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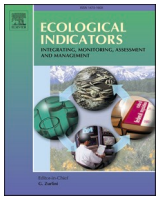

Original Articles

Assessing the suitability of EcoQS indicators to signal intertidal recreation consequences in a coastal lagoon

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ABSTRACT

Ecosystem quality status (EcoQS) indicators have been shown to be robust signallers of anthropogenic perturbations globally. Rarely, however, have they been used to assess recreation consequences in intertidal beach systems. Apart from computational differences, taxonomy knowledge gaps or those of taxon perturbation sensitivities challenge the use of EcoQS indicators to assess recreation repercussions in beach systems. We used a multi-metric approach incorporating sediment biogeochemical indicators, macrobenthic community metrics and widely used benthic EcoQS classification tools (*H'*, ABC, Bentix and M–AMBI) to quantify changes in status of intertidal sandflat ecosystems across a recreational gradient in a South African lagoon, where taxonomic and ecological knowledge gaps are evident. Highest recreation levels were associated with benthic oxygen depletion, and macrofaunal richness and abundance declines. *H'*, Bentix and M–AMBI values were strongly correlated with sediment coloration (a proxy for sediment oxygenation) and declined with increasing recreation. Since *H'* does not require detailed knowledge of species taxonomy or perturbation sensitivities, it can robustly signal EcoQS responses to recreation in regions with limited ecological and taxonomic knowledge. M–AMBI and Bentix inflated EcoQS relative to *H'*, but in the case of M–AMBI, differences were minor when sensitivities of taxa were excluded (e.g. due to coarse taxonomic resolution) or included (e.g. due to *posteriori* sensitivity assignment of taxa). M–AMBI and Bentix appear to be robust indicators of spatial changes induced by recreation, but ground-truthing scores against sediment oxygen indicators can refine EcoQS classification. Our findings offer valuable insights in the use of benthic EcoQS indicators to quantify recreation consequences, especially in regions experiencing limitations in ecological and taxonomic expertise.

1. Introduction

Coastal regions are amongst the most biologically productive ecosystems worldwide (Day et al., 1988; Burke et al., 2001). These regions also support human livelihoods and well-being, due to the provision of regulatory, supportive, cultural and recreational services, amongst others (Day et al., 1988; Costanza et al., 1997; Burke et al., 2001). Such services have contributed to coastal population growth and high human dependence on coastal resources (Defeo & Elliott, 2021). Recreational services are especially prominent in nearshore coastal systems, and noteworthy for its revenue generation regionally (Rogerson & Rogerson, 2020; Yang et al., 2021; Houston, 2024). Beaches in the USA for example, have been estimated to attract 3.4 billion visitors annually, generating $520 billion in revenue in the process (Houston, 2024). Similarly, coastal tourism generated around ¥1.8 trillion in 2019 in China (Yang et al., 2021) and R95.57 billion in 2015 in South Africa (Rogerson & Rogerson, 2020). Clearly, income generation from coastal recreational service-provision is significant, but a potential downside is the impact that recreation can have on intrinsic coastal attributes and processes such as biogeochemical cycling, ecosystem functionality, community structure and biodiversity (Wynberg & Branch, 1997; Rossi et al., 2007; Bessa et al., 2014; Lewis et al., 2022). Such impacts are likely to be most prominent in sandy beach ecosystems, considering that they are arguably the most utilised habitats for recreation globally (Lewis et al., 2022; Cabrini et al., 2025). However, the limited research afforded to sandy beach systems relative to other coastal habitats, along with a low appreciation of their intrinsic ecological relevance, contribute to impacts on sandy beaches, including from recreation,

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being poorly understood and quantified (Schlacher et al., 2008; Dugan et al., 2010; Lewis et al. 2022).

Given that sandy beaches comprise 70 % of the world's open-ocean coastlines (Dugan et al., 2010), it is key that the degree to which recreation impacts their ecosystem quality is understood. Such knowledge contributes to improvements in sandy beach management, which ultimately benefits human end-users. Notably, the benthic assemblages that inhabit beach ecosystems serve as valuable indicators that can guide decision-making (Borja & Dauer, 2008; Pinto et al., 2009), including for the detection of recreation consequences (Madell et al., 2024). Studies on recreation effects on beach systems have reported declines in richness, abundance, and diversity, associated with sediment disturbance and deoxygenation, as well as direct negative effects on organisms (Wynberg & Branch, 1997; Contessa & Bird, 2004; Rossi et al., 2007; Schlacher & Thompson, 2012; Santos et al., 2023). Globally, multimetric approaches that integrate various biological attributes of benthic macroinvertebrates have been widely employed for environmental monitoring and ecosystem quality status (EcoQS) assessments (Simboura & Argyrou, 2010; Luo et al., 2016; Liang et al., 2024). However, the utility of such approaches is challenged by various practical and contextual constraints at local levels, which have implications for detecting recreation effects on beaches. Firstly, *in situ* assessments of EcoQS across gradients of human activity run the risk of recreational effects being confounded by unrelated processes (Weston et al., 2012; Lewis et al., 2022). This point speaks to the need for careful study site selection and rigorous consideration of background co-varying processes (Lewis et al., 2022). Secondly, while multi-metric approaches have frequently been used to assess EcoQS, there has not always been agreement on their utility across different regions of the world, due to variability in ecosystem contexts and benthic community successional trajectories (Simboura & Zenetos, 2002; Simboura & Argyrou, 2010; Cabrini et al., 2025). Thirdly, some metrics are utilised for ecosystem assessments based on subjective comparisons of differences between 'impacted' and 'unimpacted' states. Examples of such indicators are community metrics such as biomass, abundance or species richness (Veloso et al., 2006; Schlacher & Thompson, 2012). In contrast, more objective EcoQS classification is achievable with metrics such as the Shannon-Wiener diversity (*H*') index, due to categorisation of *H*' levels into discrete ecosystem states (Molvær et al., 1997).

Abundance-Biomass Comparison (ABC) curves, along with associated *W*-statistics, compute the relative contribution of ranked species biomass versus abundance to infer ecosystem perturbation levels (Clarke, 1990), which like *H*', has been categorised into discrete ecosystem states (Marín-Guirao et al., 2005). More complex metrics, like M−AMBI (multivariate AZTI Marine Biotic Index; Borja et al., 2000) and Bentix (Simboura & Zenetos, 2002), require taxonomic identities of benthic invertebrates to be known, so that their pre-determined sensitivities to environmental perturbations can be explicitly included when calculating EcoQS. While these indices were originally developed for assessing benthic responses to organic pollution, they have subsequently proven valuable as EcoQS indicators for a range of anthropogenic perturbations (Borja et al., 2000; Simboura & Zenetos, 2002), including urbanisation and recreation (Liang et al., 2024; Cabrini et al., 2025). These indices are computed using freely available software with in-built species libraries containing corresponding perturbation sensitivities to assess EcoQS (Borja et al., 2000; Simboura & Zenetos, 2002). However, additional challenges arise when local species are absent in software libraries (Cabrini et al., 2025), or when taxa are identified to coarse levels (due to taxonomy knowledge gaps or lack of expertise) that are not catered for in software libraries. These problems are likely to occur in developing countries including in Africa, due to limited investments in taxonomy and ecological research (Bigot et al., 2008; Mulik et al., 2017). Nevertheless, these challenges can partially be overcome due to the flexibility in software designs, which permits manual assignment of perturbation sensitivity to taxa (Simboura & Zenetos, 2002; Borja & Muxika, 2005; Mulik et al., 2020). Here again, local knowledge gaps in the ecological sensitivity of taxa to perturbations can be problematic (Bigot et al., 2008; Mulik et al., 2017).

To address the above-mentioned challenges, we used biogeochemical-, benthic community-, and four widely used benthic indicators (*H*', ABC curves, M−AMBI, and Bentix; Shannon, 1948; Clarke, 1990; Borja et al., 2000; Simboura & Zenetos, 2002) to quantify EcoQS changes in tidal flat beach ecosystems across a recreational (trampling and benthic bait harvesting) gradient in Langebaan Lagoon, a marine-dominated system on the western coast of South Africa. We aimed to quantify EcoQS at sites most-frequented for recreation (relative to less visited sites) and to examine the robustness of these indicators in signalling EcoQS, by correlating values with benthic oxygenation (based on sediment core coloration as a proxy; Madell et al., 2024), which is a key indicator of trampling and benthic bait harvesting (Wynberg & Branch, 1994, 1997; Contessa & Bird, 2004; Madell et al., 2024). The objectives were to evaluate the degree to which recreation impacted the intertidal sedimentary ecosystem and identify metrics that could robustly signal recreation effects for future management use, especially in the context of knowledge limitations regarding benthic invertebrate taxonomy and taxon sensitivities to recreation. We also used our dataset to understand the degree of change to outcomes of M−AMBI analysis, one of the most widely used EcoQS tools globally (Borja et al., 2020), when species perturbation sensitivities were excluded (due to course taxonomic resolution) and included (through *posteriori* assignment; detailed in section 2.4.2). This approach allowed us to examine the robustness of M−AMBI in cases where both coarse and fine taxonomic resolution is used in benthic datasets, due to limitations in taxonomic expertise.

2. Materials and methods

2.1. Study sites

Langebaan Lagoon (33°10′S, 18°05′E; Fig. 1) is a permanently open, tidal lagoon located within the West Coast National Park, in the Western Cape of South Africa. It is approximately 15 km long and 4 km wide and links to Saldanha Bay in the north, wherein intertidal sandflats constitute benthic key habitats (Day, 1959; Wynberg & Branch, 1991). The lagoon has been divided into three zones (Fig. 1) to manage human usage of the system following the Sea Fisheries Act 58 of 1973. Zone A is designated as a multi-use area where recreational activities such as beach walking, kite surfing, sailing, boating, rowing, diving, swimming, water skiing, angling, bait harvesting, and other water sports are permitted (Wynberg & Branch, 1991). In this zone, sandprawns (*Kraussillichirus kraussi*), which are widely distributed and gravimetrically dominant in intertidal sandy areas, are predominantly harvested as bait using stainless-steel prawn pumps (75 cm long, 9 cm diameter) that suck up sediment and bait organisms (Wynberg & Branch, 1991). Consequently, bait harvesting and trampling, along with secondary contributions of other recreational activities listed above, disturb surface sediments in the area (Wynberg & Branch, 1991, 1994). Zone B prohibits boating, angling and bait harvesting, while Zone C is a "wilderness" area that requires permission for access (Wynberg & Branch, 1991; Nel & Branch, 2013).

2.2. Sampling design

We used a recreation gradient within Zone A of Langebaan Lagoon to achieve our objectives (Fig. 1). We focused on Zone A because permitting conditions restricted sampling in Zones B and C. While this approach limited spatial generalisation and extrapolation, it minimised confounding effects of spatial co-variance associated with sampling at greater spatial scales. For instance, sediment particle sizes vary spatially across the lagoon's zones; such variation is likely to lead to recreational effects being conflated with abiotic covariance (Nel & Branch, 2013; Madell et al., 2024). We sampled Zone A, during two seasons (winter: July/August 2022; summer: January 2023) at spring low tides, based on

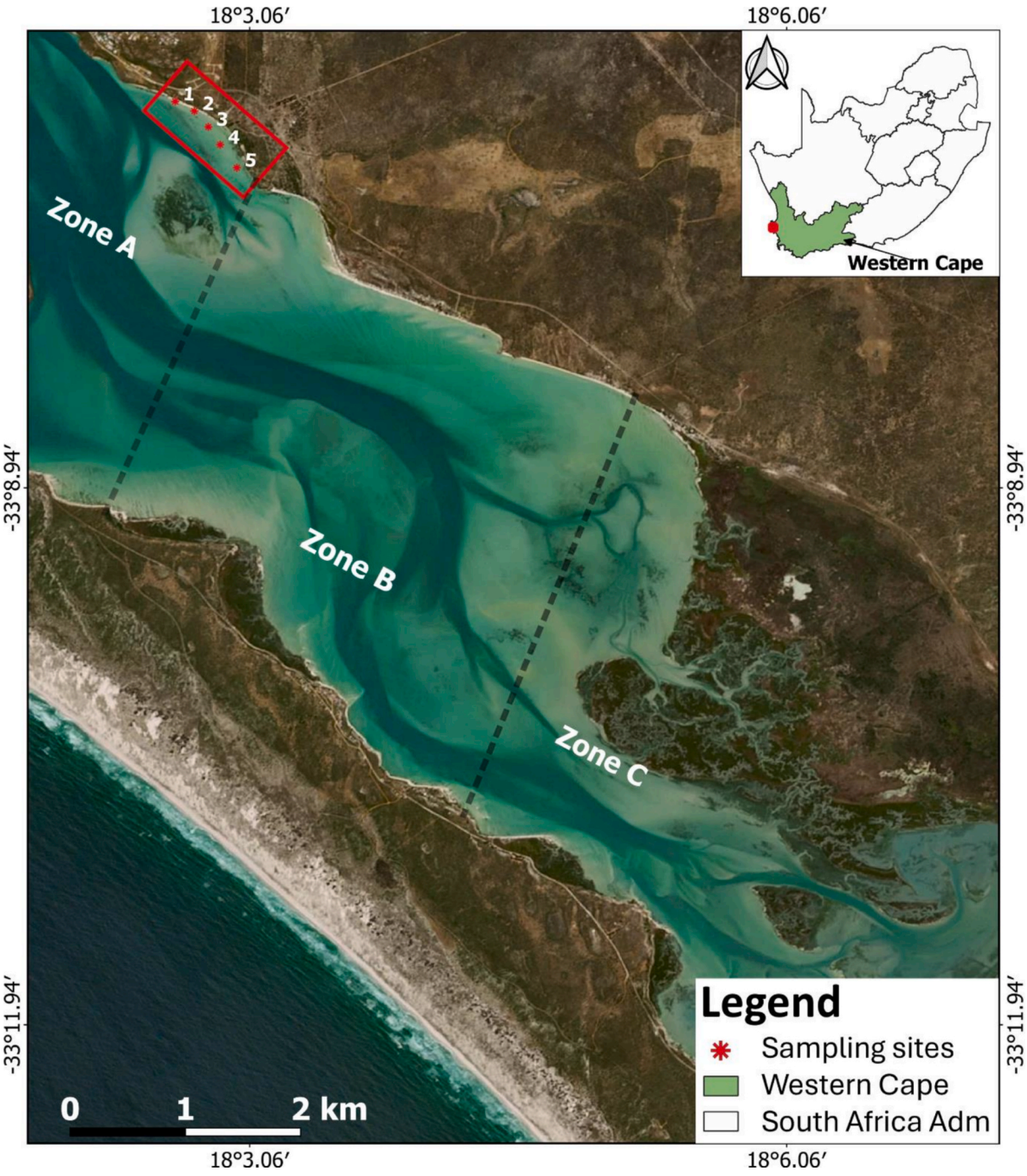

Fig. 1. Map of Langebaan Lagoon showing sampling sites 1 – 5 in Zone A (enclosed in a red rectangle) relative to Zones B and C. Inset: location of Langebaan Lagoon within the Western Cape of South Africa (red circle). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

a fixed mid-intertidal transect (start = Site 1:33°06′47″ S, 18°02′37″ E; end = Site 5: 33°07′02″ S, 18°02′57″ E) that was approximately 1 km long and ended 1 km from the start of Zone B alongshore; Fig. 1). The transect was partitioned into five equidistant (± 200 m) sampling sites and replicate samples per response variable collected randomly from a subset of ten plots (20 m apart) as described in section 2.3. For benthic microalgae, sediment particle size, organic matter content, and sediment redox conditions, sampling began at low tide. Macrofauna and phytoplankton biomass samples were collected at mid-shore during low tide while water column variables were sampled at high tide. With additional funding, sediment penetrability and porewater nutrient concentrations were measured at each site in summer 2024. All response data were collected over two days over a weekend at low tide.

2.3. Data collection

2.3.1. Recreational gradient

The number of human visitors frequenting each site was determined using a scan-sampling method (Gilby et al., 2011). This approach entailed visitors being counted every 30 min for approximately six hours (beginning at low tide) across two days over a weekend, while biotic and abiotic samples were collected.

2.3.2. Sediment biogeochemical properties

To determine sediment organic matter content and granulometry, sediment cores (diameter = 2 cm, depth = 1 cm) were collected from three (out of ten) plots per site per season. A Malvern Mastersizer (2000 SM) Auto-analyser (grain size range: 0.02 – 2000 μm) was used for granulometric analysis after sediment cores were dried (100 °C; 48 h). For organic matter content determination, cores were oven dried (100 °C; 48 h) and weighed (A&D GR-202 balance; precision: 0.00000 g) before being ignited in a muffle furnace (600 °C; 3 h). The mass change between pre- and post-combusted sediment was used as an estimate of organic matter content and expressed as a percentage of the total sediment dry weight (Contessa & Bird, 2004). Because of funding limitations, sediment colour was used as a proxy for sediment redox conditions using established methods (Rosenberg et al., 2003; Madell et al., 2024). Under reducing conditions, anoxic sediments typically appear black or dark grey due to the accumulation of iron sulphides and anaerobic decomposition of organic matter (Rosenberg et al., 2001). Using a clear Perspex corer, sediment cores (diameter = 3.2 cm, depth = ± 20 cm) were collected from five plots per site per season and photographed (iPhone 13 mini; 2340 x 1080-pixel resolution; while in the Perspex corer; Fig. A.2). Cores were held vertically and photos shot from a standard distance of one meter. All images were captured under natural light in the mid-morning, avoiding shadows and backlighting, and within a two-hour period. Image processing software (GNU Image Manipulation Program; GIMP) was used to quantify grayscale pixel numbers from zero (black) to 255 (white), using auto-colour adjusted images. Pixel frequency histograms and modal peak per sediment core were also determined using the software (The GIMP Development Team, 2023). For the image analysis, whole sediment cores were cropped (Perspex corer walls excluded) from which modal peaks were determined.

Sediment penetrability was determined per site (n = 10) using a penetrometer, which was adapted from Siebert and Branch (2005). The device comprised a steel spike (diameter = 1 cm; length = 1 m) that was raised 20 cm above the sediment surface and released to penetrate the sediment. Sediment penetrability was measured as the spike penetration depth into the sediment. A wooden frame with guide rings directed the steel spike, allowing it to be released consistently at a perpendicular angle to the sediment surface.

Porewater nutrient concentrations were determined from three replicate samples collected per site using a 50 ml syringe inserted 15 cm into the sediment, with a 63 μm filter over the inlet. Nutrient concentrations (phosphate, ammonium, nitrate and nitrite) were measured at accredited laboratory facilities of the Council for Scientific and Industrial Research (CSIR) using a nutrient autoanalyzer. However, nitrate (<5 μg/L) and nitrite (<1 μg/L) levels were below detection limits at all sites. A BenthoTorch (BBE Moldaenke; optical *in situ* probe) was used to quantify total benthic microalgal biomass at ten subsites per site.

2.3.3. Benthic macrofauna

Three sediment cores (diameter = 9.5 cm; depth = 20 cm) were collected using a stainless-steel corer at each of five plots per site and pooled (i.e. n = 5 per site). They were emptied into buckets, followed by the addition of seawater, and the contents stirred to suspend macrofauna. The supernatant was filtered using a 500 μm mesh and the process of stirring and sieving repeated five times, followed by the remaining sediment being sieved through a 2000 μm mesh to collect larger invertebrates (Cyrus & Martin, 1988). Retained macrofauna were preserved in a 70 % ethanol – Rose Bengal solution. Organisms were sorted, enumerated and identified to the lowest taxon possible. Macrofauna species names (those identified to lowest taxonomic levels) were verified using the World Register of Marine Species database (https://www.marinespecies.org).

2.3.4. Water column physico-chemical data

A CTD profiler (YSI 6600) was utilised to measure sub-surface (20 cm deep) salinity, dissolved oxygen concentration, and water temperature at each site. Phytoplankton biomass (as chl-*a*) was determined using the non-acidified *in vivo* chlorophyll *a* module on a Turner Designs Trilogy fluorometer (7200–000 Model; Turner Designs, 2019) from a 100 ml water sample at each site.

2.4. Data analyses

2.4.1. Site and seasonal trends in biotic and abiotic data

Spatio-temporal variance in visitor numbers, macrofaunal community descriptors and benthic biogeochemical variables were assessed using two-factor Analysis of Variance (ANOVA). Post-hoc testing (Tukey HSD) was employed to identify significant within-site variance where relevant. Quantile-quantile (Q-Q) plots, Shapiro-Wilk, and Levene's tests were used to evaluate the assumptions of data normality and homogeneity of variance. Response variables were log (x + 1) transformed where necessary. The data analysis software R (version 4.2.2; R Core Team, 2022) was used for all statistical analyses.

2.4.2. Macrofaunal community patterns

Since two replicate samples (from two sites in summer) were missing in the macrofauna dataset, four replicates were randomly selected (MS Excel randomise function) per site for final analysis to achieve a symmetrical dataset. To assess broad, community-level changes in macrofaunal assemblages across the recreational gradient in Zone A, the community descriptors – richness (total number of taxa), abundance (total number of organisms), biomass (total biomass of all taxa) and Shannon-Wiener diversity index (*H'*, $\log_2$), were calculated using the 'diverse' procedure in PRIMER v7. additionally, the pre-existing classification of *H'* levels into discrete ecosystem states by Molvær et al. (1997) allowed us to use it as an objective tool to assess EcoQS changes across the recreational gradient.

2.4.3. EcoQS indicators

Multiple indices were used to assess changes in EcoQS (Table 1) across the recreational gradient in Zone A. Established objective classifications of EcoQS for each metric (i.e. *H'*, ABC, Bentix and M−AMBI; Table 1) provided the basis for assessing recreation-induced changes to benthic EcoQS across the recreational gradient.

Using macrobenthic biomass and abundance data, Abundance-Biomass Comparison (ABC) curves, with *W*-statistic estimations, were performed using the 'dominance' procedure in PRIMER v7 (Clarke & Gorley, 2015). Here, we assessed two conditions: one in which all data were assessed, and a second, in which abundance and biomass data of the sandprawn (*Kraussilichirus kraussi*) were removed. We investigated the latter condition given the tolerance of sandprawns to recreation effects (Wynberg & Branch, 1991; Nel & Branch, 2013); its inclusion and exclusion allowed us to understand the degree to which their presence or absence in the dataset impacted *W*-statistics and interpretation of EcoQS trends.

We lastly applied Bentix (Bentix Microsoft Add-In v1.0; http://bent

Table 1
Ecosystem Quality Status (EcoQS) and corresponding classification scales. Molvær et al. (1997)* Shannon-Wiener diversity classification was modified in Labrune et al. (2006) for EcoQS assessment. Refer to references cited for the definitions of EcoQS.

EcoQS	*H'*	*W*-values	Bentix	M−AMBI
High	> 4	0.50–1.00	4.5–6.0	> 0.77
Good	3–4	0.15–0.49	3.5–4.5	0.53–0.77
Moderate	2–3	−0.14 to 0.14	2.5–3.5	0.38–0.53
Poor	1–2	−0.49 to −0.15	2.0–2.5	0.20–0.38
Bad	0–1	−1.00 to −0.50	0.0–2.0	< 0.20
References	Molvær et al. (1997) *	Marín-Guirao et al. (2005)	Simboura and Zenetos (2002)	Borja et al. (2020)

ix.ath.hcmr.gr; Simboura & Zenetos, 2002) and M−AMBI (AMBI v 6.0; http://ambi.azti.es; 2022 species library; Borja et al., 2000) analyses to the macrobenthic dataset. M−AMBI was run on the macrobenthic abundance dataset using two scenarios. Firstly, all macrofaunal taxa were assigned to each of the five pre-existing ecological groups (EGs) defined by Borja et al. (2000; i.e. EG I: disturbance-sensitive species; EG II: disturbance-indifferent species; EG III: disturbance-tolerant species; EG IV: second-order opportunistic species; EG V: first-order opportunistic species). This entailed accepting the default suggestions by the software to assign genera to EGs. For taxa that could not be identified to at least genus level, assignment to EGs were not performed, as recommended by Borja et al. (2000). In the second scenario, we did not use the suggested software EGs. Instead, all macrofaunal taxa, including those identified to coarse taxonomic levels, were assigned *posteriori* into EGs based on their sensitivity to recreation (hereafter referred to as perturbation or disturbance sensitivity). This was achieved by graphically (Fig. A.3) assessing abundance changes per taxon from Site 1 (most recreation) to Site 5 (no recreation or 'reference' site) and then assigning taxa to EGs I, II, III and IV. EG I taxa were defined as those that had low abundance (or were absent) at Site 1, but high abundance at Site 5 (Fig. A.3a), with an increasing trend evident in between. EG II taxa were classified as those that had similar abundances between Sites 1 to 5 (Fig. A.3b), with no obvious declining or increasing trends. EG III taxa were classified as those that had high abundances at Site 1 and low or no occurrence at Site 5 (Fig. A.3c), with a consistent decline between these sites while EG IV was classified as species exclusively present at Site 1 (Fig. A.3d). Following *posteriori* classification, 18 taxa were classified into EG I, 44 into EG II, and 5 into EG III (Table 4; Fig. A.3). Only one species (*Eurydice kensleyi*) represented by a single individual, was assigned to EG IV (Table 4; Fig. A.3d), with no assignments made to EG V (first-order opportunistic species were absent from our dataset). Running scenarios 1 (using default AMBI EGs with unassigned coarsely identified taxa) and 2 (*posteriori* assignment of all taxa into EGs) allowed us to assess differences in outcomes of M−AMBI analysis when species disturbance sensitivity is known and unknown.

For Bentix, 69 % of our macrobenthic taxa were not found in the default species library and could therefore not be assigned to its three EGs. (EG I: sensitive and indifferent species, EG II: tolerant species and second-order opportunistic species, and EG III: first order opportunistic species; Simboura & Zenetos, 2002). We therefore utilised the approach described in scenario 2 above for M−AMBI analysis in estimating Bentix values. However, we combined M−AMBI EGs I and II (i.e. sensitive and indifferent species) and assigned them to Bentix EG I while M−AMBI EG III (i.e. tolerant species and second-order opportunistic species) were assigned to Bentix EG II. Since we had no first order opportunistic taxa, none were assigned to Bentix EG III.

2.4.4. Performance of ecosystem metrics

To assess the performance of the metrics employed, we correlated (in R) indicator values with sediment coloration (Fig. A.2) to quantify relationship strengths. Specifically, mean sediment greyscale intensity per site across summer and winter were correlated with mean site EcoQS values (n = 10). We elected to use sediment coloration as a putative predictor of EcoQS given several studies highlighting darker sediment coloration (deoxygenation) and reduced redox conditions to be an indicator of recreation effects in Langebaan Lagoon and other systems (Wynberg & Branch, 1994, 1997; Contessa & Bird, 2004; Madell et al., 2024). Additionally, sediment coloration reflects decadal-scale effects of recreation to which benthic macrofauna respond, while visitor numbers that we measured per site may vary on the scale of days due to factors such as weather.

3. Results

3.1. Visitor numbers

The number of visitors in the study area varied significantly among sites, seasons, and their interaction ($p < 0.0001$; Table 2). Site 1 was most frequented for recreation, with low or no visitation at the other four sampling sites in both seasons. However, visitation was approximately 10-fold higher in summer but was highly localised to Site 1 (Table 3).

3.2. Water and sediment physico-chemical properties

Variation in water temperature, dissolved oxygen, salinity and phytoplankton biomass was minimal across sampling sites. Similarly, sediment organic matter content, and benthic microalgal biomass did not differ significantly across the recreation gradient ($p > 0.1$; Tables 2 & 3). Variance in sediment particle size among sites was detected, with Sites 1 and 2 having smaller particle sizes compared to other sites ($p = 0.0017$; Tables 2 & 3). Sediment greyscale intensity similarly varied among sites ($p = 0.0395$; Tables 2 & 3) with a significant site x season interaction ($p = 0.0129$; Table 2). Generally, lowest greyscale values were recorded at Sites 1 and 2 (indicating darker sediment), relative to Sites 3 to 5. Sediment penetrability (Summer 2024 data) varied among sites ($p = 0.00011$, Table 2) and was lowest at Site 1 (more compact) relative to the remaining sites. Porewater ammonium concentrations similarly differed among sites ($p = 0.023$, Table 2) with levels at Site 1 being approximately four times higher than Site 5 (Table 3), while there was no significant variation in porewater phosphate levels across the sites ($p = 0.971$, Table 2).

3.3. Macrofaunal community patterns

Macrofaunal taxonomic richness varied among sites and seasons ($p < 0.0001$; Table 2; Fig. 2a) and generally increased from Site 1 to 5 during both sampling seasons, but with higher values in summer relative

Table 2
Results of ANOVA testing the main and interactive effects of predictor variables on abiotic data, visitor numbers. *F* value = *F* statistic and *p*-value = significance level. Statistically significant outcomes are in **bold**.

Variables	Predictor	*F* value	*p*-value
Visitor numbers	Site	57.542	**<0.0001**
	Season	167.302	**<0.0001**
	Site x Season	25.560	**<0.0001**
Organic matter	Site	1.922	0.1460
	Season	0.713	0.4084
	Site x Season	0.739	0.5762
Particle size	Site	6.462	**0.0017**
	Season	0.037	0.8476
	Site x Season	0.842	0.5146
Greyscale intensity (mean pixel values)	Site	3.082	**0.0395**
	Season	1.999	0.1728
	Site x Season	4.171	**0.0129**
Penetrability	Site		**0.00011**
Ammonium	Site	5.52	**0.023**
Phosphate	Site	0.075	0.971
Richness	Site	28.540	**<0.0001**
	Season	33.524	**<0.0001**
	Site x Season	1.687	0.179
Abundance	Site	12.81	**<0.0001**
	Season	46.82	**<0.0001**
	Site x Season	12.55	**<0.0001**
Biomass	Site	0.366	0.831
	Season	2.864	0.101
	Site x Season	0.350	0.842
Diversity	Site	26.187	**<0.0001**
	Season	0.031	0.8625
	Site x Season	2.606	0.0555

Table 3
Variance in water column physico-chemical conditions and mean (± SE) sediment particle size, sediment organic matter content, visitor numbers at each of the five sampling sites in winter and summer. Mean (± SE) levels of sediment penetrability and porewater ammonium and phosphate are shown for 2024. Shared superscript letters denote insignificant statistical differences within a season (post-hoc Tukey tests) in cases where significant site-effects were detected (ANOVA). Sample sizes are shown per site per season. Temp = temperature, O_2 = dissolved oxygen, ‰ = salinity, PPL = phytoplankton, OM = organic matter, and MBP = microphytobenthos.

Season	Site	Temp (°C) N = 1	O_2 (µg /L) N = 1	‰ N = 1	PPL (µg Chl-a/L) N = 1	Particle size (µm) N = 3	OM (%) N = 3	Sediment coloration (greyscale intensity) N = 5	MPB Biomass (µg Chl-a/cm^2) N = 10	Visitors (/site) N = 24
Winter	1	14.50	8.90	33.40	3.80	173.20 ± 6.30^a	0.80 ± 0.20	154.00 ± 2.60^a	0.20 ± 0.00	8.30 ± 0.90^a
	2	16.00	8.60	32.60	3.40	180.30 ± 4.70^a	0.70 ± 0.00	166.60 ± 2.20^a	0.20 ± 0.00	0.10 ± 0.80^b
	3	15.80	10.10	33.90	3.50	246.60 ± 30.90^b	1.20 ± 0.10	173.60 ± 2.60^b	0.10 ± 0.00	0.00 ± 0.00^b
	4	16.90	9.10	33.50	3.60	313.10 ± 27.80^b	0.90 ± 0.10	187.30 ± 3.10^b	0.11 ± 0.00	0.00 ± 0.00^b
	5	16.00	9.30	33.50	5.30	220.00 ± 16.20^b	0.70 ± 0.10	172.60 ± 2.60^b	0.20 ± 0.00	0.00 ± 0.00^b
Summer	1	24.80	10.40	33.58	5.09	169.90 ± 1.00^a	0.60 ± 0.10	143.60 ± 1.70^a	0.20 ± 0.00	86.80 ± 14.60^a
	2	26.30	9.43	35.80	4.92	189.20 ± 12.60^a	0.40 ± 0.10	132.60 ± 1.60^a	0.20 ± 0.00	2.50 ± 0.80^b
	3	25.60	9.20	35.80	5.21	210.60 ± 2.90^b	0.60 ± 0.10	171.00 ± 1.20^b	0.20 ± 0.00	0.20 ± 0.20^b
	4	26.30	9.30	36.10	5.30	266.60 ± 13.60^b	0.80 ± 0.20	171.00 ± 1.20^b	0.20 ± 0.00	0.00 ± 0.00^b
	5	25.20	7.90	35.90	5.04	210.10 ± 16.90^b	0.60 ± 0.10	172.30 ± 1.30^b	0.20 ± 0.00	0.00 ± 0.00^b

Season	Site	Penetrability (cm) N=10	Ammonium (mg/L) N=3	Phosphate (mg/L) N=3
Summer 2024	1	8.90 ± 0.50^a	9.55 ± 2.58^a	296.60 ± 58.10
	2	10.10 ± 0.30^a	3.23 ± 0.90^b	395.00 ± 53.20
	3	13.80 ± 1.30^b	2.58 ± 1.42^b	340.00 ± 22.50
	4	12.90 ± 0.80^b	2.32 ± 0.38^b	292.30 ± 77.80
	5	13.10 ± 0.90^b	No Sample	No Sample

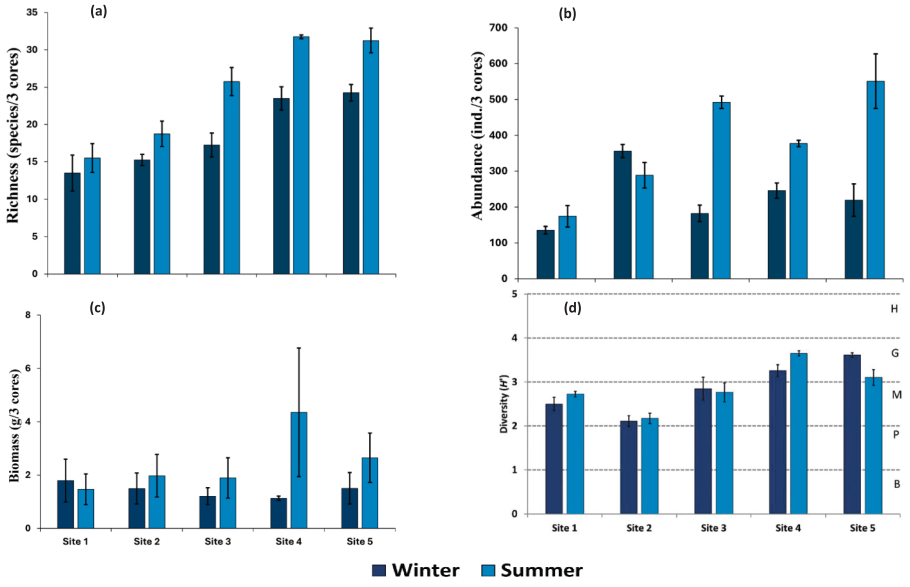

Fig. 2. Spatio-temporal variance in macrofaunal (a) richness, (b) abundance, and (c) biomass and (d) Shannon-Wiener diversity index (mean ± S.E) in sites 1 – 5. Dashed lines show ecosystem classification thresholds, where H = High, G = Good, M = Moderate, P = Poor and B = Bad.

to winter. Macrofaunal abundance similarly varied among sites and seasons, with a significant site x season interaction ($p < 0.0001$; Table 2). Abundance increased uniformly from Site 1 to 5 in summer, but this trend was not evident in winter (Fig. 2b, Table 4). For macrofaunal biomass (Fig. 2c), there was no significant variation among sites, seasons and the interaction between them, with trends being unclear.

3.4. EcoQS indicators

Shannon-Weiner (*H'*) diversity varied among sites ($p < 0.0001$; Table 2), with general increasing trends from Site 1 to 5, as was evident for richness. In terms of EcoQS (based on *H'*), Sites 1–3 were categorised as 'moderate' and Sites 4 and 5 as 'good' (Fig. 2d). Trends in variance of *W*-statistics irrespective of whether sandprawn (*K. kraussi*) abundance and biomass data were included or excluded were inconsistent across the recreation gradient (Fig. 3). However, when sandprawn data were excluded, *W*-statistics decreased, notably at Site 1 (range: 0.02 – 0.06; Fig. 3a), relative to when sandprawn data were included (range: 0.24 – 0.25; Fig. 3b). Generally, EcoQS for sampling sites was categorised as 'moderate' (the exception being Site 4, summer, categorised as 'good'), based on *W*-statistics calculated using the dataset excluding sandprawns. In contrast, EcoQS for Site 1 shifted to 'good' when sandprawn data were included.

In both seasons, Bentix values were lowest at Site 1 (Figs. 4a & 4b), with EcoQS categorised as 'good'. Bentix values increased from Sites 2 to 5, and EcoQS categorisation shifted to 'high' in winter and summer.

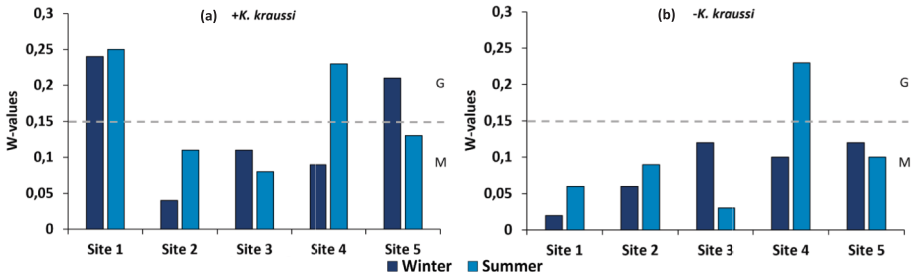

Fig. 3. Spatio-temporal variance in ABC (*W*-values) in sites 1 – 5. Dashed lines show ecosystem classification where G = Good, and M = Moderate.

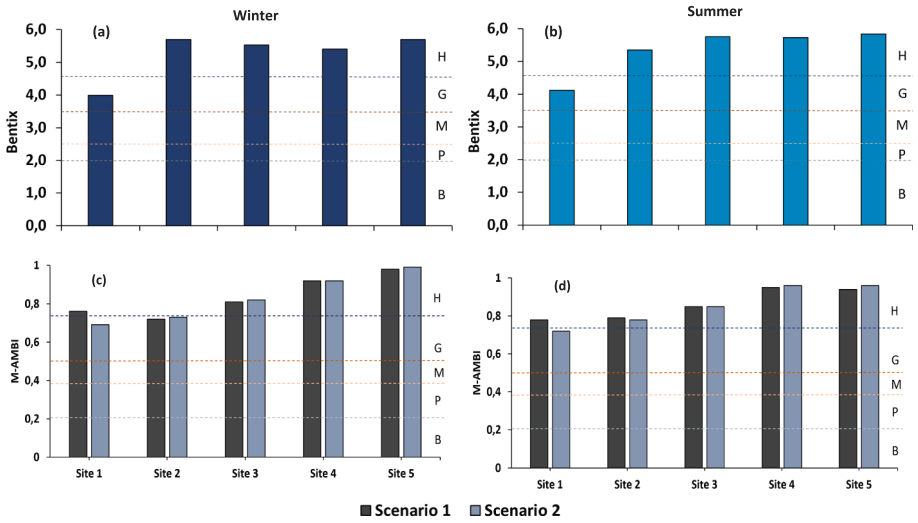

Fig. 4. (a & b) Bentix, (c & d) Scenarios 1 and 2 M−AMBI values for sites 1–5, and their respective disturbance categories (status). Dashed lines show ecosystem classification thresholds where H = High, G = Good, M = Moderate, P = Poor and B = Bad. The assignment of macrofaunal taxa to ecological groups is shown in Table 4 and Fig. A.3, along with their abundances.

Trends for M−AMBI scores mirrored those for Bentix (Figs. 4c & 4d), with values generally increasing from Site 1 to 5 in both seasons. Irrespective of the scenario, M−AMBI scores at Sites 1 and 2 were generally around the interface of 'high' and 'good' EcoQS categories, whereas for the remaining sites, scores approached the upper limits of the 'high' EcoQS category. Notably, the use of Scenario 2 data (relative to Scenario 1 data), did not appear to significantly influence spatial trends for M−AMBI scores, but it did result in a lowering of scores by 9 % and 8 % respectively in winter and summer (Table A.1) at Site 1 in particular, where EcoQS categorisation shifted from 'high' to 'good'.

3.5. *Performance of EcoQS indicators*

Greyscale intensity weakly correlated with macrofaunal biomass (*r*

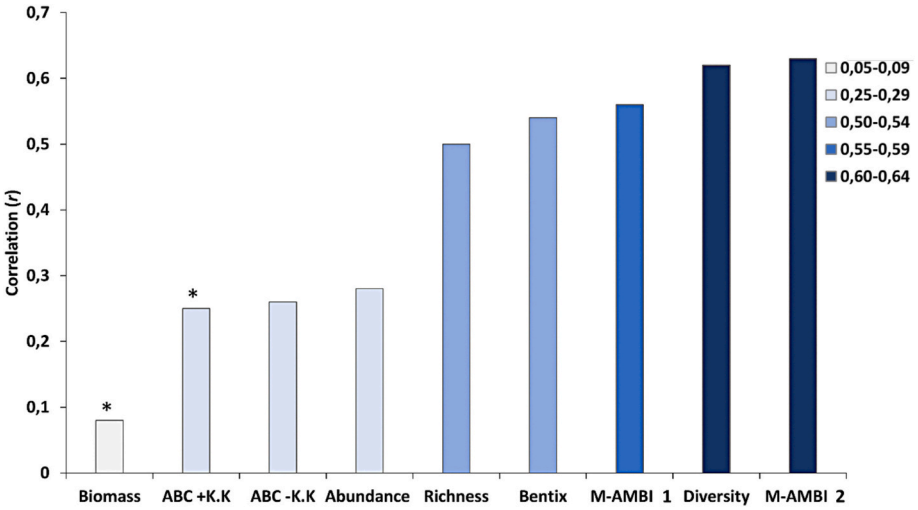

Fig. 5. Pearson correlation (*r*) of greyscale intensity with ecosystem metrics. *Denotes negative correlation, + K.K = Abundance − Biomass Comparison curve with *K. kraussi*, −K.K = Abundance − Biomass Comparison curve without *K. kraussi*, M−AMBI 1 = Scenario 1 M−AMBI and M−AMBI 2 = Scenario 2 M−AMBI.

= -0.08), abundance (r = 0.28) and *W*-statistics (+ sandprawns r = -0.25; − sandprawns r = 0.26). In contrast, stronger correlations were recorded between greyscale intensity and richness (r = 0.50), diversity (r = 0.62), Bentix (r = 0.54), and M−AMBI values (Scenario 1 r = 0.56; Scenario 2 r = 0.63; Fig. 5), with diversity and Scenario 2 M−AMBI scores being most correlated with sediment greyscale intensity (Fig. 5).

4. Discussion

4.1. Visitor numbers

Spatial trends in human visitor numbers that we recorded corroborated the presence of a recreational gradient in Zone A of Langebaan Lagoon. Specifically, visitation was highly localised to Site 1 during both winter and summer (Table 3), which is likely due to the site's proximity to human conveniences (parking area, restrooms and restaurant; Madell et al., 2024). Notably, visitor numbers were roughly 10 times higher in summer than winter, which aligns with the finding of Wynberg and Branch (1991), of a summer peak in visitation in the lagoon.

4.2. Water and sediment physico-chemical properties

Within the recreational gradient, there was limited variation in water temperature, dissolved oxygen concentration, salinity, phytoplankton biomass, benthic organic matter and microalgal biomass (Table 3), suggesting minimal co-variance with human visitation. This bears noting considering that identifying recreational effects on intertidal systems may be challenged by covariance from background environmental variability (Lewis et al., 2022; Madell et al., 2024). Nevertheless, sediment particle size did vary among sites (Table 3), with finer granulometry recorded at Site 1; sediment was classified as fine (Sites 1 and 2) to medium sand (Sites 3 to 5; Wentworth, 1922). This spatial granulometric change is likely a result of differences in recreation intensity across the sampling area, since it is unlikely for other natural processes to cause granulometric variation at the scale of 200s of meters in the lagoon (Madell et al., 2024). Moreover, Wynberg and Branch (1994) reported more fine-grained particles in areas most trampled and harvested for bait in Langebaan Lagoon, commenting that trampling-induced sediment compaction, subsequent sediment depression, and reduced sandprawn bioturbation through harvesting, were the likely drivers of this difference in sediment grain size. The low sediment penetrability that we measured at Site 1 (Table 3), coupled with qualitative indications of reduced bioturbation at the same site (Fig. A.1), support the mechanisms proposed by Wynberg and Branch (1994).

The darker sediment coloration (Fig. A.2; indications of benthic oxygen depletion) we detected at sites with greatest recreational visitor numbers aligns with findings of Wynberg and Branch (1994, 1997; Langebaan Lagoon) and Contessa and Bird (2004; Coronet Bay, Australia), and corresponds with the low sediment penetrability measured at Site 1. Sediment compaction is known to restrict oxygen diffusion into sediment, leading to anaerobic conditions, which is exacerbated by the removal of bioturbators or a depression of their activity (Wynberg & Branch, 1994; Contessa & Bird, 2004; Rossi et al., 2007; Fig. A.2). Lastly, the four-fold increase in porewater ammonium concentration at Site 1 relative to the furthest sites (Sites 3 & 4; Table 3), potentially indicates impaired nitrification, which is plausible given that nitrification is an oxygen-dependent process (Jäntti & Hietanen, 2012).

4.3. Macrofaunal community patterns

The increasing trends in macrofaunal abundance and richness (Figs. 2a & 2b) that we detected from Site 1 to Site 5 contextualise the detrimental nature of trampling and bait harvesting for benthic communities (Wynberg & Branch, 1994; Pillay & Branch, 2011). Similar declines in benthic richness and abundance in Langebaan Lagoon in response to increasing trampling and bait removal have been documented, likely due to benthic compaction, and sediment deoxygenation (Wynberg & Branch, 1994, 1997). Organismal damage, displacement or mortality, and increased susceptibility to predation of bycatch macrofauna, likely drive the declining abundance and richness trends with recreation (Wynberg & Branch, 1991, 1994, 1997). Our current findings and those of similar research from Langebaan Lagoon and elsewhere (Schlacher & Thompson 2012; Santos et al. 2023), highlight the potential for these metrics to be useful indicators of spatial changes in benthic communities in response to recreation. However, drawing conclusions about EcoQS is difficult due to the absence of objective classification criteria for these metrics as far as we are aware.

4.4. EcoQS indicators

Of the four metrics used to assess EcoQS, *W*-statistics were notable for their erratic signalling of spatial trends across the recreation gradient, relative to the other three (*H'*, M−AMBI, Bentix; see Fig. 2d to 4). However, our inclusion and exclusion of sandprawn (*K. kraussi*) biomass and abundance in the analysis revealed interesting findings, especially in the context of the utility of ABC curves and *W*-statistics to assess perturbation in ecosystems dominated by high-biomass, endobenthic engineers. The EcoQS categorisation of Site 1 as 'good' in both seasons (Fig. 3) when sandprawn data were included, and the subsequent EcoQS reduction in this metric at the same site when sandprawns were excluded (EcoQS shifted to 'moderate'), highlights the potential for high biomass taxa that are numerically rare to be highly influential determinants of EcoQS when using ABC curves and *W*-statistics. In our benthic community data (Table 4), it was evident that sandprawns were not reduced in abundance at Site 1, where most visitation occurred, relative to remaining sites. This trend indicates a degree of resilience of sandprawns to trampling and bait harvesting, which may be a product of their active, endobenthic lifestyles, which in turn facilitates emigration into deeper, less-impacted sediment. This trait has been used to explain the high sandprawn population biomass in Langebaan Lagoon (Christie & Moldan, 1977; Wynberg & Branch, 1991). Additionally, the lack of a planktonic larval stage in sandprawns (they are direct developers; Forbes, 1973) implies that populations may not be impacted by mortality associated with planktonic dispersal. While human recreation can affect early life history stages of sandprawns as higher frequencies of embryo aberrations and arrested development have been shown at sites with high intensity bait harvesting and trampling (Madell et al., 2024), the adult endobenthic crustaceans are recognised for their tolerance to benthic hypoxia, hypercapnia and high hydrogen sulphide levels (Pillay & Branch, 2011).

Irrespective of whether sandprawns were excluded or included in the analyses, *W*-statistic (Fig. 3) trends across the recreation gradient were unclear and poorly correlated with sediment coloration. These findings suggest that ABC curves and *W*-statistics may not always be robust indicators of benthic responses to recreation in systems dominated by high biomass endobenthic engineers. Evidently, the general theory underlying ABC curves and *W*-statistics, that environmental perturbations induce community switches from long-lived, high-biomass *K*-strategists to short-lived, low-biomass *r*-strategists (Wetzel et al., 2012), may not always hold true, possibly due to resilience conferring traits of high biomass taxa in relation to the perturbation intensity, or greater variability in biomass data relative to abundance data (Fig. 2).

In contrast to ABC curves and *W*-statistics, *H'*, M−AMBI and Bentix all showed consistent declining trends (Fig. 2d to 4) with increasing recreational activity and were also most strongly correlated with sediment coloration (Fig. A.2). There were however minor EcoQS categorisation disagreements, especially under maximum recreation levels. The similarity in response trends of *H'* relative to richness and abundance is unsurprising, considering that *H'* integrates both community measures in approximating diversity and EcoQS levels (Leshno et al., 2016). It is important to point out that during this study, we noted differences in *H'* ($\log_2$) values calculated by different software for the same

Table 4
Macrofaunal species abundance (average) and distribution per site during summer and winter. Superscripts denote higher taxonomic classification; values in parentheses show the ecological groups (Fig. A.3) assigned to taxa in Scenario 2 of M–AMBI analysis. A – Amphipoda, Al – Alpheidae, An – Anomura, Ax – Axiidea, Az – Anthozoa, B – Bivalvia, Br – Brachyura, C – Cumacea, G – Gastropoda, I – Isopoda, L – Leptostraca, M – Mysida, P – Polychaeta, T – Tanaidacea.

Taxon	Winter					Summer				
	Sites					Sites				
	1	2	3	4	5	1	2	3	4	5
Africofusus ocelliferus^{G} (I)										2
Ampelisca palmata^{A} (I)				1	2	1	1	6	13	63
Antinoe lactea^{P} (II)	1			1				1		
Arabella iricolor^{P} (III)	2	1				1				
Argobuccinum pustulosum^{G} (I)										1
Assiminea globulus^{G}(II)		1						1		2
Betaeus jucundus^{Al} (II)	1	2	1			1	1	1		
Capitellidae (II)		2			1					2
Carditella capensis^{B} (II)	1		4	1	3	1	2	8		2
Ceinidae (II)					1	1			3	8
Cirriformia tentaculate ^{P}(II)				4	1		1	3	4	1
Clam (II)									4	
Copepoda (III)	62	26	23	37	19	60	40	24	25	20
Cumopsis africanum^{C} (I)				1	1			1		1
Cylista ornata^{Az} (I)					1				2	
Danielella edwardsii^{Br} (II)	2	3		5			2			1
Diogenes brevirostris^{An} (II)							1			
Euclymene spp.^{P} (II)		1	3	2	4		3	15	11	7
Eurydice kensleyi^{I} (IV)						1				
Gastrosaccus psammodytes^{M} (II)			5							
Glycera tridactyla^{P} (II)			1				1	1	1	
Griffithsius latipes^{A} (I)	3	8	3	3	7	11	9	4	30	74
Hymenosoma orbiculare^{Br} (II)	1	4	2	3	1	2			1	
Idotea metallica ^{I}(II)				1		3			1	2
Kellia rotunda^{B} (II)	5	4	4	10	9	4	2	4	9	4
Kraussillichirus kraussi^{Ax} (II)	3	4	1	3	4	11	16	12	8	10
Lasaea adansoni^{B} (II)	9	20	1		6	7		3	1	3
Marphysa haemasona^{P} (I)	1	2	2	3	6		1	2	4	3
Marphysa spp.^{P} (II)						2	6	5	6	4
Mollusca A (II)	2	8		1	2	2	2	1	1	1
Nassarius capensis^{G} (I)									1	1
Nassarius niveus^{G} (I)					1					
Natatolana hirtipes^{I} (I)						1	1		1	3
Nebalia capensis^{L} (I)									20	18
Nephtys spp.^{P} (I)				1	1					
Nereididae (I)	1	1	12	6	25		4	10	8	5
Notomastus latericeus^{P} (I)	1	4	1	50	17	1	10	23	11	13
Notomastus spp.^{P} (II)	3	3	9	12	4	1	8	20	4	4
Orbinia angrapequensis^{P} (II)	22	17	12	28	15	10	10	16	8	7
Ostracoda (I)					1	1	1	1	5	22
Polychaeta A (II)			7	19	12		2	3	4	1
Polychaeta B (II)		2	11	15	17	1	13	82	83	47
Polychaeta C (III)	6	1				17	4	4	1	1
Polychaeta D (II)	1	0		2	3	3	2	7	7	4
Polychaeta E (II)	3		1,5	7	2		2	1	2	2
Polychaeta F (II)		1					1			
Polychaeta G (I)					1					1
Polychaeta H (II)			20	17	1,5		2	3	1	1
Polychaeta I (I)					3				3	11
Polychaeta J (II)				1						
Polydora spp. A^{P} (II)	1			1	1			1	1	1
Polydora spp. B^{P} (II)			5	2	2					
Prionospio cirrifera^{P} (II)			3	1					2	2
Prosphaerosyllis semiverrucosa^{P} (II)					1				3	1
Prunum capense^{G} (II)		1		2	1,5			1	3	
Scoloplos spp.^{P} (II)			1	1					1	
Simplisetia erythraeensis^{P} (II)	3	53	36	16		3		1		
Spionidae A^{P} (II)					1		0	3	19	9
Spionidae B^{P} (I)	1			3	16			1		
Spionidae C^{P} (II)				2			0	3		
Spionidae D^{P} (II)					3	3	3		2	
Spiroplax spiralis^{Br} (II)		0	1				1		1	
Tellimya trigona^{B} (III)				1	1	4	1	2	2	2
Thelepus spp.^{P} (II)	2	3	3	1	2	5	4	5	6	2
Turritella capensis^{G} (II)					1		3	5	1	3
Urothoe grimaldii^{A} (II)	18	207	55	50	58	42	171	236	93	206
Zeuxoides helleri^{T} (II)			3	1						1

dataset (our benthic macrofauna in Zone A; Table A.2). *H'* values were identical when computed by Excel and PRIMER (a widely used multivariate analysis platform) but higher when calculated by M−AMBI software (with same $\log_2$). It is beyond the scope of this paper to probe how *H'* is computed by M−AMBI software, or how it is used in EcoQS determination. Nonetheless, we suggest that manual checks of *H'* values estimated by different software will improve accuracy in reporting and promote comparability in future research.

In contrast to the *H'* EcoQS categorisation of Sites 1 and 5 as 'moderate' and 'good' respectively, Bentix categorised the same sites as 'good' and 'high', an outcome mirrored by M−AMBI, but only based on Scenario 2, in which benthic taxa were assigned using *posteriori* sensitivities. The M−AMBI EcoQS classification of Site 1 as 'high' based on Scenario 1 (sensitivities of coarsely identified taxa excluded) versus 'good' based on Scenario 2, points to the value of including perturbation sensitivities of taxa to refine EcoQS determinations, though it must be recognised that EcoQS shifts were marginal on either side of the threshold differentiating 'high' (Scenario 1) and 'good' (Scenario 2) EcoQS. Generally, across all sites, differences in outcomes of Scenarios 1 and 2 analyses were small (Figs. 4c & 4d; Table A.1). This outcome is relevant considering that about 57 % of macrofauna could not be assigned to ecological groups at Site 1, while 29–39 % were unassigned in Sites 3 to 5 in Scenario 1 of our analysis (Table A.1), due to coarse taxonomic identification. Borja and Muxika (2005) recommend that M−AMBI not be used when unassigned taxa exceed 50 % in a dataset, and that caution be applied when unclassified taxa exceed 20 %. We do not disagree with this recommendation, but the minor difference in outcomes of Scenarios 1 and 2 in our analyses suggests that even when significant uncertainties of taxon perturbation sensitivities exist, M−AMBI is still relatively robust in identifying relative changes in EcoQS between 'impacted' and 'reference' sites. This robustness may stem from M−AMBI being an integration of AMBI metrics with species richness and *H'* (Borja et al., 2008). However, the inclusion of the above-mentioned diversity metrics has been suggested to lead to misinterpretations of EcoQS by M−AMBI sometimes (Simboura & Argyrou, 2010). The integration of richness and *H'* in estimating EcoQS by M−AMBI may also explain the similarity in spatial trends we recorded for M−AMBI, richness and *H'* in our study (Figs. 2 & 4).

4.5. Performance of EcoQS indicators

Predicting the performance and robustness of EcoQS indicators under different environmental contexts is complex, due in part to computational differences among indicators (e.g. relative weighting coefficients per ecological group, selection of class boundaries) and variation in benthic successional trajectories (Simboura & Argyrou, 2010). Such complexities make it difficult to resolve the utility of EcoQS indicators in studies that adopt multi-metric approaches. Careful validation of EcoQS indicators is thus necessary, potentially against environmental data, especially when indicators differ in EcoQS assessments (Simboura & Argyrou, 2010). In our study, the EcoQS classification of Site 1 as 'moderate' based on *H'*, aligns with benthic core profiles at the same site (Fig. A.2). While sediment darkening (hence oxygen depletion) was evident at Site 1, darkening was restricted to the upper 10 cm sediment layer, suggesting that deeper layers were being oxygenated, likely due to bioturbator activity. Site 1 did not show signs of azoic benthic conditions, or skewed community dominance by first and second order opportunistic species (Table 4), that are typically characteristic of benthic systems categorised as being in 'poor' or 'bad' state (Borja et al., 2020). We therefore support the *H'* EcoQS classification of Site 1 as 'moderate' and the findings that this site was not in a 'bad' or 'poor' state, based on Bentix, M−AMBI and *H'*. Considering these outcomes, and that recreational visitation was highly localised to Site 1, recreation repercussions in Zone A are currently not of management concern. Nonetheless, it is important for recreation effects to be monitored, considering that recreation is likely to grow in parallel with coastal population growth in future (Lewis et al., 2022). Secondly, since local-scale processes such as recreation are often nested hierarchically within larger-scale global change pressures (Defeo and Elliott, 2021), recreation effects may be of significance when interacting with other non-local stressors, particularly if the interaction is synergistic.

The EcoQS classification of Site 1 as 'good' or 'high' by Bentix and M−AMBI appears misaligned with sediment core darkening at Site 1 and EcoQS categorisation by *H'* (Figs. 4 & A.2), and the notable macrofaunal richness and abundance declines from Site 1 to Site 5 in our study (Figs. 2a & 2b). Researchers have suggested that M−AMBI may overestimate EcoQS in some cases relative to Bentix (Simboura & Argyrou, 2010; Mulik et al., 2017), which appears to be the case in our study (See Fig. 4). Such overestimation has been argued to arise from differences in the computational systems used, with Bentix weighting tolerant and opportunistic taxa equally and M−AMBI down-weighting tolerant taxa relative to opportunistic ones (Simboura & Argyrou, 2010). Nonetheless, others have argued that the reduction of five ecological groups (M−AMBI) to three (Bentix) has led to Bentix being more prone to inaccurate EcoQS assessments (Borja, 2004). Such disagreements highlight the difficulty in resolving the utility of different EcoQS indicators underlying multi-metric approaches (Simboura & Argyrou, 2010), but they also signal the need for testing these metrics under different ecological contexts and anthropogenic pressures.

5. Concluding perspectives

Recreation repercussions for beach ecosystems have received little attention relative to other anthropogenic perturbations globally (Lewis et al., 2022), and multi-metric approaches that explicitly include taxon sensitivity (e.g. Bentix and M−AMBI) have rarely been applied to understand benthic responses to recreation. In Southern Africa, we are unaware of the use of such approaches to assess EcoQS responses to beach recreation. Findings from our multi-metric analysis indicate that *H'* is a robust indicator of beach EcoQS, being one of the two most strongly correlated indicators with sediment coloration (a proxy for oxygenation; Fig. 5; Fig. A.2). Since this indicator does not require detailed knowledge of species taxonomy or perturbation sensitivities for the determination of EcoQS, it can be particularly useful in assessing recreation responses in regions with taxonomy and ecology knowledge gaps. The effectiveness of diversity metrics such as *H'* has been questioned as EcoQS indicators, since they are often influenced by processes outside those of primary interest, including methodology differences such as sample size variance (Simboura & Argyrou, 2010). Such differences make comparisons across studies difficult, but this is less problematic for spatio-temporal comparisons within studies, since sampling effort is typically constant across space and time. In the long term though, investments to address the above-mentioned knowledge gaps are critical, so that species-specific responses can be monitored. The degree of colonisation by non-native species for example, or localised extirpations of rare species in response to recreation, is key information for beach management. This point highlights the need for the taxonomy of the coarsely identified taxa in our study to be resolved in future.

While Bentix values were well-correlated with sediment coloration in our study following *posteriori* taxon sensitivity assignment, 69 % of our taxa could not initially be assigned due to their absence in the species library. Bentix may therefore not always be applicable in some regions due to the limited species library, making its use challenging in cases where ecological and taxonomic knowledge gaps exist. Lastly, M−AMBI appeared robust in identifying spatial changes across the recreation gradient in our study, irrespective of whether perturbation sensitivities of coarsely identified taxa were included or excluded. This finding suggests that M−AMBI may be useful in indicating EcoQS even when species identities are unknown. However, based on our findings, M−AMBI, like Bentix to a smaller degree, may inflate EcoQS relative to *H'*, possibly due to the weighting system used for tolerant and opportunistic species by M−AMBI and Bentix. *H'* in contrast, amalgamates

species richness and evenness, without weighting contributions of tolerant and opportunistic species. The potential for M−AMBI and Bentix to overestimate EcoQS could be addressed by adjusting the weightings given to ecological groups, by correlating EcoQS values with sediment oxygenation levels for example. Sediment coloration (Fig. A.2) assessments are inexpensive and uncomplicated and can be used to adjust M−AMBI and Bentix scores in response to recreation effects.

In conclusion, we support the use of multi-metric indicators (Borja & Muxika 2005; Schéré et al., 2023) such as *H'*, M−AMBI and Bentix to understand EcoQS changes induced by beach recreation. These metrics can assist in managing beach ecosystems and thereby preserve valuable socio-ecological functions and services these ecosystems provide.

CRediT authorship contribution statement

Charles Faseyi: Writing – review & editing, Writing – original draft, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Geena Williams:** Writing – review & editing, Methodology, Investigation, Formal analysis, Conceptualization. **Ursula Scharler:** Writing – review & editing, Methodology, Funding acquisition, Conceptualization. **Candida Savage:** Writing – review & editing, Methodology, Funding acquisition, Conceptualization. **Deena Pillay:** Writing – review & editing, Writing – original draft, Validation, Supervision, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

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.ecolind.2025.114450.

Data availability

Data will be made available on request.

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