

Environmental Management

Influence of water quality on the macroinvertebrate community in a tropical estuary (Buenaventura Bay)

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Abstract

The anthropogenic discharges of inorganic nutrients impact water quality, affecting the macroinvertebrate assemblage and food safety. The main objective of this study was to examine the seawater quality and macroinvertebrate dynamics in muddy habitats of Buenaventura Bay, Colombian Pacific. Macroinvertebrates were captured using artisanal trawl nets during different seasons and along four sampling sites. Multivariate analyses (canonical correspondence analysis and generalized additive model) were used to assess the effects of variations in nitrite, nitrate, phosphate concentrations, and physicochemical variables (salinity, pH, dissolved oxygen [DO], temperature, and total dissolved solids [TDS]) of water on the macroinvertebrate assemblage. Richness was the highest at sites with high salinity and temperature and low concentrations of nitrites and TDS. The densities of the commercial shrimp species *Xiphopenaeus riveti* and *Rimapenaeus byrdi* were the highest at sites with higher DO and alkalinity, and lower nitrate concentrations. The swimming crab *Callinectes arcuatus* was dominant at sites with low water quality. In summary, in the transitional season and at the inner sites of Buenaventura Bay, it was observed the lowest water quality due to high nitrate concentration. High nitrate concentration was highlighted as the main anthropogenic factor that could decrease the capture of target macroinvertebrate species for food and livelihoods of artisanal fishermen and their families. Thus, macroinvertebrate communities may be vulnerable to increased inorganic nutrient inputs, which could affect estuarine water quality and ecosystems services. *Integr Environ Assess Manag* 2022;18: 796–812. © 2021 The Authors. *Integrated Environmental Assessment and Management* published by Wiley Periodicals LLC on behalf of Society of Environmental Toxicology & Chemistry (SETAC).

KEYWORDS: Colombian Pacific, Estuarine dynamics, Inorganic nutrients, Invertebrates

INTRODUCTION

The spatiotemporal distribution of estuarine macroinvertebrates is affected by biotic and abiotic factors. Some of the most relevant biotic factors include reproductive migrations, use of refuge to avoid predators and foraging (Costa-Dias et al., 2010). Conversely, environmental factors including, but not restricted to, salinity, temperature, dissolved oxygen (DO), and nutrients can have a strong effect on the dynamics of estuarine species (Gamboa-

García, Duque, & Cogua, 2018; Ren et al., 2018; Thompson et al., 2013). In this context, increases in human settlements, changes in production activities, and associated discharges can affect water quality by increasing the nutrient concentration, which in turn disturbs nutrient cycling, ecosystem productivity, and trophic interactions (Barletta et al., 2019; Nie et al., 2018). Thus, the decrease in water quality could affect aquatic communities and impact the sizes and abundances of target species for artisanal fisheries (Ji, 2008). In this way, preservation of water quality would be required to conserve fishing productivity (Visbeck, 2018), which is the main source of food and income for artisanal fishermen.

In developing countries, a great proportion of the production of artisanal fisheries present in estuaries and coastal waters is destined for local consumption and commercialization (Villanueva & Flores-Nava, 2019). The average per capita macroinvertebrate consumption in these zones is around 0.8 kg/week (Ardila Benavides, 2000). Thus, by

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weight, macroinvertebrates represent 23% of the total consumption of marine fish and shellfish (Gamboa-García et al., 2020), making them important fishing resources, due to their accessibility, availability, and nutritional quality.

Water quality can be considered an indicator of adequate ecosystem functioning and is defined by a set of abiotic and biotic parameters. The abiotic parameters include the nutrient concentration, while the biotic parameters include the intra- and interspecific relationships, which together support ecosystem services as artisanal fisheries production (Foley et al., 2015; Pouso et al., 2018). Other researchers have reported that estuarine waters enriched with nutrients from anthropogenic wastes and river discharge can affect invertebrate assemblages in two ways: (1) subjecting certain species to physiological stress and low DO levels, which results in a decrease in the presence, abundance, and biomass of certain species and (2) initially increasing productivity, triggering a trophic cascade, ultimately resulting in an increase in the abundance and biomass of certain dominant species (Fong & Fong, 2018; Hale et al., 2018; Kenworthy et al., 2016; de Mutsert et al., 2016; Nelson et al., 2019; Villafañe et al., 2017; Wilkerson & Dugdale, 2016).

Considering these multiple environmental impacts, studies of macroinvertebrate richness, abundance, and biomass could be useful in assessing the effects of eutrophication on estuarine ecosystems. In particular, benthic macroinvertebrate communities are sensitive to anthropogenic pressures and environmental stressors (Dauvin, 2007). However, the effect of anthropogenic stressors on macroinvertebrate biomass in tropical estuaries remains unclear because these stressors interact with strong natural variations and could thus act in isolation or synergistically (I. Martins et al., 2019). Other studies have reported that fish biomass could indicate anthropogenic disturbances, given that certain species are sensitive to eutrophication (de Mutsert et al., 2016; Duque et al., 2020). The concentrations of nitrite, nitrates, and phosphates, alkalinity, and DO, as well as nutrient eutrophication levels in water, were used to characterize anthropogenic stressors. Changes in the water salinity, temperature, pH, and total dissolved solids (TDS) were considered natural stressors.

In the Colombian Pacific, in particular, there is a strong dependency of the community on fishery resources, which are critical for food security, as fishes are exploited for consumption and commercialization (Saavedra-Díaz et al., 2016; Villanueva & Flores-Nava, 2019). In this area, a lack of handling and management of wastewater from increasing anthropogenic activities such as tourism and agroindustry has been documented (Garay-Tinoco et al., 2006). Different types of pollutants dissolved in this wastewater reach the sea without prior treatment not only through rivers but also through pipes (Vivas-Aguas et al., 2014), which could change the natural level of inorganic nutrients in the water of the bay. Sewage contains a broad range of polluting agents, which have direct and indirect effects on organisms and ecosystems (Verga et al., 2020). In particular, increased phosphate and nitrogen discharges could have indirect effects on the

macroinvertebrate community, disturbing the producer level of trophic networks and inhibiting the activity of benthic photosynthetic organisms through the commonly accepted mechanism of uncoupling photophosphorylation, thus also constraining growth (Admiraal, 1977). At the food web scale, the inhibition of producer organisms could induce a bottom-up trophic cascade effect, resulting in fluctuations of the biomass that can be used by artisanal fishermen (Duque et al., 2020). For these reasons, the livelihood of people dependent on fishery resources could be affected by changes in water quality.

In the estuaries of the Colombian Pacific, there is scientific information on the impacts of pollution. For instance, the fish assemblages are affected by the water quality (Duque et al., 2020). There is an active bioaccumulation of mercury in fish (Duque & Cogua, 2016), macroinvertebrates (Gamboa-García et al., 2020), and birds (Gamboa-García Duque, Cogua, & Freire, 2018). Additionally, the effects of organic matter on the polychaete distribution and abundance (Martínez et al., 2019) and their influence on the diet of some estuarine fish have been identified (García, 2020; Tafurt, 2020). Also, there is an increase in microplastics over time in the sediments of Buenaventura Bay (Vásquez et al., 2021). However, in these ecosystems, there was a lack of evidence on if the increase in the concentration of inorganic nutrients in the bay water could be due to the climatic season associated with increased tourism, and if the dynamics of the macroinvertebrate assemblage could be explained by the changes in the concentration of these nutrients.

Considering the relevance of monitoring the estuarine macroinvertebrate communities in relation to water quality, the aim of this study was to examine the seawater quality and macroinvertebrate dynamics in muddy habitats of Buenaventura Bay, Colombian Pacific, and explore its seasonal and spatial variability. In addition, baseline data are crucial for adequate monitoring and to provide information for public health organizations and management systems.

MATERIALS AND METHODS

Study area

This study was conducted in Buenaventura Bay (77°16'W 3°56'N, Colombia), in the eastern tropical Pacific (Figure 1). The estuary covers an area of approximately 70 km² and has a single marine inlet known as La Bocana, which is approximately 1.6 km wide; it comprises Punta Bazán in the North and Punta Soldado in the South (Castaño, 2002) and can be classified as an open bay. Buenaventura Bay includes the mouths of the Dagua (66 m³ s⁻¹) and Anchicayá (98 m³ s⁻¹) rivers (Otero, 2004) and is characterized by some of the highest humidity and rainfall levels worldwide, with an annual average rainfall of 6980 mm. This region is characterized by a dry season, between January and March, and a wet season, between September and November. The other periods correspond to transitional seasons (Cantera & Blanco, 2001).

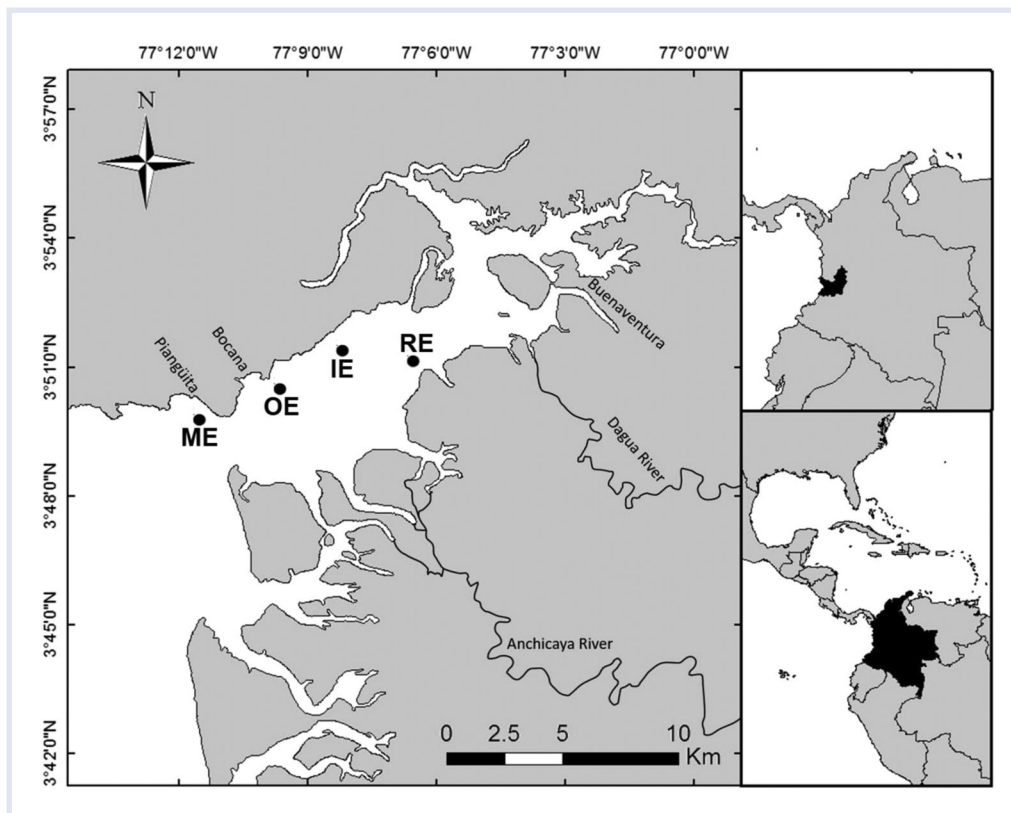


FIGURE 1 Sampling sites within Buenaventura Bay estuary. IE, inner estuary; ME, marine estuary; OE, outer estuary; RE, river estuary

The estuary is composed of two well-defined zones: the inner and outer bays. In the inner bay, port activities together with waste from fishing, logging, and mining operations, as well as the discharge of river and urban residual waters, have resulted in increased levels of several potentially polluting substances (Duque & Cogua, 2016; Gamboa-García, Duque, & Cogua, 2018; Gamboa-García et al., 2020; Martínez et al., 2019) including nitrite, nitrates, phosphates (Duque et al., 2020), sulfates, and coliforms, which in turn have resulted in a significant increase in organic matter concentration (IIAP, 2013). Moreover, a ship access channel extends all over the bay and requires maintenance dredging and canal expansion (Montenegro Arboleda & Torres Garcés, 2016), which can disrupt water nutrient dynamics. The external bay was influenced by marine conditions where the Pianguita sector is located, which is an important center for seasonal tourism (Duque et al., 2020; Escobar-Cárdenas, 2009; Gamboa-García et al., 2020).

Sampling was conducted at four sites within Buenaventura Bay (Figure 1). From east to west: river estuary (RE), inner estuary (IE), outer estuary (OE), and marine estuary (ME). RE is located at the mouth of Dagua River ($77^{\circ}6'33.1''\text{W}$ and $3^{\circ}50'51.5''\text{N}$), whereas IE is located 10 km northwest of the mouth of Dagua River ($77^{\circ}7'24.9''\text{W}$ and $3^{\circ}52'4.4''\text{N}$). These two most internal sites were the closest to the main urbanized zone within Buenaventura Bay, with approximately 300 000 inhabitants (DANE, 2019). In other investigations, this area has presented low salinity due to its

proximity to the freshwater inlet of the rivers (Gamboa-García et al., 2020; Molina et al., 2020). It has also had lower water quality due to its proximity to the populated centers (Vivas-Aguas et al., 2014). On the contrary, OE is located close to the mouth of the bay ($77^{\circ}9'35.9''\text{W}$ and $3^{\circ}50'58.7''\text{N}$), and ME is located at the mouth of the bay ($77^{\circ}12'11.4''\text{W}$ and $3^{\circ}49'52.44''\text{N}$). These two outermost sites are located close to the townships of La Bocana and Pianguita, which have a population of over 3000 inhabitants each and are strongly dependent on marine resources, exploited through artisanal fishing, and receive a large influx of tourists during the holiday season (Escobar-Cárdenas, 2009). In this area, high salinity, moderate water quality, and assemblage of macroinvertebrates determined by interaction with seawater were expected.

Field sampling

Three sampling trips were conducted representing distinct hydro-climatic conditions: wet season (November 2018, total monthly rainfall = 753.8 mm), dry season (March 2019, total monthly rainfall = 321.2 mm), and transitional season (July 2019, total monthly rainfall = 469.2 mm) (IDEAM, 2020), and four sampling sites that represented a gradient in salinity, water quality, and macroinvertebrate assemblage dynamics within the bay were selected. Sampling was carried out from 5:00 a.m. to 12:00 p.m., at a maximum depth of 8 m.

Three trawls in 1 day were performed during each season at each sampling site using an artisanal trawling net by two

men. This net was the same as that used by local people, and its dimensions were a mesh size (between knots) of 25.4 mm, a ground rope of 3.6 m, and a head rope of 3.1 m (Molina et al., 2020). Similar sampling efforts were performed at each sampling site and trawl to identify shifts in community structure. Each trawl lasted 10 min and was conducted using a trawl net with a 2.54-cm mesh size and an 8-m mouth width. The trawling area ranged between 0.086 and 0.087 ha. The trawling methods used in this study complied with mesh size regulations and were approved by the ethics committee (comité de ética, Universidad Nacional de Colombia, sede Palmira).

Physicochemical variables were measured in situ, and triplicate water nutrient samples were collected. Physicochemical variables (salinity, DO, pH, temperature, and TDS) were measured using the multiparameter probe YSI 556 MPS, whereas the concentrations of nutrients, including nitrites, nitrates, and phosphates, and alkalinity were measured using the handheld photometer YSI 9300. Both biological and seawater samples were taken in three replicates. All instruments were calibrated to certified standards before performing in situ measurements.

Specimens were identified to the species level and enumerated, and the length and weight were measured. Specific keys (Cardoso & Hochberg, 2013; Fischer et al., 1995; Lazarus Agudelo & Cantera, 2007; Lemaitre & Alvarez León, 1992; Neira & Cantera, 2005; Pineda & Madrid, 1993) and internet databases (World Register of Marine Species) were used for the taxonomic identification of macroinvertebrates.

Data analysis

Macroinvertebrate species were classified as resident or migratory according to the frequency of use of the estuary by each species (Díaz-Ochoa & Quiñones, 2008; Gamboa-García, Duque, & Cogua, 2018; Hendrickx & Sánchez-Vargas, 2005; Hernández-Noguera et al., 2016; Mora-Lara, 1973). The dynamics of the macroinvertebrate community structure were assessed using richness and biomass. Richness was the mean obtained on counting the number of species for each of the three trawl replicas. Biomass (g ha^{-1}) was calculated as the mean of the weight of fresh organisms (wet weight) for each replicate divided by the hauling area. For the density analysis (ind ha^{-1}), the most representative macroinvertebrate species were selected, which were identified using the highest average percent frequency (Equation 1), abundance (Equation 2), and weight (Equation 3) (A. C. B. Martins et al., 2015). The biomass of *Callinectes arcuatus* was analyzed as this species has been used as a bioindicator in other investigations and due to its higher comparability, which means higher frequency of occurrence, abundance, and biomass in the area.

$$\text{Frequency} = 100 \times \frac{\text{Macroinvertebrate species presence}}{\text{Total macroinvertebrate count}}, \quad (1)$$

$$\text{Relative abundance} = 100 \times \frac{\text{Number of macroinvertebrates by species}}{\text{Total number of macroinvertebrates}}, \quad (2)$$

$$\text{Weight} = 100 \times \frac{\text{Macroinvertebrate weight by species}}{\text{Total weight}}. \quad (3)$$

To determine differences in the total richness, density, and biomass of the most abundant macroinvertebrates, analysis of variance (ANOVA) and Tukey's post hoc test ($p < 0.05$) were performed with season, site, and their interaction as factors. Data normality was assessed (tukey) and data transformation was performed (squared root).

The concentrations of nitrite (mg L^{-1}), nitrates (mg L^{-1}), phosphates (mg L^{-1}), and alkalinity (mg L^{-1}), as well as nutrient eutrophication levels, were used to characterize anthropogenic stressors that might affect water quality. Nitrite and nitrate concentrations below 0.1 mg L^{-1} and phosphate concentrations below 0.01 mg L^{-1} were considered to be indicative of high water quality (i.e., oligotrophic), nitrite and nitrate concentrations between 0.1 and 1.0 mg L^{-1} and phosphate concentrations above 0.1 mg L^{-1} were considered to be indicative of intermediate water quality (i.e., mesotrophic), and nitrite and nitrate concentrations above 1.0 mg L^{-1} and phosphate concentrations above 0.1 mg L^{-1} were considered to be indicative of low water quality (i.e., eutrophic) (Lemley et al., 2015, 2017). Dissolved oxygen concentrations below 5 mg L^{-1} were considered to be indicative of low water quality, and concentrations of or above 5 mg L^{-1} were considered to be indicative of high water quality (Lemley et al., 2015). For the spatiotemporal analysis, ANOVA was performed with season, site, and their interaction as factors.

To assess variations in the density of the most representative macroinvertebrate species with respect to natural and anthropogenic stressors affecting water quality, a canonical correspondence analysis (CCA) was performed with the $\log(x+1)$ -transformed data matrix using the *vegan* library in the R software package with the *cca* and *spenvcor* functions (R Core Team, 2013). To assess variations in richness, total macroinvertebrate biomass, and swimming crab (*C. arcuatus*) biomass in relation to water quality, these biological descriptors were modeled using Bayesian generalized additive models (GAM) and different combinations of explanatory variables. These models represent an alternative to conventional statistical analyses, which are difficult to interpret in complex scenarios in which variables show non-linear relationships (Rudy et al., 2016). Akaike's information criterion (AIC) was used for model comparison and selection. For each biological descriptor, the model was selected based on a $\Delta\text{AIC} > 2$ compared with the previous model (Krause et al., 2019; A. C. B. Martins et al., 2015). Subsequently, the synergistic effects of natural and anthropogenic stressors on the richness, total biomass, and *C. arcuatus* biomass were examined. All statistical analyses were performed using the

gam, *nlme*, and *mgcv* libraries in R with the *gam* and *vis.gam* functions.

RESULTS

Macroinvertebrate species richness, biomass, and density dynamics

A total of 16 macroinvertebrate species were identified. The most representative species in descending order of relative frequency, abundance, and weight (Equations 1–3) were the resident species *C. arcuatus* (60.7%), the ontogenic migratory *Xiphopenaeus riveti* (19.1%), the resident *Squilla aculeata aculeata* (15.9%), and the commercial ontogenic migrators *Rimapenaeus byrdi* (8.3%) and *Penaeus occidentalis* (7.8%). Conversely, *Lolliguncula panamensis* (7.0%), *Penaeus brevirostris* (7.0%), *Callinectes toxotes* (4.5%), *Luidia columbia* (2.3%), *Penaeus californiensis* (2.2%), *Achelous asper* (2.1%), *Panulirus penicillatus* (1.6%), *Penaeus stylirostris* (1.0%), *Penaeus vannamei* (1.0%), *Trachysalambria brevisuturiae* (1.0%), and *Alpheus colombiensis* (1.0%) had lower average percentages and were therefore not analyzed individually.

The highest macroinvertebrate species richness (4.3 ± 0.6 species; $F = 3.15$, Tukey's $p < 0.05$) was recorded during the wet season at the ME site (Table 1). Similarly, the highest total macroinvertebrate biomass ($6149.4 \pm 442.8 \text{ g ha}^{-1}$; $F = 3.41$, Tukey's $p < 0.05$) was noted during the wet season at the ME site (Table 1). The variation in the biomass of resident species followed a spatiotemporal pattern. The biomass of *C. arcuatus* was the highest during the dry season at the OE site ($4178.5 \pm 3482.5 \text{ g ha}^{-1}$) and during the transitional season at the RE site ($3827.4 \pm 2957.3 \text{ g ha}^{-1}$; $F = 3.04$, Tukey's $p < 0.05$) (Table 1). The biomass of *S. aculeata aculeata* was the highest during the transitional season at the IE site ($711.3 \pm 281.6 \text{ g ha}^{-1}$; $F = 4.02$, Tukey's $p < 0.05$). The variation in the biomass of migratory species followed a contrasting spatiotemporal pattern: the biomasses of *X. riveti* ($3260.5 \pm 946.0 \text{ g ha}^{-1}$; $F = 129.04$, Tukey's $p < 0.05$) and *R. byrdi* ($561.8 \pm 314.8 \text{ g ha}^{-1}$; $F = 8.74$, Tukey's $p < 0.05$) were the highest during the wet season at the ME site, whereas the biomass of *P. occidentalis* was the highest during the transitional season at the RE site ($130.5 \pm 37.7 \text{ g ha}^{-1}$; $F = 6.17$, Tukey's $p < 0.05$).

In terms of resident species density, the highest density of *C. arcuatus* was recorded during the dry season at the OE site ($281.4 \pm 231.7 \text{ ind ha}^{-1}$; $F = 2.65$, Tukey's $p < 0.05$), and the highest density of *S. aculeata aculeata* was recorded during the transitional season at the IE site ($61.1 \pm 34.9 \text{ ind ha}^{-1}$; $F = 6.76$, Tukey's $p < 0.05$). In terms of migratory species density, the highest densities of *X. riveti* ($714.8 \pm 290.1 \text{ ind ha}^{-1}$; $F = 13.90$, Tukey's $p < 0.05$) and *R. byrdi* ($34.7 \pm 23.2 \text{ ind ha}^{-1}$; $F = 3.4$, Tukey's $p < 0.05$) were recorded during the wet season at the ME site, and the highest density of *P. occidentalis* was recorded during the transitional season at the RE site ($49.9 \pm 6.9 \text{ ind ha}^{-1}$; $F = 44.42$, Tukey's $p < 0.05$) (Table 1).

Spatiotemporal variations in water quality

Regarding the temporal variation in the concentration of nutrients, the concentrations of nitrites and nitrates were the highest during the transitional season ($0.13 \pm 0.03 \text{ mg L}^{-1}$, $F = 211.63$, Tukey's $p < 0.05$ and $2.15 \pm 0.47 \text{ mg L}^{-1}$, $F = 92.78$, Tukey's $p < 0.05$, respectively), but phosphate concentrations did not differ significantly across seasons ($F = 1.94$) (Table 2). Moreover, there were spatial variations in water nutrients; the most internal sites (IE, RE) had higher nutrient concentrations than the most external sites (OE, ME). The highest nitrite concentrations were recorded at the IE ($0.09 \pm 0.06 \text{ mg L}^{-1}$) and RE sites ($0.07 \pm 0.04 \text{ mg L}^{-1}$; $F = 10.83$, Tukey's $p < 0.05$), the highest nitrate concentrations were recorded at the RE ($1.73 \pm 0.65 \text{ mg L}^{-1}$) and IE sites ($1.57 \pm 0.75 \text{ mg L}^{-1}$), and the lowest nitrate concentrations were recorded at the OE ($1.46 \pm 0.18 \text{ mg L}^{-1}$) and ME sites ($1.41 \pm 0.50 \text{ mg L}^{-1}$; $F = 5.16$, Tukey's $p < 0.05$). The highest phosphate concentrations were recorded at the IE ($0.13 \pm 0.04 \text{ mg L}^{-1}$) and RE sites ($0.12 \pm 0.05 \text{ mg L}^{-1}$), whereas the lowest phosphate concentrations were recorded at the OE ($0.09 \pm 0.06 \text{ mg L}^{-1}$) and ME sites ($0.07 \pm 0.01 \text{ mg L}^{-1}$; $F = 5.36$, Tukey's $p < 0.05$) (Table 2). In terms of spatiotemporal variation, the highest nitrite and nitrate concentrations were recorded during the transitional season at the IE site ($0.17 \pm 0.01 \text{ mg L}^{-1}$, $F = 9.23$, Tukey's $p < 0.05$ and $2.56 \pm 0.03 \text{ mg L}^{-1}$, $F = 13.93$, Tukey's $p < 0.05$, respectively). In contrast, the highest phosphate concentrations were recorded during the wet season at the RE site ($0.18 \pm 0.06 \text{ mg L}^{-1}$; $F = 4.01$, Tukey's $p < 0.05$) (Table 2). In addition, the highest alkalinity was recorded during the wet season ($103.89 \pm 17.47 \text{ mg L}^{-1}$), followed by the transitional season ($83.89 \pm 6.00 \text{ mg L}^{-1}$) and the dry season ($81.94 \pm 6.02 \text{ mg L}^{-1}$; $F = 13.74$, Tukey's $p < 0.05$). However, there were no statistical differences among sites in terms of alkalinity or the interaction between site and season.

In terms of DO, the highest concentrations were recorded during the wet season ($6.92 \pm 0.54 \text{ mg L}^{-1}$), followed by the dry season ($5.65 \pm 0.67 \text{ mg L}^{-1}$) and the transitional season ($5.35 \pm 0.48 \text{ mg L}^{-1}$; $F = 115.06$, Tukey's $p < 0.05$). Overall, the highest DO concentrations were recorded during the wet season at the RE ($6.92 \pm 0.29 \text{ mg L}^{-1}$), IE ($7.17 \pm 0.39 \text{ mg L}^{-1}$), and OE sites ($7.18 \pm 0.37 \text{ mg L}^{-1}$; $F = 11.41$, Tukey's $p < 0.05$), whereas the lowest DO concentrations were recorded during the transitional season at the IE ($5.00 \pm 0.14 \text{ mg L}^{-1}$) and OE sites ($4.93 \pm 0.11 \text{ mg L}^{-1}$; $F = 6.18$, Tukey's $p < 0.05$). These results suggest that the water quality was low during the transitional season due to the low concentrations of DO recorded (Table 2).

Within Buenaventura Bay, a spatial salinity gradient was observed, wherein the most internal sites (RE and IE) had lower salinities than the most external sites (OE and ME; $F = 139.51$, Tukey's $p < 0.05$). Moreover, the interaction between seasons and sites showed that the highest salinity occurred during the dry season at the ME site ($25.56 \pm 0.15 \text{ PSU}$; $F = 31.78$, Tukey's $p < 0.05$) (Table 2). A spatial temperature gradient was also observed, with lower

TABLE 1 Species richness, total biomass (g ha⁻¹) wet weight, and density (ind ha⁻¹) of resident macroinvertebrates, density of the most representative migratory macroinvertebrates across seasons, and interaction between season and site (bold, mean ± standard deviation)

Season	Site	Species richness	Total biomass	Resident			Migratory		
				<i>Callinectes arcuatus</i> Density	<i>Squilla aculeata</i> Density	<i>Squilla aculeata</i> Biomass	<i>Xiphopenaeus riveti</i> Density	<i>Rimopenaeus byrdi</i> Density	<i>Penaeus occidentalis</i> Density
Nov 2018 (wet)	RE	1.3 ± 1.5 ^{ab}	1483.8 ± 1614.3 ^{abc}	11.5 ± 11.6 ^a	3.8 ± 6.7 ^a	65.3 ± 113.1 ^{ab}			
	IE	2.7 ± 2.5 ^{ab}	752.5 ± 917.6 ^{ab}	15.2 ± 17.4 ^{ab}	3.8 ± 6.6 ^a	42.1 ± 73.0 ^{ab}		19.0 ± 17.4 ^{ab}	
	OE	1.7 ± 0.6 ^{ab}	2472.6 ± 2043.0 ^{abc}	30.6 ± 23.9 ^{ab}	11.4 ± 19.7 ^{ab}	288.9 ± 500.3 ^{ab}			
Total wet	ME	4.3 ± 0.6 ^b	6149.4 ± 442.8 ^c	23.1 ± 11.6 ^{ab}		1221.0 ± 788.3 ^{ab}	714.8 ± 290.1 ^b	34.7 ± 23.2 ^b	
	RE	2.5 ± 1.8	2714.6 ± 2473.5	20.1 ± 16.3	4.8 ± 10.3	99.1 ± 250.0	178.7 ± 346.1	13.4 ± 19.6	
	IE	2.7 ± 0.6 ^{ab}	3344.3 ± 1316.5 ^{bc}	153.0 ± 8.2 ^{bc}	23.0 ± 11.7 ^{ab}	267.1 ± 250.4 ^{ab}		3.9 ± 6.7	7.7 ± 6.7 ^a
March 2019 (dry)	RE	3.3 ± 1.5 ^{ab}	1091.9 ± 295.6 ^{abc}	27.0 ± 24.1 ^{ab}		281.1 ± 251.4 ^{ab}			
	OE	3.0 ± 2.6 ^{ab}	4320.1 ± 3500.6 ^{bc}	281.4 ± 231.7 ^c	19.1 ± 17.6 ^{ab}	107.5 ± 123.8 ^{ab}			3.8 ± 6.6 ^a
	ME	2.3 ± 1.5 ^{ab}	220.9 ± 153.6 ^{ab}	15.3 ± 6.5 ^{ab}	3.8 ± 6.6 ^a	41.7 ± 72.3 ^{ab}			
Total dry	RE	2.8 ± 1.5	2244.3 ± 2355.2	119.2 ± 150.4	11.5 ± 13.9	1899.3 ± 2402.3	1.0 ± 3.3	2.9 ± 5.2	
	IE	3.0 ± 1.0 ^{ab}	4054.3 ± 2954.0 ^{bc}	69.0 ± 30.3 ^{abc}	3.9 ± 6.7 ^a	27.0 ± 46.8 ^a			49.9 ± 6.9 ^b
	OE	3.3 ± 1.5 ^{ab}	3498.3 ± 2652.3 ^{bc}	56.9 ± 71.0 ^{ab}	61.1 ± 34.9 ^b	711.3 ± 281.6 ^b	3.8 ± 6.6 ^a	3.8 ± 6.6 ^a	
Total transitional	RE	2.7 ± 0.6 ^{ab}	2705.9 ± 1153.7 ^{abc}	64.8 ± 28.7 ^{abc}	7.6 ± 13.2 ^{ab}	2403.3 ± 1158.1 ^{ab}			3.8 ± 6.6 ^a
	ME	0.3 ± 0.6 ^a	15.5 ± 26.8 ^a	3.9 ± 6.7 ^a		15.5 ± 26.8 ^a			
	OE	2.3 ± 1.5	2568.5 ± 2393.4	48.7 ± 44.6	18.1 ± 30.7	200.7 ± 335.3	1.0 ± 3.3	1.0 ± 3.3	13.4 ± 22.4

Note: For each biological variable, the results of pairwise Tukey's post hoc tests ($p \leq 0.05$) are represented with letters. Means were calculated using three replicates, for a total of 36 samples. Abbreviations: IE, inner estuary; ME, marine estuary; OE, outer estuary; RE, river estuary.

TABLE 2 Concentration of nutrients and natural parameters across seasons (bold) and sampling sites (mean ± standard deviation)

Season	Site	Nitrites (mg L ⁻¹)	Nitrates (mg L ⁻¹)	Phosphates (mg L ⁻¹)	Alkalinity (mg L ⁻¹)	Dissolved oxygen (mg L ⁻¹)	Salinity (PSU)	Temperature (°C)	pH	Total dissolved solids (g L ⁻¹)
Nov 2018 (wet)	RE	0.08 ± 0.01 ^{bc}	1.64 ± 0.17 ^{cd}	0.18 ± 0.06 ^b	91.67 ± 12.02	6.92 ± 0.29 ^d	14.53 ± 0.27 ^a	27.97 ± 0.03 ^c	7.47 ± 0.02 ^a	15.63 ± 0.29 ^a
	IE	0.06 ± 0.02 ^{ab}	0.96 ± 0.14 ^a	0.15 ± 0.03 ^{ab}	111.67 ± 7.26	7.17 ± 0.39 ^d	15.96 ± 0.50 ^b	28.13 ± 0.08 ^{cd}	7.78 ± 0.06 ^{cde}	17.02 ± 0.49 ^b
	OE	0.03 ± 0.01 ^a	1.27 ± 0.05 ^{abc}	0.07 ± 0.01 ^a	101.67 ± 23.33	7.18 ± 0.37 ^d	16.16 ± 0.18 ^b	28.21 ± 0.01 ^d	7.89 ± 0.03 ^{ef}	17.08 ± 0.05 ^b
	ME	0.06 ± 0.01 ^{ab}	1.14 ± 0.21 ^{abc}	0.08 ± 0.02 ^a	110.56 ± 23.41	6.43 ± 0.81 ^{bcd}	16.68 ± 0.21 ^b	28.06 ± 0.02 ^{cd}	7.44 ± 0.01 ^a	17.65 ± 0.30 ^b
Total wet	0.05 ± 0.02	1.26 ± 0.29	0.12 ± 0.06	103.89 ± 17.47	6.92 ± 0.54	15.83 ± 0.87	28.09 ± 0.10	7.65 ± 0.20	7.65 ± 0.20	16.84 ± 0.82
March 2019 (dry)	RE	0.03 ± 0.01 ^a	1.04 ± 0.03 ^{ab}	0.12 ± 0.01 ^{ab}	83.89 ± 2.55	5.24 ± 0.09 ^{ab}	20.54 ± 0.24 ^{cde}	27.17 ± 0.07 ^b	7.56 ± 0.01 ^{ab}	21.41 ± 0.26 ^{cd}
	IE	0.04 ± 0.01 ^a	1.18 ± 0.10 ^{abc}	0.15 ± 0.04 ^a	79.44 ± 5.09	5.35 ± 0.24 ^{abc}	21.32 ± 0.39 ^{de}	27.20 ± 0.09 ^b	7.66 ± 0.01 ^{bc}	22.16 ± 0.41 ^{cd}
	OE	0.03 ± 0.01 ^a	1.60 ± 0.17 ^{cd}	0.06 ± 0.02 ^{ab}	76.67 ± 5.77	5.51 ± 0.13 ^{abc}	21.53 ± 0.13 ^e	27.23 ± 0.10 ^b	7.72 ± 0.02 ^{cd}	22.23 ± 0.04 ^d
	ME	0.03 ± 0.01 ^a	1.07 ± 0.10 ^{ab}	0.08 ± 0.01 ^{ab}	87.78 ± 5.09	6.50 ± 0.94 ^{cd}	25.56 ± 0.15 ^g	26.42 ± 0.02 ^a	7.87 ± 0.08 ^{def}	26.05 ± 0.13 ^f
Total dry	0.03 ± 0.01	1.22 ± 0.25	0.10 ± 0.04	81.94 ± 6.02	5.65 ± 0.67	22.24 ± 2.05	27.01 ± 0.36	7.7 ± 0.12	7.7 ± 0.12	22.96 ± 1.91
July 2019 (transitional)	RE	0.11 ± 0.02 ^{cd}	2.52 ± 0.01 ^{ef}	0.08 ± 0.01 ^{ab}	86.11 ± 5.09	5.48 ± 0.30 ^{abc}	21.49 ± 0.87 ^{de}	29.00 ± 0.11 ^f	7.88 ± 0.03 ^{def}	22.26 ± 0.88 ^d
	IE	0.17 ± 0.01 ^e	2.56 ± 0.03 ^f	0.08 ± 0.01 ^{ab}	86.11 ± 8.55	5.00 ± 0.14 ^a	20.24 ± 0.83 ^{cd}	28.83 ± 0.03 ^f	7.76 ± 0.04 ^{cde}	21.13 ± 0.76 ^{cd}
	OE	0.12 ± 0.02 ^d	1.51 ± 0.13 ^{bcd}	0.13 ± 0.08 ^{ab}	82.78 ± 6.94	4.93 ± 0.11 ^a	19.95 ± 0.24 ^c	28.96 ± 0.02 ^f	7.81 ± 0.04 ^{def}	20.91 ± 0.26 ^c
	ME	0.10 ± 0.02 ^{cd}	2.02 ± 0.35 ^{de}	0.07 ± 0.02 ^a	80.56 ± 4.19	6.00 ± 0.17 ^{abcd}	23.01 ± 0.39 ^f	28.47 ± 0.09 ^e	7.96 ± 0.13 ^f	23.73 ± 0.37 ^e
Total transitional	0.13 ± 0.03	2.15 ± 0.47	0.09 ± 0.05	83.89 ± 6.00	5.35 ± 0.48	21.17 ± 1.38	28.82 ± 0.23	7.85 ± 0.10	7.85 ± 0.10	22.01 ± 1.29

Note: Results from Tukey's pairwise comparisons (two-way $p \leq 0.05$) are represented with letters for each water quality parameter, which are read vertically from a to g. Abbreviations: IE, inner estuary; ME, marine estuary; OE, outer estuary; RE, river estuary.

TABLE 3 Canonical correspondence analysis (CCA) of the density of the most representative estuarine macroinvertebrate species and water quality parameters

Total inertia	1.24		
Model p value	0.001		
	CCA1	CCA2	
% of variation explained	40.9	11.0	
% of variation explained (cumulative)	40.9	51.9	
Species–environment correlations	0.93	0.67	
p value	0.001	0.132	
Water quality indicator variables	p value		
Salinity	0.60	−0.30	0.001
pH	0.72	−0.13	0.002
Temperature	−0.05	−0.42	0.002
Phosphates	0.14	0.27	0.011
Nitrates	0.41	−0.48	0.040
Nitrites	0.08	−0.21	0.120
Dissolved oxygen	−0.45	0.30	0.213
Total dissolved solids	0.61	−0.29	0.327
Alkalinity	−0.59	0.20	0.532

Note: Water quality parameters are listed by level of significance. Large and significant contributions of water quality parameters to the axes are indicated in bold.

temperatures recorded at the most external sites (ME) than at the most internal sites ($F=96.05$, Tukey's $p<0.05$). In this study, the highest mean temperature was recorded during the transitional season at the RE site ($29.00 \pm 0.11^\circ\text{C}$; $F=27.10$, Tukey's $p<0.001$) (Table 2).

Moreover, there was a spatial variation in pH, with the most internal sites having lower pH values (RE = 7.63 ± 0.19 , IE = 7.73 ± 0.07) than the most external sites (ME = 7.76 ± 0.25 , OE = 7.81 ± 0.08) ($F=17.47$, Tukey's $p<0.05$). When considering interactions between seasons and sites, the lowest pH was recorded during the wet season at the ME (7.44 ± 0.01) and RE sites (7.47 ± 0.02 ; $F=32.01$, Tukey's $p<0.05$) (Table 2). Finally, when considering the interactions between seasons and sites, TDS concentrations were the highest during the dry season at the ME site ($26.05 \pm 0.13 \text{ g L}^{-1}$) and the lowest during the wet season at the RE site ($15.63 \pm 0.29 \text{ g L}^{-1}$; $F=16.49$, Tukey's $p<0.05$).

Effects of water quality on the density of the most representative macroinvertebrate species

The density of the most representative resident and migratory macroinvertebrate species and water quality parameters were significantly correlated with the first ordination axis ($r=0.93$). The first two ordination axes (CCA1 and CCA2)

explained 51.9% of the variation in macroinvertebrate density associated with water quality (Table 3). The permutation test showed that the relationship between macroinvertebrate distribution and the water quality parameters within Buena-ventura Bay was statistically significant ($F=6.19$, $p<0.001$).

The first ordination axis (CCA1) explained 40.9% of the variance and was strongly and positively correlated with salinity, pH, and TDS; moderately positively correlated with nitrates; and negatively correlated with alkalinity and DO. CCA1 represented a combination of anthropogenic stressors such as nutrient eutrophication and natural stressors such as salinity and pH, which affected macroinvertebrate density. The migratory species *P. occidentalis* was positively correlated with CCA1. However, the assemblage composed of the migratory species *R. byrdi* and *X. riveti* was negatively correlated with CCA1. These results suggest that the macroinvertebrate species that were positively correlated with CCA1 tolerated higher nutrient concentrations, particularly during the transitional season at the RE and OE sites. In contrast, the macroinvertebrate species that were negatively correlated with CCA1 were associated with higher quality habitats, particularly during the wet season at the ME and IE sites (Figure 2 and Table 3).

The second ordination axis (CCA2) explained 11% of the variance and was moderately and negatively correlated with temperature and nitrate concentration. CCA2 represented the effects of high temperature and high nitrate and phosphate concentrations. The mantis shrimp *S. aculeata aculeata* was positively correlated with CCA2, which suggests that this species tolerates low-water-quality environments (Figure 2 and Table 3).

The migratory species assemblage composed of *R. byrdi* and *X. riveti* was associated with habitats with low salinities,

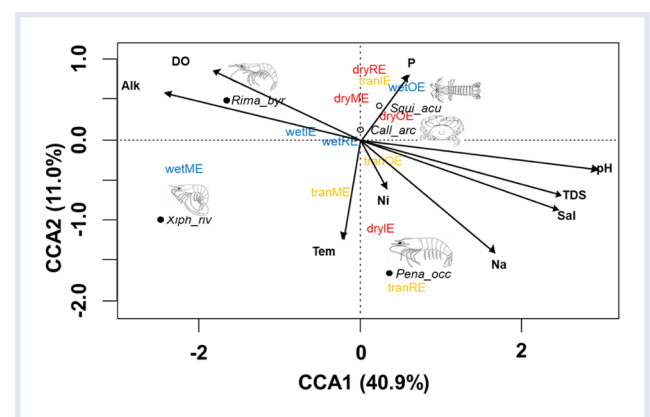


FIGURE 2 Canonical correspondence analysis (CCA) ordination plot illustrating the relationships between the density of the most representative macroinvertebrate species and water quality parameters, seasons, and sampling sites. Resident and migratory macroinvertebrate species are represented by empty and filled circles, respectively. The arrows indicate water quality parameters, abbreviated as follows: Alk, alkalinity; DO, dissolved oxygen; Ni, nitrites; Na, nitrates; P, phosphates; Sal, salinity; TDS, total dissolved solids; Tem, temperature. Season abbreviations: dry, dry season; tran, transitional season; wet, wet season. Sampling site abbreviations: IE, inner estuary; ME, marine estuary; OE, outer estuary; RE, river estuary

pH, and nitrate concentrations and higher DO concentrations. These environmental conditions are indicators of higher water quality and were recorded during the wet season at the ME and IE sites. In contrast, *P. occidentalis* was associated with predominantly marine waters with high nitrate concentrations and low phosphate concentrations. These habitats were also characterized by higher salinities and were recorded during the dry season at the IE site and during the transitional season at the RE site. By comparison, the resident species *S. aculeata aculeata* was associated with high phosphate concentrations during the dry and wet seasons at the OE site. The resident species *C. arcuatus* was located close to the origin of the ordination plot, suggesting that this species was not affected by the water quality gradient reported in this study (Figure 2).

Effects of water quality on macroinvertebrate species richness and total biomass

All measured physicochemical and water quality parameters were included in the multivariate GAM analysis and assessment of GAM interactions. The best model to explain macroinvertebrate species richness variation as a function of water quality included salinity, TDS concentrations, and DO concentrations. This model explained 54.1% of the variance (adjusted $R^2 = 0.35$) (Table 4). Macroinvertebrate species richness was the highest in habitats with higher salinities and lower TDS concentrations (Table 4). Moreover, species richness was nonlinearly related to DO, with higher species richness values being recorded in habitats with DO concentrations between 5 and 5.5 mg L⁻¹ (Figure 3A).

The GAM interactions indicated that macroinvertebrate species richness was affected by the interaction between nitrate concentration and DO concentration; species richness was the lowest in habitats with high nitrate concentrations and high DO concentrations (Figure 4A). Species richness was also affected by the interaction between phosphate concentration and pH; species richness was the lowest in habitats with lower phosphate concentrations and higher pH values (Figure 4B). Lastly, species richness was also affected by the interaction between nitrite concentration and temperature; it was the lowest at sites with high nitrite concentrations and low temperatures (Figure 4C). Species richness was the lowest in habitats with low water quality, characterized by high nitrite and nitrate concentrations.

Macroinvertebrate biomass was better explained by pH and by the concentration of phosphates, TDS, and DO (Table 4). Total macroinvertebrate biomass was negatively correlated with TDS and showed non-linear relationships with pH and phosphate and DO concentrations. Macroinvertebrate biomass was the highest in habitats with lower phosphate concentrations (approximately 0.05 mg L⁻¹, Figure 3B), lower pH values (less than 7.50, Figure 3C), and lower DO concentrations (less than 5.00 mg L⁻¹, Figure 3D). This model explained 73.0% of the variance (adjusted $R^2 = 0.60$) (Table 4).

The biomass of the most abundant resident species, *C. arcuatus*, was primarily explained by DO concentrations. In this study, the range of DO was between 5.00 and

7.5 mg L⁻¹. *C. arcuatus* biomass was the highest in habitats with DO concentrations of around 5.00 mg L⁻¹ (Figure 3E). These results match the trends recorded for total macroinvertebrate biomass, suggesting that *C. arcuatus* is a dominant species of the macroinvertebrate assemblage in habitats with low DO concentrations.

Moreover, the interaction between anthropogenic factors and natural stressors negatively affected *C. arcuatus* biomass. In particular, nitrate and phosphate concentrations had an effect on *C. arcuatus* biomass. The biomass of *C. arcuatus* was the highest in habitats with high nitrate concentrations (between 2.56 and 1.32 mg L⁻¹) and low phosphate concentrations (between 0.09 and 0.04 mg L⁻¹) (Figure 4D). In addition, *C. arcuatus* biomass was affected by the concentration of phosphates and DO and was the highest in habitats with low concentrations of phosphates and DO (between 4.85 and 5.36 mg L⁻¹) (Figure 4E). *C. arcuatus* biomass was also affected by the interaction between phosphate concentration and temperature and was the highest in habitats with low phosphate

TABLE 4 Summary model statistics for the multivariate generalized additive models analysis to assess variations in estuarine macroinvertebrate species richness, total biomass, and *C. arcuatus* biomass

	Species richness	Total biomass	<i>Callinectes arcuatus</i> biomass
N	36	36	36
Adjusted R^2	0.35	0.60	0.31
Deviation explained (%)	54.1	73.0	48.1
AIC	119.5	642.18	542
Coefficient or polynomial grade			
Nitrites	(-0.34)	(-7728.7)	(-8391.1)
Nitrates	—	1.7	(+1166.0)
Phosphates	(+7.44)	2.3**	(-6538.7)
Alkalinity	(+0.04)	—	—
Salinity	(+11.27)**	—	(+5622.5)
Temperature	(+0.19)	—	(-284.3)
Total dissolved solids	(-11.99)**	(-466.8)**	(-6160.5)
pH	—	2.4**	(526.9)
Dissolved oxygen	4.16**	2.27***	1.8**

Note: Number of species (N), model fit (adjusted R^2 and Akaike's information criterion [AIC]), percentage of deviance explained by each variable, and linear coefficient or polynomial grade associated with each variable. The linear coefficient is shown in parentheses and specifies the direction of linear relationships. The polynomial grade is shown without parentheses; — denotes removed during model selection.

* $p < 0.05$.

** $p < 0.01$.

*** $p < 0.001$.

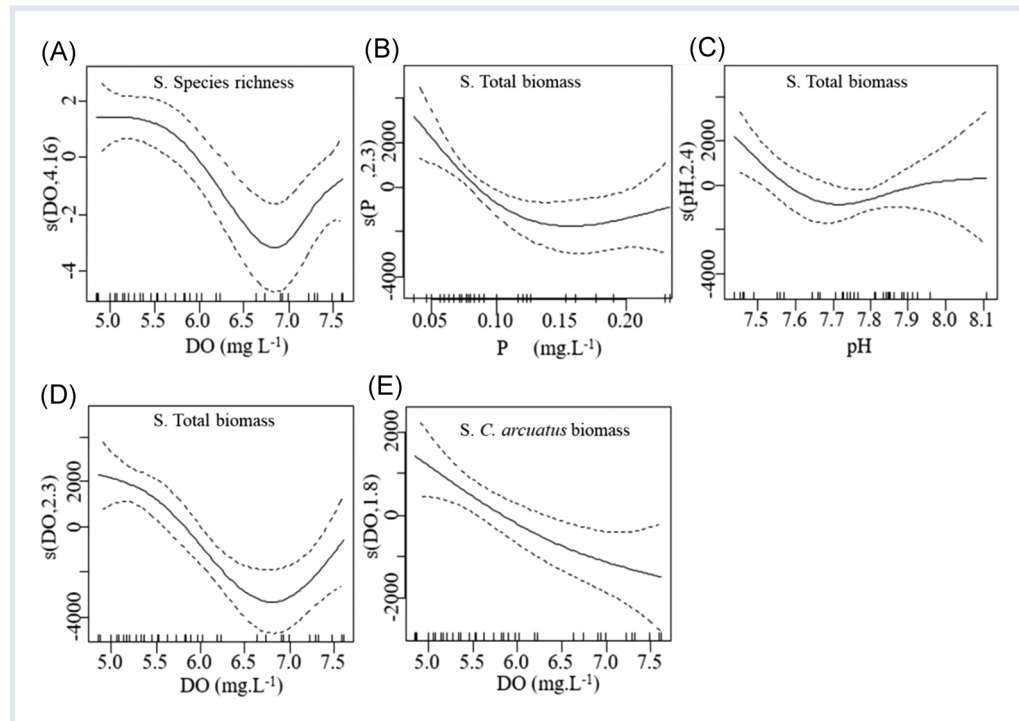


FIGURE 3 Effect of water quality parameters on smoothed macroinvertebrate species richness, total biomass, and swimming crab biomass using generalized additive models (GAMs). (A) Effect of variation in dissolved oxygen (DO) concentration on smoothed species richness (S. Species richness). (B–D) Effect of water quality predictor variation on smoothed total biomass (S. Total biomass). P, phosphate concentration. (E) Effect of variation in DO concentration on smoothed *Callinectes arcuatus* biomass (S. *C. arcuatus* biomass). Plots represent the relationships indicated by the best-fitting GAM (Table 4). Smoothed functions are presented as solid lines; dashed lines denote two standard errors

concentrations and low temperature (Figure 4F). These results suggest that *C. arcuatus* biomass could indicate low-quality habitats, particularly in terms of the concentration of nitrates and DO.

DISCUSSION

This study represents the first assessment of estuarine water that provides physicochemical and macroinvertebrate data in the most inhabited shore area of the Colombian Pacific. The macroinvertebrate species in this research are important because they represent fast and accessible sources of food to humans. According to the results of this research, the abundance and biomass of these organisms were affected by water quality in the estuary. This may suggest that the increase in nutrient discharges can decrease water quality and threaten one of the main sources of food and income for artisanal fishermen and their families.

Nevertheless, estuary ecosystems are subject to strong spatiotemporal changes, which interfere with our ability to differentiate the effect of natural stressors from that of anthropogenic impacts. This challenge is referred to as “the estuarine quality paradox” because of the similarities in the characteristics of organisms and assemblages between relatively unaffected and anthropogenically impacted estuaries (Elliott & Quintino, 2007). Accordingly, the Buenaventura Bay macroinvertebrate assemblage was affected by the convergence of anthropogenic pollution and natural stressors (Abessa et al., 2019; Dauvin & Ruellet, 2009; Elliott &

Quintino, 2007). The most relevant anthropogenic stressors were nitrate and phosphate concentrations, whereas the most relevant natural stressors were salinity, pH, and water temperature. DO was not an indicator of eutrophication, which was a counterintuitive result. This could be due to the fact that DO was highly influenced by water temperature. Furthermore, the photosynthetic activity of the plankton could affect this variable. However, DO was a significant driver of the density of commercial crustacean species.

This suggests that if untreated wastewater discharges increase due to population growth and amplified tourism, the water quality and ecosystem services of Buenaventura Bay might be susceptible to eutrophication. Also, the synergia between natural changes and anthropogenic stressor highlights the complexity of ecological processes in this estuary.

Macroinvertebrate species richness and biomass were the highest during the wet season and the lowest during the transitional and dry seasons at the outer ME site. Although water quality could explain this trend, ontogenetic migration patterns could also be at play. The wet season could coincide with the reproductive season of the titi (*X. riveti*) and tiger (*R. byrdi*) shrimps, which could result in an increase in species richness and biomass in this region (Hernández-Noguera et al., 2016; Mora-Lara, 1973). However, the higher species richness and biomass recorded during the wet season at the outer sites could also be attributed to these sites being farther away from domestic, industrial, and agricultural discharges (Duque et al., 2020). Conversely, during

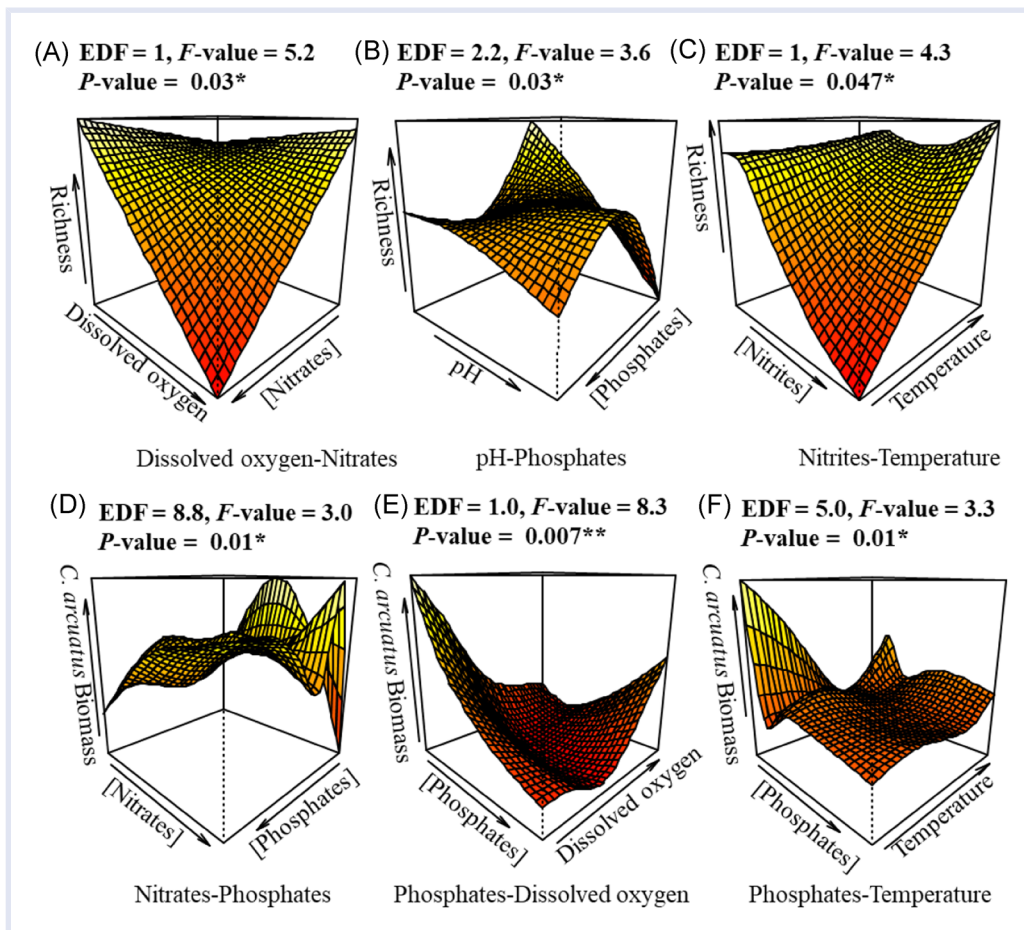


FIGURE 4 Three-dimensional graphs of interacting factors that affect the variation in macroinvertebrate (A–C) species richness and (D–F) *Callinectes arcuatus* biomass. Hypothesis test of the interaction generalized additive models. Green represents the highest richness and *C. arcuatus* biomass values, whereas red represents the lowest values. EDF, estimated degree of freedom

the dry and transitional seasons, the outermost sites were characterized by extreme salinities, which could be tolerated by only a few species (de Oliveira Gomes & Bernardino, 2020). In addition, higher water temperatures could generate hypoxic zones, which could result in assemblages with lower species richness, abundance, and biomass (Jiang et al., 2014).

In contrast, a previous study in Buenaventura Bay reported higher macroinvertebrate species richness and abundance during the dry season at the outermost sites (Gamboa-García, Duque, & Cogua, 2018). However, that study reported higher average monthly rainfall values during the dry and transitional seasons than those recorded in the present study (dry \approx 450 mm and transitional \approx 550 mm compared with 321.2 and 469.2 mm in the present study, respectively) (Gamboa-García, Duque, & Cogua, 2018; Molina et al., 2020).

These results suggest that macroinvertebrate species richness and biomass may have complex intrinsic relationships with abiotic variables in Buenaventura Bay, and future studies should focus on larger spatiotemporal scales to determine how the macroinvertebrate community is affected by climatic events, such as ENSO-El Niño events and

their associated rainfall patterns (Edwards, 2016; Francisco & Netto, 2020).

Water nutrient concentrations showed spatial variations within Buenaventura Bay, with the innermost sites having higher nutrient concentrations than the outermost sites. These results agree with those of previous studies in other human-impacted estuaries, which were characterized by higher phosphate and nitrogen oxide concentrations at the innermost sites, where salinities were lower (Balls et al., 1995; Duque et al., 2020). Moreover, different nutrients followed distinct patterns of temporal variation. Nitrite and nitrate concentrations were the highest during the transitional season at the innermost sites. This could be attributed to the combination of three factors: first, the transitional season (July) coinciding with the beginning of the main tourist season, which could result in an increase in anthropogenic nutrient discharge into the estuary (Herrera et al., 2007; Ospina Niño, 2017), second, the volume of water entering the estuary could decrease during the transition from the dry to the wet season, resulting in an increase in the concentration of nitrites and nitrates (Balls et al., 1995), and finally, the nutrients discharged into the estuary were concentrated because Buenaventura Bay has shown high

residence time, which could be increased until 12 days in some areas, due to a dredging process (García Rentería & González Chirino, 2019). These results highlight the role played by rainfall and residence time of particles in the body of water in diluting nitrite and nitrate concentrations in Buenaventura Bay.

In contrast, phosphate concentrations were the highest during the wet season at the mouth of the Dagua River. The pattern recorded for phosphates was the opposite of that recorded for nitrites and nitrates; a similar pattern has been reported in other estuaries (Balls et al., 1995). Organic matter and fertilizers can be sources of phosphates (Nie et al., 2018), which could get eroded and leached during the wet season. Therefore, there could be an increase in organic matter from mangrove forests and anthropogenic discharges entering river basins (Barletta et al., 2019). These results suggest that the larger river flow during the wet season could contribute to the increase in phosphate concentrations within Buenaventura Bay.

In summary, considering the DO and nutrient concentrations recorded in Buenaventura Bay, a large part of the sampling sites and seasons had moderate-to-low water quality, as proposed by Lemley et al. (2017). In particular, one of the lowest DO concentrations was recorded during the transitional season at the OE site ($4.93 \pm 0.11 \text{ mg L}^{-1}$); phosphate concentrations recorded at the OE site ($0.13 \pm 0.08 \text{ mg L}^{-1}$) also suggested that this site had low water quality (Lemley et al., 2015, 2017). Previous studies have reported that optimal phosphate concentrations during the summer should be below 0.015 mg L^{-1} (Lemley et al., 2015, 2017; Smith, 2003). Nevertheless, phosphate concentrations were higher than that threshold across all sampled sites in Buenaventura Bay, especially during the transitional season at the OE site, when the phosphate concentration was 15.33 times the reference concentration. Moreover, other studies have reported nitrite concentrations of 0.03 mg L^{-1} in highly eutrophic estuaries (Watson et al., 2019). However, 78% of the sampling sites were above this threshold in Buenaventura Bay, and the worst water quality was recorded during the transitional season at the IE site, where nitrite concentrations were six times the threshold value. This excess of inorganic nutrients could inhibit the photosynthetic activity of benthic algae (Admiraal, 1977) and affect the assembly of macroinvertebrates through a bottom-up trophic cascade effect. These results highlight how susceptible the Buenaventura Bay estuary is to eutrophication, which results in low water quality.

Macroinvertebrate species richness and total biomass were affected by water quality in the Buenaventura Bay estuary, which might be explained by the susceptibility of macroinvertebrates to physicochemical variations (Marshall et al., 2008; Proum et al., 2018). Thus, the adaptations of certain macroinvertebrate species have enabled them to occupy specific estuarine niches, which include a wide range of salinities as well as natural stressors and anthropogenic impacts.

The most representative macroinvertebrate species in Buenaventura Bay estuary were classified into three groups

according to the response of each species to the water quality: relatively unaffected species, species associated with low water quality, and species associated with high water quality. The density of the resident and dominant species, *C. arcuatus*, was relatively unaffected by water quality. This species was captured in 81% of the trawls in varying conditions of water quality within the estuary, which suggests a tolerance of this species to a wide range of water quality conditions. Previous studies reported the same pattern for resident species in Buenaventura Bay (Duque et al., 2020; Gamboa-García, Duque, & Cogua, 2018; Molina et al., 2020) and other estuaries around the world (Jiang et al., 2014; Ke et al., 2016; Rezende et al., 2019). These species are characterized as having wide physiological tolerance ranges, enabling them to predominate in these highly variable ecosystems. In particular, *C. arcuatus* is classified as a euryhaline species (Hernández & Arreola-Lizárraga, 2007; Norse & Estevez, 1977), and other studies have reported species from the same genus as being the dominant species in their assemblages (López-Martínez et al., 2014). These results match those reported for Buenaventura Bay, where these species accounted for most of the abundance, especially during the dry and transitional seasons, and were tolerant of both natural and anthropogenic stressors.

Within the group of species affected by water quality, *S. aculeata aculeata* was associated with higher phosphate concentrations. In previous studies in Buenaventura Bay, this species was collected abundantly at inner sites (Gamboa-García, Duque, & Cogua, 2018); however, the associations between their abundance and water quality have not been measured. The high phosphate concentrations within the estuary (Lemley et al., 2015, 2017) suggest that this species could be tolerant of low water quality. These results could represent a baseline to evaluate the monitoring of *S. aculeata aculeata* density as a potential bioindicator of water quality.

Within the group of species associated with high water quality, the shrimp species *X. riveti* and *R. byrdi*, which have high market value, were associated with low nitrate concentrations. Other studies have reported an association between nitrate concentrations of 0.23 mg L^{-1} and larval mortality rates between 31% and 37% for certain species of *Penaeus* (Camargo et al., 2005; Muir et al., 1991). This observation agrees with the results reported in the present study, which suggests that the highest densities recorded for the migratory species *X. riveti* and *T. byrdi* were associated with better water quality, characterized by low nitrate concentrations, salinity, and pH and high DO concentrations, specifically during the wet season at the ME and IE sites. According to these results and additional studies, to protect the estuarine fauna and the livelihood of the artisanal fishermen, one of the goals of the monitoring and control programs of discharge of wastewater could be “keeping low the nitrates levels in these discharges.”

A decrease in species richness might be indicative of low water quality within the estuary, which agrees with previous studies reporting the effects of water nutrients on macroinvertebrate communities (Costa-Dias et al., 2010; Watson

et al., 2019). In Buenaventura Bay, macroinvertebrate species richness was affected by the interaction between nitrite and nitrate concentrations and other factors. The lowest species richness was associated with the highest nitrite, nitrate, and DO concentrations and cooler temperatures, which was confirmed using GAM interaction models. The habitats with the highest nitrate and DO concentrations corresponded with the wet season at the RE site and with the transitional season at the ME site, which matches the results of previous studies in the region (Duque et al., 2020; Gamboa-García, Duque, & Cogua, 2018; Gamboa-García et al., 2020; Molina et al., 2020). The discharge of nutrients and pollutants into the estuary can increase during these seasons at these sampling sites. For example, the RE site is located close to the mouth of the Dagua River and could receive increased erosion and runoff, as well as nitrate and organic matter leaching during the wet season (Nie et al., 2018). Conversely, the ME site is close to a tourist destination (Escobar-Cárdenas, 2009; Herrera et al., 2007; Ospina Niño, 2017) and could be a recipient of increased nitrate discharges during the transitional season (July).

The multivariate GAM confirmed that the lowest macroinvertebrate species richness was associated with low salinity and high TDS concentrations. These results are in agreement with those of previous studies in the region (Gamboa-García, Duque, & Cogua, 2018) and of studies from other tropical estuaries, where distinct macroinvertebrate assemblages were formed depending on the habitat salinity (Hossain et al., 2019). Other researchers have reported that TDS concentrations above 32 g L^{-1} could cause 50% mortality in certain shrimp species (Wilber & Clarke, 2001) and that extremely high TDS concentrations could be barely tolerable for species inhabiting sandy bottoms (McFarland & Peddicord, 1980). Finally, the results obtained in the present study agree with those of previous studies reporting a decrease in the abundance of macrozoobenthos species in habitats with higher DO concentrations (Yan et al., 2019), which could be a result of an increased number of predators (i.e., demersal fish). An increase in DO concentration and an increase in the *Cathorops* spp. resident catfish population has been reported in Buenaventura Bay during the wet season (Duque et al., 2020; Molina et al., 2020); catfish are benthophagous and could thus have an effect on macroinvertebrate populations.

In this study, the biomass of certain dominant macroinvertebrates was unexpectedly large in low-water-quality habitats. However, the tolerance of these macroinvertebrates to poor water quality might enable them to evade predators, such as fish, that avoid low-quality habitats (Yan et al., 2019). However, it could also be due to the dominance and proliferation of tolerant species to low water quality, which leads to an increase in population due to the increase in usable resources left by nontolerant species. Accordingly, some studies have reported that an increase in macroinvertebrate biomass could be an indicator of low water quality within estuaries (Currie et al., 2011; Elliott & Quintino, 2007; Jiang et al., 2014; Villnäs et al., 2019).

Conversely, other researchers have reported a decrease in macroinvertebrate biomass in low-water-quality habitats, particularly hypoxic habitats (Ren et al., 2019). These observations highlight the complexity of using macroinvertebrate biomass as an indicator of water quality and that biomass may depend on the most relevant natural and anthropogenic stressors within each ecosystem at a regional scale (I. Martins et al., 2019).

The *C. arcuatus* biomass could be used as an indicator of water quality in Buenaventura Bay because it was associated with habitats with low DO and high nitrate concentrations. Experiments conducted on other *Callinectes* species showed that adults could withstand low DO concentrations (Eggleston et al., 2005). Moreover, certain species have physiological adaptations to regulate hemolymph pH under low oxygen and hypercapnic conditions, which allows them to increase their movements when foraging and avoiding predators (Lehtonen & Burnett, 2016; Stover et al., 2013). These adaptations could explain an increase in the biomass of these organisms in lower-quality habitats. Regarding the tolerance of *C. sapidus* to nitrite and nitrate concentrations, toxicity bioassays showed that the LC50 of this species was 73 mg L^{-1} for nitrites and that concentrations of 10 mg L^{-1} resulted in 4.5% mortality (Ary & Poirrier, 1989). Moreover, an efficiency of 9.6% in the removal of nitrates in the treatment of hospital wastewaters using *C. sapidus* has been reported (Foroutan et al., 2019). This suggests that this species is highly tolerant of nitrates, although, within Buenaventura Bay, nitrite and nitrate concentrations did not exceed 0.18 and 2.58 mg L^{-1} , respectively. However, a positive correlation between nitrogen isotopes and water nitrate concentrations has been reported for *C. sapidus*, which suggests that this species forages in habitats with high nitrate concentrations (Bucci et al., 2007). Hence, monitoring of *C. arcuatus* biomass could be useful as an indicator of water quality in Buenaventura Bay and could aid in diagnosing various scenarios of nutrient pollution. On the contrary, the results of this study highlight how vulnerable the macroinvertebrate community is to environmental change, which could result in pH reduction due to ocean acidification and increased nutrient leaching, ultimately affecting DO concentrations in estuaries (Hossain et al., 2019; Marshall et al., 2008; Proum et al., 2018).

CONCLUSIONS

On monitoring of the environmental variables in this estuary, a decrease in water quality due to nutrient levels was observed. The entry of nutrients into the estuary could be related to the emission of industrial and domestic effluents, and activities related to the port and tourism. Monitoring is of global interest to evaluate anthropogenic impacts on water quality and their influence on food resources such as the resident and migratory shrimps and swimming crabs.

The density of commercial macroinvertebrates was influenced by anthropogenic nutrient inputs and natural factors. The highest density of *X. riveti* and *R. byrdi* was found in waters with low concentrations of nitrates, pH,

and salinity, and high alkalinity and DO. These species are the main capture targets for food and the livelihood of artisanal fishermen and their families. Thus, nitrate concentration was highlighted as an important anthropogenic disturbance factor in the set of environmental variables that influence artisanal fishery catches. From a socio-economic point of view, the decrease in water quality due to high nitrates could cause a decrease in fishing productivity. Therefore, one of the main goals of management could be to regulate the nitrate levels in the anthropogenic discharges.

Additionally, the higher density of *S. aculeata aculeata* was mainly associated with high concentrations of phosphates. On the contrary, the lowest species richness was associated with the highest nitrite, nitrate, and DO concentrations and cooler temperatures. The biomass of dominant *C. arcuatus* was unexpectedly large in low-water-quality habitats, with low DO and high nitrate concentrations, which may be a response from highly disturbed ecosystems.

The high dominance in the macroinvertebrate assemblage combined with the low water quality in the estuary suggests that anthropogenic impacts associated with coastal cities and harbor activities are present. The consequences of poor water quality on estuarine macroinvertebrate health and ecosystem services have not been determined in this area, and this topic should be addressed in future studies. It is clear, nonetheless, that macroinvertebrates are useful as indicators of estuarine water quality and that a subset of the macroinvertebrate species and community indicators used in this study could be included in estuarine monitoring.

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CONFLICT OF INTEREST

The authors declare that there are no conflicts of interest.

DATA AVAILABILITY STATEMENT

The data in this study are available upon request from corresponding author Guillermo Duque (gduquen@unal.edu.co).

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