Ecological Indicators 160 (2024) 111804

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

Developing a bioassessment framework to inform tidal stream management along a hydrologically variable coast

Alexis J. Neffinger, Natasha J. Breaux, Abraham D. Margo, Terence A. Palmer, Stacy N. Trackenberg, Jennifer Beseres Pollack *

Harte Research Institute for Gulf of Mexico Studies, Texas A&M University–Corpus Christi, 6300 Ocean Dr., Corpus Christi, TX 78412, United States

ARTICLE INFO

Keywords:
Bioindicator
Gulf of Mexico
Index of biotic integrity
Macrobenthos
Nekton
Texas

ABSTRACT

Tidal streams are spatiotemporally varying areas that encompass tidally influenced limnetic and oligohaline zones within estuaries. These areas are important for many biogeochemical processes and for the life cycles of many fishery species. However, tidal streams are also susceptible to impairment from coastal development and watershed-derived runoff, which potentially affects faunal assemblages within the ecosystem. This study developed indices of biotic integrity (IBIs) for nekton and benthic macroinfauna in tidal streams along the southern Texas coast. Fifteen tidal stream sites with mean salinities ranging from 0.4 to 11.9 were classified as degraded if their surrounding land use was > 20 % urban or agricultural, watershed population density was > 50 km^{-2}, and nutrient and chlorophyll concentrations exceeded specific screening limits. Otherwise, sites were classified as reference. Nekton and benthic macroinfauna communities were then sampled at these fifteen stream sites in 2020 and 2021. Historical metrics and metrics derived from multivariate analyses were considered for inclusion in the IBIs, and were assessed for collinearity, redundancy, suitability for score assignment, and agreement with historical literature. Nine univariate nektonic metrics (including total abundance, number of invertebrate taxa, and the percent abundance of five species, one family, and one functional group) and six benthic macroinfauna metrics (including Shannon's diversity, total abundance and biomass, and the percent abundance of two taxa and one functional group) were incorporated into separate nektonic and benthic IBIs. Mean IBI scores of reference sites were greater than degraded sites by 42 % for nekton and 30 % for benthic macroinfauna. Seven out of eight reference sites had greater mean nekton IBI scores than the mean scores of all seven degraded sites, while four of eight reference sites had greater benthic IBI scores than all degraded sites. However, overlap in the ranges of scores calculated for degraded and reference sites occurred, which is likely caused by spatiotemporal variability, including stream size variation and the changing climatic and biogeographical gradient along the southern Texas coast. The IBIs developed in this study represent an important preliminary step in bioassessment development for Texas tidal streams, and will help to provide a useful tool for coastal environmental management.

1. Introduction

Human development in coastal areas has fundamentally changed the structure and functioning of estuarine systems (Kennish, 2002; Lotze et al., 2006). Overexploitation of coastal resources, habitat destruction, and eutrophication are linked to increased development and have affected estuarine faunal biodiversity, trophic structure, and habitat utilization, among other ecological effects (Beach, 2002; Lotze et al., 2006; Freeman et al., 2019). Costs of development are not limited to the faunal communities that utilize estuaries – human health, industry, and safe recreation in coastal areas are also being threatened. Important ecosystem services, such as those provided by healthy fisheries (Houde and Rutherford, 1993) and shoreline-protecting habitats (Chowdhury et al., 2021), are being depleted, and health concerns, including degraded water quality (Freeman et al., 2019) and toxic algal blooms (Anderson, 2009), are becoming more prevalent. Today, the Gulf of Mexico is the fastest growing coastal region in the United States of America (USA), with a population growth rate of 26.1 % between 2000 and 2017 (Cohen, 2019). Resource managers can better assess the ecological impacts of the rapid development in Gulf of Mexico coastal

* Corresponding author.
E-mail address: jennifer.pollack@tamucc.com (J. Beseres Pollack).

https://doi.org/10.1016/j.ecolind.2024.111804
Received 18 September 2023; Received in revised form 9 January 2024; Accepted 24 February 2024
Available online 2 March 2024

areas by utilizing bioassessment tools (Karr, 1981).

Across the Gulf of Mexico and broader USA, resource managers have developed robust physicochemical monitoring programs to assess anthropogenic impacts on water quality. However, more recent shifts to incorporate ecological health (=integrity) into assessment of water bodies have identified a need to also develop standardized bioassessment protocols (Karr, 1991; Latham, 2010). The inherent spatiotemporal variation in estuarine systems has posed problems for resource managers developing bioassessment protocols (Elliott and Quintino, 2007; Tweedley et al., 2015). Tidal streams – tributaries situated in the most upstream portion of estuaries that span from tidally influenced limnetic (salinity < 0.5) to oligohaline (salinity 0.5 – 5.0) waters (McLusky, 1993; Jones et al., 2020) – are particularly dynamic portions of the estuary. Various characteristics of tidal streams can complicate the development of bioassessment strategies for these systems. Firstly, these systems are generally understudied within estuaries compared to both the upstream freshwater and the downstream, more saline zones (McLusky, 1993). Secondly, the response of tidal stream biota to anthropogenic stress can be difficult to detect because of natural, wide-ranging spatiotemporal variation which may lead to natural stress on organisms within the ecosystem (Elliott and Quintino, 2007; Mabe and Moring, 2008; Tolan and Nelson, 2009; Barendregt and Swarth, 2013). Lastly, coastal development along tidal streams has made pristine conditions nearly impossible to define; rather, tidal streams often exist along a continuum of degradation with no clear baseline conditions (Barendregt and Swarth, 2013). Because of the latter two characteristics, tidal streams often lack sensitive indicator species that, in other systems, are usually relied upon to identify areas of degradation (Tolan and Nelson, 2013). Therefore, a more holistic approach is needed for bioassessment of tidal stream systems.

Despite factors that may complicate the development of a bioassessment protocol for tidal streams, management of these systems is of interest because of their ecological and economic importance, and susceptibility to degradation. Tidal stream ecosystems provide key nursery habitat for freshwater and saltwater fishery species (Hackney et al., 1976; Mallin and Lewitus, 2004; Wessel et al., 2021), mediating nutrient transformation and transfer into the downstream estuary (Barendregt and Swarth, 2013), and sequestering carbon (Loomis and Craft, 2010). Tidal streams have increased potential for impairment because they are situated at the receiving zone for upstream pollution loads, are fringed by the rapid land development associated with the coast, and have a limited area when compared to other locations within the estuary (Bergquist et al., 2011; Barendregt and Swarth, 2013). To ensure that ecosystem services provided by tidal streams are sustained, development of bioassessment protocols is warranted to supplement physicochemical monitoring and facilitate more robust management strategies.

One way resource managers have implemented successful bioassessment protocols is through the development of multi-metric indices of biotic integrity (IBIs) for specific habitats and biological community types (Karr et al., 1986; Simon and Lyons, 1995). The creation of an IBI involves identifying and combining a suite of ecological metrics into a single index that respond uniquely to a broad range of human impacts (Karr et al., 1986; Simon and Lyons, 1995; Capmourteres et al., 2018). Methodologies used to integrate metrics into an IBI range from simple investigations of candidate metrics adapted from previous studies (Simon and Lyons, 1995) to more statistically intensive and multi-pronged evaluation processes (Stoddard et al., 2008). The resulting index scores, which are easily understood and used by resource managers, can be used to connect anthropogenic influence and aquatic life use after linking these metrics to scoring criteria developed from baseline condition data. An IBI is a versatile bioassessment tool that can be (1) adapted for use in different aquatic systems, such as streams, lakes, wetlands, and estuaries (Karr, 1981; Deegan et al., 1997; Burton et al., 1999; Lyons et al., 2000); (2) developed for different taxa groups, such as nekton, benthic macroinvertebrates, or phytoplankton (Karr, 1981; Weisberg et al., 1997; Gómez et al., 2012); and (3) implemented throughout different global regions (Borja et al., 2000; Lyons et al., 2000; Raburu et al., 2009; Qadir and Malik, 2009). Previous studies have created IBIs for estuarine systems in the Gulf of Mexico, including the seminal benthic condition IBI that spans the Gulf of Mexico (Engle and Summers, 1999), a benthic index to detect freshwater inflow in a Texas estuary (Beseres Pollack et al., 2009), and a nekton-based IBI for tidal streams and bayous in the northern Gulf of Mexico (Guillen, 1996), but none have been developed for tidal streams in southern Texas.

The aim of this study is to create IBIs for nektonic and benthic macrofauna communities that discriminate between degraded and reference (least-impacted) tidal streams, using the southern Texas coast as a study area. Historical monitoring programs have attempted to use depressed dissolved oxygen levels as an indicator of Texas tidal stream impairment. However, studies show that other physicochemical variables are more influential in structuring tidal stream faunal communities than dissolved oxygen in Texas (Tolan and Nelson, 2009). This contradiction exemplifies the issue of utilizing a single metric as a proxy for predicting ecological degradation and suggests a multi-metric approach is warranted. This study combines historical methods and metrics (Karr, 1981; Weisberg et al., 1997; Van Dolah et al., 1999; Raburu et al., 2009; Margo, 2020) with metrics derived from data specific to the study region to develop the IBIs. Results from this study can be used in future research to refine the developed IBIs and test their effectiveness on a validation dataset. Implementation of a tidal stream IBI can enhance current management programs by identifying streams most in need of management attention, standardizing an assessment protocol for biological communities in tidal streams, and providing a tool to monitor community change over time.

2. Methods

2.1. Site Selection and Designation

The southern Texas coast extends from northern Matagorda Bay to the Rio Grande (river) on the USA-Mexico border (Fig. 1). Twenty-three candidate tidal streams were evaluated for inclusion in this study based on geomorphological and biogeographical criteria. Narrow drainage ditches and intermittent streams were first removed from consideration. A subset of 11 tidal streams were then selected from the remaining streams to encompass a broad spatial distribution across the southern Texas coast to minimize the effects of biogeography (**Table S1**). Sites within the 11 streams were then evaluated to identify sampling locations that (1) were accessible for collecting seine hauls and benthic cores, (2) had relatively shallow edge depths of < 1 m to allow sampling at consistent depths, and (3) were within the upper 30–50 % of the tidal stream segment to minimize the effects of salinity fluctuations on the nekton and macroinfauna communities being sampled.

Defining degraded and reference designations is a critical step in the creation of a multi-metric IBI because the biological metric selection process is based upon these *a priori* designations. Tidal streams were considered impaired for a variable if it exceeded threshold values for at least one land-use variable and at least one water quality variable (**Table S2**). Threshold values for impaired land use included > 20 % urban or cultivated land cover within 1000 m of the tidal stream segment, and a watershed population density > 50 km^{-2} (**Table S3**). Land cover within each 1000 m buffer region was calculated using the U. S. Geological Survey's 2016 National Land Cover Database (www.mrlc.gov), which provides land use/land cover data of the USA at a 30 m resolution (Yang et al., 2018) in ArcGIS v.10.6.1 (Environmental Systems Research Institute (ESRI), 2017). Population density was calculated by summing 2010 census block-level point data within each watershed (United States Census Bureau, 2010) and dividing by the watershed area (km^2).

Threshold values for impaired water quality were exceeded if the 85th percentile of measurements taken in the 11 tidal streams from 2011 to 2021 for ammonia (NH_3), nitrate + nitrite (NO_2 + NO_3), total

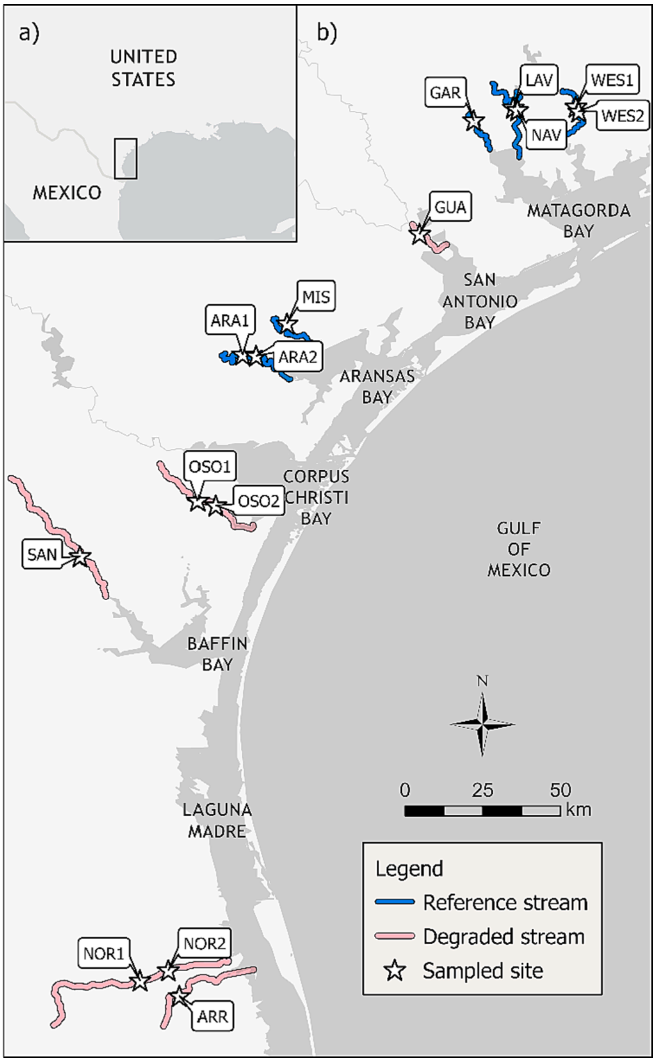

Fig. 1. Extent of study area, including a) the Gulf of Mexico region and b) tidal streams and sites sampled in this study. Tidal streams are colored based on degraded or reference classifications. Sampling sites are labeled by site code (see Table 1).

phosphorus (TP), and chlorophyll-a (Chl) was greater than the Texas Commission on Environmental Quality (TCEQ) defined screening limit (Table 1; **Table S2**; **Table S3**). The TCEQ identifies a concern for water quality if concentrations surpass the screening limit value greater than 20 % of the time (Texas Commission on Environmental Quality (TCEQ), 2020; **Table S3**). Dissolved oxygen concentration was not considered as a potential impairment indicator in this study because it does not appear to be a major structuring factor for biotic communities in Texas tidal streams (Tolan and Nelson, 2009). Sediment contamination was not used as a potential impairment variable due to lack of access to a publicly available dataset. However, increased sediment contamination levels are related to urban and agricultural land use in estuaries (United States Environmental Protection Agency, 1994; Dauer et al., 2000; Garner et al., 2009; Parker et al., 2023), which were variables considered in site classification.

The tidal segments of five streams were classified as having degraded conditions (Guadalupe River [GUA], Oso Creek [OSO], San Fernando Creek [SAN], North Floodway [NOR], Arroyo Colorado [ARR]), and the tidal segments of six streams were classified as having reference conditions (West Carancahua Creek [WES], Navidad River [NAV], Lavaca River [LAV], Garcitas Creek [GAR], Mission River [MIS], Aransas River [ARA]) (Table 1). Two sites (2——9 km apart) were selected in Oso Creek ([OSO1, OSO2], degraded), North Floodway ([NOR1, NOR2], degraded), Aransas River ([ARA1, ARA2], reference), and West Carancahua Creek ([WES1, WES2], reference) to investigate the impact of salinity on tidal stream faunal communities. One site was selected in each of the remaining seven streams.

2.2. Field Sampling

Field sampling was conducted at each site twice per year in 2020 and 2021, for a total of four sampling events. Sampling was standardized to occur before (spring) and during (summer) the time of typical minimum streamflows, maximum temperatures, and minimum dissolved oxygen concentrations (Texas Commission on Environmental Quality (TCEQ), 2012). Spring samples were collected each year between 18 March and 21 April, and summer samples were collected each year between 1 July and 27 August.

Nekton were collected during spring and summer sampling periods using three consecutive 10 m seine (4.6 m wide, 4.76 mm mesh) hauls conducted parallel to each stream bank (30 m length per stream bank) for a total of six seine hauls per site (276 m^2 total area). Individuals > 0.3 m in length were identified, photographed, and released. Smaller individuals were fixed in 10 % buffered formalin and sorted in the laboratory. Benthic macroinfauna were collected during the summer using six replicate benthic cores (6.7 cm diameter x 10 cm deep; 35.4 cm^2 area) along the shoreline in approximately 1 m water depth. All organisms were fixed with 10 % buffered formalin for processing in the laboratory. One additional benthic core was collected during the summer period for sediment grain size and organic matter analyses.

Water temperature, dissolved oxygen concentration, salinity, pH, turbidity, and water transparency were measured during each sampling event using a handheld YSI Pro DSS multiparameter water quality meter (YSI Incorporated, Yellow Springs, Ohio) and a secchi disc. Water samples were collected at a depth of 0.3 m deep during each sampling event and chilled on ice before being analyzed for nutrients and chlorophyll in the laboratory. Triplicate stream cross sections for measurements of stream width and depth were made each spring using either a measuring tape (depth and width), sonar (Garmin GT24UHD-TM Transducer for depth) or a laser rangefinder (width).

2.3. Laboratory Processing

Benthic macroinfauna were washed on a 500 μm sieve and sorted under a microscope. Nekton and benthic macroinfauna were identified to the lowest practical taxon (usually species) and counted. Benthic macroinfauna biomass was measured after being dried at 60 °C for ≥ 24 h.

Nutrient and chlorophyll samples were analyzed by EASTEX Environmental Laboratory or Ana-Lab Corporation for $NO_2 + NO_3$ (EPA 9056 or SM 4500, NO3 F), TP (EPA 365.3 or EPA 200.7), Chl (EPA 445.0 or EPA 446.0), NH_3 (EPA 350.1 or SM 4500; NH3 G), total Kjeldahl nitrogen (TKN; EPA 351.2), total organic carbon (TOC; EPA 9060 or SM 5310C), and total suspended solids (TSS; SM2540D) using standardized United States Environmental Protection Agency (USEPA) methods (United States Environmental Protection Agency, 2022). Relative quantities of sand (63–2000 μm), silt (4–63 μm), and clay (<4 μm) in the sediment were determined with a Beckman Coulter LS 13 320 Laser Particle Sizing Analyzer (Beckman Coulter Life Sciences, Indianapolis, IN) after digestion in ≤ 30 % hydrogen peroxide to remove any organic content. The proportion of sediment organic matter by weight was calculated using the loss on ignition method (after combustion at 450 °C for 4 h).

2.4. Data analysis

Physiochemical and biotic characteristics of the sampled streams were determined using multivariate analyses. Water quality and

Table 1
Variables considered to make degraded or reference classifications for tidal streams. The table includes the tidal stream name (arranged from north to south), codes for sites on each tidal stream, and station type (degraded or reference). Land use characteristics include population (from 2010) by watershed size, and percent urban and percent cultivated crops landcover based on 1000 m buffer regions around the tidal segment. Water quality characteristics include the 85th percentile concentrations of ammonia as nitrogen (NH_3), nitrate + nitrite ($NO_2 + NO_3$), total phosphorus (TP), and chlorophyll-a (Chl) derived from this study and 2011–2021 historical TCEQ data (Texas Commission on Environmental Quality, 2022). The number of observations from each sampling source can be found in **Table S2**. Bolded values indicate that the threshold value was exceeded for the tidal segment (**Table S3**).

Tidal Stream	Sites on Tidal Stream	Station Type	Land Use			Water Quality (85th percentile)			
			Population Density (km^{-2})	Urban (%)	Cultivated Crops (%)	NH_3 (mg L^{-1})	$NO_2 + NO_3$ (mg L^{-1})	TP (mg L^{-1})	Chl (ppb)
West Carancahua Creek	WES1, WES2	Reference	2.0	1.4	14.4	0.27	**2.15**	0.53	**82.95**
Navidad River	NAV	Reference	5.2	1.3	0.6	0.16	0.69	0.21	14.14
Lavaca River	LAV	Reference	7.9	1.8	3.5	0.15	0.82	0.15	16.37
Garcitas Creek	GAR	Reference	5.3	1.6	8.4	0.21	0.13	0.27	20.77
Guadalupe River	GUA	Degraded	**91.1**	3.0	1.1	0.10	**4.27**	0.49	14.00
Mission River	MIS	Reference	3.6	2.0	6.7	0.05	0.08	0.27	**36.13**
Aransas River	ARA1, ARA2	Reference	21.3	1.9	10.3	0.06	0.69	**1.08**	**36.91**
Oso Creek	OSO1, OSO2	Degraded	**241.6**	18.7	**55.9**	**0.51**	**14.01**	**2.93**	**80.36**
San Fernando Creek	SAN	Degraded	20.6	5.9	**35.2**	0.14	**6.28**	**3.02**	**46.55**
North Floodway	NOR1, NOR2	Degraded	**401.0**	6.0	**73.2**	0.21	**4.45**	0.46	**69.90**
Arroyo Colorado	ARR	Degraded	**217.1**	8.5	**37.1**	0.40	**4.75**	**0.73**	**73.46**

sediment data were analyzed separately using Principal Components Analysis (PCA) with no rotation of axes. Water quality variables were examined for approximate normality and transformed when skewed (square root-transformed: temperature, $\log_e$-transformed: salinity, secchi, Chl, NH_3, $NO_2 + NO_3$, TKN, TP, TSS) before standardization (Clarke and Gorley, 2015). Non-metric Multi-Dimensional Scaling analyses (nMDS; Clarke and Warwick, 1994) using Bray-Curtis similarity matrices were used to characterize nekton and benthic macroinfauna community composition among sites and sampling dates. Nekton data were summed over six replicate seine hauls per sampling event, then dispersion weighted and square root-transformed prior to multivariate analyses. Benthic macroinfauna data were dispersion weighted and fourth root transformed (Clarke et al., 2006). Species richness (S), J' species evenness, Shannon's H' diversity, Simpson's 1- λ diversity, and Hill (N1) species diversity were calculated using nekton and benthic macroinfaunal abundance data individually. All multivariate analyses and calculations of diversity indices were conducted using PRIMER v7.0.13 (Clarke and Gorley, 2015).

2.5. Metric Development

Developing our nektonic and benthic IBIs involved identifying univariate metrics that discriminate between degraded and reference sites in the southern Texas coast from historic studies, and statistical analyses of our nektonic and benthic community data (Fig. 2). Twenty-seven potential nektonic metrics (Table 2) and 19 potential benthic metrics (Table 3) were derived from seminal IBI studies (Karr, 1981; Weisberg et al., 1997), and IBI studies of Texas tidal streams (L. Broach [TCEQ] pers. comm.; Margo, 2020). These previously identified metrics include measures of taxonomic diversity and abundance, trophic groups, stress tolerance, and endemicity. The ability of metrics to discriminate between degraded and reference site conditions was evaluated using a Mann-Whitney *U* test ($\alpha = 0.05$; Weisberg et al., 1997; Van Dolah et al., 1999) using R 4.1.2 software (RStudio Team, 2018).

New metrics using our collected nektonic and benthic community data were determined using the similarity percentages (SIMPER) procedure (Clarke, 1993). Nekton and benthic macroinfauna taxa, families, and functional groups with a dissimilarity/standard deviation (Diss./SD) of ≥ 1 between degraded and reference sites were selected as potential IBI metrics. Functional group categorization of fishes was assigned by categorizing species based on estuarine habitat utilization as described in Elliott et al. (2007) (**Table S4**), and then subdividing these groups by feeding guild as described in Gonzalez et al. (2020) (**Table S5**). Benthic macroinfauna were assigned to feeding functional groups based on previous research (**Table S6**). Potential metrics from SIMPER analyses were removed from consideration for the IBI if: (1) the 10th and 50th percentiles of the reference data were both 0 or, (2) the directionality of the data distribution was opposite of that in published literature. Total abundance and biomass measurements of benthic macroinfauna were analyzed with and without Ostracoda due to the high proportion of the taxa in benthic samples, particularly at degraded sites. Metrics identified in SIMPER analyses were also removed if there was taxonomic redundancy, e.g., the "collector gatherers-scavenger" functional group was removed because it contained only Ostracoda. Linear (Pearson's) correlations were calculated among the selected metrics to identify potential sources of collinearity among metrics. Metrics were not weighted because this would introduce unknown bias into the IBI.

2.6. Index Creation

An IBI combines multiple ecological metrics into a single IBI score (Morehead et al., 2008). IBI scores in this study were assigned using percentile values derived from reference condition data, using methodology adapted from Van Dolah et al. (1999). Each metric was assigned a score of 5, 3, or 1, with scores decreasing as the metric deviated further from reference site conditions. For metrics elevated in reference site conditions (disturbance sensitive metrics, e.g., total abundance), samples were scored using the 10th and 50th percentile values (5 if > 50th percentile, 3 if between the 10th and 50th percentiles, 1 if < 10th percentile; **Table S7**). For metrics depressed in reference site conditions (disturbance indicator metrics, e.g., % Oligochaeta), samples were scored using the 50th and 90th percentile values (5 if < 50th percentile, 3 if between the 50th and 90th percentiles, 1 if > 90th percentile; **Table S7**).

Index scores were calculated for each sampling event by summing the scores for each metric in the nekton or benthic macroinfauna IBI then dividing by the total number of metrics. This scaling results in a standardized minimum (1) and maximum (5) site score for each IBI. The range of index scores from sampling events was categorized into discrete "integrity classes," which are more easily interpretable classifications that range from very poor to excellent (Raburu et al., 2009). Percentile values from the reference and degraded condition data were used separately for defining integrity classes to account for differences in the

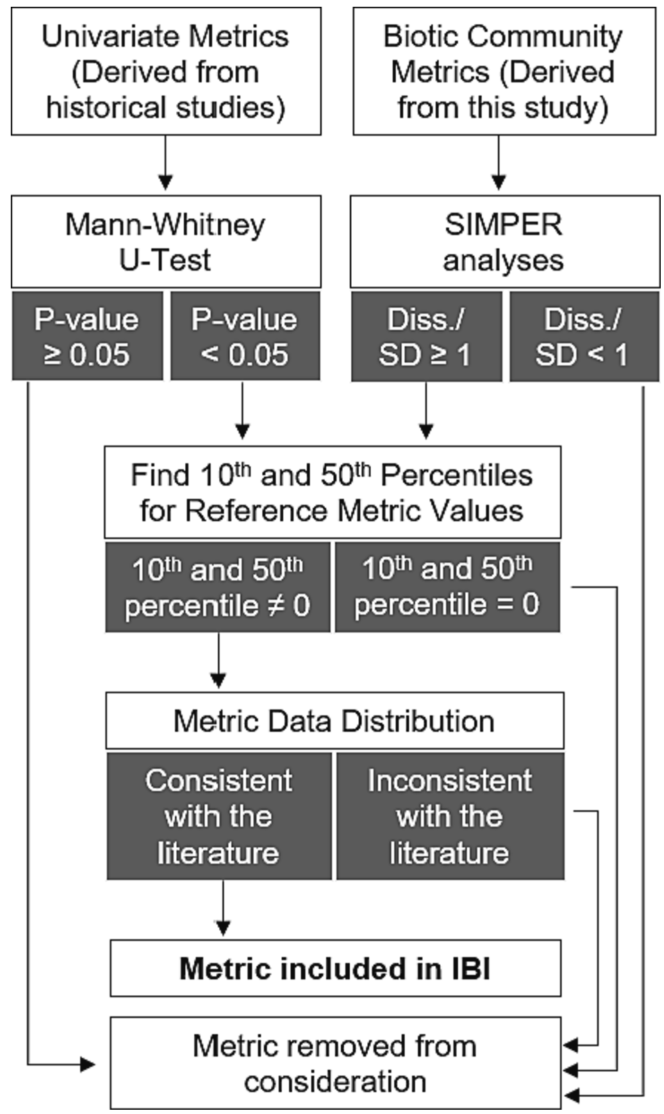

Fig. 2. Flowchart describing methodology used to evaluate candidate metrics for inclusion in the nekton and benthic macroinvertebrate IBIs. Diss./SD = dissimilarity divided by standard deviation.

range of IBI scores between reference and disturbed samples. Threshold values for each integrity class were derived using the 75th (excellent), 50th (good), and 25th (moderate) percentiles of reference sample index scores and the 50th percentile from degraded sample index scores (poor if > degraded 50th percentile, very poor if < degraded 50th percentile).

Linear correlations were calculated among mean nekton scores, mean benthic IBI scores, mean stream width, mean thalweg depth, and site latitude. Differences in nekton and benthic IBI scores between years and nekton IBI scores between seasons were evaluated using a Mann-Whitney *U* test (α = 0.05).

3. Results

3.1. Environmental Variables

The first and second principal components (PC1 and PC2) of the water quality PCA explained 29.7 % and 20.3 % of the variation within the water quality data set (Fig. 3; Table S8, Table S9). Degraded sites had higher PC1 scores than most reference sites, which corresponded to higher concentrations of nitrate + nitrite, total Kjeldahl nitrogen, and total suspended solids, and lower water transparency (secchi disk

Table 2
Metrics examined to evaluate differences in nekton communities between degraded and reference tidal streams. Metric values are per 6 seine hauls (276 m^2 area) unless otherwise indicated. P-values for each metric were obtained using a Mann-Whitney U-Test comparing degraded and reference data from this study. A single asterisk (*) indicates p < 0.05. A double asterisk (**) indicates p < 0.05 and inclusion in the IBI. % = percentage of total abundance per 6 seine hauls.

Nekton Metric	P-value	Mean ± SD Values		Source
		Reference	Degraded	
Taxa richness (S)	0.348	11.88 ± 2.86	11.25 ± 3.89	Karr, 1981; TCEQ 2014; L. Broach, TCEQ (pers. comm.)
Shannon's (log e-transformed) diversity (H')	< 0.001*	1.04 ± 0.50	1.51 ± 0.45	Margo, 2020
Hill's diversity (N1)	< 0.001*	3.20 ± 1.68	4.94 ± 2.04	Margo, 2020
Pielou's evenness (J')	< 0.001*	0.43 ± 0.21	0.64 ± 0.17	Margo, 2020
Gini-Simpson diversity (1-λ)	0.001*	0.46 ± 0.23	0.65 ± 0.17	Margo, 2020
Total abundance	< 0.001**	880.72 ± 1196.16	186.57 ± 253.18	Karr, 1981; TCEQ 2014; L. Broach, TCEQ (pers. comm.)
Number of intolerant taxa	0.011*	0.38 ± 0.55	0.07 ± 0.26	Karr, 1981; TCEQ 2014
Number of fish taxa	0.643	8.81 ± 2.16	9.43 ± 3.21	TCEQ 2014
Number of invertebrate taxa	0.002**	3.00 ± 1.37	1.79 ± 1.64	L. Broach, TCEQ (pers. comm.)
Number of native Cyprinidae taxa	0.019*	0.16 ± 0.37	0.46 ± 0.58	TCEQ 2014
Number of Centrarchidae taxa	0.717	0.69 ± 1.09	0.71 ± 0.94	TCEQ 2014
Number of Centrarchidae taxa (sans *Lepomis cyanellus*)	0.938	0.63 ± 0.94	0.61 ± 0.92	Karr, 1981
Tolerant (%)	0.002*	7.27 ± 10.76	20.54 ± 21.23	L. Broach, TCEQ (pers. comm.)
Tolerant (sans *Gambusia* sp.) (%)	0.002*	4.96 ± 12.35	12.53 ± 21.52	TCEQ 2014
Introduced (%)	0.111	0.01 ± 0.07	0.34 ± 0.99	TCEQ 2014
Omnivores (%)	0.378	64.02 ± 37.25	62.18 ± 26.97	Karr, 1981; TCEQ 2014
Invertivores (%)	0.519	34.06 ± 36.06	34.82 ± 27.68	TCEQ 2014
Piscivores (%)	0.039*	1.15 ± 3.59	2.33 ± 3.82	Karr, 1981
Invertebrates (%)	0.411	24.19 ± 0.30	19.97 ± 0.25	L. Broach, TCEQ (pers. comm.)
Invertivorous Cyprinidae (%)	0.019*	0.03 ± 0.15	0.62 ± 1.23	Karr, 1981
Flatfish (%)	0.254	0.23 ± 0.61	0.52 ± 2.46	L. Broach, TCEQ (pers. comm.)
Gobies (%)	0.299	0.11 ± 0.48	0.88 ± 2.99	L. Broach, TCEQ (pers. comm.)
Centrarchidae (%)	0.645	2.13 ± 5.08	1.50 ± 3.13	Margo, 2020
Lepomis cyanellus (%)	0.547	0.19 ± 0.90	0.17 ± 0.51	Karr, 1981
Abundance tolerant	0.44	29.53 ± 81.54	27.25 ± 34.94	Margo, 2020
Abundance of introduced taxa	0.122	0.03 ± 0.18	0.21 ± 0.63	L. Broach, TCEQ (pers. comm.)

(*continued on next page*)

Table 2 (*continued*)

Nekton Metric	P-value	Mean ± SD Values		Source
		Reference	Degraded	
Abundance Centrarchidae	0.96	3.03 ± 6.88	2.68 ± 8.66	Margo, 2020

Table 3
Metrics examined to evaluate differences in benthic macroinfauna communities between degraded and reference tidal streams. Metric values are per core replicate (35.4 cm^2 area) unless otherwise indicated. P-values for each metric were obtained using a Mann-Whitney U-Test comparing degraded and reference data from this study. A single asterisk (*) indicates $p < 0.05$. A double asterisk (**) indicates $p < 0.05$ and inclusion in the IBI. % = percentage of total abundance per core.

Metric	P-value	Metric Mean ± SD		Source
		Reference	Degraded	
Taxa richness (S)	0.101	3.51 ± 1.58	3.27 ± 2.08	L. Broach, TCEQ (pers. comm.)
Shannon's (log e-transformed) diversity (H')	0.006**	0.80 ± 0.39	0.62 ± 0.84	Weisberg et al., 1997; L. Broach, TCEQ (pers. comm.)
Hill's diversity (N1)	0.006*	2.38 ± 0.84	2.10 ± 1.11	Margo, 2020
Pielou's evenness (J')	0.016*	0.63 ± 0.26	0.50 ± 0.34	Margo, 2020
Gini-Simpson diversity (1-λ)	0.008*	0.44 ± 0.21	0.34 ± 0.26	Margo, 2020
Total abundance (n m^{-2})	0.010*	6919.63 ± 5712.40	18352.18 ± 23842.20	Weisberg et al., 1997
Total abundance sans Ostracoda (n m^{-2})	< 0.001**	6588.72 ± 5400.90	4936.69 ± 7938.00	This study
Total biomass (g/ m^{-2}(−|-))	0.005*	2.19 ± 6.72	2.16 ± 3.11	Weisberg et al., 1997
Total biomass sans Ostracoda (g/ m^{-2}(−|-))	< 0.001**	2.15 ± 6.73	0.52 ± 1.09	This study
Ostracoda (g/ m^{-2}(−|-))	< 0.001*	0.04 ± 0.12	1.64 ± 2.89	This study
Number tolerant taxa	0.025	1.60 ± 0.61	1.35 ± 0.78	Margo, 2020
Number intolerant taxa	0.003*	0.19 ± 0.42	0.04 ± 0.19	Margo, 2020
Tolerant (%)	< 0.001*	68.15 ± 30.42	37.86 ± 34.76	Weisberg et al., 1997; L. Broach, TCEQ (pers. comm.)
Intolerant (%)	0.004*	1.07 ± 2.80	0.66 ± 3.87	Weisberg et al., 1997; L. Broach, TCEQ (pers. comm.)
Crustacea (%)	< 0.001*	4.73 ± 10.52	53.22 ± 38.70	L. Broach, TCEQ (pers. comm.)
Crustacea sans Ostracoda (%)	0.045*	1.86 ± 5.67	0.52 ± 3.67	This study
Mollusca (%)	0.987	9.79 ± 24.82	4.39 ± 14.32	L. Broach, TCEQ (pers. comm.)
Insecta sans Chironomidae (%)	0.013*	2.48 ± 10.47	0.55 ± 1.92	Margo, 2020
Insecta sans Chironomidae (n m^{-2})	0.028*	115.23 ± 223.82	145.20 ± 729.47	Margo, 2020

measurements) than reference sites. Summer sampling events usually had higher PC2 scores than spring sampling events, which corresponds to higher temperatures and TOC concentrations, but lower salinities than spring sampling events. The lower relative summertime salinities occurring in this study were contrary to the anticipated higher salinities, which are expected during typical summers.

The first and second principal components (PC1 and PC2) of the sediment quality PCA explained 93.5 % and 5.2 % of the variation within the sediment quality data set (**Fig. S1**; **Table S10**). Site-year combinations with high PC1 scores had greater organic, clay, and silt content but lower sand content. There was no discernable trend in grain size (along PC1) between degraded and reference site-year combinations. The proportion of clay was negatively related to percent organic material along PC2, but there was minimal separation of samples along this axis.

Stream width varied from 4 m (OSO1) to 118 m (NAVI; **Table S11**; Fig. 1). Maximum depths ranged from 0.3 m (OSO1) to 6.2 m (GUA). Both mean stream width and maximum depth were greater (both $p < 0.001$) in reference sites (52.0 m wide, 3.0 m deep) than degraded sites (28.4 m wide, 2.2 m deep; **Table S11**).

3.2. Nekton

A total of 33,452 nektonic individuals from 67 taxa were collected within the two-year study (**Table S12**). Degraded sites had a mean of 186 individuals and 11 taxa per six seine hauls (276 m^2 area), while reference sites had a mean of 882 individuals and 12 taxa.

Non-metric MDS analysis identified a difference in nekton community composition between reference and degraded sites, with no discernable trend related to sampling period (stress = 0.26; Fig. S2a). A similar differentiation between reference and degraded sites was observed whether the nekton nMDS plot was plotted in a higher stress (0.26) two-dimensions (2D) or a lower stress (0.19) three dimensions (3D), meaning that the 2D nMDS represents major trends in actual nekton community composition well. *Anchoa mitchilli*, *Menidia beryllina*, *Palaemonetes pugio*, *Gambusia affinis*, and *Callinectes sapidus* were more abundant in reference sites than degraded sites, while *Poecilia formosa* was more abundant in degraded sites (Diss./SD ≥ 1 from SIMPER analysis; **Table S13**). SIMPER results for nekton abundance at the family level yielded 11 metrics with Diss./SD ≥ 1 (**Table S14**). Of the nekton families, Sciaenidae and Clupeidae were more abundant in reference sites, and Poecilidae, Fundulidae, Mugilidae, Characidae, and Cyprinodontidae were more abundant in degraded sites (Diss./SD ≥ 1; **Table S14**). Four nekton families (Engraulidae, Palaemonidae, Atherinopside, and Portunidae) were taxonomically redundant as a potential IBI metric because only one species was represented in each family and were therefore excluded from metric selection. The estuarine carnivores (EC) and freshwater migrant carnivores (FMC) functional groups were more abundant in reference sites, while freshwater migrant omnivores (FMO) and freshwater straggler carnivores (FSC) were more abundant in degraded sites (Diss./SD ≥ 1; **Table S5**; **Table S15**). Estuarine carnivore and FMC abundances were considered redundant because these functional groups were dominated by individual species that were already identified as metrics (EC was dominated by *A. mitchilli*, FMC was dominated by *G. affinis* and *M. beryllina*). After excluding redundant metrics, additional metrics (*P. formosa,* Sciaenidae, Poecilidae, Fundulidae, Mugilidae, Characidae, and FMO) were removed from consideration for the IBI since the 10th and 50th percentiles of the reference data were both 0 (**Table S16**). The relative abundances of remaining selected taxa and functional groups were moved forward in the metric selection process.

3.3. Benthic Macroinfauna

A total of 7,777 benthic individuals from 40 taxa were collected within the 2-year study (**Table S17**). Degraded sites had a mean of 65 individuals and 3 taxa per core (35.4 cm^2 area), while reference sites had a mean of 24 individuals and 4 taxa.

The benthic community composition differed between reference and degraded sites, as identified by nMDS ordination (stress = 0.19;

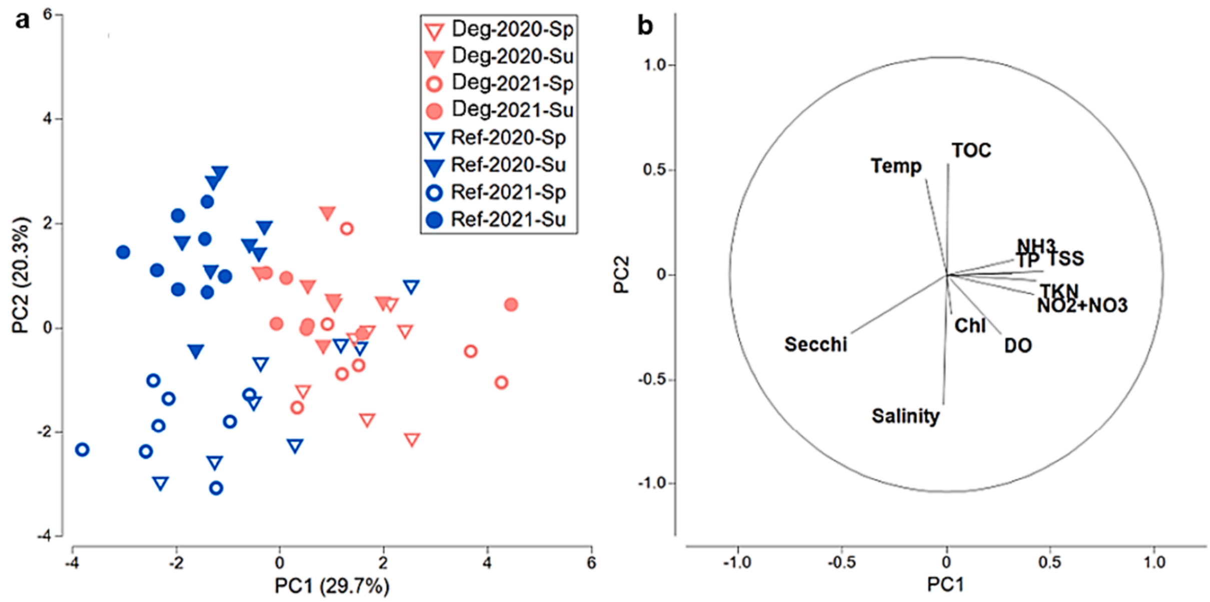

Fig. 3. Principal component analysis of water quality variables with a) sampling event scores and b) vector plot of water quality variables. Temp = temperature, NH3 = ammonia as nitrogen, TP = total phosphorus, TSS = total suspended solids, TKN = total Kjeldahl nitrogen, NO2 + NO3 = nitrate plus nitrite, DO = dissolved oxygen, Chl = chlorophyll-a. Ref = reference, Deg = degraded. Sp = spring, Su = summer.

Fig. S2b). Chironomidae were more abundant in reference sites, and Ostracoda and Oligochaeta were more abundant in degraded sites (Diss./SD ≥ 1; **Table S18**). The collector-gatherer (CG) functional group was more abundant at degraded sites (Diss./SD ≥ 1; **Table S6**, **Table S19**). The collector-gatherer/scavenger (CG-SCAV) and predator/collector-gatherer/filtering-collector (P-CG-FC) functional groups had different abundances between degraded and reference sites but were taxonomically redundant with taxonomic group abundance because only one taxon was represented in each group (Ostracoda and Chironomidae, respectively). After excluding redundant metrics (**Table S20**), relative abundances of selected taxa and functional groups were moved forward in the metric selection process.

3.4. Metric Selection

Nine metrics were selected for inclusion in the nekton IBI and six metrics were selected for inclusion in the benthic macroinfauna IBI. Two previously identified nekton IBI metrics (total abundance, and number of invertebrate taxa) and three previously identified benthic metrics (Shannon's H' diversity [$\log_e$-transformed], total abundance sans Ostracoda, and total biomass sans Ostracoda) successfully discriminated between degraded and reference sites in the current study ($p < 0.05$ in Mann-Whitney U-tests; Table 4, Table 5). Ostracoda was excluded in the total abundance and biomass metrics, as it rendered results more consistent with historical bioassessment studies which show a positive correlation between total abundance and biomass with site condition. Seven newly identified nekton IBI metrics (percent *A. mitchilli,* percent *M. beryllina,* percent *P. pugio*, percent *G. affinis,* percent *C. sapidus,* percent Clupeidae, and percent FSC) and three newly identified benthic metrics (percent Chironomidae, percent Oligochaeta, and percent CG) successfully discriminated between degraded and reference sites in the current study (Diss./SD ≥ 1 in SIMPER analyses). The highest Pearson correlation (R^2) among nekton metric pairs was 0.71 between percent Clupeidae and total abundance (**Table S21**), and the highest correlation (R^2) among benthic metric pairs was 0.75 between percent Oligochaeta and percent CG (**Table S22**).

3.5. Index Creation

IBI scores were derived for each sampling event by summing the metric scores for each of the nekton and benthic metrics then dividing by the total number of metrics per IBI. Integrity classes for the nekton IBI were assigned based on 25th, 50th and 75th percentile values from the degraded and reference score data (excellent: > 3.7, good: > 3.5 – 3.7, moderate: > 3.1 – 3.5, poor: > 2.3 – 3.1, very poor: ≤ 2.3; **Table S23**). Mean nekton IBI scores for each site ranged from 1.6 at Oso Creek 1

Table 4
Nekton index of biotic integrity (IBI) and corresponding threshold values for metric scores. Scores correspond to percentile values extracted from reference stream data (**Table S6**). Metric values are per 6 seine hauls (276 m^2 area) unless otherwise indicated. FSC = freshwater straggler carnivore. % = percentage of total abundance per 6 seine hauls.

Nekton Metrics	Score		
	5	3	1
Total abundance	> 351	97—351	< 97
Number of invertebrate taxa	> 3	> 0 – 3	0
A. mitchilli (%)	> 8.92	0.40 – 8.92	< 0.40
M. beryllina (%)	< 0.96	0.96 – 9.79	> 9.79
P. pugio (%)	> 2.98	> 0 – 2.98	0
G. affinis (%)	< 0.59	0.59 – 7.70	> 7.70
C. sapidus (%)	< 0.15	0.15 – 1.80	> 1.80
Clupeidae (%)	> 3.30	> 0 – 3.30	0
FSC (%)	< 0.23	0.23 – 3.08	> 3.08

Table 5
Benthic macroinfauna index of biotic integrity (IBI) and corresponding threshold values for metric scores. Metric values are per core replicate (35.4 cm^2 area) unless otherwise indicated. Scores correspond to percentile values extracted from reference stream data (**Table S6**). % = percentage of total abundance per core.

Benthic Macroinfauna Metrics	Score		
	5	3	1
Shannon's (log e-transformed) diversity (H')	> 0.90	0.20 – 0.90	< 0.20
Total abundance sans Ostracoda (n m^{-2})	> 5,815	1,135 – 5,673	< 1,135
Total biomass sans Ostracoda (g/m^{-2}(−|-))	> 0.27	0.06 – 0.27	< 0.06
Chironomidae (%)	> 49.14	7.52 – 49.14	< 7.52
Oligochaeta (%)	< 7.42	7.42 – 53.57	> 53.57
Collector-gatherers (%)	> 26.09	> 0 – 26.09	0

(OSO1) to 3.8 at Mission River (MIS) out of a maximum score of 5 and minimum score of 1 (**Table S23**). The mean metric score for reference sites was 3.7, and for degraded sites was 2.4 (Fig. S3a). All reference sites except Garcitas Creek (GAR) had greater mean nekton IBI scores than the mean scores of the seven degraded sites (Fig. 4a; **Table S23**).

Integrity classes for the benthic IBI were assigned based on 25th, 50th and 75th percentile values from the degraded and reference score data (excellent: > 4.0, good: > 3.7 – 4.0, poor/moderate: > 2.7 – 3.7, very poor: ≤ 2.7; **Table S24**). Mean benthic IBI scores for each site ranged from 1.9 at North Floodway 1 (NOR1) to 4.2 at Aransas River 2 (ARA2) out of a maximum score of 5 and minimum score of 1 (**Table S24**). The mean metric score for reference sites was 3.7, and for degraded sites was 2.9 (Fig. S3b). Half of the reference sites, Aransas River 2 (ARA2), West Carancahua 2 (WES2), Navidad River (NAV), Aransas River 1 (ARA1), had greater mean benthic IBI scores than the mean scores of the seven degraded sites (Fig. 4b; **Table S24**).

Mean nekton IBI scores were positively correlated with site latitude, mean stream width, and mean thalweg depth (all $p < 0.01$) (**Fig. S4**). Mean benthic IBI scores were positively correlated with site latitude ($p < 0.01$) but not mean stream width ($p = 0.17$), or mean thalweg depth ($p = 0.14$) (**Fig. S5**). IBI scores did not generally differ between year (nekton IBI: $p = 0.77$, benthic IBI: $p = 0.86$) or spring vs. summer sampling periods (nekton IBI: $p = 0.07$). Nekton IBI site scores were more stable between years (mean difference = 0.2) than between sampling seasons(mean difference = 0.4). There was also a decrease in nekton IBI score from the spring to summer sampling periods at most (11 out of 15) sites (**Table S25**). Differences in site score between years was greater in the benthic IBI (mean difference = 0.5) than the nekton IBI (**Table S24**).

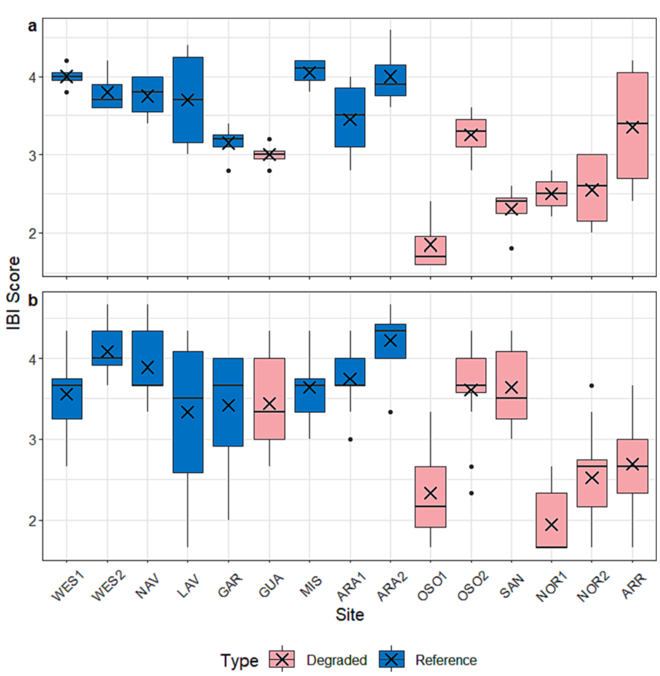

Fig. 4. Box plot of a) nekton and b) benthic macroinfauna index of biotic integrity scores for each tidal stream. Sites along the x-axis are arranged from northernmost (left) to southernmost (right). The horizontal line in each box represents median, while "X" represents mean. The lower bars represent the minimum values above (1.5*Inter-quartile range - lower quartile). The upper bars represent the maximum values below (1.5*Inter-quartile range + upper quartile). Site codes along x-axis correspond to those listed in Table 1.

4. Discussion

4.1. Metric Selection and Interpretation

The majority of metrics that were significantly different between degraded and reference conditions were developed from data collected in this study, rather than from metrics derived from historical studies. Only 2 (abundance, number of invertebrate taxa) of the 27 historical nekton metrics and 3 (Shannon H' [$\log_e$] diversity, total abundance sans Ostracoda, total biomass sans Ostracoda) of the 19 historical benthic macroinfauna metrics were included in the IBIs developed for this study. Tidal stream systems, as is typical with other parts of estuaries, have natural environmental fluctuations that drive the dominance of ubiquitous euryhaline taxa (Hackney et al., 1976; Tolan and Nelson, 2009; Barendregt and Swarth, 2013; Tolan and Nelson, 2013) and often exist at the edge of coastal development, which makes true baseline conditions difficult to identify (Bergquist et al., 2011; Barendregt and Swarth, 2013). Therefore, metrics common to freshwater IBIs are often not as successful in estuarine systems (Deegan et al., 1997; Weisberg et al., 1997; Engle and Summers, 1999). Modifications of the traditional freshwater IBI approach to improve use in estuarine systems have included: (1) developing metrics or scoring criteria specific to estuarine zones (Weisberg et al., 1997; Van Dolah et al., 1999; Breine et al., 2010), (2) including new metrics that describe taxa by estuarine use categorizations (nursery species, estuarine spawners, etc.; Deegan et al., 1997, Llansó et al., 2002), or (3) adding metrics across multiple taxa groups to address differences related to space or time (Gómez er al. 2012). This study utilized the latter two approaches to incorporate additional metrics specific to conditions in southern Texas tidal streams, resulting in indices that were somewhat successful in differentiating between degraded and reference sites. Benthic and nekton taxa that were most dissimilar between degraded and reference sites were geographically wide-ranging estuarine taxa that are adapted to variable environmental conditions. Selected nekton metrics included three fish species (*A. mitchilli, M. beryllina*, and *G. affinis*), one fish family (Clupeidae), and two nektonic macroinvertebrates (*P. pugio* and *C. sapidus*), the majority of which are estuarine taxa that commonly utilize tidal streams on the US East Coast and Gulf of Mexico (Hackney et al., 1976; Ogburn-Matthews and Allen, 1993; Tolan and Nelson, 2009; Tolan and Nelson, 2013). Two benthic metrics selected at the lowest identified taxonomic level were Oligochaeta and Chironomidae, each considered a cosmopolitan taxon for their ability to tolerate wide environmental shifts (Porinchu and MacDonald, 2003; Rodriguez and Reynoldson, 2011). Greater abundances of Oligochaeta in degraded conditions corroborate previous studies which state that Oligochaetes can be found in degraded conditions, such as in areas of high organic matter or hypoxia (Rodriguez and Reynoldson, 2011).

The association of Chironomidae with reference sites exemplifies some of the difficulty in interpretation of metrics because individual Chironomidae species can have differing ranges of tolerances (Molineri et al. 2020). It is possible that Chironomidae taxa collected in this study represent less tolerant species. However, further taxonomic specificity in future studies could allow for better interpretation of this metric. In the benthic macroinfauna samples, Ostracoda often occurred in high abundances relative to other taxa and were the dominant taxon at many of the degraded sites. Therefore, the historical metrics, total abundance, and total biomass were assessed with and without Ostracoda. When Ostracoda were excluded, total abundance and biomass were greater in reference conditions; the opposite was true when Ostracoda were included. While other Ostracoda measures were assessed for inclusion as a metric in the IBI, the taxon was found in such low abundances at reference sites that scoring thresholds could not be defined because the 10th and 50th percentiles of the reference data were both 0. Despite this, the association of Ostracoda with degraded sites suggests the taxon may be a useful bioindicator for degraded conditions in future tidal stream studies. Ostracoda have been utilized as bioindicators in different aquatic systems, with certain species dominating the Ostracoda community in the presence of pollution (Alvarez Zarikian et al., 2000; Ruiz et al., 2005; Tan et al., 2021).

Functional categorizations allowed for inclusion of sensitive taxa in the benthic IBI, but the included nekton functional group metric may possibly be indicative of biogeographical changes along the southern Texas coast. The collector-gatherer (CG) benthic group was more abundant in reference conditions and consisted of some of the more sensitive benthic taxa, particularly polychaetes (Rakocinski et al., 1997; Dean, 2008; Díaz-Castañeda and Reish, 2009), the mayfly genus *Hexagenia* (Howland et al., 2019), and the gastropod *T. sphinctostoma* (Rakocinski et al., 1997). Without grouping taxa into functional categorizations, these taxa would not have been represented in the IBI due to low individual abundances but, together, contribute an important metric that allows for sensitive taxa to be represented within the IBI. The freshwater straggler carnivore (FSC) functional group was included in our nekton IBI and was more abundant in degraded conditions. Gonzalez et al. (2020) found that freshwater functional groups were more sensitive to freshwater inflow volumes than estuarine or marine functional groups, which suggests that they could serve as bioindicators for environmental variation associated with inflow changes. Despite this association, FSC was numerically dominated (~75 %) by *Astyanax mexicanus* and *Herichthys cyanoguttatus*, both species endemic to waters south of Corpus Christi Bay, an area of the coast where only degraded sites are found. These species are introduced to and occur in much lesser numbers in watersheds north of Corpus Christi Bay (Hubbs et al., 1978), where our reference sites are concentrated. Therefore, it is unclear if increased abundances of FSC indicates degraded conditions or geographic location of the degraded sites. Grouping nekton and benthic macroinfauna taxa into functional categorizations allowed for more holistic community information to be brought into the IBIs by grouping taxa with similar life histories or responses to environmental change and by incorporating more sensitive or rare taxa representation.

By creating IBIs based on both nekton and benthic macroinfaunal communities, a more holistic picture of aquatic life use across different spatiotemporal scales is produced. The relatively sessile nature of benthic macroinfauna makes them particularly useful indicators of conditions in their immediate environment, especially changes in sediment contaminants and hypoxia (Pearson and Rosenberg, 1978; Weisberg et al., 1997; Pinto et al., 2009). Conversely, nekton are motile and represent conditions at a larger spatial scale, such as the entire tidal stream system (Whitfield and Elliott, 2002). Benthic macroinfauna have shorter life histories compared to nekton, especially in stressed conditions (Odum, 1985; Borja et al., 2012). Taxa of both short and long life histories are useful as bioindicators because they can reflect changes with time and bioaccumulation or the rapid response by a population to change, respectively (Resh, 2008). Despite the high natural and anthropogenic variability in tidal stream systems (Dauvin and Ruellet, 2009; Tweedley et al., 2015), using the nekton and benthic macroinfauna IBIs in tandem may help tease apart some of the natural variation across space and time better than using an IBI for one of these groups alone.

Baseline conditions along the southern Texas coast were difficult to identify because all study sites were impacted by anthropogenic land use in some way, existing along a continuum of degradation rather than in clear degraded and reference conditions. Land alongside tidal streams is often considered “prime real estate” for residential, industrial, tourism, or recreational-related development (Mallin and Lewitus, 2004). Identifying indicator taxa in tidal streams is difficult because the inherent environmental variation in these systems leads to dominance of ubiquitous estuarine taxa that can handle wide environmental shifts. Lacking clear reference conditions further complicates this process because sensitive taxa decrease substantially even in moderate levels of disturbance (Raburu et al., 2009; Lubanga et al., 2021). While seminal IBI studies define reference conditions based on water bodies that have been minimally affected by anthropogenic degradation (Karr et al., 1986), unimpacted baseline conditions may be impossible to define in most tidal stream studies. Limited or absent long-term monitoring data, shifting theoretical baselines, and the lack of standardized methodology in defining reference conditions in estuaries also convolutes how baseline conditions are chosen (Thrush et al., 2013). Using historical baselines may also not be appropriate to establish reference conditions in systems that are not considered “pristine.” Rather, using contemporary community data from least impaired sites establishes more realistic reference, or best attainable, conditions and can minimize potential issues associated with shifting baselines (Stoddard et al., 2006). Future tidal stream studies should consider the limitations associated with their method of defining reference conditions, whether baselines are derived from current least-impacted conditions or models based on long-term historical data (Stoddard et al., 2006).

4.2. IBI Applications and Future Refinement

Upon future refinements and testing on an independent dataset, the IBIs developed in this study can likely be incorporated as a standardized bioassessment tool to identify streams most in need of management attention, and monitor faunal community changes, including those in response to management actions. The process of implementation could include summing metric scores for a tidal stream, identifying sites that are lowest scoring (falling into the “poor” or “very poor” integrity classes), investigating sources of degradation for candidate sites, and developing management strategies based on agency priorities and capacity. Measuring the success of management actions, such as the establishment of total maximum daily loads (TMDLs), bank stabilization, or in-stream habitat restoration (Castillo et al., 2016), can likely be evaluated with continued biomonitoring using the sampling methods outlined in this study to determine any increases to the IBI score. By using the refined and tested IBIs, managers can determine if biotic communities in degraded systems are becoming more similar to reference conditions after management actions have been implemented (Karr, 1991). Overall, incorporating successful IBIs into tidal stream management can complement existing physicochemical monitoring programs, allowing managers to target tidal streams based both on degraded biological conditions and water quality standards.

The TCEQ’s most recent surface water quality assessment of the same tidal segments generally reports high aquatic life use at sites with higher IBI scores, but discrepancies exist among mid- to low-scoring sites. The 2022 *Texas Integrated Report of Surface Water Quality for Clean Water Act Sections 305(b) and 303(d)* (Report) assesses the condition of tidal waterbodies, including concerns for public health, aquatic life use, water quality impairments, and possible pollutant sources (TCEQ 2022b). In the Report, all sites that scored in the “excellent” or “good” integrity classes using the nekton and benthic IBIs were also given a “high” designation for aquatic life use and had two or fewer water quality parameters of concern or impairment. The only sites that received a “limited” designation for aquatic life use were the low-scoring sites NOR1 and NOR2, which exist along a stream with water quality concerns surrounding bacteria, chlorophyll-a, and nitrate, and has significant non-point source pollution related to agriculture(TCEQ 2022b). Some mid- and low-scoring sites were given “high” (OSO1, OSO2, SAN) and “exceptional” (GUA) aquatic life use designations, despite concerns surrounding bacteria, chlorophyll-a, nitrate, and/or total phosphorus, and known municipal point-source discharges and/or non-point source pollution from urban and stormwater runoff and adjacent grazelands (TCEQ 2022b). ARR, another low-scoring site using the IBIs, also was given a “high” aquatic life use designation, despite having the most parameters of concern among sites in the Report. Water quality concerns (depressed dissolved oxygen, chlorophyll-a, nitrate, and total phosphorus) and impairments (bacteria, depressed dissolved oxygen, Mercury in edible tissue, and PCBs in edible tissue) from municipal and industrial point source pollution and non-point source pollution related to crop production, urban and stormwater runoff, and atmospheric deposition were all cited at ARR. Contrasting this study's results with the Report show that the IBIs may be a better, more conservative assessment for aquatic life use suitability of southern Texas tidal streams.

Consideration of stream size may be useful in future tidal stream studies because stream size can affect both the actual and sampled community composition. Individual species' abundances and measures of diversity have been shown to differ for freshwater streams of different orders, widths, and depths for fish (Harrel et al., 1967; Gorman and Karr, 1978; Walrath et al., 2016) and benthic macroinvertebrate communities (Harrel and Doris 1968). However, these relationships between biotic variables and physical characteristics are inconsistent among studies. In contrast one tidal stream study found that species assemblages among a tidal freshwater marsh system did not differ by stream order in Virginia, USA (Rozas and Odum, 1987). In this study, stream size appears only to be a potentially biasing characteristic of the nekton IBI developed in this study because both stream size and thalweg depth are significantly correlated with the nekton IBI scores. It is possible that differences in community composition in this study and previous ones may be perceived rather than actual because of differences in sampling efficiency. Seining sampling efficiency becomes less reliable as stream depth, width, and bottom structural complexity increase because seines can only reasonably profile the near-bank and near-surface community (Karr et al., 1986; Portt et al., 2006). However, communities collected by both seine and otter trawl consisted of generally similar and consistent sets of dominant taxa in several northeastern Texas tidal streams (Tolan and Nelson, 2013). Stream size does not likely affect our benthic IBI because benthic communities were sampled at similar depths regardless of the thalweg depth in this study. These findings warrant future studies to explore methodologies to reduce bias of scoring schema to stream size parameters. Some IBIs (Karr et al., 1986; Hughes et al., 1998; Harris and Silveira, 1999) utilize maximum species richness analyses to determine their scoring schema, where threshold values that correspond with IBI scores will differ with stream order or size. With future analyses, the bioassessment potential of the nekton IBI may be improved by incorporating data from tidal streams with greater size disparities within degraded and reference sites, or by implementing scoring criteria adjustments based on stream size.

In future refinements, further regionalizing the IBIs may minimize impacts that the Texas coast climatic gradient and species biogeographical range limits have on community assemblages within the study area. Precipitation rates decrease by a factor of two from the northeast to the southwest along the Texas coast, with much greater decreases in inflow occurring along the same gradient (Montagna et al., 2013; Montagna et al., 2018). It is difficult to discern whether some selected IBI metrics were truly indicative of degraded versus reference conditions or merely reflected the biogeography of species along the coastal gradient because degraded conditions were concentrated in the southern, drier part of our study area. Both the nekton and benthic IBI scores were significantly correlated with site latitude, where higher scores were generally given to sites in the northern portion of the study area. The inherent climatic variation appears to have influenced biological community assemblages because most degraded sites were concentrated in the southern half of the study area. An additional confounding factor is that the northernmost native range of some nekton species (such as *Poecilia formosa, Herichthys cyanoguttatus,* and *Astyanax mexicanus*) is the Nueces River drainage. These species were inherently found more often and in higher abundances in our southernmost streams, where degraded sites were concentrated. While these species were not represented as metric taxa within the IBIs, their presence did contribute to variation between degraded and reference condition nekton assemblages. Variation between major climactic subregions and species biogeography along a coastal gradient should be considered in future studies assessing communities in tidal streams.

When developing tidal stream IBIs, resource managers should consider incorporating multiple sampling sites along the lengthwise (upstream–downstream) gradient that may minimize the effects of salinity on collected fauna, allowing for a more characteristic analysis of the entire tidal stream reach. Previous studies determined that salinity was a major structuring factor for nekton assemblages when characterizing biological communities in Texas tidal streams (Tolan and Nelson, 2009; Tolan and Nelson, 2013; Margo, 2020). This indicates that it may be warranted to consider multiple sampling sites along the salinity gradient within the tidal stream segment. Four streams in this study (West Carancahua, Aransas River, Oso Creek, and North Floodway) had two sites per tidal segment to discern if location along this gradient affected nekton and benthic macroinfauna community composition. For the nekton IBI, three (Aransas River, Oso Creek, and North Floodway) of the four streams had lower mean scores at upstream sites when compared to the downstream counterparts, while all four downstream sites had higher mean scores in the benthic IBI. Although the sample size in this study was too small to sufficiently test for lengthwise effects, this indicates that the IBIs are potentially biased to assign higher scores to assemblages with higher salinity levels. This separation of scores for sites on the same stream indicates that, in refining tidal stream IBIs, multiple sampling sites along the same tidal segment should be included to incorporate the effects of salinity on the metrics derived for the index.

In addition to accounting for lengthwise spatial differences, temporal variation associated with these systems may be accounted for by incorporating long-term monitoring into the construction of tidal stream IBIs. Wide variations in salinity and tidal amplitude, often connected to temporal tidal and climatic events, are major structuring factors for the biotic and abiotic components of tidal streams (Odum et al., 1984; Odum, 1988). The nekton IBI appeared to have relatively stable site scores between years, while scores more notably decreased between the spring and summer seasons, indicating seasonal variation may be a more important structure factor in nekton communities than year-to-year variation. The summer season, typically a time of minimum stream flows, maximum temperatures, and minimum dissolved oxygen concentrations (Texas Commission on Environmental Quality (TCEQ), 2012), represents a higher stress portion of the year that influences nekton community composition and, therefore, IBI scores. The benthic IBI had a less predictable score response by site between years; for example, LAV's mean site score increased by 1.4, while SAN decreased 0.9. This inconsistency shows that benthic macroinfaunal assemblages may be more responsive to site-specific variation than to major climatic changes between years; however, additional data through time would solidify relationships between faunal assemblages and temporal impacts. Future studies may benefit from incorporating longer-term monitoring data into IBI construction to reduce the impact on temporal variation in IBI scoring results.

5. Conclusion

Indices of biotic integrity have been lauded for their ability to be implemented across multiple aquatic systems, regions, and taxa, to easily communicate results, and to be implemented efficiently and at a low cost (Karr, 1981; Simon and Lyons, 1995). The IBIs developed for both nekton and benthic macroinfauna communities in southern Texas tidal streams were somewhat successful in differentiating between degraded and reference streams. However, the natural environmental variation associated with tidal streams along the southern Texas coast makes the interpretation of some selected metrics difficult, because these systems are largely dominated by cosmopolitan taxa adapted to variable environmental conditions, and are influenced by biogeographic shifts in faunal assemblages along the coast's natural climate gradient. Regardless, this study provides an important first step in development of bioassessment protocols to monitor these coastal resources, which often lie at the edge of coastal development. In refining the IBI for tidal streams along the Texas coast, future studies should consider: (1) establishing a separate IBI for distinct climatic regions along the Texas coast latitudinal gradient, (2) utilizing long-term monitoring data to establish baseline conditions that diminish the effects of seasonal and inter-annual variation on biotic community structure, (3) establishing multiple sampling sites along the tidal segment to account for the effects of salinity on metric development and scoring, (4) analyzing stream size for scoring criteria adjustments or supplemental sampling methodologies, and (5) testing the IBI on an independent data set not used in the creation of the IBI. The indices developed in this study represent improvement upon previous tidal stream monitoring programs in Texas, which relied on only univariate physicochemical measures to list streams as impaired. With refinements and validation, the IBIs could enhance current tidal stream monitoring programs with a refined bio-assessment approach, allowing resource managers to assess the ability of tidal streams to support aquatic life more accurately and identify areas most in need of management attention.

CRediT authorship contribution statement

Alexis J. Neffinger: Writing – review & editing, Writing – original draft, Visualization, Investigation, Formal analysis. **Natasha J. Breaux:** Writing – review & editing, Supervision, Methodology, Investigation. **Abraham D. Margo:** Writing – review & editing, Visualization, Methodology, Investigation. **Terence A. Palmer:** Writing – review & editing, Visualization, Validation, Supervision, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Stacy N. Trackenberg:** Writing – review & editing. **Jennifer Beseres Pollack:** Writing – review & editing, Supervision, Software, Resources, Project administration, Methodology, Funding acquisition, 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.

Data availability

Data are available at doi: 10.7266/tc4p6mbx; The Gulf of Mexico Research Initiative Information and Data Cooperative (GRIIDC)

Acknowledgements

This project has been funded by the United States Environmental Protection Agency under assistance agreement 66.419 to Texas Commission on Environmental Quality (TCEQ). The contents of this document do not necessarily reflect the views and policies of the Environmental Protection Agency, nor does the EPA endorse trade names or recommend the use of commercial products mentioned in this document. Staff at the TCEQ were instrumental in providing support for this project, especially Ryan Seymour, Kate Lavelle, Sarah Eagle, and Linda Broach. This publication was also made possible by the National Oceanic and Atmospheric Administration, Office of Education Educational Partnership Program with Minority Serving Institutions awards (NA21SEC4810004 and NA16SEC4810009). Its contents are solely the responsibility of the award recipient and do not necessarily represent the official views of the U.S. Department of Commerce, National Oceanic and Atmospheric Administration. We would like to thank various other funding sources for support of AN over the course of this project, including: the Crutchfield Fellowship Endowment, Texas Sea Grant, the Coastal Conservation Association Science of Conservation Marine Scholarship Program, the Millicent Quammen Memorial Research Award, the Coastal and Estuarine Research Federation, Texas A&M University-Corpus Christi's Coastal and Marine System Science Program and Provost Fund. We thank James Tolan and Kim Withers for their contributions in reviewing early versions of this manuscript. Thank you to the members of the Coastal Conservation and Restoration Ecology Lab for their enthusiastic assistance throughout the course of field sampling. We would also like to thank Daunte Gaiter for his hard work sorting nekton and benthic macroinfauna samples in the laboratory. Lastly, we would like to thank two anonymous reviewers, whose constructive criticisms improved this manuscript.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2024.111804.

References

Alvarez Zarikian, C. A., P. L. Blackwelder, T. Hood, T. A. Nelsen, and C. Featherstone. 2000. Ostracods as indicators of natural and anthropogenically-induced changes in coastal environments. In Coasts at the Millennium, 896–905. Portland.

Anderson, D.M., 2009. Approaches to monitoring, control and management of harmful algal blooms (HABs). Ocean Coast. Manag. 52, 342–347. https://doi.org/10.1016/j.ocecoaman.2009.04.006.

Barendregt, A., Swarth, C.W., 2013. Tidal freshwater wetlands: variation and changes. Estuar. Coasts 36, 445–456. https://doi.org/10.1007/s12237-013-9626-z.

Beach, D., 2002. Coastal sprawl: the effects of urban design on aquatic ecosystems in the United States. Pew Oceans Commission, Arlington.

Bergquist, D., A. Blair, G. Riekerk, E. Wirth, L. Webster, J. Felber, T. Washburn, and A. F. Holland. 2011. *Gulf of Mexico tidal creeks serve as sentinel habitats for assessing the impact of coastal development on ecosystem health.* NOAA Technical Memorandum NOS NCCOS 136. Charleston: National Ocean Service of the National Oceanic and Atmospheric Administration.

Beseres Pollack, J., Kinsey, J.W., Montagna, P.A., 2009. Freshwater inflow biotic index (FIBI) for the Lavaca-Colorado estuary, Texas. Environ. Bioindic. 4, 153–169. https://doi.org/10.1080/15555270902986831.

Borja, A., Franco, J., Pérez, V., 2000. A marine biotic index to establish the ecological quality of soft-bottom benthos within european estuarine and coastal environments. Mar. Pollut. Bull. 40, 1100–1114. https://doi.org/10.1016/S0025-326X(00)00061-8.

Borja, A., Basset, A., Bricker, S., Dauvin, J., Elliot, M., Harrison, T., Marques, J., Weisberg, S., West, R., 2012. Classifying ecological quality and integrity of estuaries. In: Wolanski, E., McLusky, D. (Eds.), Treatise on Estuarine and Coastal Science. Academic Press, Waltham, pp. 125–162.

Breine, J., Quataert, P., Stevens, M., Ollevier, F., Volckaert, F.A.M., Van den Bergh, E., Maes, J., 2010. A zone-specific fish-based biotic index as a management tool for the zeeschelde estuary (Belgium). Mar. Pollut. Bull. 60, 1099–1112. https://doi.org/10.1016/j.marpolbul.2010.01.014.

Burton, T.M., Uzarski, D.G., Gathman, J.P., Genet, J.A., Keas, B.E., Stricker, C.A., 1999. Development of a preliminary invertebrate index of biotic integrity for Lake Huron coastal wetlands. Wetlands 19, 869–882. https://doi.org/10.1007/BF03161789.

Capmourteres, V., Rooney, N., Anand, M., 2018. Assessing the causal relationships of ecological integrity: a re-evaluation of karr's iconic index of biotic integrity. Ecosphere 9, 1–19. https://doi.org/10.1002/ecs2.2168.

Castillo, D., Kaplan, D., Mossa, J., 2016. A synthesis of stream restoration efforts in Florida (USA): Florida stream restoration. River Res. Appl. 32, 1555–1565. https://doi.org/10.1002/rra.3014.

Chowdhury, M.S.N., La Peyre, M., Coen, L.D., Morris, R.L., Luckenbach, M.W., Ysebaert, T., Walles, B., Smaal, A.C., 2021. Ecological engineering with oysters enhances coastal resilience efforts. Ecol. Eng. 169, 1–12. https://doi.org/10.1016/j.ecoleng.2021.106320.

Clarke, K.R., 1993. Non-parametric multivariate analyses of changes in community structure. Aust. J. Ecol. 18, 117–143. https://doi.org/10.1111/j.1442-9993.1993.tb00438.x.

Clarke, K.R., Gorley, R.N., 2015. Getting started with PRIMER v7. PRIMER-E, Plymouth.

Clarke, K.R., Warwick, R.M., 1994. Similarity-based testing for community pattern: the two-way layout with no replication. Mar. Biol. 118, 167–176. https://doi.org/10.1007/BF00699231.

Clarke, K.R., Chapman, M.G., Somerfield, P.J., Needham, H.R., 2006. Dispersion-based weighting of species counts in assemblage analyses. Mar. Ecol. Prog. Ser. 320, 11–27. https://doi.org/10.3354/meps320011.

Cohen, D., 2019. About 60.2M live in areas Most vulnerable to hurricanes. Accessed April 2022 United States Census Bureau. https://www.census.gov/library/stories/2019/07/millions-of-americans-live-coastline-regions.html.

Dauer, D.M., Ranasinghe, J.A., Weisberg, S.B., 2000. Relationships between benthic community condition, water quality, sediment quality, nutrient loads, and land use patterns in Chesapeake Bay. Estuaries 23, 80–96.

Dauvin, J.-C., Ruellet, T., 2009. The estuarine quality paradox: is it possible to define an ecological quality status for specific modified and naturally stressed estuarine ecosystems? Mar. Pollut. Bull. 59, 38–47. https://doi.org/10.1016/j.marpolbul.2008.11.008.

Dean, H.K., 2008. The use of polychaetes (annelida) as indicator species of marine pollution: a review. Rev. Biol. Trop. 56, 11–38. https://doi.org/10.15517/RBT.V56I4.27162.

Deegan, L.A., Finn, J.T., Ayvazian, S.G., Ryder-Kieffer, C.A., Buonaccorsi, J., 1997. Development and validation of an estuarine biotic integrity index. Estuaries 20, 601–617. https://doi.org/10.2307/1352618.

Díaz-Castañeda, V., Reish, D.J., 2009. Polychaetes in environmental studies. In: Shain, D. H. (Ed.), Annelids in Modern Biology. Wiley-Blackwell, Hoboken, pp. 203–227.

Elliott, M., Quintino, V., 2007. The estuarine quality paradox, environmental homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. Mar. Pollut. Bull. 54, 640–645. https://doi.org/10.1016/j.marpolbul.2007.02.003.

Elliott, M., Whitfield, A.K., Potter, I.C., Blaber, S.J.M., Cyrus, D.P., Nordlie, F.G., Harrison, T.D., 2007. The guild approach to categorizing estuarine fish assemblages: a global review. Fish Fish. 8, 241–268. https://doi.org/10.1111/j.1467-2679.2007.00253.x.

Engle, V.D., Summers, J.K., 1999. Refinement, validation, and application of a benthic condition index for northern Gulf of Mexico estuaries. Estuaries 22, 624–635. https://doi.org/10.2307/1353050.

Environmental Systems Research Institute (ESRI), 2017. ArcGIS Desktop version 10.6.1. Environmental Systems Research Institute, Redlands, CA.

Freeman, L.A., Corbett, D.R., Fitzgerald, A.M., Lemley, D.A., Quigg, A., Steppe, C.N., 2019. Impacts of urbanization and development on estuarine ecosystems and water quality. Estuar. Coasts 42, 1821–1838. https://doi.org/10.1007/s12237-019-00597-z.

Garner, T.R., Weinstein, J.E., Sanger, D.M., 2009. Polycyclic aromatic hydrocarbon contamination in South Carolina salt marsh-Tidal Creek systems: relationships among sediments, biota, and watershed land use. Arch. Environ. Contam. Toxicol. 57, 103–115. https://doi.org/10.1007/s00244-008-9256-9.

Gómez, N., Licursi, M., Bauer, D.E., Ambrosio, E.S., Rodrigues Capítulo, A., 2012. Assessment of biotic integrity of the coastal freshwater tidal zone of a temperate estuary of South America through multiple indicators. Estuar. Coasts 35, 1328–1339. https://doi.org/10.1007/s12237-012-9528-5.

Gonzalez, L.A., Quigg, A., Steichen, J.L., Gelwick, F.P., Lester, L.J., 2020. A new approach to functionally assess estuarine fish communities in response to hydrologic change. Estuar. Coasts 44, 1118–1131. https://doi.org/10.1007/s12237-020-00824-y.

Gorman, O.T., Karr, J.R., 1978. Habitat structure and stream fish communities. Ecology 59, 507–515. https://doi.org/10.2307/1936581.

Guillen, G.J., 1996. Development of a rapid bioassessment method and index of biotic integrity for tidal streams and bayous located along the Northwest Gulf of Mexico. Texas Natural Resource Conservation Commission, Austin.

Hackney, C.T., Burbanck, W.D., Hackney, O.P., 1976. Biological and physical dynamics of a Georgia tidal creek. Chesapeake Sci. 17, 271–280. https://doi.org/10.2307/1350514.

Harrel, R.C., Davis, B.J., Dorris, T.C., 1967. Stream order and species diversity of fishes in an intermittent Oklahoma stream. Am. Midl. Nat. 78, 428–436. https://doi.org/10.2307/2485240.

Harris, J.H., Silveira, R., 1999. Large-scale assessments of river health using an index of biotic integrity with low-diversity fish communities. Freshw. Biol. 41, 235–252. https://doi.org/10.1046/j.1365-2427.1999.00428.x.

Houde, E.D., Rutherford, E.S., 1993. Recent trends in estuarine fisheries: predictions of fish production and yield. Estuaries 16, 161. https://doi.org/10.2307/1352488.

Howland, J., Alexander, A., Milani, D., Peru, K., Culp, J., 2019. Risks of mixtures of oil sands contaminants to a sensitive mayfly sentinel. *Hexagenia*. *Diversity* 11, 118. https://doi.org/10.3390/d11080118.

Hubbs, C., Lucier, T., Garrett, G., 1978. Survival and abundance of introduced fishes near San Antonio, Texas. Tex. J. Sci. 30, 369–376.

Hughes, R.M., Kaufmann, P.R., Herlihy, A.T., Kincaid, T.M., Reynolds, L., Larsen, D.P., 1998. A process for developing and evaluating indices of fish assemblage integrity. Can. J. Fish. Aquat. Sci. 55, 1618–1631. https://doi.org/10.1139/cjfas-55-7-1618.

Jones, A.E., Hardison, A.K., Hodges, B.R., McClelland, J.W., Moffett, K.B., 2020. Defining a riverine tidal freshwater zone and its spatiotemporal dynamics. Water Resour. Res. 56, 1–17. https://doi.org/10.1029/2019WR026619.

Karr, J.R., 1981. Assessment of biotic integrity using fish communities. Fisheries 6, 21–27. https://doi.org/10.1577/1548-8446(1981)006<0021:aobiuf>2.0.co;2.

Karr, J.R., 1991. Biological integrity: a long-neglected aspect of water resource management. Ecol. Appl. 1, 66–84. https://doi.org/10.2307/1941848.

Karr, J.R., Fausch, K.D., Angermeier, P.L., Yant, P.R., Schlosser, I.J., Creek, J., 1986. Assessing biological integrity in running waters a method and its rationale. Illinois National History Survey, Champaign.

Kennish, M.J., 2002. Environmental threats and environmental future of estuaries. Environ. Conserv. 29, 78–107. https://doi.org/10.1017/CBO9780511751790.018.

Latham, M.A., 2010. (Un)restoring the chemical, physical, and biological integrity of our nation's waters: the emerging clean water act jurisprudence of the Roberts court. Virginia Environmental Law Journal 28, 411–482.

Llansó, R.J., Scott, L.C., Hyland, J.L., Dauer, D.M., Russell, D.E., Kutz, F.W., 2002. An estuarine benthic index of biotic integrity for the mid-Atlantic region of the United States. II. Index Development. *Estuaries* 25, 1231–1242. https://doi.org/10.1007/BF02692220.

Loomis, M.J., Craft, C.B., 2010. Carbon sequestration and nutrient (nitrogen, phosphorus) accumulation in river-dominated tidal marshes, Georgia, USA. Soil Sci. Soc. Am. J. 74, 1028–1036. https://doi.org/10.2136/sssaj2009.0171.

Lotze, H.K., Lenihan, H.S., Bourque, B.J., Bradbury, R.H., Cooke, R.G., Kay, M.C., Kidwell, S.M., Kirby, M.X., Peterson, C.H., Jackson, J.B.C., 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science 312, 1806–1809. https://doi.org/10.1126/science.1128035.

Lubanga, H.L., Manyala, J.O., Sitati, A., Yegon, M.J., Masese, F.O., 2021. Spatial variability in water quality and macroinvertebrate assemblages across a disturbance gradient in the Mara River basin, Kenya. Ecohydrol. Hydrobiol. 21, 718–730. https://doi.org/10.1016/j.ecohyd.2021.03.001.

Lyons, J., Gutierrez-Hernandez, A., Diaz-Pardo, E., Soto-Galera, E., Medina-Nava, M., Pineda-Lopez, R., 2000. Development of a preliminary index of biotic integrity (IBI) based on fish assemblages to assess ecosystem condition in the lakes of Central Mexico. Hydrobiologia 418, 57–72. https://doi.org/10.1023/A:1003888032756.

Mabe, J. A., and J. B. Moring. 2008. *Variation in biotic assemblages and stream-habitat data with sampling strategy and method in tidal segments of Highland and Marchand Bayous, Galveston County, Texas, 2007*. Scientific Investigations Report 5151. Reston: U.S. Geological Survey.

Mallin, M.A., Lewitus, A.J., 2004. The importance of tidal creek ecosystems. J. Exp. Mar. Biol. Ecol. 298, 145–149. https://doi.org/10.1016/S0022-0981(03)00356-3.

Margo, A., 2020. Analyses of biological communities and development of indices of biotic integrity for monitoring tidal streams along the upper Texas coast. Texas A&M University-Corpus Christi, Corpus Christi. Master's thesis,.

McLusky, D.S., 1993. Marine and estuarine gradients—an overview. Neth. J. Aquat. Ecol. 27, 489–493. https://doi.org/10.1007/BF02334809.

Montagna, P.A., Palmer, T.A., Beseres Pollack, J., 2013. Hydrological changes and estuarine dynamics. Springer, New York.

Montagna, P.A., Hu, X., Palmer, T.A., Wetz, M., 2018. Effect of hydrological variability on the biogeochemistry of estuaries across a regional climatic gradient. Limnol. Oceanogr. 63, 2465–2478. https://doi.org/10.1002/lno.10953.

Morehead, S., Montagna, P., Kennicutt II., M.C., 2008. Comparing fixed-point and probabilistic sampling designs for monitoring the marine ecosystem near McMurdo Station, Ross Sea, Antarctica. Antarct. Sci. 20, 471–484. https://doi.org/10.1017/S0954102008001326.

Odum, E.P., 1985. Trends expected in stressed ecosystems. Bioscience 35, 419–422. https://doi.org/10.2307/1310021.

Odum, W.E., 1988. Comparative ecology of tidal freshwater and salt marshes. Annu. Rev. Ecol. Syst. 147–176, 0066–4 162/8811120-0147$02.00.

Odum, W.E., Smith III, T.J., Hoover, J.K., McIvor, C.C., 1984. Ecology of tidal freshwater marshes of the United States east coast: a community profile. U.S. Fish and Wildlife Service, Washington, DC.

Ogburn-Matthews, M.V., Allen, D.M., 1993. Interactions among some dominant estuarine nekton species. Estuaries 16, 840–850. https://doi.org/10.2307/1352443.

Parker, C., Sanger, D., Wirth, E., 2023. An assessment of Southeast United States headwater Tidal Creek sediment contamination over a twenty-year period in relation to coastal development. Environ. Manag. 72, 883–901. https://doi.org/10.1007/s00267-023-01835-8.

Pearson, T.H., Rosenberg, R., 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanogr. Mar. Biol. 16, 229–311.

Pinto, R., Patrício, J., Baeta, A., Fath, B.D., Neto, J.M., Marques, J.C., 2009. Review and evaluation of estuarine biotic indices to assess benthic condition. Ecol. Ind. 9, 1–25. https://doi.org/10.1016/j.ecolind.2008.01.005.

Porinchu, D.F., MacDonald, G.M., 2003. The use and application of freshwater midges (chironomidae: insecta: diptera) in geographical research. Progress in Physical Geography: Earth and Environment 27, 378–422. https://doi.org/10.1191/0309133303 pp388ra.

Portt, C. B., G. A. Coker, D. L. Ming, and R. G. Randall. 2006. *A review of fish sampling methods commonly used in Canadian freshwater habitats*. Canadian Technical Report of Fisheries and Aquatic Sciences Report No. 2604. Burlington: Fisheries and Oceans Canada.

Qadir, A., Malik, R.N., 2009. Assessment of an index of biological integrity (IBI) to quantify the quality of two tributaries of river chenab, Sialkot, Pakistan. Hydrobiologia 621, 127–153. https://doi.org/10.1007/s10750-008-9637-0.

Raburu, P.O., Masese, F.O., Mulanda, C.A., 2009. Macroinvertebrate index of biotic integrity (M-IBI) for monitoring rivers in the upper catchment of Lake Victoria Basin, Kenya. Aquat. Ecosyst. Health Manag. 12, 197–205. https://doi.org/10.1080/14634980902907763.

Rakocinski, C.F., Brown, S.S., Gaston, G.R., Heard, R.W., Walker, W.W., Summers, J.K., 1997. Macrobenthic responses to natural and contaminant-related gradients in northern Gulf of Mexico estuaries. Ecol. Appl. 7, 1278–1298. https://doi.org/10.1890/1051-0761(1997)007[1278:MRTNAC]2.0.CO;2.

Resh, V.H., 2008. Which group is best? attributes of different biological assemblages used in freshwater biomonitoring programs. Environ. Monit. Assess. 138, 131–138. https://doi.org/10.1007/s10661-007-9749-4.

Rodriguez, P., Reynoldson, T.B., 2011. The pollution biology of aquatic oligochaetes. Springer, Dordrecht.

Rozas, L.P., Odum, W.E., 1987. Use of tidal freshwater marshes by fishes and macrofaunal crustaceans along a marsh stream-order gradient. Estuaries 10, 36–43. https://doi.org/10.2307/1352023.

RStudio Team, 2018. RStudio: Integrated Development for R. RStudio, Inc. http://www.rstudio.com/.

Ruiz, F., Abad, M., Bodergat, A.M., Carbonel, P., Rodríguez-Lázaro, J., Yasuhara, M., 2005. Marine and brackish-water ostracods as sentinels of anthropogenic impacts. Earth Sci. Rev. 72, 89–111. https://doi.org/10.1016/j.earscirev.2005.04.003.

Simon, T.P., Lyons, J., 1995. Application of the index of biotic integrity to evaluate water resource integrity in freshwater ecosystems. In: Davis, W.S., Simon, T.P. (Eds.), Biological Assessment and Critera: Tools for Water Resource Planning and Decision Making. Lewis, Boca Raton, pp. 245–262.

Stoddard, J.L., Larsen, D.P., Hawkins, C.P., Johnson, R.K., Norris, R.H., 2006. Setting expectations for the ecological condition of streams: the concept of reference condition. Ecol. Appl. 16, 1267–1276. https://doi.org/10.1890/1051-0761(2006)016[1267:SEFTEC]2.0.CO;2.

Stoddard, J.L., Herlihy, A.T., Peck, D.V., Hughes, R.M., Whittier, T.R., Tarquinio, E., 2008. A process for creating multimetric indices for large-scale aquatic surveys. J. N. Am. Benthol. Soc. 27, 878–891. https://doi.org/10.1899/08-053.1.

Tan, C.W.J., Gouramanis, C., Pham, T.D., Hoang, D.Q., Switzer, A.D., 2021. Ostracods as pollution indicators in lap an lagoon, Central Vietnam. Environ. Pollut. 278, 116762 https://doi.org/10.1016/j.envpol.2021.116762.

Texas Commission on Environmental Quality (TCEQ), 2012. *Surface water quality monitoring procedures Volume 1: Physical and chemical monitoring methods.* RG-415. Austin, TX: Texas Commission on Environmental Quality. 202 pp. https://www.tceq.texas.gov/downloads/publications/rg/swqm-procedures-volume-1.pdf.

Texas Commission on Environmental Quality (TCEQ), 2020. *2020 Guidance for Assessing and Reporting Surface Water Quality in Texas, In Compliance with Sections 305(b) and 303(d) of the Federal Clean Water Act.* Austin: Texas Commission for Environmental Quality. 169 pp. https://www.tceq.texas.gov/assets/public/waterquality/swqm/assess/20txir/2020_guidance.pdf.

Texas Commission on Environmental Quality, 2022. Surface water quality web reporting tool. Accessed Dec 2021. https://www80.tceq.texas.gov/SwqmisPublic/index.htm.

Thrush, S.F., Townsend, M., Hewitt, J.E., Davies, K., Lohrer, A.M., Lundquist, C., Cartner, K., 2013. The many uses and values of estuarine ecosystems. In: Dymond, J. R. (Ed.), Ecosystem Services in New Zealand – Conditions and Trends. Manaaki Whenua Press, Lincoln, New Zealand, pp. 226–237.

Tolan, J.M., Nelson, J.M., 2009. Relationships among nekton assemblage structure and abiotic conditions in three Texas tidal streams. Environ. Monit. Assess. 159, 15–34. https://doi.org/10.1007/s10661-008-0609-7.

Tolan, J.M., Nelson, J.M., 2013. Spatial assessment of a biocriteria applied to Texas tidal streams. Journal of Ecosystems 2013, 1–16. https://doi.org/10.1155/2013/726594.

Tweedley, J.R., Warwick, R.M., Potter, I.C., 2015. Can biotic indicators distinguish between natural and anthropogenic environmental stress in estuaries? J. Sea Res. 102, 10–21. https://doi.org/10.1016/j.seares.2015.04.001.

United States Census Bureau, 2010. 2010 census – block maps. Accessed Jun 2022. https://www.census.gov/geographies/reference-maps/2010/geo/2010-census-block-maps.html.

United States Environmental Protection Agency, 1994. *Chesapeake Bay basin toxics loading and release inventory*. CPB/TRS 102/94.

United States Environmental Protection Agency, 2022. Guidelines Establishing Test Procedures for the Analysis of Pollutants. In: *Title 40 of the Code of Federal Regulations Part 136*. (7–1–22 Edition). 416 pp. https://www.ecfr.gov/current/title-40/part-136.

Van Dolah, R.F., Hyland, J.L., Holland, A.F., Rosen, J.S., Snoots, T.R., 1999. A benthic index of biological integrity for assessing habitat quality in estuaries of the southeastern USA. Mar. Environ. Res. 48, 269–283. https://doi.org/10.1016/S0141-1136(99)00056-2.

Walrath, J.D., Dauwalter, D.C., Reinke, D., 2016. Influence of stream condition on habitat diversity and fish assemblages in an impaired upper Snake River basin watershed. Trans. Am. Fish. Soc. 145, 821–834. https://doi.org/10.1080/00028487.2016.1159613.

Weisberg, S.B., Ranasinghe, J.A., Dauer, D.M., Schaffner, L.C., Diaz, R.J., Frithsen, J.B., 1997. An estuarine benthic index of biotic integrity (B-IBI) for Chesapeake Bay. Estuaries 20, 149–158. https://doi.org/10.2307/1352728.

Wessel, M.R., Leverone, J.R., Beck, M.W., Sherwood, E.T., Hecker, J., West, S., Janicki, A., 2021. Developing a water quality assessment framework for Southwest Florida tidal creeks. Estuar. Coasts. https://doi.org/10.1007/s12237-021-00974-7.

Whitfield, A.K., Elliott, M., 2002. Fishes as indicators of environmental and ecological changes within estuaries: a review of progress and some suggestions for the future. J. Fish Biol. 61, 229–250. https://doi.org/10.1111/j.1095-8649.2002.tb01773.x.

Yang, L., Jin, S., Danielson, P., Homer, C., Gass, L., Bender, S.M., Case, A., et al., 2018. A new generation of the United States National Land Cover Database: requirements, research priorities, design, and implementation strategies. ISPRS J. Photogramm. Remote Sens. 146, 108–123. https://doi.org/10.1016/j.isprsjprs.2018.09.006.