et al., 2012; Ziegler et al., 2021; Billah et al., 2022). These communities play a critical role in coastal food webs by transporting nutrients across ecosystems and supporting offshore fisheries (Ziegler et al., 2019; Waltham and Connolly, 2022). Given their importance in mediating trophic linkages, mobile creek faunal communities provide an opportunity to investigate how saltmarsh restoration influences ecosystem structure across multiple trophic levels (Raposa et al., 2003; Ziegler et al., 2021; Keppeler et al., 2023, 2024). Many projects that monitor mobile creek fauna use common metrics such as abundance or richness but fail to account for the species' ecological attributes such as their trophic role, origin, or residency patterns (Rozas et al., 1988; Chamberlain and Barnhart, 1993; Sanger et al., 2004; Platell and Freewater, 2009). Ecological attributes are valuable to estuarine monitoring because rather than simply detecting if diverse communities are present in the ecosystem, they reveal what roles various species play in the ecosystem (Mathieson et al., 2000; Pérez-Domínguez et al., 2012; Prado et al., 2017). A critical ecological attribute is trophic role (Mathieson et al., 2000; Green et al., 2009) which describes a species' position in the food web (Luczkovich et al., 2003). Understanding the distribution of trophic roles within a community is critical for assessing a site's ability to support a complex food web (Muñoz and Ojeda, 1997). Monitoring species origin (native vs. non-native) can provide insight into ecosystem health (Kennard et al., 2004) and ensure that high richness or abundance driven by non-native species, such as the European green crab (*Carcinus maenas*), is not mistaken for a beneficial contribution to community composition. Additionally, a healthy saltmarsh should support fauna with various patterns of residency as one of the key functions of a saltmarsh is as nursery grounds and nutrient export (Boorman, 2003; Ziegler et al., 2019). Thus, the composition of residency patterns in tidal creek communities can also serve as an indicator of ecological function (Sheaves, 2009). Ultimately, evaluating traditional community metrics and ecological attributes requires methods that reliably sample mobile creek fauna.

Restoration managers typically desire rapid, low-cost methods for assessing restoration success. Traps, seines and baited remote underwater videos (BRUVs) are useful techniques for estimating abundance and richness, metrics essential for evaluating ecosystem function and resilience (Duffy et al., 2016; Winfree et al., 2025). Traps and seines have been useful tools for monitoring fish and crustaceans in saltmarshes for decades (Rozas et al., 1988; Buchsbaum et al., 2006; Steele et al., 2006). Traps are often most efficient at catching benthic species (Layman and Smith, 2001) while seines are efficient at sampling species found higher in the water column (Nordby and Zedler, 1991). Underwater video is a widely used method for detecting species in marine environments like coral reefs but are far less common in estuarine systems (Whitmarsh et al., 2017; Walker et al., 2023). One possible advantage of using underwater cameras is their ability to detect fast and large species that traps and seines are unable to catch. Further knowledge is needed of the effectiveness of BRUVs in estuarine systems and of the performance of these three methods in California saltmarshes.

Elkhorn Slough is a small estuary on the central coast of California, extending 11.4 km inland from Monterey Bay and encompassing approximately 3.25 km^2 of wetland habitat within a 182 km^2 watershed. The estuary has semi-diurnal tides with a tidal range of roughly 2.5 m during spring tides. It is the only major estuary along a 350 km stretch of coastline, making it a critical biodiversity hotspot and an important nursery for marine fauna (Caffrey et al., 2002). The Elkhorn Slough Reserve has a single large marsh recently restored via sediment addition, a single large, diked marsh planned for restoration, and several natural marshes. Because Reserve staff are preparing to invest substantially in a forthcoming second restoration, they requested this study to clarify ecological differences among marsh types and inform decision-making. The restored marsh presents a unique opportunity to better understand how mobile creek fauna respond to sediment addition projects, a topic which has received little attention (Raposa et al., 2022; Keppeler et al., 2023). To our knowledge, no previous study has examined how mobile creek fauna respond to sediment-addition restoration in California. In addition, the responses of mobile creek fauna to marsh restoration projects along the North American Pacific coast are understudied (Chamberlain and Barnhart, 1993; Larkin et al., 2008; Grimaldo et al., 2012).

We directly address these gaps in the literature by conducting a rapid field assessment comparing mobile creek faunal communities across natural, degraded (diked) and sediment-addition-restored (formerly diked) marshes during the summer in Elkhorn Slough, California. By focusing on a region where only limited mobile creek fauna monitoring has been conducted, this work contributes to a regional knowledge gap (Williams and Zedler, 1999; Desmond et al., 2000). Additionally, by combining traditional metrics such as abundance, richness, and diversity with analyses of ecological attributes, this study aims to provide a more comprehensive assessment of restoration success. This approach not only determines whether diverse communities return following restoration but also evaluates whether restored marshes support similar ecological functions to natural systems, such as providing nursery habitat and facilitating the export of productivity to the ocean. This investigation also offers methodological insights by comparing tidal creek community characterizations provided by baited minnow traps, seines, and BRUVs. In doing so, this study provides insight into the ecological value of saltmarsh restoration and offers a framework for future monitoring efforts seeking to capture both taxonomic and functional responses of mobile creek fauna. We predicted indicators such as richness and abundance to be highest at the natural sites, lowest at the degraded site, and intermediate at the restoration site.

2. Methods

2.1. Study sites

We examined the mobile creek fauna (fish and decapod crustaceans >1 mm in total length, including nektonic and benthic species) in four marshes in Elkhorn Slough (Figs. 1 and 2): one restored, one degraded

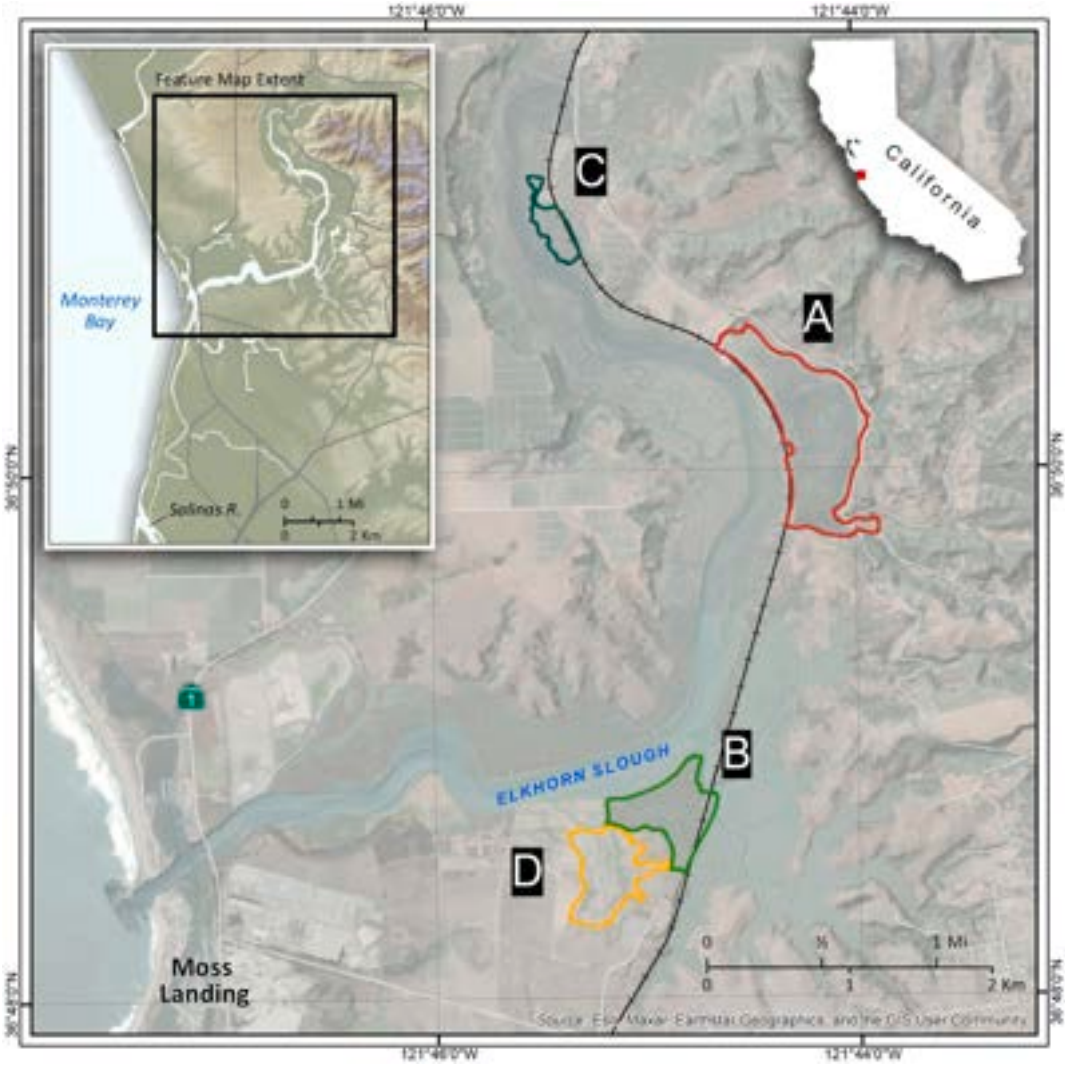


Fig. 1. Map of study sites in Elkhorn Slough: Degraded 1 (A), Natural 1 (B), Natural 2 (C), and Restored 1 (D). Background imagery is courtesy of ESRI, Earthstar, and GeoGraphics and entirely consists of material in the public domain. Color coding of sites is the same in all subsequent figures. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

by tidal restriction and two natural. Our "Restored 1" site, Hester Marsh, was diked in the mid-1900s. By the 1980s, the dikes had broken down, restoring tidal exchange. However, the wetland was too low to support a healthy saltmarsh (Fountain et al., 2024). In response, between 2018 and 2021, over 375,000 m^3 of sediment were used to raise the marsh to an elevation suitable for resilience to sea level rise (Fountain et al., 2024). During restoration, tidal creeks, excluding the main channel and a few secondary channels, were filled with sediment and re-excavated. The high elevation and compacted sediments pose challenging conditions for marsh colonization, and cover was around 25% at the time of our study (Wasson et al., 2025). "Degraded 1" site is Pelican Marsh, which was diked in the 1900s and still has very limited tidal exchange through culverts in the dike, limiting tidal range to only about 15 cm during spring tide and $<$ 10 cm during neap tide (Elkhorn Slough National Estuarine Research Reserve (ESNERR) monitoring data, https://cdmo.baruch.sc.edu/). The shallow, standing water is characterized by nightly anoxic and eutrophic conditions (Hughes et al., 2011). Tidally restricted marshes like Degraded 1 are hypereutrophic (Hughes et al., 2011) and support less biodiversity (Ritter et al., 2008) than marshes with natural tidal exchange. "Natural 1" site is Yampah Marsh, located adjacent to Hester. Yampah has never been restricted by tidal barriers. "Natural 2" site is Azevedo Marsh, located farthest upstream in the estuary. Almost all the study area in Natural 2 was never diked; the most northern portion was briefly diked but remained vegetated saltmarsh during this period. Reserve staff requested this study across all four marshes to better understand how restoration and tidal restriction influence marsh ecology before investing in the restoration of Degraded 1.

At each site, we aimed to sample creeks from 0.1 m to 1.1 m of elevation above mean lower low water (MLLW). Physical and environmental attributes of each site can be found in Table 1.

Ideally, to study the general effects of sediment addition or tidal restriction, one could sample many replicates of each category of marsh. However, in the Elkhorn Slough estuary, Restored 1 is the only location where saltmarsh restoration has occurred. Degraded 1 is the only extensive tidally restricted wetland under ownership of the Elkhorn Slough Reserve and the only one being considered for future restoration. So, these two categories could not be replicated. Two natural reference sites were chosen at varying distances along the estuarine gradient from mouth to head, but it did not make sense to have many more reference sites than restored/degraded sites. Each site was relatively large, and we selected multiple replicate sampling locations within them. This is the rationale for our study's focus on comparing four contrasting sites - an approach shaped by the goals of evaluating restoration outcomes at Restored 1 and assessing the ecological condition of Degraded 1 to inform future restoration decisions.

Fig. 2. Study sites. The left column shows the overall habitat, the right an example of a tidal creek where sampling occurred.

2.2. Sampling overview

Three complementary sampling methods were used to rapidly assess diversity and abundance of mobile creek fauna over the course of two summer months: minnow trapping, seining, and baited remote underwater videos. Nine minnow traps were deployed 3 times at each marsh every other week from early June to early July 2025 during neap tide totaling 27 replicates per marsh. BRUV deployments were conducted opportunistically three times per site in the same period. Seining was conducted as a single event at each site from July 14–17, 2025. Below, we provide an overview of the methods for each of the three approaches; more detail on all of them is provided in the supplemental information.

2.3. Minnow trapping

To sample small fishes and crabs moving on or near the bottom, we deployed plastic minnow traps baited with anchovies at the bottom of tidal creeks three times at each site from June to July 2025, for a total of 27 trapping unit deployments in each marsh. Each site was sampled with nine minnow traps per visit: six in shallow creeks (mostly between 0.7 m and 1.1 m of elevation above MLLW) and three in deeper creeks (mostly between 0.1 m and 0.6 m of elevation above MLLW). Elevations of each trap can be found in Supplemental Table 4. Shallow creeks were more heavily sampled because two sites (Natural 2, Degraded 1) had few deep creek areas, and several deep creeks at another site (Natural 1) are

Table 1
Environmental conditions and distance from estuary mouth of each site studied (National Estuarine Research Reserve System Centralized Data Management Office, 2025). Values in the lowest 10% and highest 10% columns were obtained by taking the mean of the lowest and highest 10% of all 2025 measurements. Values were calculated from the seven months up to and including the study period, from January 1 through July 11. No data were available from Natural 2.

Metric	Site	2025 Average	Lowest 10%	Highest 10%
Temperature (°C)	Degraded 1	16.26	10.58	21.08
	Natural 1	13.58	10.98	17.99
	Natural 2	-	-	-
	Restored 1	16.11	10.97	19.69
Salinity (psu)	Degraded 1	32.26	25.27	37.27
	Natural 1	32.86	25.55	39.35
	Natural 2	-	-	-
	Restored 1	36.23	34.35	39.68
DO (mg/L)	Degraded 1	7.07	0.01	11.58
	Natural 1	7.88	6.23	9.69
	Natural 2	-	-	-
	Restored 1	6.08	3.36	9.97
pH	Degraded 1	8.05	7.23	8.36
	Natural 1	7.81	7.57	8.02
	Natural 2	-	-	-
	Restored 1	7.97	7.8	8.16
Distance from estuary mouth (km)	Degraded 1	6.40	-	-
	Natural 1	3.62	-	-
	Natural 2	8.59	-	-
	Restored 1	3.16	-	-

frequented by sea otters (*Enhydra lutris*), which we wanted to avoid disturbing. Each trapping unit included one trap with a small (25 mm in diameter) opening and one with a large (51 mm in diameter) opening, allowing a wider variety of individuals to be caught than traps of a single size. Traps had a mesh size of 3x4 mm.

Traps were placed during low tide. Traps were typically deployed from the end of low tide on one day to the start of low tide on the following day during a neap tide series; trap submergence averaged 16.2 h (1.75 - 26.00). Submergence time of the traps was calculated using trap elevation and water level measurements from the Vierra water quality monitoring station (ESNERR monitoring data, https://cdmo.baruch.sc.edu/). During retrieval, all individuals were identified in the field to the species level and counted. On select weeks, the total length (TL) in mm of the smallest and the largest individual of each species from each trap was recorded to characterize size trends of common species at each site. We quantified this minimum and maximum so that we could analyze differences by site, for example to determine whether some sites have representation of tiny, juvenile size classes or large, adult size classes absent from other sites. During those weeks, each crab's sex was identified by abdominal shape and recorded to assess if there were differences in the sex ratios between sites.

2.4. Seining

Seining was conducted as a single-event sampling effort from July 14–17 2025, with one site sampled per day to further characterize fish community diversity, including species that may not respond to bait. Each sampling event began shortly after slack high tide and lasted 3 to 4 h, when water level ranged roughly from 1.0 to 1.4 m above MLLW. We used a 4.27 × 1.83 m beach seine net with a 1.22 x 1.22 × 1.22 m bag and 0.32 cm mesh size to sample mobile creek faunal communities in relatively shallow creeks (29 - 102 cm water depth during seining; mean 48 cm). Five creeks were selected per site, and one 15 m seine pull was conducted per creek.

All individuals captured were identified to species level and counted. The smallest and largest fish of each species was measured per seine, as an indicator of whether different size classes use the different sites. During seining, we had the capacity to also haphazardly select eight individuals per species per seine and measure them to characterize the average size caught. To determine nursery function of each site, an important metric when assessing fish use of marsh habitat (Rountree and Able, 2007), we visually estimated the proportion of individuals below and above species-specific size thresholds during each seine pull. Size cutoffs used to define young-of-year (YOY) status included <30 mm for gobies (Kent and Marliave, 1997; Grossman, 1979), <102 mm for topsmelt (Vejar, 2011) and northern anchovy (Baxter, 1966).

2.5. Baited remote underwater video (BRUV)

To better characterize the tidal creek communities, especially fish that would not enter baited traps, BRUVs were deployed in each site during flood and high tides. Deployments were 90-120 min. BRUV deployments were scheduled across all four sites within one week, repeated over four sampling intervals in summer 2025. While the goal was to deploy at each site within the same week to minimize temporal variation, water clarity conditions (Fig. 11 supplemental information) prevented this during some sampling intervals, necessitating slight adjustments to deployment timing (see supplemental information).

BRUV deployment procedures were loosely based on the California Estuary Marine Protected Area (EMPA) Monitoring Program's standard operating procedures (Walker et al., 2023). A GoPro camera was mounted to a PVC stake driven into the substrate in a deep creek area (~0.7 - 1.5 m deep in the center (Table 5 supplemental information)) near the bank, with the camera facing toward the center of the creek and towards the main estuary channel. When an individual appeared in the video, the timestamp was recorded, and the individual was identified to the lowest taxonomic level. The most numerous appearance of each species (or the clearest appearance if only one individual was observed throughout the recording) was screenshotted to document the observation. Abundance was estimated using the MaxN metric defined as the maximum number of individuals of a given species in a single frame (Willis and Babcock, 2000; Schobernd et al., 2014).

2.6. Statistical analyses

To analyze the results from each method we used a similar statistical approach, though each was analyzed separately since replication and distributions of data differed. Trap data were standardized by dividing the catch by hours of inundation resulting in catch per unit effort (CPUE) equaling the number of individuals caught per hour. BRUV data were standardized by dividing MaxN by the number of minutes of video analyzed and multiplying by 60 resulting in observation per unit effort (OPUE) equaling the number of individuals observed per hours of video. Seine hauls at each site were of equal length and duration. We used a Kruskal-Wallis test to determine whether creek depth differed among sites and found no significant differences ($p > 0.05$). Because seine effort was standardized and depth did not vary among sites, no additional CPUE standardization was required. For each method, we conducted univariate analyses of total abundance. richness, abundance of the most common species, and size of the most common species. Because our sampling gear does not capture all individuals present, CPUE and OPUE were used as proxies for relative abundance. Differences among sites were evaluated using statistical tests chosen to match data distribution, including Type II Wald χ^2 tests for generalized linear models (GLMs), Kruskal–Wallis tests for nonparametric median comparisons and two sample t-tests. Post-hoc pairwise comparisons were performed using estimated marginal means contrasts with Sidak adjustment for GLMs and Dunn's multiple-comparison tests with Sidak adjustment following

significant Kruskal–Wallis results. All univariate analyses were performed in R v4.5.1 using R-Studio (R Core Team, 2025) with some R scripts being partially written using ChatGPT GPT-4o (OpenAI, 2025) and manually reviewed. More detail on statistical analyses can be found in the supplemental information.

We used a suite of nonmetric multidimensional scaling (nMDS) approaches in PRIMER (Plymouth Routines In Multivariate Ecological Research) v.7 to examine whether faunal communities differed across the four sites. To analyze the community composition visually, an nMDS ordination was performed on a Bray–Curtis resemblance matrix of data square-root transformed to down-weight the contribution of the most abundant species. For traps, an initial visualization was attempted using all 27 trap-pairs per site, but patterns were obscured because most traps clustered on top of each other while a few were widely separated. So, for ease of visualizations, the 9 traps per sampling date were averaged, yielding 3 replicate dates per site. For BRUVs, each of the four deployments was used as a replicate; for seines, each of the five seines per site. A statistical analysis of similarities (ANOSIM) was performed with the Bray-Curtis resemblance matrix of square-root transformed data to determine whether sites differed significantly in community composition. Lastly, a similarity percentages (SIMPER) analysis was applied to determine which species contributed to 70% of the differences among the sites and to calculate Bray-Curtis dissimilarity among them (Clarke and Gorley, 2015).

2.7. Qualitative classifications

To better assess marsh function, we also conducted statistical and visual assessments of the distribution of ecological attributes present at each site (Raposa and Talley, 2012; Lechêne et al., 2018a). We recorded data on species origin, residency pattern and trophic role. Origin was classified as native or non-native (Table 2). When assigning residency patterns, all fish species were classified as in Yoklavich et al. (2002) into four categories: 1) residents (entire life-cycle in estuary), 2) partial residents (primarily live and feed in the slough but migrate to the ocean during some life stages or seasons), marine immigrants (use the slough for spawning or nursery grounds) and 4) freshwater species (species that can live in estuaries but are typically associated with freshwater) (Table 2). All crustaceans fell into the first category (Willason, 1981; Morgan et al., 2006; Béguer et al., 2011; Burke et al., 2025). Finally, we recorded the trophic role because the distribution of trophic roles in a community is a key indicator of recovery following restoration (Mahoney et al., 2021). Post-collection classification placed all species into one of seven trophic guilds: zooplankton and flora feeders (Yoklavich et al., 2002), generalist omnivores (Blanco et al., 2004; Ricketts et al., 1985; Lorda et al., 2016; Thiessen et al., 2018), benthic predators (Grosholz et al., 2000; Fisher et al., 2024), epifaunal crustacean feeders (Yoklavich et al., 2002), benthic–pelagic invertebrate feeders (Sitts and Knight, 1979; Jakubavičiūtė et al., 2017), mollusk and worm feeders (Caffrey et al., 2002) and mobile crustacea and fish predators (Yoklavich et al., 2002) (Table 2). All three ecological attributes were only qualitatively assessed.

3. Results

A total of 4833 individuals representing 14 different species were captured or observed on video over 32 sampling events across all methods in the summer of 2025. The five most numerous species were topsmelt (*Atherinops affinis*, n = 2999), hairy shore crabs (*Hemigrapsus oregonensis*, n = 668), green crabs (n = 489), gobies (arrow gobies, bay gobies and longjaw mudsuckers - *Oxudercidae* spp., n = 372) and Pacific staghorn sculpins (*Leptocottus armatus*, n = 186) (Table 2). Additional results details are provided in the supplemental information.

Below, we provide the results for each metric, summarizing findings for each of the three methods in turn. Because our dataset required many individual comparisons, we report only the p-values from the global tests in the main text. Rather than presenting fold differences for every site comparison or adhering strictly to a 0.05 significance threshold, we used a tiered reporting system, highlighting only the most pronounced differences. For results with $p < 0.05$, we explicitly state the fold

Table 2
Total mobile creek fauna abundance detected throughout the study, their origin, residency and trophic roles (ZF = zooplankton and flora feeder, GO = generalist omnivores, BP = benthic predator, EC = epifaunal crustacean feeder, BPI = benthic-pelagic invertebrate feeder, MW = mollusk and worm feeder, MP = mobile crustacea and fish predator). In the columns listing each site, the total number captured by any method is listed, as well as the detection method employed (T = trap, S = seine, B = BRUV). The final column shows the total number detected across all sites. For BRUVs all gobies including mudsuckers were identified as "goby" and appear in the arrow/bay goby row.

Species	Taxon	Origin	Residency	Trophic Role	Degraded 1	Natural 1	Natural 2	Restored 1	Total
Topsmelt *Atherinops affinis*	Fish	Native	Partial resident	ZF	99 (T, S)	534 (T, S, B)	1665 (S)	701 (S, B)	2999
Hairy Shore Crab *Hemigrapsus oregonensis*	Crustacean	Native	Resident	GO	109 (T, S)	120 (T, S)	191 (T, S)	248 (T, S)	668
Green Crab *Carcinus maenas*	Crustacean	Non-Native	Resident	BP	27 (T)	133 (T, S)	171 (T)	158 (T, S)	489
Arrow/Bay Goby *Clevelandia ios/Lepidogobius lepidus*	Fish	Native	Resident	EC	87 (T,S,B)	159 (T, S, B)	10 (T, S, B)	81 (T, S, B)	337
Staghorn Sculpin *Leptocottus armatus*	Fish	Native	Resident	EC	1 (B)	59 (T, B)	3 (T, B)	123 (T, B)	186
Asian Shrimp *Palaemon macrodactylus*	Crustacean	Non-Native	Resident	BPI		1 (T)	38 (T)		39
Longjaw Mudsucker *Gillichthys mirabilis*	Fish	Native	Resident	EC	15 (T, S)	1 (S)	19 (T, S)		35
Lined Shore Crab *Pachygrapsus crassipes*	Crustacean	Native	Resident	GO		1 (T)	4 (T)	14 (T)	19
Northern Anchovy *Engraulis mordax*	Fish	Native	Marine immigrant	ZF		5 (S)		12 (T, S, B)	17
Three-spine stickleback *Gasterosteus aculeatus*	Fish	Native	Freshwater	BPI	15 (T, S, B)			1 (T)	16
Mosquitofish *Gambusia spp.*	Fish	Non-Native	Freshwater	GO	15 (T, S)				15
Bat Ray *Myliobatis californica*	Fish	Native	Partial resident	MW		1 (B)		1 (B)	2
Yellowfin Goby *Acanthogobius flavimanus*	Fish	Non-Native	Resident	EC			2 (T)		2
Leopard Shark *Triakis semifasciata*	Fish	Native	Partial resident	MP				1 (B)	1

difference between sites regardless of magnitude. For results with p values between 0.05 and 0.10, we describe sites as "somewhat different," independent of fold difference. For results with $p > 0.10$, we report sites as similar. Graphical results are provided in accompanying figures, and statistical outputs of the models for each metric and method are provided in Table 6 in the supplemental information. Additional details and results are provided in the supplemental information.

3.1. Abundance

Total abundance varied across sites, with patterns differing among the three sampling methods (Fig. 3). Total abundance differed markedly among sites for minnow traps ($p < 0.0001$). Mean abundance in minnow traps was higher at all sites compared to Degraded 1, with Natural 1, Natural 2, and Restored 1 supporting roughly 6–8 times more individuals (Fig. 3a). There were no differences among the natural and

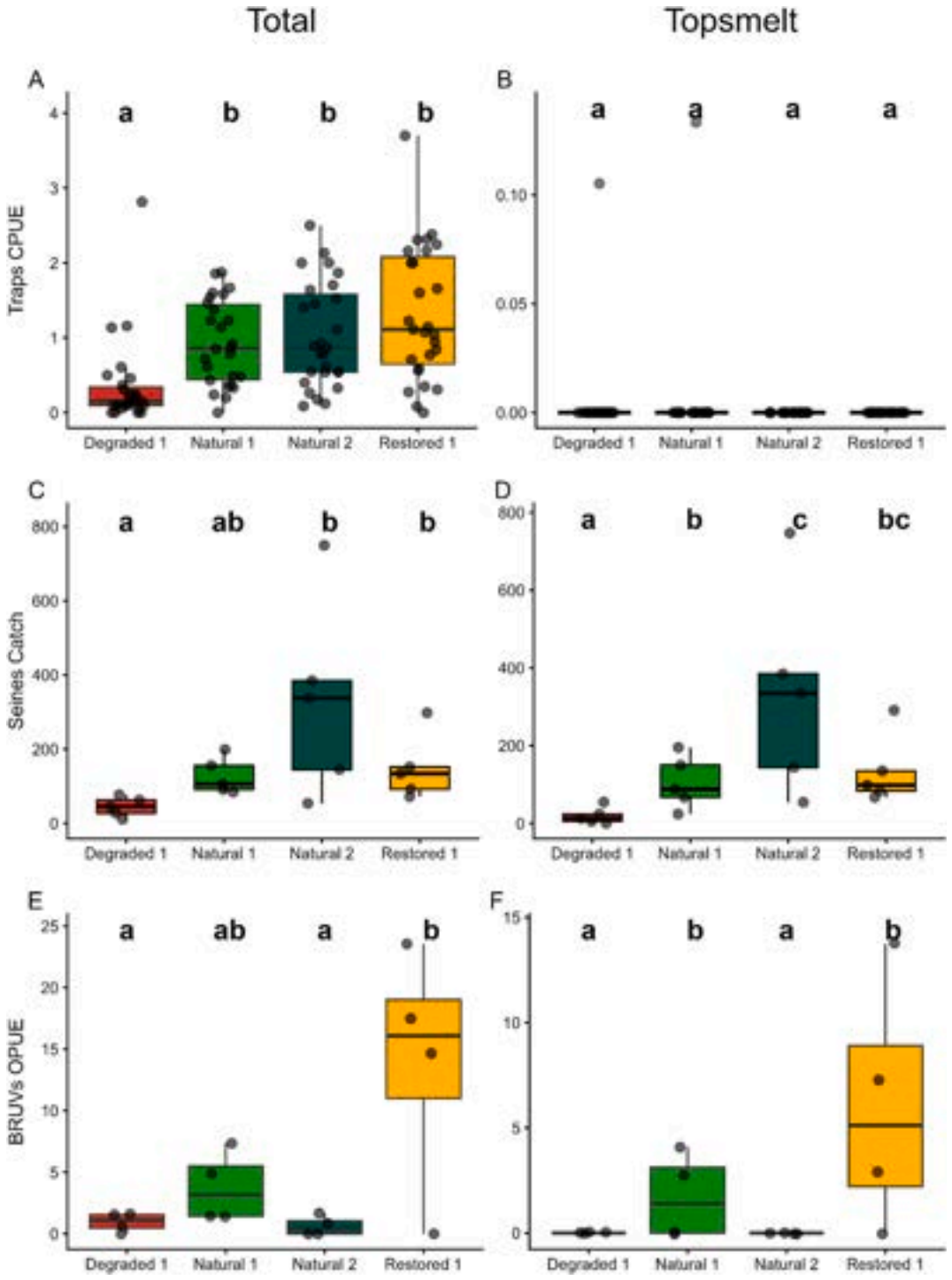


Fig. 3. Boxplots of total abundance and topsmelt abundance at each site detected by different methods. For this and all subsequent box plots, the box shows the interquartile range (25th to 75th quartile), with horizontal line for median; whiskers show points within 1.5 times the interquartile range; all samples are shown as dots, with jitter added to avoid points overlaying each other. Letters denote differences between sites where $p < 0.05$. CPUE is the number of individuals caught per hour and OPUE is the number of observations per hour of video.

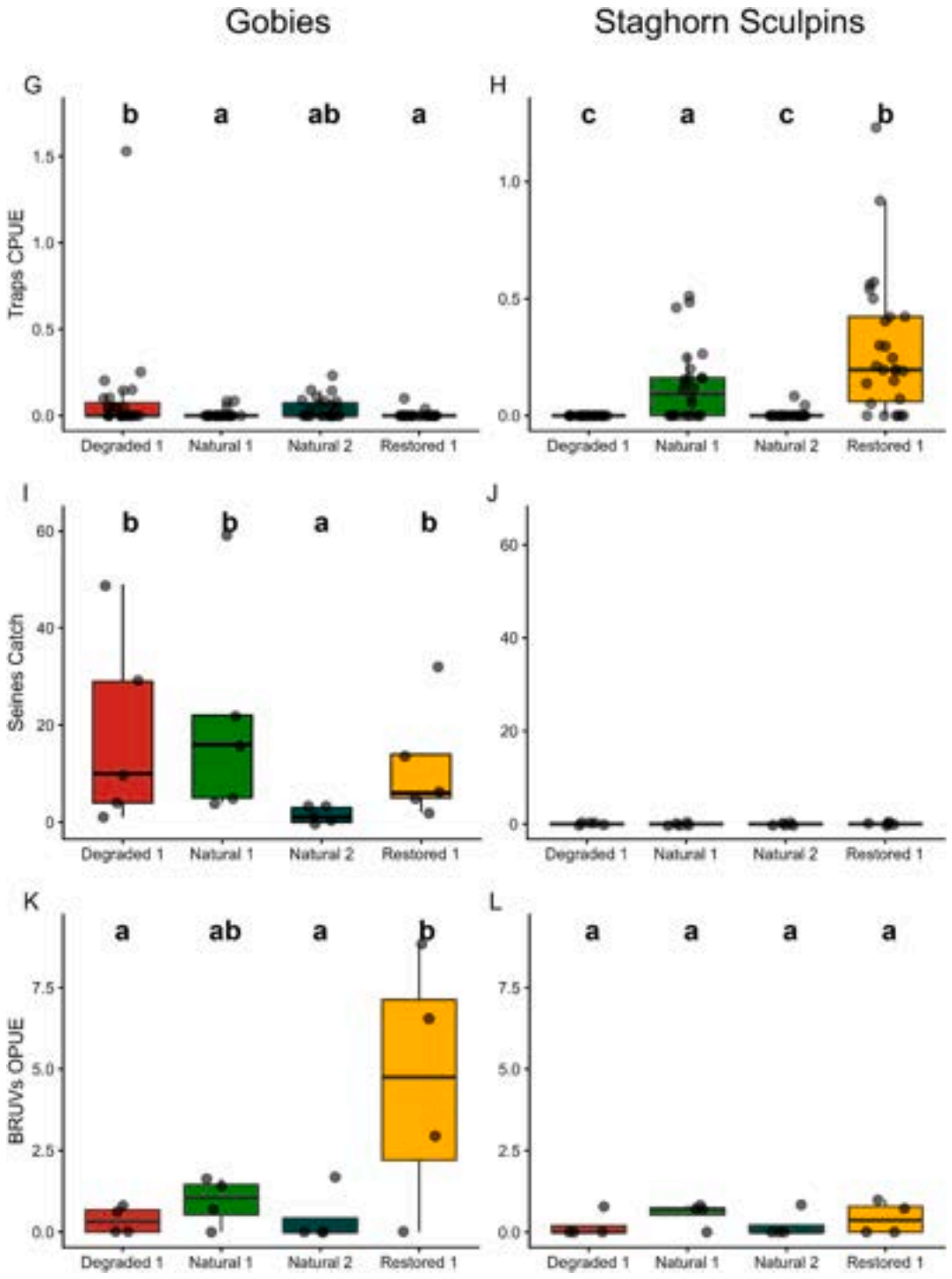
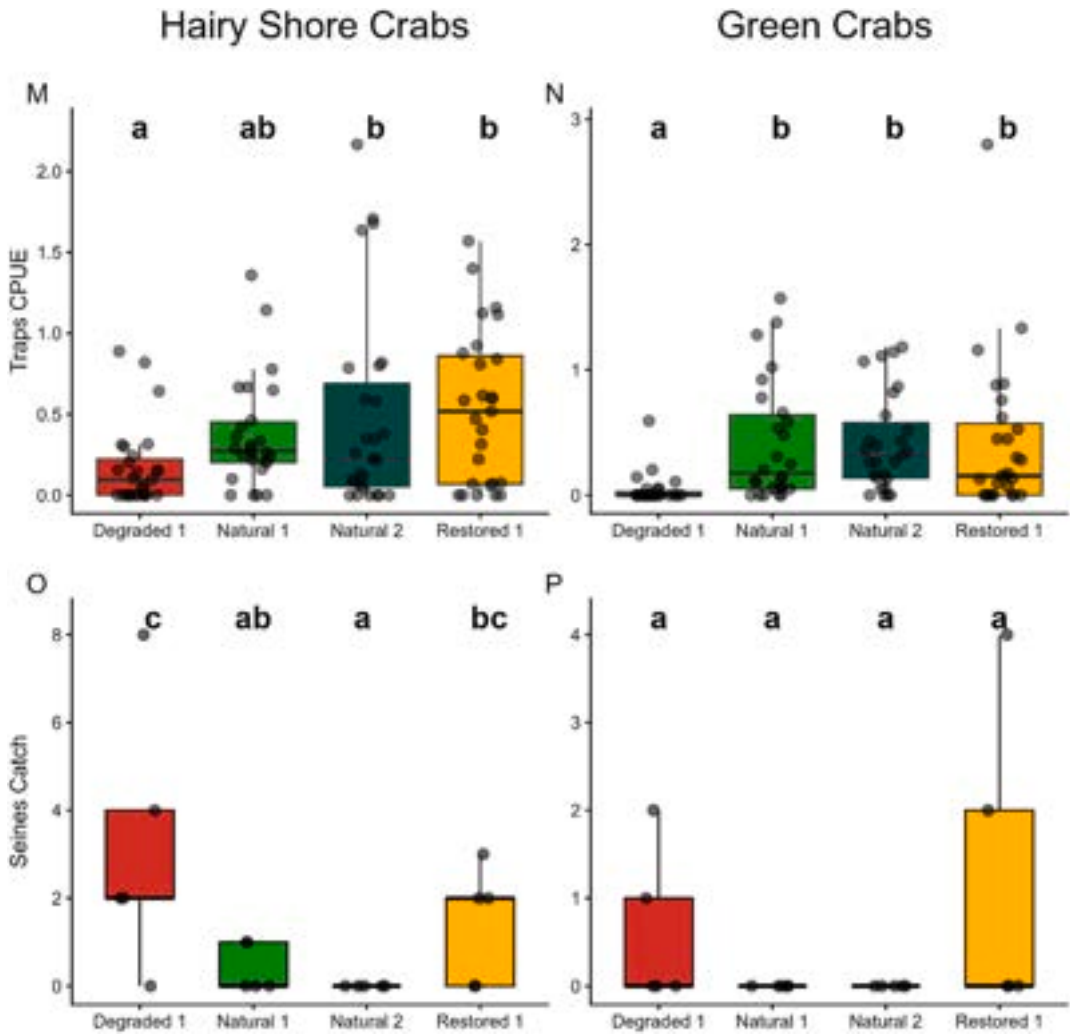


Fig. 3. (*continued*).

restored sites. Total abundance differed markedly among sites for seines also ($p < 0.0001$) but with much greater overall abundance. Per-haul abundance was somewhat higher at Natural 1 while Restored 1 and Natural 2 had roughly 3 and 7 times greater abundance than Degraded 1, respectively (Fig. 3c). Seines sampled the most topsmelt of the three methods by a large margin. BRUVs showed a different pattern ($p < 0.0001$), with Restored 1 supporting the greatest abundance, approximately 15 times higher than both Natural 2 and Degraded 1 (Fig. 3e). Natural 1 also had somewhat higher abundance than Natural 2, while other sites were similar.

We analyzed species-specific abundances for the five most common species across all methods (Fig. 3). The most abundant species, topsmelt, was rarely captured in traps ($n = 3$) (Fig. 3b), so no differences between sites were observed. *Topsmelt abundance* detected by seines showed that Degraded 1 supported the fewest topsmelt ($p < 0.0001$) (Fig. 3d). Natural 2 had 17 times more topsmelt than Degraded 1 and 3 times more topsmelt than Natural 1. Natural 1 and Restored 1 had 5 and 7 times more topsmelt than Degraded 1, respectively. BRUVs revealed a different trend ($p < 0.0001$) since Natural 2 and Degraded 1 had no topsmelt (Fig. 3f). Restored 1 and Natural 1 had markedly more topsmelt than both sites. Restored 1 and Natural 1 were similar.

Goby (arrow/bay goby and longjaw mudsucker) abundance trends



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among sites varied considerably depending on the method used (Fig. 3). Goby abundance detected by traps differed strongly by site ($p < 0.0001$) with Degraded 1 and Natural 2 being hotspots for gobies (Fig. 3g). Degraded 1 had roughly 12 times higher goby abundance than Natural 1 and 20 times higher goby abundance than Restored 1. Natural 2 supported somewhat more gobies than Restored 1. All other sites were similar. Seines detected a strong difference in goby abundance between sites ($p = 0.00056$), with a different trend than traps: Degraded 1, Natural 1, and Restored 1 had 13, 15 and 8 times more gobies than Natural 2, respectively (Fig. 3i). The three most abundant sites were similar to each other. Using BRUVs ($p = 0.0013$), Restored 1 had roughly 13 times more gobies than Degraded 1 and 11 times more than Natural 2 (Fig. 3k). All other sites were similar to each other.

Staghorn sculpin abundance patterns varied by site and sampling method. Traps ($p < 0.0001$) showed that Restored 1 and Natural 1 supported 60 times and 28 times more sculpins than Natural 2, respectively (Fig. 3h). No sculpins were caught at Degraded 1. Natural 2 and Degraded 1 were similar as were Natural 1 and Restored 1. BRUVs revealed no site differences ($p = 0.36$), and no sculpins were captured in seines (Fig. 3l).

Hairy shore crab abundance detected by traps also differed across sites ($p = 0.0024$) (Fig. 3). In traps, Natural 2 and Restored 1 supported roughly 3 times more crabs than Degraded 1 (Fig. 3m) All other sites were similar to each other. In contrast, seines ($p < 0.0001$) indicated that Degraded 1 had high hairy shore crab abundance with roughly 5 and 7 times more hairy shore crabs than Natural 1 and Natural 2, respectively (Fig. 3o). Restored 1 also had high hairy shore crab abundance with 3 times higher abundance than Natural 2. Other sites were similar.

Green crab abundance detected by traps differed across sites ($p < 0.0001$) (Fig. 3). Restored 1 and both Natural sites had roughly 10 times more green crabs than Degraded 1 (Fig. 3n). All other sites were similar. Seines did not detect any differences in green crab abundance between sites ($p = 0.19$) (Fig. 3p).

3.2. Richness

Species richness varied by site in all three sampling methods, though the patterns differed by method (Fig. 4). Traps generally detected the most species of any method and detected strong differences across sites ($p < 0.0001$) (Fig. 4a). Traps showed that Degraded 1 had the lowest mean richness per trap of all sites with roughly half as many species per trap as the other three sites. The other three sites were similar. Seines revealed a contrasting pattern ($p < 0.0001$): richness was greatest at Degraded 1, which supported approximately 3 times more species per haul than Natural 2 and 2 times more species per haul than Natural 1 and Restored 1 (Fig. 4b). Natural 2 was the least species-rich site, with richness somewhat lower than both Natural 1 and Restored 1. Richness detected by BRUVs also showed site differences ($p = 0.019$), with Restored 1 supporting somewhat higher richness than Natural 2, while all other sites were similar (Fig. 4c).

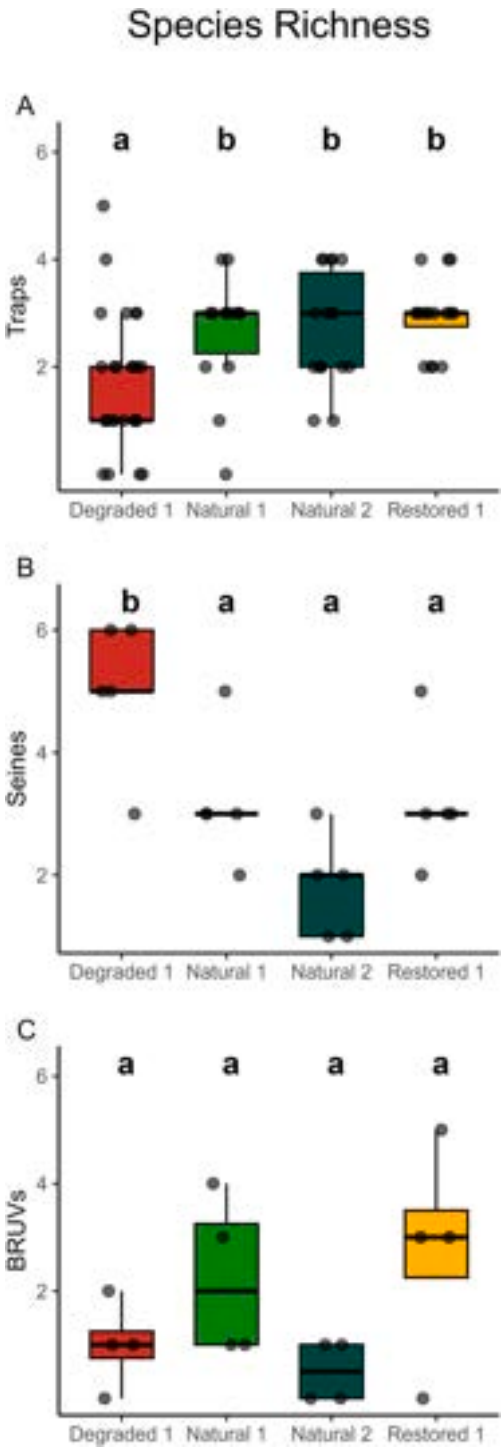

Fig. 4. Boxplots of species richness at each site detected by different methods. For traps, richness is plotted unstandardized, but the letters represent statistics from standardized data.

3.3. *Community composition (nMDS)*

Multivariate analysis of *community composition* revealed differences among sites. The ordinations were effective at separating sites for traps and seines, but not BRUVs (Fig. 5). The nMDS plots for trap and seine data indicate that the community at Degraded 1 was most dissimilar from those at other sites. In contrast, Restored 1 and Natural 1 communities show considerable overlap, more than do Natural 1 and Natural 2 (Fig. 5a and b).

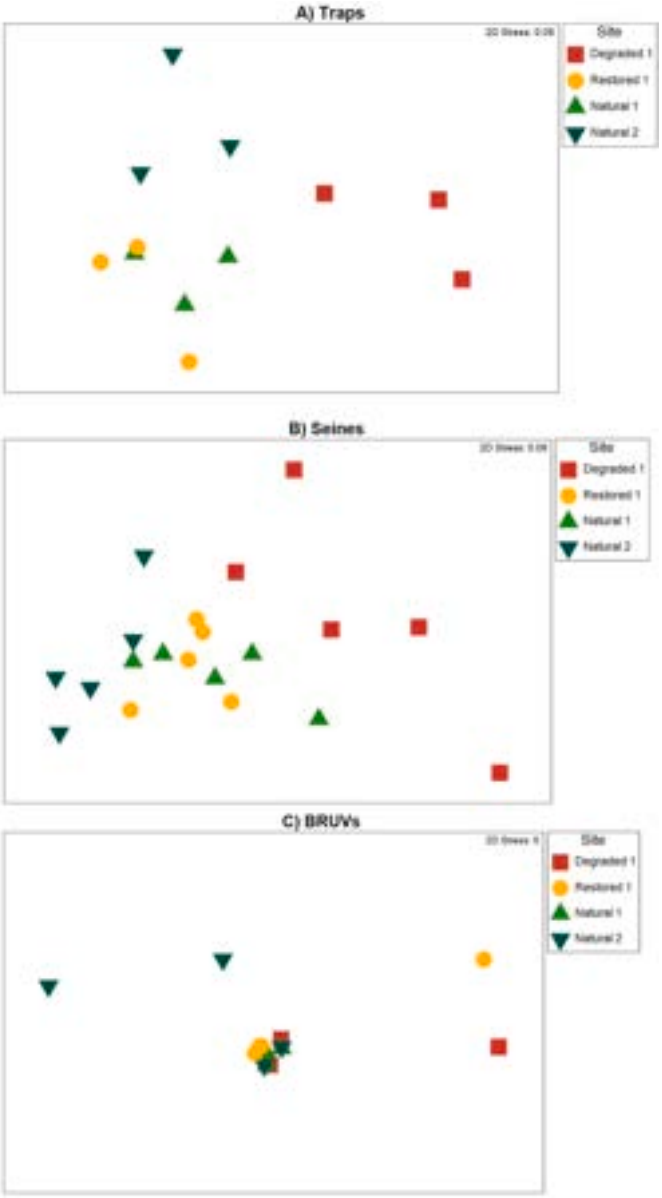

Fig. 5. Multivariate community analyses. Ordinations show non-metric multivariate ordinations of species abundance data from A) Traps (to simplify, the average of all traps per sampling date are shown, so n = 3), B) Seines (n = 5), C) BRUVs (n = 4).

Similar to the ordinations, ANOSIM showed communities differed by site for traps (Global R = 0.197, p = 0.001) and seines (Global R = 0.326, p = 0.003) but not BRUVs (Global R = 0.003, p = 0.420). Pairwise comparisons showed somewhat different patterns for traps vs. seines (Table 7 supplemental information). For traps, every pair except Restored 1 and Natural 1 were markedly different from each other. For seines, any pairwise comparison involving Degraded 1 showed a difference, but no other pairs differed. SIMPER yielded the same contrasts among methods. For traps, dissimilarity between Restored 1 and Natural 1 was lower than for all other pairs. For seines, there was greater dissimilarity for pairs involving Degraded 1 than other pairs.

The species that contributed the most to dissimilarity among pairs identified by SIMPER differed by method (Table 7 supplemental information). For trapping, the community at Degraded 1 was distinguished from the others by lower sculpin, green and hairy shore crab abundance. The community at Restored 1 was distinguished from the two Natural sites by higher sculpin and hairy shore crab, and lower green crab

abundance. For seining, the community at Degraded 1 was differentiated from other sites by lower topsmelt abundance; it also had

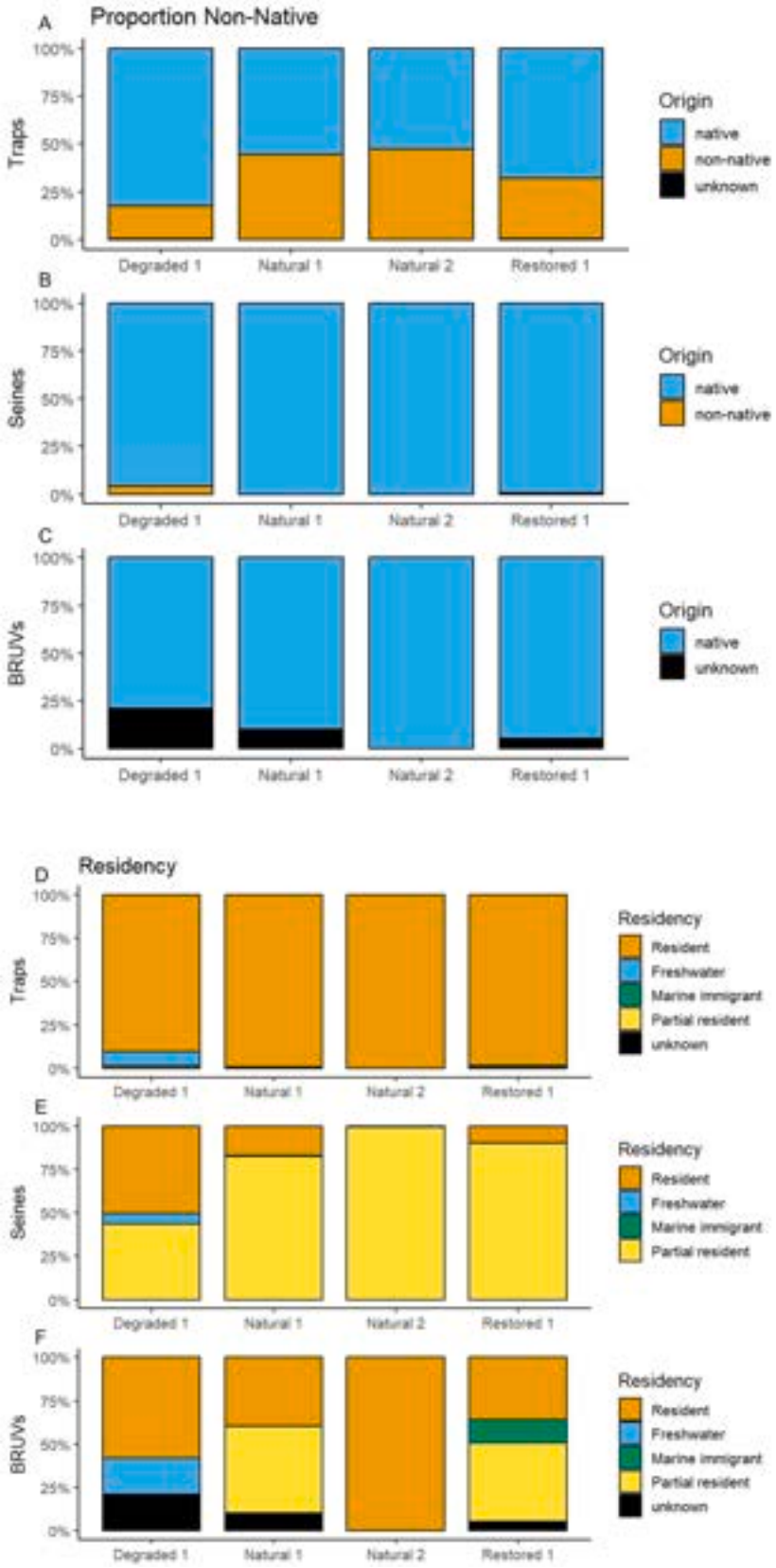

Fig. 6. Proportions of ecological attributes present at each site detected by different methods, including origin (A-C), residency (D-F) and trophic role (G-H). (ZF = zooplankton and flora feeder, GO = generalist omnivores, BP = benthic predator, EC = epifaunal crustacean feeder, BPI = benthic-pelagic invertebrate feeder, MW = mollusk and worm feeder, MP = mobile crustacea and fish predator).

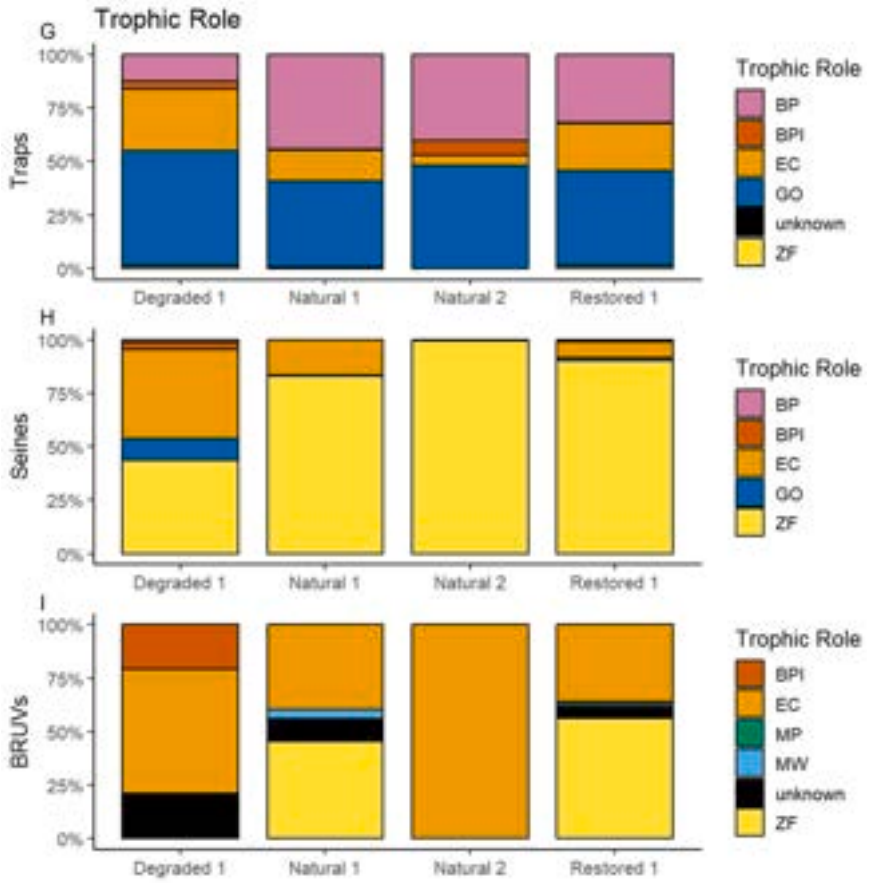

Fig. 6. (*continued*).

higher mudsucker abundance than found at Restored 1. The community at Restored 1 did not have consistent patterns of difference with the two Natural sites. Natural 1 was differentiated from Natural 2 by higher goby and lower topsmelt abundance. BRUVs showed generally similar patterns to seining, but since sample sizes were low, patterns are less robust.

3.4. Ecological attributes

The *proportion of non-native to total abundance* in minnow traps qualitatively differed among sites (Fig. 6a,b,c). Degraded 1 had the lowest proportion of non-native catch in traps and the largest proportion of non-native catch in seines. (*Palaemon macrodactylus*) (Fig. 6a). (Fig. 6b). No non-native individuals were observed in any BRUV deployments (Fig. 6c).

Patterns of *residency* qualitatively varied across sites and methods (Fig. 6d,e,f). Natural 2 showed the smallest number of residency patterns. Nearly all catch were either residents or partial residents. Degraded 1 was the only site in which we consistently detected freshwater species. Natural 1 and Restored 1 supported similar residency patterns. Of the three methods, traps were most effective at detecting marsh resident species (Fig. 6d), whereas seines were most effective at detecting partial residents (Fig. 6e). BRUVs detected a broad mix of residency types (Fig. 6f).

The distribution of *trophic roles* qualitatively varied across sites and methods (Fig. 6g,h,i). Natural 2, again, had the least diverse distribution of trophic roles, being dominated by zooplankton and flora feeders and epifaunal crustacean feeders for seines and BRUVs respectively. Natural 1 and Restored 1 had similar trophic role distributions across methods. Meanwhile, Degraded 1 supported a higher proportion of benthic-pelagic invertebrate feeders and generalist omnivores than the other three sites (Fig. 6h and i). Seines and traps effectively detected trophic roles associated with lower levels of the food chain, while BRUVs detected a broader spectrum of trophic roles spanning both lower and higher levels.

3.5. Size trends and sex ratio

We found that small green crabs and small hairy shore crabs varied in size between sites (both $p < 0.0001$) (Fig. 15a,b). Out of all four sites, Degraded 1 supported the largest invasive green crabs and the smallest native hairy shore crabs. Restored 1 supported the smallest invasive crabs and the second largest native crabs.

Topsmelt size patterns varied by site, though all topsmelt caught were young-of-year size indicating some level of nursery function in all sites (Fig. 15c,d). The general trend observed between both haphazardly selected ($p < 0.0001$) and the smallest topsmelt ($p = 0.036$) was that Degraded 1 had the smallest individuals, Natural 2 had the largest individuals, and the other two sites were intermediate.

See the supplemental information for results on sex ratio and for figures and more detail on size comparisons.

4. Discussion

4.1. Tidal creek community in a California estuary

Fish and crustaceans are crucial parts of the estuarine community which connect primary producers to higher trophic levels (Coffey et al., 2025), transport nutrients and energy between land and sea (Bronk and Steinberg, 2008; Boswell et al., 2019) and contribute to nutrient cycling through excretion and bioturbation (Haertel-Borer et al., 2004; Bronk and Steinberg, 2008; Jones, 2009; Allen et al., 2013). These communities have been extensively studied in regions such as the US east coast

(Rochlin et al., 2012; Boutin and Targett, 2013) and Gulf of Mexico (Minello and Webb, 1997; Piazza and La Peyre, 2007). However, relatively few characterizations of estuarine fish and crustacea communities have been conducted in California, especially outside of San Francisco Bay, so our results are important for characterizing California communities.

Relatively few species were detected in this tidal creek community assessment. We caught only 14 species (10 fish, 4 crustacean) during this relatively short sampling period, which is consistent with the summer catch in other California estuaries (Chamberlain and Barnhart, 1993; Grimaldo et al., 2012). Additional fish species are present elsewhere in the estuary, detected in other seasons or habitats or with different gear (e.g. trawls in deeper water) (Yoklavich et al., 1991), but our study focused on identifying dominant species in creeks during the summer.

Along the coast of California, small estuarine fishes tend to dominate shallow marsh habitats such as tidal creeks. In the present study, the most abundant fish species were topsmelt and arrow/bay gobies, both of which have been found in abundance in Elkhorn Slough in the past (Hughes et al., 2012). Staghorn sculpins were only abundant in Natural 1 and Restored 1 sites in this study, whereas past surveys reported this fish as the most abundant species throughout the slough (Hughes et al., 2012). Studies in southern California indicate that estuarine fish assemblages there are dominated by California killifish (*Fundulus parvipinnis*) and longjaw mudsuckers (Nordby and Zedler, 1991; Williams and Zedler, 1999; Desmond et al., 2000; Larkin et al., 2008), while staghorn sculpins were found relatively less frequently than in our study.

We captured three crab species in the slough, with hairy shore crabs most abundant, followed by green crabs. Although in the past lined shore crabs were found to be the most numerically dominant crab species in the slough (Beheshti et al., 2021) and abundant along the West coast (Wasson et al., 2019; Beheshti et al., 2021), they were captured only rarely by our sampling methods. Other West coast studies have found green crabs and hairy shore crabs in abundance (Grosholz et al., 2000; Fisher et al., 2024).

In total, four non-native species were detected during this study: the green crab, Asian shrimp, yellowfin goby and mosquitofish. Green crabs were found to be highly abundant in this study after years of being relatively rare (Jeppesen et al., 2025). The non-native Asian shrimp was abundant at Natural 2 and detected once at Natural 1. Although first recorded in 1981 (Standing, 1981) and absent from a 2001 survey (Wasson et al., 2001), our results show that it persists in the slough. Natural 2 had the only two yellowfin gobies, a much smaller number than had been found in studies over the last few decades (Hughes et al., 2012). Mosquitofish, which are stocked in Degraded 1 to control mosquito populations, were not found at the other three sites.

4.2. Management implications

The post-restoration monitoring of fish and crustaceans is essential for evaluating ecosystem recovery (Billah et al., 2022) and provides a way to capture biological function and detect changes across multiple trophic levels (Whitfield and Elliott, 2002). Previous studies have demonstrated that restoration can yield positive effects on mobile creek fauna, reinforcing their value as indicators of restoration success (Kimball and Able, 2007; Boys and Pease, 2017; Weinstein et al., 2019; Mahoney et al., 2021). Our findings demonstrate that mobile creek fauna can effectively indicate sediment-addition restoration performance and tidal-restriction effects in a California estuary, where both the region and this restoration strategy have received limited scientific attention (Raposa et al., 2022; Keppeler et al., 2023).

When evaluated using the methods applied in this study, the tidally restricted site, Degraded 1, was clearly different from the other sites in terms of abundance, community composition and ecological attributes present. Degraded 1 supported the lowest or near lowest abundance across all three methods. Despite its low abundance, it showed relatively similar species richness, driven in part by freshwater species not found elsewhere, and higher species richness found using seines. Degraded 1 primarily supported freshwater and resident species. Across all methods, it had the lowest proportion of transient species with roughly 50% of individuals caught in seines being residents and nearly all BRUV and trap detections being residents. This finding suggests a lower nursery value compared to the other three sites as fewer partial residents and marine immigrants use the marsh. In addition, Degraded 1 supported the fewest topsmelt of any site, further suggesting reduced nursery function, though all these topsmelt were YOY-sized. Surprisingly, despite supporting both green crabs and mosquito fish, the proportion of non-native species at Degraded 1 was the lowest of the three sites due to the high green crab abundance at the other three sites and relatively low abundance of mosquitofish at Degraded 1. In the past, tidally restricted sites typically had higher non-native species abundance, apparently due to lower otter presence and predation (Jeppesen et al., 2025).

Our results are generally similar to others from the East coast and elsewhere showing strong effects of tidal restriction (Raposa and Roman, 2001; Able et al., 2002; Roman et al., 2002; Moreno-Valcárcel et al., 2016; Boys and Pease, 2017). Tidally restricted marshes have been found to support lower abundance but similar species richness to natural or restored marshes, consistent with our results (Bowron et al., 2011; Raposa and Talley, 2012; Mahoney et al., 2021). Other literature suggests that tidally restricted systems host higher fish densities, but findings related to species richness are similar (Raposa and Roman, 2001; Moreno-Valcárcel et al., 2016). Studies have found that tidal restriction both alters community composition of fish communities and promotes freshwater and non-native species (Moreno-Valcárcel et al., 2016; Lechêne et al., 2018b).

Overall, the site restored seven years earlier with sediment addition (Restored 1) had similar fish and crab communities to the natural sites and exceeded them in some metrics including higher native species richness and lower non-native abundance. Analysis of community composition revealed that Restored 1 was indistinguishable from the adjacent natural site and exhibited some overlap with the second natural site. Total and species-specific abundances were similar between the restored site and the adjacent natural site for all species except the staghorn sculpin, which was more abundant at Restored 1. Abundance patterns in Restored 1 were mostly similar to the Natural 2, though gobies and sculpins were more abundant at Restored 1. Richness patterns between the restored and natural sites were similar. Restored 1 supported the most native species of any site and hosted a lower proportion of non-native species compared to both natural sites. Restored 1 functions as valuable nursery habitat for topsmelt, which were all young-of-year and highly abundant at the site. The ecological attributes observed at Restored 1 closely mirrored those of Natural 1, which aligns with the similarities in community composition between the two sites. BRUVs showed that both sites supported high trophic level fishes. Though detections of high trophic level fishes were infrequent, this finding is important as the movement of nutrient-transporting fauna into and out of the habitat enhances food web complexity and strengthens ecosystem resilience (Boys and Pease, 2017). Food web studies in the nearby San Francisco estuary discovered that restored and natural sites support similar trophic structures indicating that restoration can reestablish some level of food web complexity, though food chain lengths are generally shorter in restored sites (Pagliaro et al., 2025). The presence of marine immigrants at Restored 1 highlights the marsh's ability to export productivity beyond the estuary. Additionally, Restored 1 and Natural 1 both supported a high proportion of partial residents. This similarity further reinforces the restored site's function as a nursery habitat and ability to transport productivity to nearby ecosystems. While sediment addition changes marsh geomorphology (Fang et al., 2025) and temporarily displaces submerged aquatic vegetation which impacts hydrological flow (Zhu et al., 2026), mobile fish and crustacean communities rapidly became similar to reference marshes.

Monitoring mobile creek fauna after sediment addition has received less attention than other restoration strategies, but the few studies

available suggest positive outcomes for these communities (Keppeler et al., 2023). Raposa et al. (2022) reported that mobile creek fauna density initially declined following sediment placement but rebounded strongly within three years, exceeding pre-restoration levels. Species-specific responses were also observed, with sand-associated fishes and crustaceans increasing in abundance. Our results showed a similar trend: although we did not conduct a before–after comparison, Restored 1 supported high overall abundance and a notable prevalence of staghorn sculpin, a sand-associated species (Eschmeyer and Herald, 1983). In some cases, marshes with artificially introduced sediment have not functioned like natural ones. In Texas, Minello and Webb (1997) concluded that high-elevation saltmarshes may never reach functional equivalency with natural marshes. However, our data align more closely with the findings of Raposa et al. (2022) and Keppeler et al. (2023) suggesting that restored, high-elevation marshes may reach functional equivalence for mobile creek fauna within a few years.

We observed strong contrasts among the two natural reference sites at different positions along the estuarine gradient from mouth to head, suggesting distance to the ocean may have influenced our results. Natural 2 exhibited lower diversity of ecological attributes and a different community composition than Natural 1, suggesting that not all natural marshes provide equivalent benchmarks for restoration success. Natural 2 supported a notably high abundance of the non-native Asian shrimp and was the only site where the non-native yellowfin goby was detected. Natural 2 was also a hotspot for topsmelt which were less abundant at Natural 1. The differences between the two reference sites highlight the importance of including multiple reference sites for comparison (Underwood, 1992; Raposa and Talley, 2012), even if there is only one restored or degraded site to evaluate, as in our case.

4.3. Methodological comparisons

Characterizing animal use of estuarine habitats is important, but can be time-consuming and difficult, since mobile animals can be patchy in space and time (Boswell et al., 2019). Estuarine managers often have limited budget and time to conduct such surveys. A variety of methods are available for assessing estuarine communities but selecting those that most effectively capture the desired metrics is essential for maximizing efficiency and conserving limited time and resources. This study presented us with the opportunity to compare three different rapid sampling methods of monitoring California mobile creek fauna. The three sampling methods revealed distinct patterns in community composition and representation of ecological attributes (Table 3). Below, we summarize the strengths and weaknesses of each approach, in our investigation and in other similar studies, and end with recommendations.

Trapping provided a balanced mix of abundance and richness, with a strong representation of benthic fauna. Consistent with the literature, we found that minnow traps offer a replicable and efficient method for comparing tidal creek communities (Raposa et al., 2003), however, they fail to capture large fish and flatfish and overrepresent benthic species, limiting their ability to fully characterize estuarine assemblages (Rozas and Minello, 1997; Layman and Smith, 2001). Furthermore, baited minnow traps add bias since only certain species respond to bait (Layman and Smith, 2001). Nonetheless, in our study, traps were especially effective for distinguishing site differences due to higher species encounter rates and high replicability.

Seining detected variable richness and the highest overall abundance, particularly of transient water column fish species lower on the food chain. Seines are often preferred because they tend to capture greater species richness (Samarasin et al., 2017; Walker et al., 2023). However, similar to our findings, it has been shown that they primarily target water column species and often fail to capture burrowing species such as gobies (Nordby and Zedler, 1991; Desmond et al., 2000) and benthic species which can pass underneath the lead line (Layman and Smith, 2001; Steele et al., 2006). In addition, the capture efficiency of seines has been reported as low and variable (Raposa et al., 2003). Seines were far more efficient at detecting small, transient taxa than both BRUVs and traps, underscoring their importance in assessing the nursery value of California saltmarshes.

Table 3
Comparison of effort and results for the three methods employed in this study.

Replicate	Traps	BRUVs	Seines
	2 traps deployed from stake for ~20 h	1 video camera deployment for ~90 min	1 seine dragged for 15 m
Person-hours per replicate	~0.75	~5.5	~4.2
Ideal field team size	2 people	1 person	6-10 people
Average # species detected per rep	2.5	2.2	3.3
Average # species detected per person-hour	3.4	0.4	0.8
Average # individuals detected per rep	13.6	5.3	164.0
Average # individuals detected per person-hour	18.1	1.0	39.1
Especially good for	Crabs, benthic fish	Large fish	Small schooling fish
Rank for rapid assessment of diversity and multivariate community analyses	1	3	2
Rank for rapid assessment of abundance or biomass	2	3	1

Though BRUVs performed poorly in terms of overall abundance and richness, they were notable for detecting two elasmobranch species, the leopard shark and bat ray. To our knowledge these species have not been detected by similar saltmarsh tidal creek studies along the West Coast, highlighting BRUV's utility for detecting large, elusive species. While BRUVs can effectively sample large, water-column (Gladstone et al., 2012) or fishery-important species in estuaries (Unsworth et al., 2014), they are ineffective for small or benthic species. Similar to Unsworth et al. (2014), we found that a lack of water clarity was a consistent issue when using BRUVs, necessitating several redeployments which cost time and money. Alternative approaches not constrained by water clarity include 3-D acoustic imaging (Zhao et al., 2024) and eDNA (Yao et al., 2022). Additionally, recent advances in deep-learning approaches for automated underwater organism detection (Li et al., 2026; Gu et al., 2026) may improve the scalability of BRUV-based monitoring in future large-scale studies. While BRUVs are capable of detecting unique species, they required considerably more effort in our study than traps or seines and failed to detect high abundance.

If resources are limited and only one method can be employed in a rapid assessment of mobile creek fauna in an estuary similar to Elkhorn Slough, we recommend trapping, which proved effective for distinguishing site differences while remaining logistically simple and replicable. Additionally, the logistical ease of trapping makes it well-suited for rapid assessments. However, if the goal of a study is to fully describe the mobile creek faunal community, one should employ a variety of methods, each sampling a niche complementary to the others. Although the specific performance of each sampling method will vary among estuaries, the rapid assessment framework presented here is broadly applicable. By combining complementary sampling methods with analyses of community metrics and ecological attributes, estuary managers can evaluate restoration success, identify how degradation is impacting mobile creek faunal communities, and prioritize management actions.

CRediT authorship contribution statement

Carter Newsome: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Project administration, Supervision, Visualization, Writing – original draft, Writing – review & editing. **Sadie Kolackovsky:** Investigation. **Michael J. Polito:** Investigation, Methodology, Project administration, Supervision, Writing – review & editing. **Monique Fountain:** Investigation, Methodology, Project administration, Resources, Supervision, Writing – review & editing. **Awantha Dissanayake:** Conceptualization, Methodology, Supervision, Writing – review & editing. **Kerstin Wasson:** Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Project administration, Resources, Supervision, Visualization, Writing – original draft, Writing – review & editing.

Ethics

Approval was obtained from the ethics committee of the University of Gibraltar.

Declaration of generative AI and AI-assisted technologies in the manuscript preparation process:

During the preparation of this work the authors used ChatGPT in order to assist with R script development and to improve writing clarity. After using this tool/service, the authors reviewed and edited the content as needed and take full responsibility for the content of the published article.

Declaration of competing interests

The authors 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.ecss.2026.110178.

Data availability

Data are available at https://doi.org/10.5061/dryad.zpc866tpj.

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