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**Seasonal structure of the fish community associated to
artificial reefs and natural bottoms in Ojo de Liebre lagoon,
B.C.S., Mexico**

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Presenta:

Jaime Perrin López

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Tesis defendida por
Jaime Perrin López

y aprobada por el siguiente Comité

Dr. Jorge Adrián Rosales Casián
Director de tesis

Dr. Rafael Andrés Cabral Tena

Dra. Patricia Juárez Camacho



Dr. Jorge Adrián Rosales Casián
Coordinador del Posgrado en Ecología Marina

Dra. Rufina Hernández Martínez
Directora de Estudios de Posgrado

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Abstract of the thesis presented by **Jaime Perrin López** as a partial requirement to obtain the Master of Science degree in Marine Ecology.

Seasonal structure of the fish community associated to artificial reefs and natural bottoms in Ojo de Liebre lagoon, B.C.S., Mexico

Abstract approved by:

Dr. Jorge Adrián Rosales Casián
Thesis Director

Artificial reefs (AR) have played an important role in fisheries for centuries, be it to enhance habitat productivity in low productivity areas, to create an aggregation of fishes, to increase fisheries yields, or to facilitate the location of fish. Starting in January 2018, artificial reefs (AR) were installed at Ojo de Liebre lagoon, B.C.S., Mexico, to increase lobster shelter and boost in season captures. A comparison of the fish communities at the site with the AR, a soft bottom (SB) site, a natural rocky reef (RR) site, and a site on the main channel (MC) near the mouth of lagoon were conducted using two monitoring methods, trap capture and visual observations of transects with SCUBA. There were 25 species identified belonging to 18 families and 9 orders, with *Labrisomus xanti* being recorded inside the lagoon for the first time. Seasonally, there were no differences in abundance and biomass between seasons in trap capture data, while with the SCUBA monitoring spring 2018 had significantly higher abundance than autumn 2018 and winter 2019, and biomass was significantly higher in spring 2018 than winter 2019. Between sites, the MC site had significantly higher abundance and biomass than all other sites with the trap capture data. With the SCUBA observations, there were no significant differences in abundance and biomass between the AR site, the RR site, and the SB site. *Paralabrax maculatofasciatus* was the species of highest importance during all seasons using the index of community importance (ICI) and the index of relative importance (IRI) the same species has the highest value in the majority of seasons except summer and autumn of 2018 when *Paralabrax nebulifer* has the highest values of IRI. This differs from a study by Cívico Collados (2016), which found the species of most importance in ICI and IRI as *Calamus brachysomus*. Differing environmental conditions, such as an El Niño Southern Oscillation event during Cívico Collados' study could have contributed to the differences. Spearman's rank correlation determined that the trap capture abundance and biomass was negatively correlated to seawater temperature while with visual observations using SCUBA, there was a positive correlation of seawater temperature with abundance and biomass as well as positive correlation of salinity with biomass and abundance; a negative correlation of abundance with dissolved oxygen was found using SCUBA observations. SHE analysis of the observations with SCUBA determined there were different communities between the AR site and RR site from spring 2018 onward, as well as different seasonal communities between winter 2018, summer 2018, and winter 2019. The RR community had higher species richness (S), diversity index (H) and Buzas-Gibson's evenness (E) than the AR community, while seasonally the summer community had higher S and H than the winter communities, yet lower E. Shannon-Weiner's diversity index (H') for the trap capture data was highest in spring and autumn of 2018 and lowest in summer 2018, while the SB site presented the highest index and the AR site the lowest.

Keywords: Artificial reefs, fish communities, Ojo de Liebre, SHE analysis

Resumen de la tesis que presenta **Jaime Perrin López** como requisito parcial para la obtención del grado de Maestro en Ciencias en Ecología Marina.

Estacionalidad de la estructura de la comunidad de peces asociada a arrecifes artificiales y fondos naturales en laguna Ojo de Liebre, B.C.S., México

Resumen aprobado por:

Dr. Jorge Adrián Rosales Casián
Director de tesis

Los arrecifes artificiales (AR) han desempeñado un papel importante en la pesca durante siglos, ya sea para mejorar la productividad del hábitat en áreas de baja productividad, para crear una agregación de peces, para aumentar el rendimiento de pesquerías o para facilitar la ubicación de peces. A partir de enero de 2018, se instalaron arrecifes artificiales (AR) en la laguna Ojo de Liebre, B.C.S., México, para incrementar el refugio de langosta y aumentar la captura durante la temporada. Se realizó una comparación de las comunidades de peces entre el sitio con AR, un sitio de fondo blando (SB), un sitio de arrecife rocoso (RR) y un sitio en el canal principal (MC) cercano a la boca de la laguna utilizando dos métodos de monitoreo, captura en trampas y observaciones visuales de transectos con buceo. Se identificaron 25 especies pertenecientes a 18 familias y 9 órdenes, y se registró a *Labrisomus xanti* dentro de la laguna por primera vez. Estacionalmente, no hubo diferencias en la abundancia y biomasa entre estaciones con los datos de captura en trampas, mientras que en el monitoreo por buceo, la primavera de 2018 tuvo una abundancia significativamente mayor que en el otoño de 2018 y el invierno de 2019 y la biomasa fue significativamente mayor en la primavera de 2018 que en el invierno de 2019. Entre sitios, el sitio MC tuvo una abundancia y biomasa significativamente mayor que todos los otros sitios usando los datos de captura con trampas. Con las observaciones de buceo, no hubo diferencias significativas en la abundancia y la biomasa entre el sitio con AR, el sitio con RR y el sitio con SB. *Paralabrax maculatofasciatus* fue la especie de mayor importancia durante todas las estaciones utilizando el índice de importancia de la comunidad (ICI) y el índice de importancia relativa (IRI), la misma especie tiene el valor más alto en la mayoría de las estaciones, excepto en el verano y el otoño de 2018, cuando aparece *Paralabrax nebulifer* con los valores más altos de IRI. Esto difiere del estudio realizado por Cívico Collados (2016) que obtuvo que *Calamus brachysomus* fue la especie con los valores más altos de ICI e IRI. Las condiciones ambientales diferentes, como un evento de Oscilación del Sur de El Niño durante el estudio de Cívico Collados, podría haber contribuido a las diferencias encontradas. Por medio de la correlación de Spearman se determinó que la abundancia y biomasa de las capturas en trampa se correlacionó negativamente con la temperatura del agua, mientras que con las observaciones visuales de buceo, hubo una correlación positiva de la temperatura del agua con la abundancia y la biomasa, así como una correlación positiva de la salinidad con biomasa y abundancia; se encontró una correlación negativa de la abundancia con oxígeno disuelto utilizando observaciones de buceo. El análisis SHE de las observaciones con buceo determinó que había diferentes comunidades entre el sitio con AR y el sitio con RR desde la primavera de 2018 en adelante, así como diferentes comunidades estacionales entre el invierno de 2018, el verano de 2018 y el invierno de 2019. La comunidad de RR presentó una mayor riqueza de especies (S), índice de diversidad (H) y equidad de Buzas-Gibson (E) que la comunidad de AR, mientras que estacionalmente la comunidad de verano tuvo

S y H más altas que las comunidades de invierno, pero más baja E. El índice de diversidad de Shannon-Weiner (H') para los datos de captura de trampa fue más alto en primavera y otoño de 2018 y más bajo en verano de 2018, mientras que el sitio con SB presentó el más alto índice de diversidad y el sitio con AR el más bajo.

Dedication

To my parents, for always being in my corner, no matter the difficulties. This work was not possible without your support.

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Chapter 1. Introduction

Artificial reefs (AR) have played an important role in fisheries for centuries, be it to enhance habitat productivity in low productivity areas, to create an aggregation of fishes, to increase fisheries yields, or to facilitate the location of fish (NOAA, 2007). However, there is no consensus into what exactly is the function of AR, whether it be to increase fish production or biomass, or to create an area for the congregation of disperse fish (Bohnsack, 1989; Bohnsack and Sutherland, 1985; Solonsky, 1985). Using Japan as his leading example due to the large-scale use of artificial reefs, Polovina (1989) states that no noticeable increase in fisheries stock has been noted that can evidence AR as more than just fish aggregators. Considering this view and doing a review on various work on AR, Bohnsack (1989) proposed that each artificial reef falls within a gradient between production and aggregation, on what spectrum of this gradient depends on local physical and biological factors. More recently, it has been shown that reef design and management strategies contribute to the functioning of AR as either aggregating devices or to increase production, and there have been more cases in which artificial reefs have been used to enhance stock by increasing recruitment of fertilized eggs onto artificial structures (Kim *et al.*, 2011; Pickering and Whitmarsh, 1997).

AR have been successful in recruiting lobsters and have been deployed at various locations across the globe in order to increase lobster production (Bortone *et al.*, 2011; Jensen *et al.*, 1994). In Mexico, the lobster fishery is considered of high value, and licenses for a number of boats and traps are given to individual fishing cooperative societies, who then, in turn, manage their quotas (Vega Velázquez *et al.*, 1996). In order to increase catch, the report by Vega Velázquez *et al.* (1996) proposes the use of AR over sandy bottoms to increase both the lobster's refuge and catch. Baja California Sur catches about 50% of the total lobster fishery in Mexico, and Ojo de Liebre lagoon (OLL) is home to a fishing cooperative society (Luis Gómez Z.) allowed to fish for the California Spiny lobsters (*Panulirus interruptus*). In order to increase their catch capture in OLL (Figure 1), and given the suggestions of the report by Vega Velázquez *et al.* (1996) of using AR to increase lobster catch, the fishing cooperative society Luis Gómez Z. decided to deploy AR (Figure 2) in OLL in January 2018.

OLL is a coastal lagoon, these lagoons are, in general, shallow bodies of water with depths between one and five meters, exceptions being the channels, which can be deeper (Kjerfve, 1994). Due to coastal lagoons having high rates of primary production because they are filters and sinks of sediment and organic matter they are essential to the life cycles of various fish and other valuable fisheries species (Knoppers *et*

al., 1991; Robbins, 2006; Yáñez-Arancibia *et al.*, 1994). Because of the reasons mentioned above, fisheries yield in coastal lagoons are high and require management that not only considers the biological aspects of the fisheries but also the socio-economic factors of the region (Kapetsky, 1981).

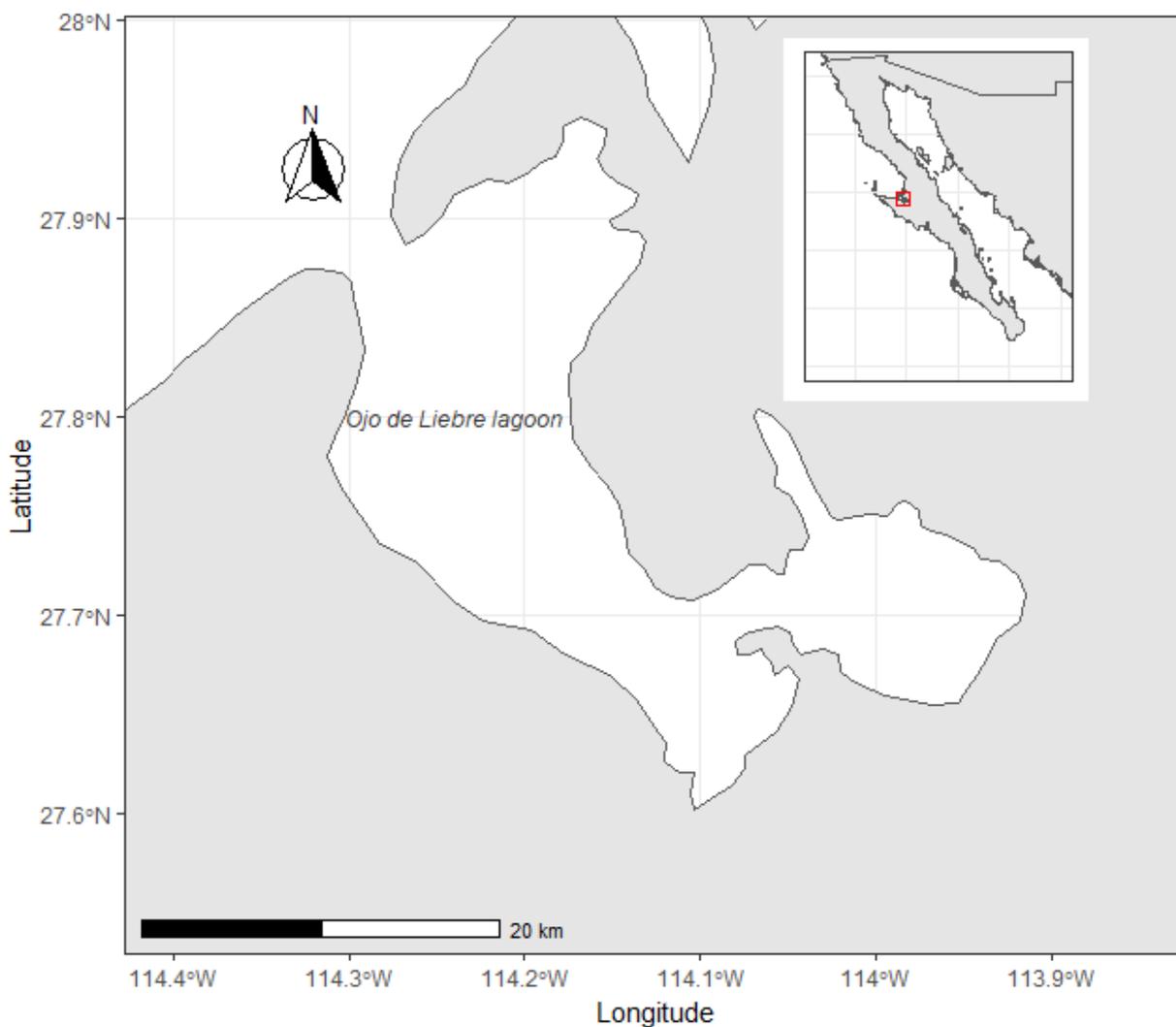


Figure 1. Ojo de Liebre lagoon. Location of Ojo de Liebre (Scammon's) lagoon in the central-western Baja California peninsula, Mexico (latitudes 27.6 N and 28.0 N, longitudes 114.4 W and 113.9 W).

The present work was done in OLL, South Baja California, which is one of 22 lagoon systems that exist on the Baja California peninsula, where most studies have been directed towards aquaculture (Acevedo Cervantes, 1997; Contreras, 1985). As mentioned previously, coastal lagoons have high productivity and function as havens and nurseries to various species; due to this, its conservation and management are of primary importance. This lagoon was declared a World Heritage site by the UNESCO's Man and Biosphere (MAB) program (De La Cruz-Agüero *et al.*, 1996). OLL is within a natural protected area called El Vizcaíno

Biosphere Reserve whose laws are outlined in Mexico's General Law of Ecological Equilibrium and Environmental Protection and whose management plans are detailed in the biosphere reserve program (Diario Oficial de la Federación, 2015; Programa de Manejo Reserva de la Biosfera El Vizcaíno, 2000). These management plans are very restrictive in terms of resource usage allowed within the reserve, which leaves the local human population in Guerrero Negro with few activities they are allowed to carry out. Because of this and taking advantage of the AR to increase lobster production, the aim of this study is to assess the fish communities gathered around the AR and comparing it to other natural sites around OLL to assess its use for the different fish fisheries other than lobster production. It is important to compare AR to natural rocky reefs because of the similarities in their structural complexities; natural rocky reefs have some of the highest levels of diversity, abundance, and biomass in the habitats of the northeastern Pacific (L. G. Allen, 1985; Carr and Hixon, 1997; Perkol-Finkel *et al.*, 2006).

In addition, we seek to identify any relationship between three environmental variables (dissolved oxygen, salinity, and seawater temperature) and fish abundance and biomass. These environmental variables were chosen due to the annual variability of seawater temperature and the vast change in salinity that can occur in coastal lagoons. Dissolved oxygen was chosen because of its importance to respiration.

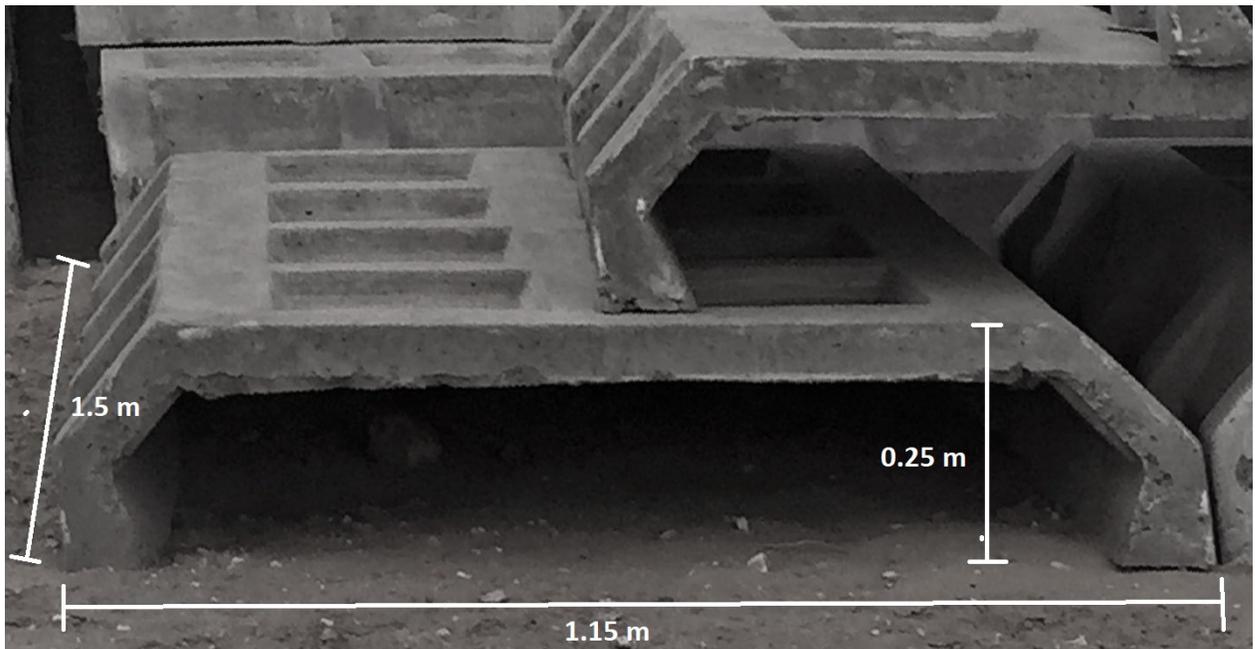


Figure 2. Artificial reefs. Artificial reef units made of Portland cement installed to increase the shelter of California Spiny lobster (*Panulirus interruptus*) at various locations of Ojo de Liebre lagoon.

1.1 Background

In OLL, there are various studies about its environmental conditions, such as seasonal water temperature variations, density, visibility and depth as they relate to the movement of grey whales, *Eschrichtius robustus* (Alvarado B. *et al.*, 1986; Alvarez Borrego and Granados Guzman, 1992); studies on littoral sediment transport (L. Marinone and Lizárraga Arciniega, 1982); the population and incident mortality of grey whale (Fuentes, 1983); about important local resources such as bivalves (Arellano-Martínez *et al.*, 2004; Hernández-Olalde *et al.*, 2007; Quiñones Arreola, 2003); and about the state of the population of the green sea turtle *Chelonia mydas* (Hernández Cruz, 2013); yet very few studies on the fish communities of the lagoon.

The few studies that have been done about the fish community at OLL are De La Cruz-Agüero *et al.* (1996) who published a marine fish species list of Ojo de Liebre and Guerrero Negro lagoons in Baja California Sur, identifying a total of 59 fish species belonging to 50 genera and 36 families. Acevedo Cervantes (1997) who sampled the lagoon with gill nets, otter trawl nets, and beach seine nets bimonthly from January to November 1995 concluded that OLL has low fish diversity with fish species of small size and that the most substantial diversity is found near the mouth of the lagoon due to the seasonal transit of various species. Fish trophic interactions within the lagoon were studied by Bocanegra Castillo (1998) by analyzing stomach content of the most abundant fish species and found that the six most abundant species are specialist predators that feed on benthic organisms, and due to the availability of such prey (mollusk, polychaetes, crustaceans, and some fish) interspecific competition for prey is low. Most recently, a master's thesis was written by Cívico Collados (2016), which identified 27 fish species from 19 families by sampling with two methods, traps and visual monitoring using SCUBA, at various sites (potential sites for deployment of AR) along OLL; she also found that *Paralabrax nebulifer* and *Calamus brachysomus* were the two most important species using the index of community importance and index of relative importance; peak abundance and biomass were in spring while lowest values of these were in summer. In Cívico Collados (2016) study, abundance and biomass were both positively correlated with temperature with both methods of sampling, traps and SCUBA monitoring.

In a study sampling a site with AR, a natural rocky reef site, and a sandy-mud bottom site off the coast of Japan with bottom trammel nets, Fujita *et al.* (1996) found that the most abundant species at the AR and natural reef sites were the same, and this differed from the sandy-mud bottom site. However, they also found that even though species richness was similar at all three sites, at the AR reef site, equitability, and thus diversity was lower. Granneman and Steele (2015) did a similar study off the coast of California using

visual underwater transect with SCUBA comparing fish assemblages between natural reefs and AR, they found that species richness was similar between AR and natural reefs yet biomass and density was higher at the AR sites even though fish had slightly smaller body sizes.

A synthesis of various studies of habitats in California by L. G. Allen (1985) found that inshore habitats like bays and estuaries and nearshore sandy bottoms had some of the lowest diversities and species richness in the California coast, while open coast natural rocky reefs, kelp beds, and deep sandy bottom habitats had the highest diversities and species richness. OLL is an inshore body of water, yet it contains mainly shallow sandy bottoms and some natural rocky reefs.

1.2 Justification

OLL is part of an important ecosystem for many species of migratory and resident species, which use the area for nursery, feeding, and shelter (Fuentes, 1983; Programa de Manejo Reserva de la Biosfera El Vizcaíno, 2000). Recent evidence of this area as an important white shark nursery and migratory corridor has added more value to the area as critical habitat (Oñate-González *et al.*, 2017). The main town inside El Vizcaino Biosphere Reserve is Guerrero Negro, B.C.S. in which the inhabitants depend economically on OLL (INEGI, 2010); the livelihood of the majority of the population is either in salt extraction, fishing, and ecotourism, and since they are within a natural protected area under the category of biosphere reserve it is challenging to diversify the economic activities (Kachok *et al.*, 2012; Oñate-González *et al.*, 2017). Due to these reasons, the placement of AR modules inside the lagoon to increase recruitment of the California Spiny Lobster by increasing the availability of shelter; the AR should also increase other fisheries, therefore creating a benefit for the community that commercially exploits the lagoon. Also, the data collected in these habitats should help to better understand the relationship between fish communities, environmental factors, and artificial structures.

1.3 Hypothesis

Main Hypothesis

The artificial reef site (AR: Conchalito) will have equal abundance, biomass, and diversity parameters (Shannon-Wiener diversity index, Buzas-Gibson's Evenness, and species richness) as the rocky reef site (RR: El Borbollon), which will all be higher than the soft bottom site (SB: La Ventana). Seasonally, spring will have the highest values for abundance, biomass, and diversity. There will be differences in all these parameters seasonally.

Secondary Hypothesis

1. The main channel site near the mouth (MC: Carros Viejos) will have the highest abundance and biomass from all sites.
2. Salinity will increase from the Carros Viejos (MC) site (near the inlet) towards all other sites, with
3. *Calamus brachysomus* and *Paralabrax nebulifer* will be the two species with the highest level of importance based on two indexes: the index of community importance (ICI) and the index of relative importance (IRI).
4. The seawater temperature will be positively correlated with abundance and biomass with both survey methods, as was found in the previous study by Cívico Collados (2016).

1.4 Objectives

1.4.1 Main objective

To determine the seasonal variation of the structure of the fish community at different sites (AR, RR, SB, MC), and compare conclusions with previous studies at OLL.

1.4.2. Specific objectives

1. Determine the seasonal variation of seawater temperature, salinity, and dissolved oxygen in the sites being monitored at OLL.
2. Identify the fish species at the different sites.
3. Determine the seasonal variation of abundance and biomass at the different sites in OLL. Determine the variation between sites of abundance and biomass.
4. Determine the relationship between the seasonal variation in fish abundance and biomass with seawater temperature, salinity, and dissolved oxygen.
5. Determine the seasonal and between site variations in diversity, evenness, and species richness.
6. Identify any changes in community structure between sites and seasons.
7. Compare the conclusion of this study with the conclusion of previous studies at OLL.

Chapter 2. Methodology

2.1 Study area

OLL is found inside an alluvial plain due to the movement of sediment originally from the Sierra San Borja in the northeast and the Sierra Vizcaino in the south (Eberhardt, 1966). Guerrero Negro, Baja California Sur, is the nearest human settlement to the lagoon complex; OLL is connected to Sebastián Vizcaino Bay by a 4 km wide mouth; it is found between latitudes N 27°59' and N 27°54' and longitudes W 113°55' and W 114°31'; the lagoon has a surface area of 360 km², a length of 40 km and an average width of 6 km (Hernández Cruz, 2013). The tide heights vary between 1 and 3.5 m and have a delay of three hours with the open ocean; inside the lagoon, there is a channel system with an area of 171 km² that have a depth range of 3 to 15 m (Sánchez-Pacheco, 1991). The sediment in the lagoon is mostly fine, with mud and clays in the interior plains of the lagoon, and medium sand in the channels near the mouth; the lagoon has three islands, Isla Piedra in the narrowest part, Isla Brosas further south, and Isla Conchas 6 km from the mouth (Phleger and Ewing, 1962). The presence of vegetation in the lagoon plains acts as a sediment trap, in its shallow areas is *Salicornia*, below that there are areas of seagrass of *Spartina*, and in the deepest channels, there is *Zostera* (Phleger and Ewing, 1962).

The region presents an average annual seawater temperature between 18 and 22 °C with a dry-semiarid climate with intermediate rainfall regime (Salinas Zavala *et al.*, 1990). Variations in the lagoon's salinity and seawater temperature have been related to tidal oscillations; during August, salinity values of 34 have been recorded near the mouth and up to 47 PSU within the lagoon (Phleger and Ewing, 1962). Current speeds of 2.5 knots were recorded in the mouth of the lagoon, which is twice the amount compared to the inner area, with the maximum velocities occurring when the tide rises or falls and the lowest velocities being in the crest and trough of the tidal waves (Phleger and Ewing, 1962).

2.2 Fieldwork

The present study was based on the methodology used by Cívico Collados (2016) and counted with the logistical support of the fishing cooperative society S.C.P.P. Luis Gómez Z. de R.L. of C.V., who provided two boats two crew members per boat, two engines, gasoline, and four fish traps. Four sites within the

lagoon were surveyed during this study (Figure 3, Table 1). Three sites were surveyed via SCUBA census, and all four sites were monitored with trap capture. Five sampling days were conducted: winter 2018 (March 8), spring 2018 (June 8), summer 2018 (August 18), autumn 2018 (November 15), and winter 2019 (February 13).

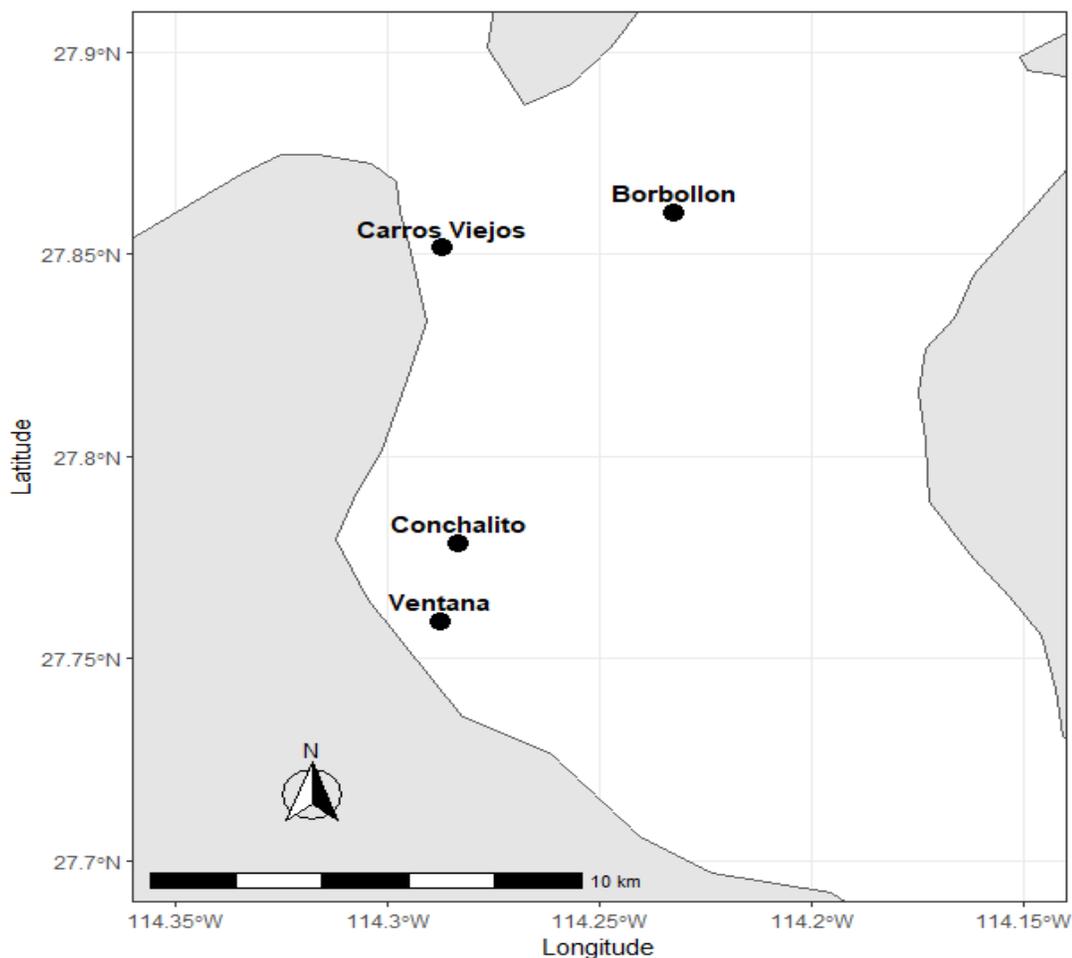


Figure 3. Study sites. Study sites at Ojo de Liebre lagoon, Baja California Sur (Mexico). El Conchalito: artificial reefs, El Borbollon: natural rocky reef, La Ventana: Sandy bottom, Carros Viejos: mouth and main channel.

The sites are (Table 1):

Borbollon is the site with the Natural rocky reef (El Borbollon) (RR). The RR is made of an extension of stones that look like an underwater mountain range that could be over 100m in length and about 10 m wide, and extend in the water column about 2-3 m. The average depth of the site is between 6-12 m.

Conchalito is the site with the artificial reefs (AR). This site has a sandy bottom with the occasional sedimentary rocks protruding but is mostly just clay, sand, and shellfish fragments. The AR are deployed with clusters of two or three in a radius of around 20 m. The AR are somewhat scarce. The average depth of the site is between 4-8 m. The AR deployed (Figure 2) are made of Portland cement and are shaped in the form of an arc, with the arc width being 1.15 m and standing at 0.25 m in height. The average thickness of the cement is around 0.07 m. The length of the units is 1.5 m and have 16 0.03 m deep 0.2 x 0.25 m rectangular cavities on the exterior part of the AR units.

Ventana is the soft bottom (SB) site. The sediment is similar if not equal to the AR site, just without the AR. This location is a control site for the AR. Its depth range is between 4-12 m due to its closeness to the lagoon's main channel.

Carros Viejos, which is near the inlet, and in a deep section of the lagoon's main channel (MC). This site was the only site not surveyed with SCUBA due to the dangers of the currents there. This site was chosen to compare the importance of the inlet with the other sites, as mentioned by Acevedo Cervantes (1997). The site has a depth between 10-20 m.

2.2.1 Physical and chemical parameters

The salinity, seawater temperature ($^{\circ}\text{C}$), and dissolved oxygen (mg L^{-1}) of the lagoon were measured at the sites where traps were deployed during each seasonal monitoring using the PRO2030 YSI multi-analyzer. The measurement was done at the surface of the lagoon (<1 m) and at depth (4-5 m).

2.2.2 Fish identification

Fish species were identified with keys for the coasts of Baja California, Mexico, and California, USA (D. J. Miller and Lea, 1972), and with the Peterson Field Guide for Pacific Coast Fish (Eschmeyer *et al.*, 1983). For fish that are found in tropical waters such as the Sea of Cortez, a guide to Sea of Cortez animals (Gotshall, 1998) was used as well as the guide Reef fish identification: tropical Pacific (G. R. Allen *et al.*, 2003).

Table 1. Latitude and longitude of the study sites. Names of the study sites at Ojo de Liebre lagoon, Baja California Sur, Mexico, with their coordinates, substrate types, and their in-text reference

SITE NAME	LATITUDE	LONGITUDE	SUBSTRATES	REFERRED AS:
BORBOLLON	27.8603167°	-114.2322167°	Rocky reef/sand	Rocky reef (RR)
CONCHALITO	27.7784833°	-114.2830833°	Artificial reef/sand/sedimentary rock	Artificial reef (AR)
VENTANA	27.7592667°	-114.2873500°	Sand/ sedimentary rock	Soft bottom (SB)
CARROS VIEJOS	27.8515667°	-114.2871833°		Main channel entrance (MC)

2.2.3 Visual census by SCUBA

The visual censuses were carried out by SCUBA at the AR site and the RR site. A third site was included in the census when the tidal currents allowed it, which were in August 2018 and February 2019. The third site was the SB site. Due to problems, the November 2018 census was only done at the RR site. Three belt transects of 30 m in length, 2 m in width, and 2 m high were done at each site. These transects were set up with a separation of 5 m and were parallel to each other. The census in each transect was done with an average time of seven minutes. Two divers were required as a minimum; the first diver identified the species of fish inside the transect, estimated the total length, and the abundance of the fish. The second diver recorded with a camera the fish seen inside and outside the transect, as well as using the buddy system for security; a third diver was used when possible to assist with videotaping. This methodology based on Reef Check California Fish transect survey (Freiwald *et al.*, 2015).

2.2.4 Trap capture data

Commercial traps were used that measured 122 cm x 92 cm x 36 cm, with openings of 5 cm x 5 cm (Figure 4). The traps were deployed at four different sites: the RR site, the AR site, the SB site, and finally, the MC site. At each site, four traps were deployed using the Pacific Sardine (*Sardinops sagax caeruleus*) as bait. The traps remained submerged at the bottom of the lagoon for an average of 45 minutes. Once the traps were retrieved, each captured fish was identified, measured to obtain its total length (TL, cm), standard

length (SL, cm), and weighed (g). The weight was measured on a digital spring scale that measured up to 12 kg in 20-gram increments.



Figure 4. Fish traps. Commercial fish traps stacked one above the other with conical openings into the two sections that divide the traps. Also in the image is a member (Mario Hermida) of the two-person helper crew per boat provided by the commercial fishing cooperative society.

2.3 Data analysis

2.3.1 Environmental parameters

The statistical analyses were done using the program RStudio version 1.0.153 for Windows. Normality of data was tested on seawater temperature, salinity, and dissolved oxygen using the Shapiro-Wilks test (Zar, 2010) for: 1) Between seasons, 2) Between sites, 3) between seasons within each site, and 4) between the sites within each season.

Homogeneity of variance was tested using Bartlett's test of equal variance (Zar, 2010). Because the data did not show either normality ($p < 0.05$) or homogeneity of variance ($p < 0.05$), a non-parametric Kruskal-Wallis (K-W) test was run to determine if there were differences in each of the comparisons (Zar, 2010). When statistical significance was achieved ($p < 0.05$), the non-parametric Dunn's *post hoc* test was run (Zar, 2010).

2.3.2 Species listing

A list of fish species was made by grouping them into class, order, family, genus, and species (R. Froese and Pauly, 2016). The species listing also breaks down the method in which the species was identified as well as their general distribution as temperate, subtropical, or tropical (R. Froese and Pauly, 2016). Also included in the list of species are those fish identified outside the transects.

2.3.3 Abundance and biomass

The total number of fish was calculated from the visual census and those captured with traps. From this data, the relative and cumulative abundances (% Rel; Eq. 1 and 2) and frequencies of occurrence (%FO; Eq. 2) per species were obtained (Rosales-Casián, 2004). Biomass was obtained from the weighing of the fish from the traps, and the estimated biomass for the observed visual census fish was calculated using weight-length relationship models obtained either from literature or calculated from the trap data (Table 6). Relative biomass was calculated from the total biomass on both the visual census and the trap data. Differences between a) seasons, b) sites, c) seasons within each site, and d) sites within each season, were checked for statistical significance for both abundance and biomass using a one-way ANOVA once checked and approved for normality with the Shapiro-Wilks test and homocedastic variance with Bartlett's test (Zar, 2010). For the cases in which normality or homocedastic variances were not achieved after the transformation, a non-parametric Kruskal-Wallis test was used (Zar, 2010). For cases of statistical significance, Tukey's parametric *post hoc* test or Dunn's non-parametric *post hoc* test was performed (Zar, 2010). When comparing two sites (AR and RR) or two seasons at the SB site from the visual SCUBA census, a t-test (Zar, 2010) was performed or a non-parametric Wilcoxon test (Zar, 2010).

2.3.4 Community indexes

Two community indexes were used, first is the index of community importance (ICI), and second is the index of relative importance (IRI). The ICI is an addition of rank of relative abundance with the rank of the frequency of occurrence (Eq. 1; Stephens and Zerba, 1981). Similarly, IRI also involves both relative abundance (P_i) and frequency of occurrence (%FO), yet it adds relative biomass(%W) to the equation, and it is not a ranking system (Eq. 2; Pinkas *et al.*, 1971). Each of these indexes was calculated for all species identified in both census types for each of the sampling seasons.

$$ICI = \text{rank}(P_i) + \text{rank}(\%FO) \quad (1)$$

$$IRI = (P_i + \%W) * \%FO \quad (2)$$

where:

$P_i = (\text{number of individuals of one species} / \text{total number of individuals}) \times 100$

$\%FO = (\text{number of samples where species occurred} / \text{total number of samples}) \times 100$

$\%W = (\text{biomass of one species} / \text{total biomass}) \times 100$

2.3.5 Correlation between environmental and biological factors

To determine the correlation between biological factors (abundance and biomass) and environmental factors (seawater temperature, salinity, and dissolved oxygen), we ran the non-parametric Spearman's rank correlation coefficient (Zar, 2010).

2.3.6 Diversity

Diversity was analyzed using SHE analysis (Buzas and Hayek, 1996, 1998, 2005) for the SCUBA census data. The analysis is based on the Shannon-Weiner diversity index (3); an evenness index (E) called Buzas-Gibson's evenness (Eq. 4; Buzas and Hayek, 1996, 1998, 2005); and the number of species (S). The decomposition equation of the SHE analysis (5) connects these three elements of the SHE analysis.

$$H' = \sum P_i \cdot \ln P_i \quad (3)$$

where:

$P_i = n_i/N$ (relative abundance)

n_i = number of fish of a particular species

N = total number of fish

H' = Shannon-Weiner diversity index

$$E = e^{H'} / S \quad (4)$$

where:

S = total number of species or species richness

H' = Shannon-Weiner diversity index

E = Buzas-Gibson's Evenness index

$$H = \ln S + \ln E \quad (5)$$

Where:

H = the diversity index derived from Shannon-Weiner's diversity

S= species richness

E= species evenness

SHE analysis was used to determine the statistical distribution tendencies of relative abundance (P_i) in order to determine if there is “within” community sampling, or “between” community sampling. The three statistical distributions for P_i are the log series, the log normal, and the broken stick (Buzas and Hayek, 1996, 1998, 2005).

For “within” community sampling, the statistical distributions and their associated parameters are a constant value. On the other hand, for “between” communities sampling, the statistical distributions are different, or if they have the same distribution, the parameters are different.

In a log series distribution, as the number of individuals increases, Shannon’s H' (3), Fisher’s α (6), and Simpson’s index $1/\lambda$ (7) remain constant, while Buzas-Gibson’s E (4) and Pielou’s J' (8) decrease. The slopes for the linear regressions of $\ln S$ versus $\ln N$, and $\ln E$ versus $\ln N$ are of equal magnitude, but opposite sign and the slope of the linear regression of $\ln S$ versus $\ln E$ is -1 with the intercept being an estimate of the mean H .

For a log normal distribution, J' remains constant, while E decreases, and H , $1/\lambda$, and α increase, in this distribution, the ratio $\ln E/\ln S$ remains constant.

For the broken stick distribution, E remains constant, while J' , H , α , and $1/\lambda$ increase. This part of the analysis is called SHE analysis for community structure identification (SHECSI).

A useful tool of the SHE analysis is that a $\ln E$ versus $\ln N$ graph can be used to distinguish between the three statistical distributions mentioned previously; a community from a single statistical distribution will have a linear tendency. Any deviation from this tendency is a change in either the parameter of the statistical distribution or a change in the type of distribution. If there is a break in the linear sequence, the samples preceding the break are designated as a distinct community. The remaining samples are reanalyzed. An extensive breakdown of this method is found in Buzas and Hayek (2005).

$$S = \alpha * \ln(1 + n/\alpha) \quad (6)$$

Where:

S = species richness

α = Fisher's α index

n = number of individuals

$$\lambda = \sum P_i^2 \quad (7)$$

Where:

$P_i = n_i/N$ (relative abundance)

n_i = number of fish of a particular species

N = total number of fish

λ = Simpson's index parameter

$$J' = H' / \ln S \quad (8)$$

Where:

J' = Pielou's Evenness Index

H' = Shannon-Weiner diversity index

S = species richness

$$E(S) = S * \ln(N) / \ln(n) \quad (9)$$

Where:

$E(S)$ = expected number of species

S=number of species in the observations

N= number of individuals in the estimate of S

n= number of individuals in the observations

Once communities and their statistical distributions were identified using SHE analysis, they were compared between each other at equal sample sizes to compare S, H, and E between them. To estimate species richness (S) for log series, the equivalent alpha diversity for rarefaction and abundance methods were used (6) with the known α estimate from the observations, and for log normal and broken stick the species predictions with a ratio of log relationships (9) was used (Hayek and Buzas, 2010). Regression equations from the SHE analysis were used to obtain the values of $\ln E$ and $\ln S$ for the decomposition equation (5).

The trap capture data was analyzed using Shannon-Weiner's diversity index (3) per season, and Pielou's equitability index J' (8). A SHE analysis was not used on the trap capture data due to the low species richness (S) and sample bias (size, food selection). For the calculation of all indexes and parameters, the program PAST (PAleontological STatistics) version 3.25 was used as well as Microsoft Excel 2013.

Chapter 3. Results

3.1 Environmental parameters

Seawater temperature, salinity, and dissolved oxygen were measured at each of the sites during the sampling days then compared for differences.

3.1.1 Between seasons

Seawater temperature had a mean of 20.9 ± 0.3 °C for all sampling seasons, a maximum mean value of 26.6 ± 0.1 °C in the summer 2018 and a minimum mean of 16.9 ± 0.1 °C in winter 2019 (Figure 5). There was a significant difference between the sampling dates (K-W, $p < 0.001$). Dunn *post hoc* test showed differences between winter 2019 and all other seasons except winter 2018 (spring 2018, summer 2018, and autumn 2018, $p < 0.001$). Summer 2018 is significantly different from all other seasons as well: winter 2018 ($p < 0.001$), spring 2018 ($p < 0.001$), and autumn 2018 ($p = 0.003$). There was also significant difference in seawater temperature between winter 2018 and autumn 2018 ($p < 0.001$).

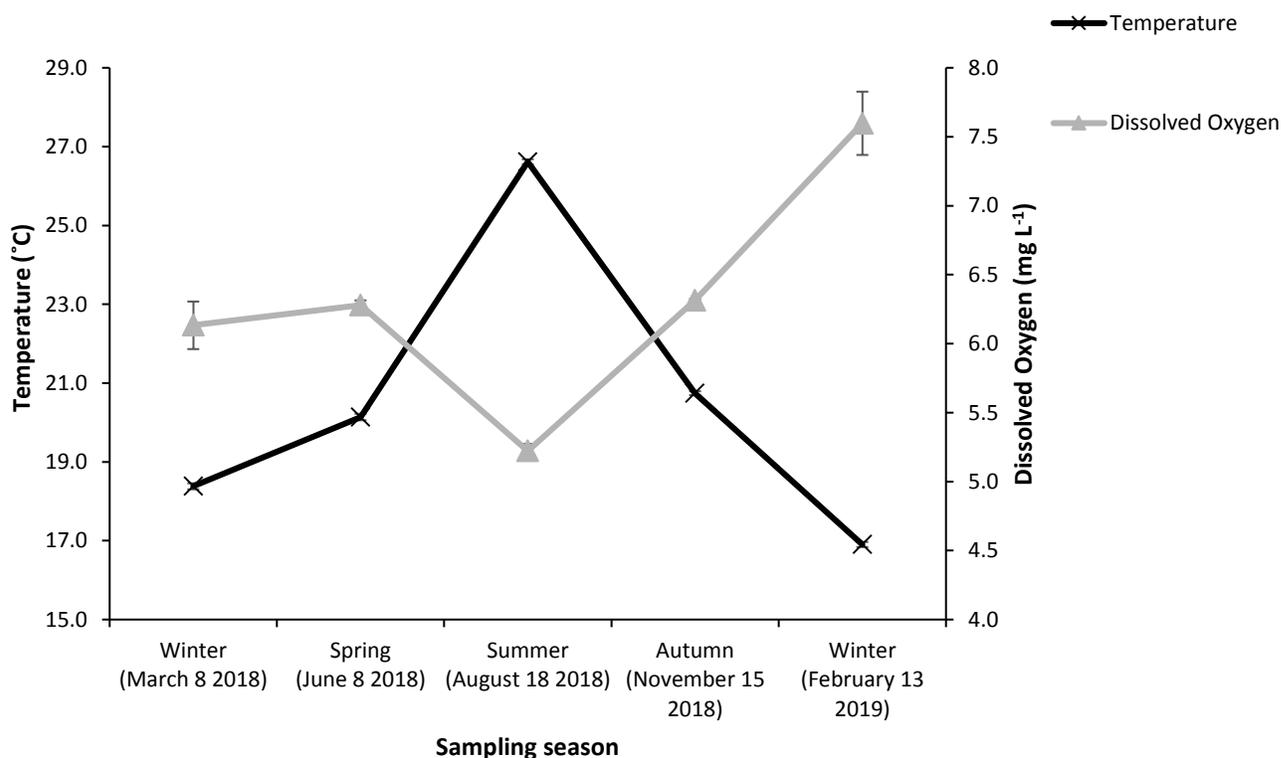


Figure 5. Seasonality of temperature and dissolved oxygen. Seasonality of the means of temperature (°C), and dissolved oxygen (mg L⁻¹). The bars represent the standard error. Significant differences in temperature are between winter 2019 and all other seasons except winter 2018 ($p < 0.001$). There are also significant difference with summer 2018 and all other season, winter and spring 2018 ($p < 0.001$), and autumn ($p = 0.003$). Differences in seawater temperature are also between spring and autumn 2018 ($p < 0.001$). Dissolved oxygen was significantly different between winter 2019 and all other seasons ($p < 0.001$), as well as summer 2018 with all other seasons ($p < 0.001$).

Salinity had a mean of 33.2 ± 0.6 PSU for all sampling seasons, the maximum mean value was in spring 2018 of 35.2 ± 0.2 PSU, and a minimum mean in winter 2019 of 26.9 ± 3.2 PSU (Figure 6). There was a significant difference in salinity between sampling seasons (K-W; $p=0.035$). A *post hoc* Dunn test showed a difference in salinity between the sampling seasons of spring 2018 and autumn 2018 ($p=0.049$).

Mean dissolved oxygen for all sampling seasons was 6.2 ± 0.1 mg L⁻¹, the maximum mean for dissolved oxygen was in winter 2019 of 7.6 ± 0.2 mg L⁻¹ and a minimum mean during summer 2018 of 5.2 ± 0.1 mg L⁻¹ (Figure 5). The Kruskal-Wallis test showed that there was a significant difference ($p<0.001$) between the sampling seasons. The *post hoc* Dunn test showed difference between the dissolved oxygen in winter 2019 and all other months: winter 2018 ($p<0.001$), spring 2018 ($p<0.001$), summer 2018 ($p<0.001$), autumn 2018 ($p<0.001$). There were also differences ($p<0.001$) in dissolved oxygen between summer 2018 and the other seasons: winter 2018, spring 2018, and fall 2018.

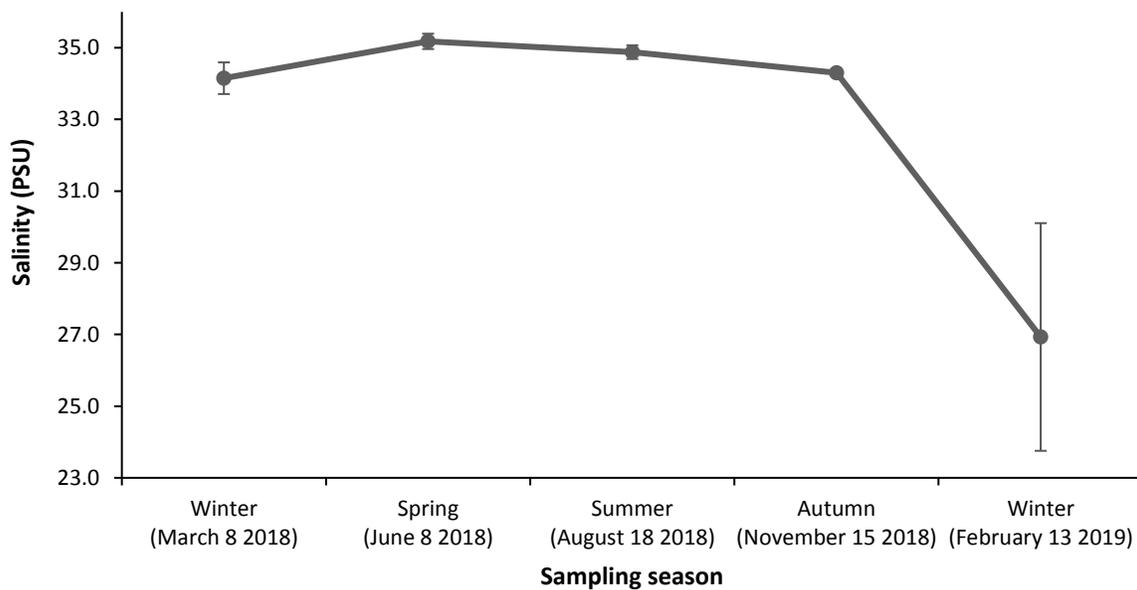


Figure 6. Seasonality of salinity. Seasonality of the mean of salinity (PSU). Significant differences are between the spring and autumn seasons ($p=0.049$). The bars represent the standard error.

3.1.2 Between sites

Seawater temperature between sites during all seasons had a mean of 20.9 ± 0.3 °C, with a maximum mean at the SB site of 21.3 ± 0.7 °C and a minimum mean at the RR site of 20.6 ± 0.6 °C (Figure 7). No significant difference (K-W; $p=0.395$) in the seawater temperature between sites was found.

Salinity had a mean of 33.2 ± 0.6 PSU Between sites, with maximum mean at the AR site of 35.4 ± 0.1 PSU and a minimum mean at the SB site of 27.9 ± 2.7 PSU (Figure 8). There were significant differences between the site samples (K-W; $p<0.001$) in mean salinity. Dunn's *post hoc* test for differences in salinity shows that there were significant differences between the AR site with all other sites: with the RR site ($p<0.001$), with the SB site ($p=0.04$), and the MC ($p<0.001$).

Finally, dissolved oxygen had a mean of 6.2 ± 0.1 mg L⁻¹ for the sites during the sampling seasons, with a maximum mean at the SB site of 6.7 ± 0.3 mg L⁻¹, and a minimum mean of 6.0 ± 0.1 mg L⁻¹ at the RR site (Figure 7). There was no significant difference (K-W; $p=0.17$) between the sites for all the sampling seasons.

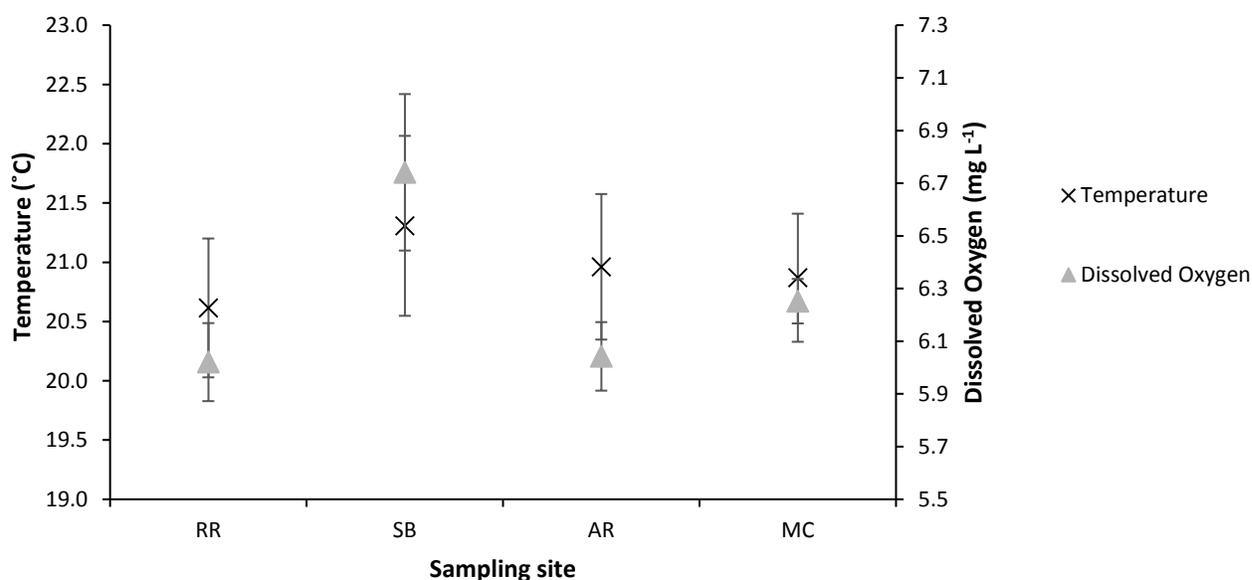


Figure 7. Mean temperature and dissolved oxygen by site. Distribution of the means in seawater temperature (°C) and dissolved oxygen (mg L⁻¹) at the study sites: rocky reef (RR), soft bottom (SB), artificial reefs (AR), and main channel (MC). No significant differences between sites in seawater temperature ($p=0.395$) or dissolved oxygen ($p=0.17$) were found. The bars represent the standard error.

3.1.3 Between seasons within each site

Natural rocky reef (El Borbollon)

This site had a mean seawater temperature of 20.6 ± 0.6 °C with a maximum mean during summer 2018 of 26.15 ± 1.4 °C, and a minimum of 16.42 ± 0.02 °C in winter 2018. A significant difference (K-W; $p < 0.001$) in seasonal temperature at the rocky reef was found. Dunn's *post hoc* test showed differences between winter 2019 and both summer 2018 ($p < 0.001$) and autumn 2018 ($p = 0.004$); there were also differences in seawater temperature between summer 2018 and winter 2018 ($p = 0.001$) and between summer 2018 and spring 2018 ($p = 0.049$; Table 2).

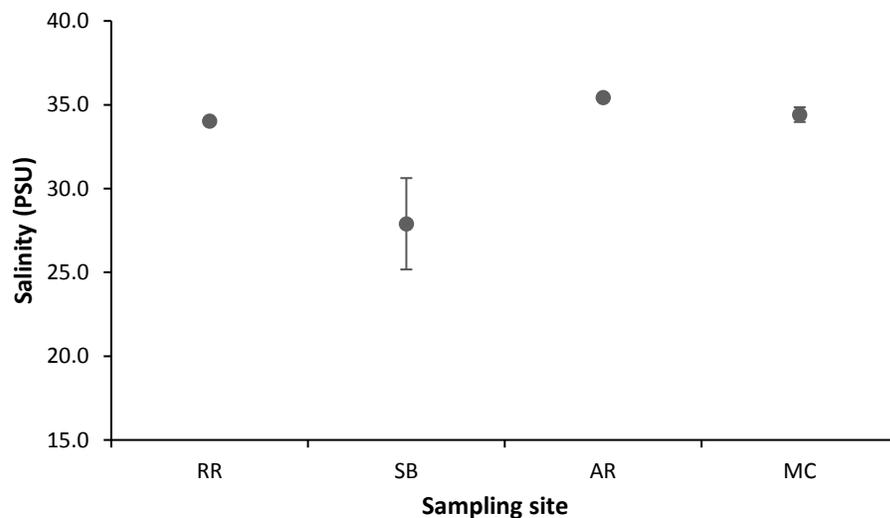


Figure 8. Mean salinity by site. Distribution of the mean salinity (PSU) at the study sites: rocky reef (RR), soft bottom (SB), artificial reefs (AR), and main channel (MC). Differences in salinity shows that there were significant differences between the AR site with all other sites: with the RR site ($p < 0.001$), with the SB site ($p = 0.04$), and the MC ($p < 0.001$). The bars represent the standard error.

Salinity had a mean of 34.01 ± 0.08 °C, a maximum mean value during spring 2018 of 34.75 ± 0.02 PSU, and a minimum mean during winter 2019 of 33.19 ± 0.01 PSU (Table 2). There was a significant difference in salinity (K-W; $p < 0.001$) between seasons at this site. Dunn's *post hoc* test showed a difference in salinity means at this site between winter 2019 and both winter 2018 ($p = 0.003$) and spring 2018 ($p < 0.001$). There was also a significant difference in salinity between spring 2018 and summer 2018 ($p = 0.005$; Table 2).

Dissolved oxygen at the site had a mean of 6.02 ± 0.15 mg L⁻¹ with a maximum mean value (7.07 ± 0.04 mg L⁻¹) during winter 2019, and minimum mean values of 4.94 ± 0.01 mg L⁻¹ (Table 2) during the summer 2018

sampling. A significant difference between the seasonal dissolved oxygen means (K-W, $p < 0.001$) was found. Dunn's *post hoc* test showed the differences at this site in dissolved oxygen were between winter 2019 and both spring 2018 ($p = 0.035$) and summer 2018 ($p < 0.001$), as well as between summer 2018 and autumn 2018 ($p = 0.035$; Table 2).

Soft bottom (La Ventana)

Seawater temperature at the SB site showed a mean of 21.3 ± 0.76 °C with a maximum mean of 27.10 ± 0.05 °C during summer 2018, and a minimum (17.13 ± 0.08 °C) during winter 2019 (Table 2). A significant difference (K-W, $p < 0.001$) between the seasonal mean temperatures at the SB site was found. Differences in seawater temperature at this site were observed during the winter 2019 and both summer 2018 (Dunn; $p < 0.001$) and autumn 2018 ($p = 0.019$), as well as differences between winter 2018 and summer 2018 ($p = 0.004$; Table 2).

Salinity presented a mean of 27.9 ± 2.73 PSU with a maximum mean during summer 2018 of 35.99 ± 0.04 PSU and a minimum (0.80 ± 0.17 PSU) during winter 2019 (Table 2). There were significant differences (K-W; $p < 0.001$) between salinities during the different sampling seasons at this site. The *post hoc* test shows that salinity differed between the samples from summer 2018 and both autumn 2018 (Dunn; $p = 0.004$) and winter 2019 ($p < 0.001$). There was also a difference in salinity between both the sampling winters ($p = 0.018$; Table 2).

Dissolved oxygen had a mean of 6.74 ± 0.3 mg L⁻¹ with a maximum mean during winter 2019 of 9.18 ± 0.54 mg L⁻¹, and a minimum mean (5.17 ± 0.04 mg L⁻¹) during summer 2018 (Table 2). There were significant differences (K-W; $p < 0.001$) between seasons at this site. Dunn's *post hoc* test showed differences between the samples of summer 2018 and both winters (2018: $p = 0.007$, 2019: $p < 0.001$; Table 2).

Artificial reef (El Conchalito)

This site showed a mean seawater temperature of 20.96 ± 0.61 °C with a maximum mean seawater temperature during summer 2018 of 26.8 ± 0.0 °C, and minimum (16.97 ± 0.02 °C) during winter 2019 (Table 2).

A significant difference between the seasonal seawater temperature means at the AR site was found (K-W, $p < 0.001$). Dunn's *post hoc* test showed the differences being between seasonal mean temperature in

winter 2018 with summer 2018 ($p < 0.001$) as well as between winter 2019 with both summer 2018 ($p < 0.001$) and autumn 2018 ($p = 0.015$; Table 2).

Salinity showed a mean of 35.43 ± 0.11 PSU with maximum mean values during spring 2018 of 36.4 ± 0.0 PSU, and minimum values (34.7 ± 0.0 PSU) during autumn 2018 (Table 2). There were significant differences (K-W; $p < 0.001$) between seasonal mean salinities at this site. Dunn's *post hoc* test showed a difference in salinity means between summer 2018 and autumn 2018 ($p < 0.001$), and between spring 2018 and both autumn 2018 ($p < 0.001$) and winter 2019 ($p = 0.004$; Table 2).

Dissolved oxygen at this site had a mean of 6.04 ± 0.13 mg L⁻¹ for all sampling periods with a maximum mean in winter 2019 of 6.99 ± 0.05 mg L⁻¹, and minimum mean in summer 2018 of 5.12 ± 0.06 mg L⁻¹ (Table 2). There were significant differences (K-W; $p < 0.001$) in dissolved oxygen between the different seasonal measurements at this site. Dunn's *post hoc* test showed the differences between the samples of summer 2018 with both autumn 2018 ($p = 0.042$) and winter 2019 ($p < 0.001$; Table 2).

Main channel entrance (Carros Viejos)

Seawater temperature at the MC site had a mean of 20.87 ± 0.54 °C, with a maximum mean during the summer of 2018 of 26.4 ± 0.0 °C, and a minimum during winter 2019 of 17.10 ± 0.07 °C (Table 2). There were significant differences ($p < 0.001$) in seawater temperature between seasons at this site. Dunn's *post hoc* test showed differences in the samples from winter 2019 with both summer 2018 ($p < 0.001$) and autumn 2018 ($p = 0.002$), as well as differences between winter 2018 and summer 2018 ($p < 0.001$; Table 2).

Salinity at the site had a mean of 34.41 ± 0.44 PSU for all seasons, with a maximum mean of 38.93 ± 0.08 PSU during winter 2019, and a minimum mean (32.22 ± 1.18 PSU) during winter 2018 (Table 2). There were significant differences between (K-W; $p < 0.001$) the seasons in salinity at this site. Dunn's *post hoc* test showed differences in salinity between the sample of winter 2018 and both spring 2018 ($p < 0.001$) and autumn 2018 ($p = 0.022$), as well as between the samples of winter 2019 and both winter 2018 ($p < 0.001$) and summer 2018 ($p = 0.002$; Table 2).

Table 2. Mean physical parameters by site and season. Means of seawater temperature (°C), salinity (PSU), and dissolved oxygen (mg L⁻¹) and their standard error during seasons from winter 2018 to winter 2019 at each study site.

Sampling Site/Season	Seawater temperature (°C)	Salinity (PSU)	Dissolved Oxygen (mg L⁻¹)
Natural rocky reef			
<i>Winter 2018</i>	18.3±0.1	34.17±0.03	5.85±0.49
<i>Spring 2018</i>	20.0±0.0	34.75±0.02	6.19±0.05
<i>Summer 2018</i>	26.1±1.4	33.96±0.02	4.94±0.01
<i>Autumn 2018</i>	20.4±0.0	34.02±0.04	6.31±0.01
<i>Winter 2019</i>	16.4±0.0	33.19±0.01	7.07±0.04
Soft bottom*			
<i>Winter 2018</i>	18.1±0.0	35.35±0.04	6.94±0.17
<i>Summer 2018</i>	27.1±0.1	35.99±0.04	5.17±0.04
<i>Autumn 2018</i>	21.0±0.0	34.55±0.02	6.33±0.01
<i>Winter 2019</i>	17.1±0.1	0.80±0.17	9.18±0.54
Artificial reef			
<i>Winter 2018</i>	18.0±0.0	35.50±0.00	5.82±0.43
<i>Spring 2018</i>	20.6±0.0	36.40±0.00	6.21±0.04
<i>Summer 2018</i>	26.8±0.0	35.87±0.02	5.12±0.06
<i>Autumn 2018</i>	20.6±0.0	34.70±0.00	6.29±0.01
<i>Winter 2019</i>	16.9±0.0	34.80±0.00	6.99±0.05
Main channel entrance			
<i>Winter 2018</i>	18.9±0.0	32.22±1.18	5.97±0.07
<i>Spring 2018</i>	19.8±0.0	34.38±0.02	6.43±0.04
<i>Summer 2018</i>	26.4±0.0	33.70±0.00	5.66±0.01
<i>Autumn 2018</i>	20.9±0.1	33.94±0.02	6.32±0.03
<i>Winter 2019</i>	17.1±0.1	38.93±0.08	7.15±0.01

* No data was taken in spring 2018.

Dissolved oxygen at this site had a mean of 6.25 ± 0.08 mg L⁻¹ for all sampling seasons, with a maximum value reached in winter 2019 of 7.15 mg L⁻¹, and a minimum mean (5.66 ± 0.01 mg L⁻¹) in summer 2018 (Table 2). There were significant differences (K-W; $p < 0.001$) in dissolved oxygen between the seasons. Dunn's *post hoc* test of dissolved oxygen at this site showed differences between the samples of summer 2018 with the following three: spring 2018 ($p = 0.002$), autumn 2018 ($p = 0.011$) and winter 2019 ($p < 0.001$). There was also a difference between the samples of winter 2018 with winter 2019 ($p = 0.002$; Table 2).

3.1.4 Between sites within each season

Winter 2018

The maximum mean seawater temperature was at the MC site, and the minimum mean was at the AR site (Table 2). There were significant differences (K-W; $p < 0.001$) between sites in seawater temperature. Dunn's *post hoc* test showed a significant difference between the MC and both the SB site ($p = 0.018$) and the AR site ($p < 0.001$).

Salinity had a maximum mean at the AR site, and a minimum mean at the MC site (Table 2). There were significant differences (K-W; $p < 0.001$) in salinity between the sites this season. The differences were between the MC site and both the SB site ($p = 0.005$) and AR site ($p < 0.001$).

Maximum mean dissolved oxygen was at the SB site and minimum mean at the AR site (Table 2). There were significant differences (K-W; $p = 0.011$) in dissolved oxygen between sites this season. Dunn's *post hoc* test showed there was a difference in dissolved oxygen in this data set between the MC site and SB site ($p = 0.007$).

Spring 2018

Seawater temperature had a maximum at the AR site and a minimum at the MC site (Table 2). There was a significant difference (K-W; $p < 0.001$) in seawater temperature between sites. Dunn's *post hoc* test showed there to be differences between ($p < 0.001$) the MC and AR sites.

Maximum mean salinity was at the AR site, and minimum mean salinity was at the MC site. There was a significant difference (K-W; $p < 0.001$) in salinity between sites this season. Salinity's *post hoc* test showed there to be a difference ($p < 0.001$) between the MC and AR site.

The maximum mean dissolved oxygen was at the MC site and the minimum mean at the RR (Table 2). There was a significant difference (K-W; $p = 0.011$) between the dissolved oxygen of the different sites. Dissolved oxygen showed differences between the MC and both the RR (Dunn; $p = 0.022$) and AR ($p = 0.035$).

Summer 2018

Maximum mean seawater temperature was at the SB site while the minimum mean was at the RR site (Table 2). There was a significant difference (K-W; $p < 0.001$) between the sites this season. Seawater temperature differences were between the RR and both the AR ($p = 0.003$) and the SB ($p < 0.001$) sites. There were also differences between the MC and SB ($p = 0.003$) sites.

The maximum mean salinity was at the SB site and a minimum mean at the MC site (Table 2). There were significant differences (K-W; $p < 0.001$) in salinity between the sites this season. Salinity differed between the MC site and both the AR (Dunn; $p < 0.001$) and SB ($p < 0.001$) sites.

Maximum mean dissolved oxygen was at the MC site and the minimum mean at the RR site (Table 2). There were significant differences (K-W; $p < 0.001$) between sites this season. Dissolved oxygen was different between the MC and both the RR (Dunn; $p < 0.001$) and AR ($p = 0.022$) sites; and between the RR and SB ($p = 0.033$) sites.

Autumn 2018

The maximum mean seawater temperature was at the SB site, and the minimum mean was at the RR site (Table 2). Seawater temperature was significantly different (K-W; $p < 0.001$) among sites. Dunn's test showed differences in seawater temperature between the data from the RR site and both MC ($p < 0.001$) and SB ($p < 0.001$) sites as well as between the AR and SB ($p = 0.021$) sites.

Maximum mean salinity was at the AR site, and a minimum mean was at the MC site (Table 2). Salinity was significantly different (K-W; $p < 0.001$) between sites. Dunn's *post hoc test* showed differences between the RR and AR ($p < 0.001$) sites and between the MC site and both the AR ($p < 0.001$) and SB ($p = 0.013$) sites.

Maximum mean dissolved oxygen was at the SB site, and the minimum mean was at the AR site (Table 2). There were no significant differences (K-W; $p = 0.150$) between sites in dissolved oxygen.

Winter 2019

The maximum mean seawater temperature was at the SB site, and the minimum mean was at the RR site (Table 2). Seawater temperature was significantly different (K-W; $p = 0.002$) between sites. Seawater

temperature had differences between the data sets of the RR site and both the SB (Dunn; $p=0.003$) and MC ($p=0.008$) sites.

Maximum mean salinity was registered at the MC site, and the minimum mean at the SB site (Table 2). There were significant differences (K-W; $p<0.001$) between sites in salinity this season. Dunn's *post hoc* test showed the differences in salinity between the MC site and both the RR ($p=0.017$) and SB ($p<0.001$) sites, and differences between the AR and SB ($p=0.017$) sites.

Dissolved oxygen had a maximum mean at the SB site, and a minimum mean at the AR site (Table 2). There were significant differences (K-W; $p=0.006$) among sites. The differences were among two sites in this sampling date, between the AR and SB ($p=0.006$) sites.

3.2 Species listing

Recall that species seen by trap capture and the visual transects were identified and recorded each season. During the entire survey period, a total of 25 fish species were identified, belonging to nine orders and 18 families (Table 3). Of all the species surveyed, 23 species of fish were identified by SCUBA and nine with traps. A single species of fish presented an affinity to the temperate zone, while six presented tropical distribution, and 18 species to the subtropical zone. Summer 2018 was the season that showed the largest number of fish species (17).

Winter 2018

During this sampling, nine species were identified, five were seen in the traps, and eight species were seen during the SCUBA survey (Table 3). The Garibaldi (*Hypsypops rubicundus*) was only seen during this sampling. Also of interest is the presence of a tropical species during a winter sampling, as observed with the Burrito Grunt (*Anisotremus interruptus*). All sites had similar species, with the California Sheephead (*Semicossyphus pulcher*) only appearing in MC site, *H. rubicundus* only in the RR site, and the Kelpbass (*Paralabrax clathratus*) appearing in SB and MC sites. The Rock Wrasse (*Halichoeres semicinctus*) was only observed with SCUBA in both sites (RR and AR). The Horn shark (*Heterodontus francisci*) was recorded outside the transect at the RR site. The Spotted sandbass (*Paralabrax maculatofasciatus*), Barred sandbass

(*Paralabrax nebulifer*), and Pacific Porgy (*Calamus brachysomus*) were recorded at all four sites. Most species observed have a subtropical distribution (Table 3).

Table 3. Species listing. List of fish species identified for the whole study (March 2018 through February 2019). Species are separated by class, order, family, and sampling method: trap (T), SCUBA (C). The distribution (Dist.) of the species is listed as subtropical (S), tropical (Tr), and temperate (Te). Seasons in which the species were observed are listed as winter 2018 (W), spring 2018 (P), summer 2018 (U), autumn 2018 (F), winter 2019 (N), and for all sampling seasons (A).

Class/Order	Family	Species	T	C	Dist.	Season	
Chondrichthyes							
Heterodontiformes	Heterodontidae	<i>Heterodontus francisci</i> (Girard, 1855)	X	X	S	W, P, U	
Myliobatiformes	Urolophidae	<i>Urolophus halleri</i> (Cooper, 1863)		X	S	U	
Torpediniformes	Narcinidae	<i>Narcine entemedor</i> (Jordan and Starks, 1895)		X	Tr	U	
Rajiformes	Rajidae	<i>Beringraja binoculata</i> (Girard, 1855)		X	Te	U	
Rhinopristiformes	Rhinobatidae	<i>Pseudobatos productus</i> (Ayres, 1854)		X	Tr	P, N	
Osteichthyes							
Perciformes	Serranidae	<i>Paralabrax clathratus</i> (Girard, 1854)	X	X	S	A	
		<i>Paralabrax maculatofasciatus</i> (Steindachner, 1868)	X	X	S	A	
		<i>Paralabrax nebulifer</i> (Girard, 1854)	X	X	S	A	
		<i>Mycteroperca xenarcha</i> (Jordan, 1888)		X	S	U	
	Kyphosidae	<i>Girella nigricans</i> (Ayres, 1860)		X	S	U	
	Haemulidae	<i>Anisotremus davidsonii</i> (Steindachner, 1876)		X	S	P, U	
		<i>Anisotremus interruptus</i> (Gil, 1862)	X	X	Tr	W, P, F	
	Sparidae	<i>Calamus brachysomus</i> (Lockington, 1880)	X	X	S	A	
	Sciaenidae	<i>Pareques viola</i> (Gilbert, 1898)		X	Tr	U	
		<i>Menticirrhus undulatus</i> (Girard, 1854)		X	S	F	
		<i>Atractoscion nobilis</i> (Ayres, 1860)		X	S	P	
	Labridae	<i>Halichoeres semicinctus</i> (Ayres, 1859)		X	S	W, P, U, N	
		<i>Semicossyphus pulcher</i> (Ayres, 1854)	X		S	W, P	
		Gerreidae	<i>Eucinostomus currani</i> (Zahuaranec, 1980)		X	Tr	U
		Labrisomidae	<i>Labrisomus xanti</i> (Gill, 1860)		X	Tr	U
	Tetraodontiformes	Pomacentridae	<i>Hypsypops rubicundus</i> (Girard, 1854)		X	S	W
		Tetraodontidae	<i>Sphoeroides annulatus</i> (Jenyns, 1842)	X	X	S	P, U, N
		Balistidae	<i>Balistes polylepis</i> (Steindachner, 1876)	X		S	U
Pleuronectiformes	Pleuronectidae	<i>Hypsopsetta guttulata</i> (Girard, 1856)		X	S	F	
Atheriniformes	Atherinopsidae	<i>Atherinopsis californiensis</i> (Girard, 1854)		X	S	F	
TOTAL	18	25	9	23			

Spring 2018

Twelve species were identified during the spring 2018 campaign. Eight species were observed in the traps, and 11 were observed by SCUBA (Table 3). The White seabass (*Atractoscion nobilis*) was recorded only in this sampling period, and it was observed outside the SCUBA transect at the RR site. The Sargo

(*Anisotremus davidsonii*) was only recorded at the RR site, and similarly to Winter 2018, *H. semicinctus* was only recorded by SCUBA in both the RR and AR sites. The Shovelnose guitarfish (*Pseudobatos productus*) was observed only at the AR site. *A. interruptus* was observed just in the MC site. *P. clathratus* and *H. francisci* were observed in all sites except RR. The Bullseye puffer (*Sphoeroides annulatus*) was recorded only at the SB site. *S. pulcher* was only observed at the MC site. *C. brachysomus*, *P. maculatofasciatus*, and *P. nebulifer* were recorded in all sites.

During this sampling period, we identified two species with tropical distribution, both *A. interruptus*, and *P. productus*, all other species identified had a subtropical distribution (Table 3).

Summer 2018

A total of 17 species were identified during summer 2018. The vast majority of these were identified with SCUBA (16); five species were recorded with the trap method (Table 3).

The Pacific Flagfin Mojarra (*Eucinostomus currani*), Broomtail Grouper (*Mycteroperca xenarcha*), Round Stingray (*Urolophus halleri*), Finescale Triggerfish (*Balistes polylepis*), Electric Giant Ray (*Narcine entemedor*), Big Skate (*Beringraja binoculata*), Opaleye (*Girella nigricans*), Gungo Highhat (*Pareques viola*), and the Largemouth Blenny (*Labrisomus xanti*) were only observed during this sampling period. Of these, *L. xanti* was not in any historical reference as a species encountered inside OLL. From the species observed only in this season, *B. polylepis* was observed by traps, all others except *E. currani* were observed by SCUBA at the SB site. *B. polylepis* was captured at the MC site.

H. semicinctus was seen at both the AR and SB sites by SCUBA. *S. annulatus* and *E. currani* were recorded only at the RR site. Of the three *Paralabrax* species *P. maculatofasciatus* and *P. nebulifer* appeared at all sites, *P. clathratus* appeared at the MC and SB sites. *C. brachysomus* also appeared at all sites.

Four species from this sampling period have a tropical distribution: *N. entemedor*, *P. viola*, *E. currani*, and *L. xanti*; there is also one species with temperate distribution (*B. binoculata*) with all others having a subtropical distribution (Table 3).

Autumn 2018

Eight species were identified during the autumn 2018 campaign, of which five were viewed with SCUBA and five were also identified from trap capture (Table 3). Only two species were identified inside the transects with SCUBA: *C. brachysomus*, and *P. maculatofasciatus*. Species that appeared at all sites were: *C. brachysomus*, *P. maculatofasciatus*, and *P. nebulifer*. *A. interruptus* was captured only at the RR site. The other *Paralabrax* species, *P. clathratus*, was recorded at all sites except the AR site. The Jack silverside (*Atherinopsis californiensis*), the California kingcroaker (*Menticirrhus undulatus*), and the Diamond turbot (*Hypsopsetta guttulata*), which were only observed this sampling period, were observed outside the transects with SCUBA by the surf zone near the MC site. During this season, we identified one species with a tropical distribution (*A. interruptus*), while all other species have a subtropical distribution (Table 3).

Winter 2019

During the winter 2019 sampling, seven fish species were identified. Of these, five fish species were observed inside traps, and five during the SCUBA census (Table 3). Only *P. maculatofasciatus* was recorded at all four sites. *H. semicinctus* was only observed with SCUBA in two sites, AR and RR. *S. annulatus* was observed at the SB and AR sites. *P. productus* was observed at the RR site and was the only species with a tropical distribution recorded this sampling day, all other species observed were from a subtropical distribution. *P. nebulifer*, *P. clathratus*, and *C. brachysomus* were observed at all sites except the AR site.

3.3 Abundance and biomass

For the trap capture, all fish were counted and weighed in order to obtain abundance and biomass, while for the visual transects, the fish were counted inside the transects, their sizes were estimated while their weight was obtained using linear regressions done using the trap capture data or with models found in literature (Table 8).

3.3.1 Between seasons

Trap capture

There was a mean in abundance of 8.1 ± 0.8 fish trap⁻¹ for the entirety of the trap capture, with the highest mean abundance (11.8 ± 2.5 fish trap⁻¹) during spring of 2018 and the lowest mean (5.1 ± 1.0 fish trap⁻¹) in the autumn of 2018 (Figure 9). There were significant differences (K-W; $p=0.028$) in abundance between the seasons. However, when Dunn's *post hoc* test was performed, there were no differences among seasons in mean abundance for the trap captures. The closest seasons with differences were between winter 2018 with both summer 2018 ($p=0.161$) and autumn 2018 ($p=0.131$).

Biomass showed a general mean of 3.6 ± 0.4 kg trap⁻¹, with maximum mean biomass of 5.6 ± 1.2 kg trap⁻¹ in the spring of 2018 and a minimum of 2.2 ± 0.4 kg trap⁻¹ in autumn 2018 (Figure 9). There were significant differences in biomass between mean biomass of the seasons (K-W; $p=0.010$). Dunn's *post hoc* test showed no differences between seasons, yet many were close to having differences, such as winter 2018 with both summer 2018 ($p=0.114$) and autumn 2018 ($p=0.129$), and between spring 2018 and both summer 2018 ($p=0.101$) and autumn 2018 ($p=0.114$).

SCUBA census

An overall mean of abundance for the SCUBA census of 16.0 ± 2.3 fish transect⁻¹ was found. The highest mean was 29.8 ± 3.5 fish transect⁻¹ in the spring of 2018, and the lowest mean was 5.0 ± 2.6 fish transect⁻¹ during the autumn of 2018 (Figure 10). There was a significant difference among the seasonal samples for the log[abundance per transect] (ANOVA; $p=0.001$). Tukey's *post hoc* test showed differences between spring 2018 and both autumn 2018 ($p=0.017$) and winter 2019 ($p=0.003$). As well as between winter 2018 and winter 2019 ($p=0.020$).

Biomass had a mean of 2.3 ± 0.5 kg transect⁻¹ for all transects surveyed in this study, with a maximum mean of 5.3 ± 1.2 kg transect⁻¹ for the spring of 2018 and a minimum in both autumn 2018 and winter 2019 of 0.7 ± 0.4 kg transect⁻¹ (Figure 10).

The One-Way ANOVA of log[Biomass per transect] showed there were significant differences between the seasons ($p=0.017$). Tukey's *post hoc* test showed differences between spring 2018 and winter 2019 ($p=0.010$).

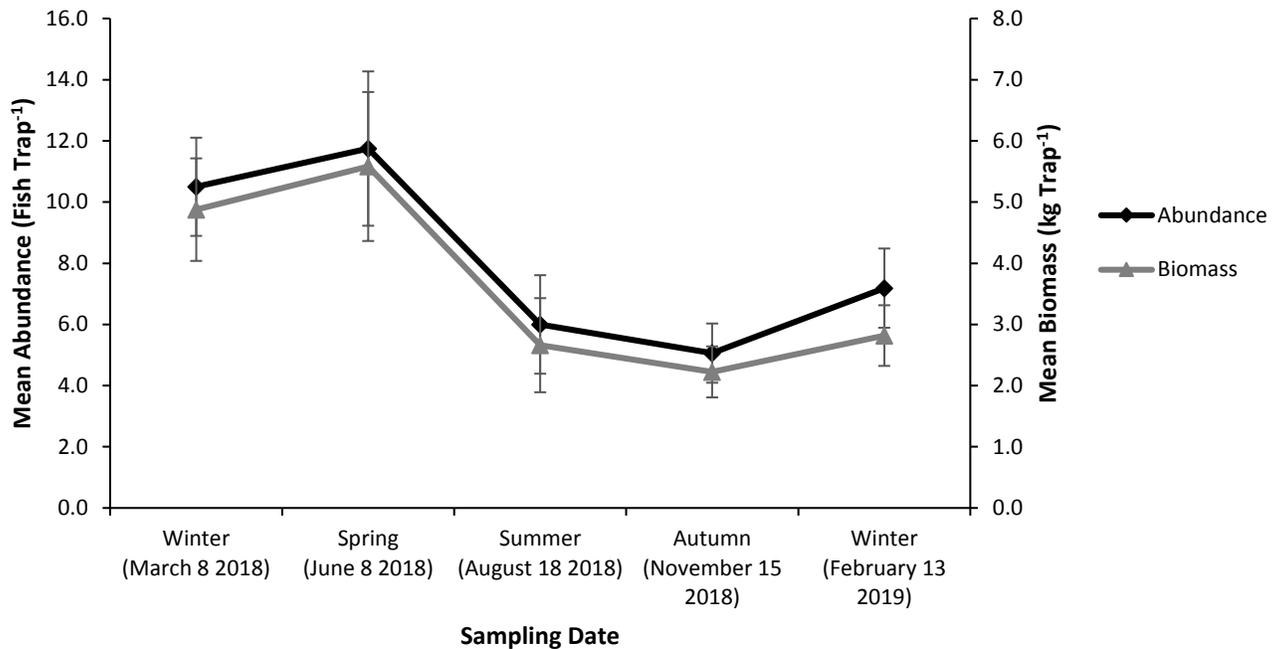


Figure 9. Trap capture seasonal abundance and biomass. Seasonal means of abundance (fish trap⁻¹) and biomass (kg trap⁻¹) captured with traps at Ojo de Liebre lagoon. Dunn's *post hoc* test showed no significant differences between the seasons in abundance and biomass. The bars represent the standard error.

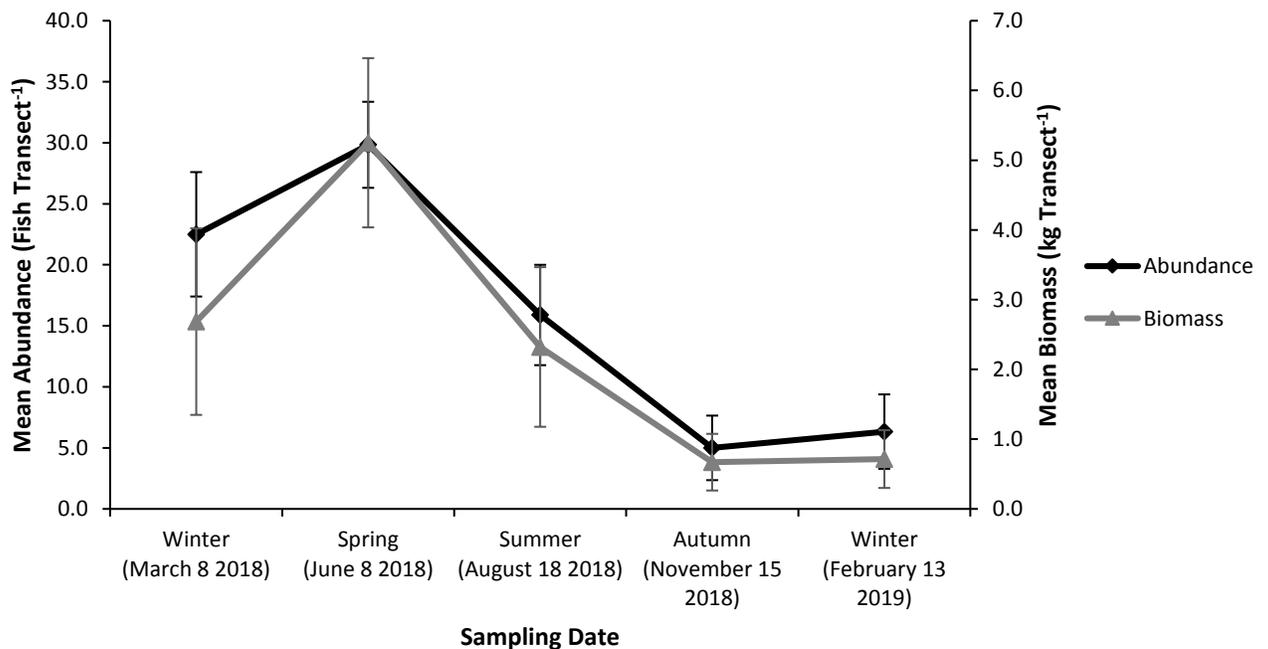


Figure 10. SCUBA census seasonal abundance and biomass. Seasonal means of abundance (fish transect⁻¹) and biomass (kg transect⁻¹) with SCUBA census at Ojo de Liebre lagoon. Differences in abundance are between spring 2018 and both autumn 2018 ($p=0.017$) and winter 2019 ($p=0.003$) as well as between winters ($p=0.020$). Differences in biomass are between spring 2018 and winter 2019 ($p=0.010$). The bars represent the standard error.

3.3.2 Between sites

Trap capture

Mean abundance between sites showed the highest mean (15.3 ± 1.8 fish trap⁻¹) at the MC site, and the lowest mean (4.2 ± 0.8 fish trap⁻¹) at the SB site (Figure 11). There was a significant difference between sites in the trap capture data (K-W; $p < 0.001$). The *post hoc* test showed differences between the MC site and all other sites. The MC site had differences in the mean abundance data with the RR site, with the AR site, and with the SB site ($p < 0.001$).

Biomass presented a higher mean (6.7 ± 0.9 kg trap⁻¹) at the MC site and a minimum of 1.9 ± 0.5 kg trap⁻¹ at the SB site (Figure 11). There were significant differences in biomass between sites (K-W; $p < 0.001$). Biomass was different for the trap capture data between the MC site and the other sites. The MC site biomass was significantly different with the RR ($p = 0.001$) site, with the AR ($p = 0.005$) site, and with the SB ($p < 0.001$) site.

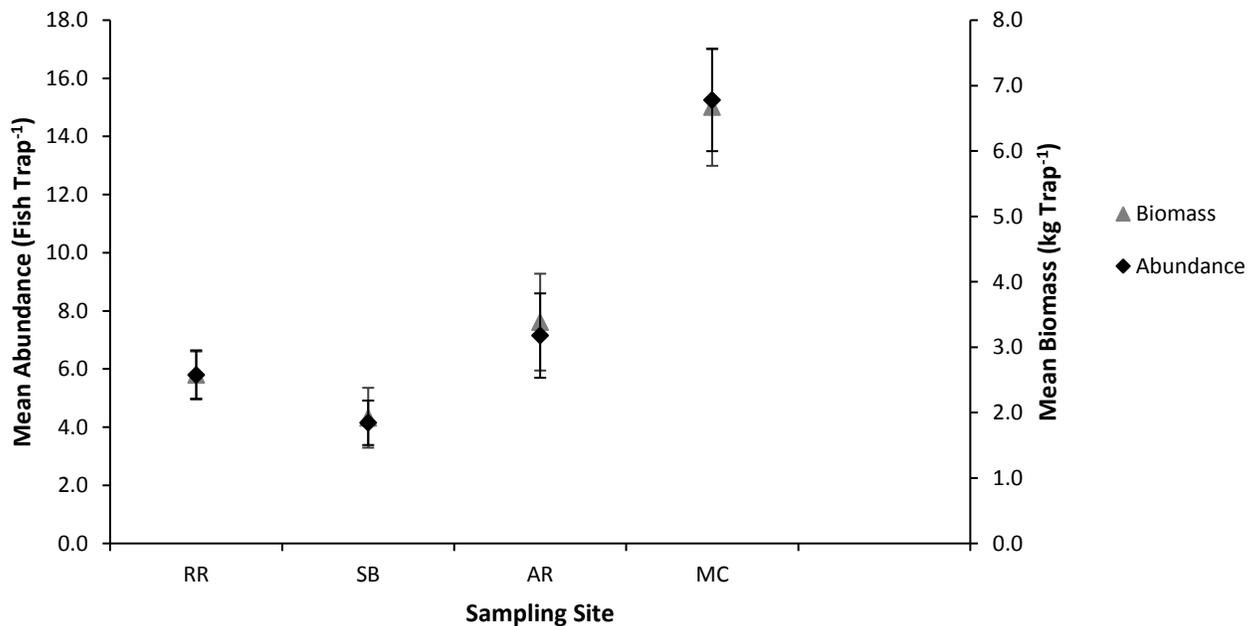


Figure 11. Abundance and biomass for trap capture per site. Means of abundance (fish trap⁻¹) and biomass (kg trap⁻¹) per study sites: rocky reef (RR), soft bottom (SB), artificial reefs (AR), and main channel (MC). Significant differences in abundance were between the MC site and all other sites ($p < 0.001$). Biomass was significantly different between the MC site and all other sites: RR ($p = 0.001$), AR ($p = 0.005$), and SB ($p < 0.001$) sites. The bars represent the standard error.

SCUBA census

Mean abundance per transect for the entire study showed a maximum at the AR site with 18.2 ± 3.5 fish transect⁻¹, and a minimum of 12.3 ± 6.4 fish transect⁻¹ at the SB site (Figure 12). A t-test showed no significant difference ($p=0.219$) between the RR and AR sites. No significant difference (K-W; $p=0.312$) between the mean abundances of fish per transect was found.

Concerning the seasonal biomass per transect, a highest mean (2.9 ± 1.7 kg transect⁻¹) at the SB site was found, with the lowest (2.1 ± 0.9 kg transect⁻¹) at the AR site (Figure 12). The t-test between the AR and the RR site of log[biomass] showed no significant difference ($p=0.392$). No significant difference (K-W; $p=0.845$) between the mean biomass of the AR, RR and SB sites.

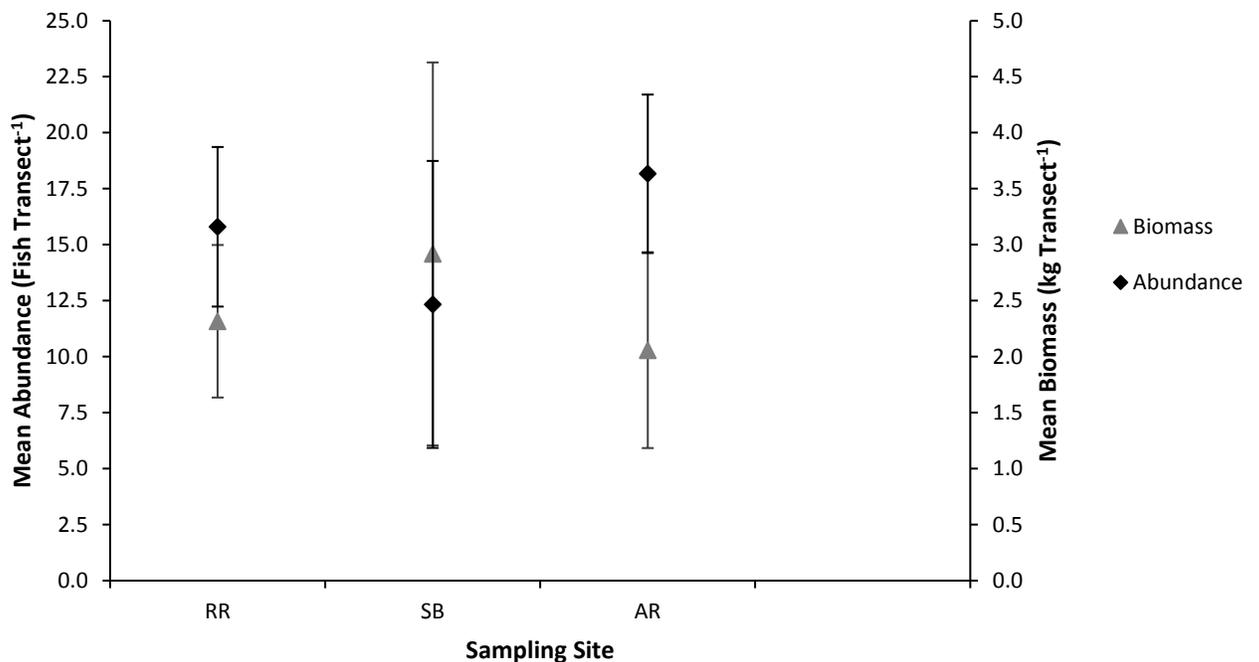


Figure 12. Abundance and biomass for the SCUBA census per site. Means of abundance (fish transect⁻¹) and biomass (kg transect⁻¹) for the SCUBA census between the rocky reef (RR), soft bottom (SB), and artificial reefs (AR) sites for all season. No significant differences in abundance and biomass between the sites was found. The bars represent the standard error.

3.3.3 Between seasons within each site

Natural rocky reef (El Borbollon)

This site showed a mean abundance of 5.8 ± 0.8 fish trap⁻¹, with a maximum abundance of 9.8 ± 1.5 fish trap⁻¹ during winter 2019 and a minimum of 2.3 ± 0.8 fish trap⁻¹ for the autumn of 2018 (Figure 13). There was a significant difference in the abundance per trap between seasons (K-W; $p=0.015$). Dunn's *post hoc* test showed significant differences between autumn 2018 and winter 2019 ($p=0.042$).

Biomass for trap capture showed a mean of 2.6 ± 0.4 kg trap⁻¹ at the RR site for the whole study, with maximum biomass of 3.9 ± 0.7 kg trap⁻¹ in the winter of 2019 and a minimum of 1.1 ± 0.2 kg trap⁻¹ in the autumn of 2018 (Figure 13). Biomass per trap did not show a significant difference (K-W; $p= 0.163$) between seasons at the RR site.

Abundance in the scuba census at this site presented a mean of 15.8 ± 3.6 fish transect⁻¹, with a maximum of 30 ± 5.2 fish transect⁻¹ in the spring of 2018, and a minimum of 5.0 ± 2.6 fish transect⁻¹ in autumn 2018 (Figure 14). A significant difference in the abundance means of fish per transect between seasons (ANOVA; $p=0.047$) was found. However, Tukey's *post hoc* test did not detect a significant difference between the seasonal abundance means at the RR. The closest seasons to have significant differences were spring 2018 with both summer 2018 ($p=0.088$) and autumn 2018 ($p=0.088$).

Biomass for the SCUBA census at the RR site showed a mean of 2.3 ± 0.7 kg transect⁻¹, with a maximum in winter 2018 of 4.5 ± 2.4 kg transect⁻¹, and a minimum of 0.2 ± 0.1 kg transect⁻¹ (Figure 14). A One-Way ANOVA showed there were significant differences in the log[biomass] at this site during the different seasons ($p=0.018$). Tukey's *post hoc* test showed significant differences in log[biomass] between summer 2018 and both winter 2018 ($p=0.031$) and spring 2018 ($p=0.024$).

Soft bottom (La Ventana)

Abundance of the trap capture data at the SB site presented a mean of 4.2 ± 0.8 fish trap⁻¹, with the highest mean (6.8 ± 2.2 fish trap⁻¹) in spring 2018, and the lowest (1.8 ± 0.5 fish trap⁻¹) in summer 2018 (Figure 13). No significant differences (ANOVA; $p=0.404$) in the means of abundance per trap among the different seasons at the SB site.

Biomass for trap capture data at this site had a mean of 1.9 ± 0.5 kg trap⁻¹ for the entire study period, with a maximum 3.6 ± 1.3 kg trap⁻¹ during the spring 2018 sampling, and a minimum of 0.5 ± 0.2 kg trap⁻¹ in the summer of 2018 (Figure 13). There was no significant difference (K-W; $p=0.160$) in the biomass data from trap capture between seasons at this site.

At the SB site, two seasons were sampled with SCUBA. The mean abundance from the SCUBA census for this site was 12.3 ± 6.4 fish transect⁻¹, with summer 2018 showing the highest abundance of 23.3 ± 9.2 fish transect⁻¹, and winter 2019 with the lowest abundance of 1.3 ± 0.0 fish transect⁻¹ (Figure 14). Wilcoxon test was performed for abundance between summer 2018 and winter 2019 that showed a significant difference in the abundance data between summer 2018 and winter 2019 ($p=0.038$).

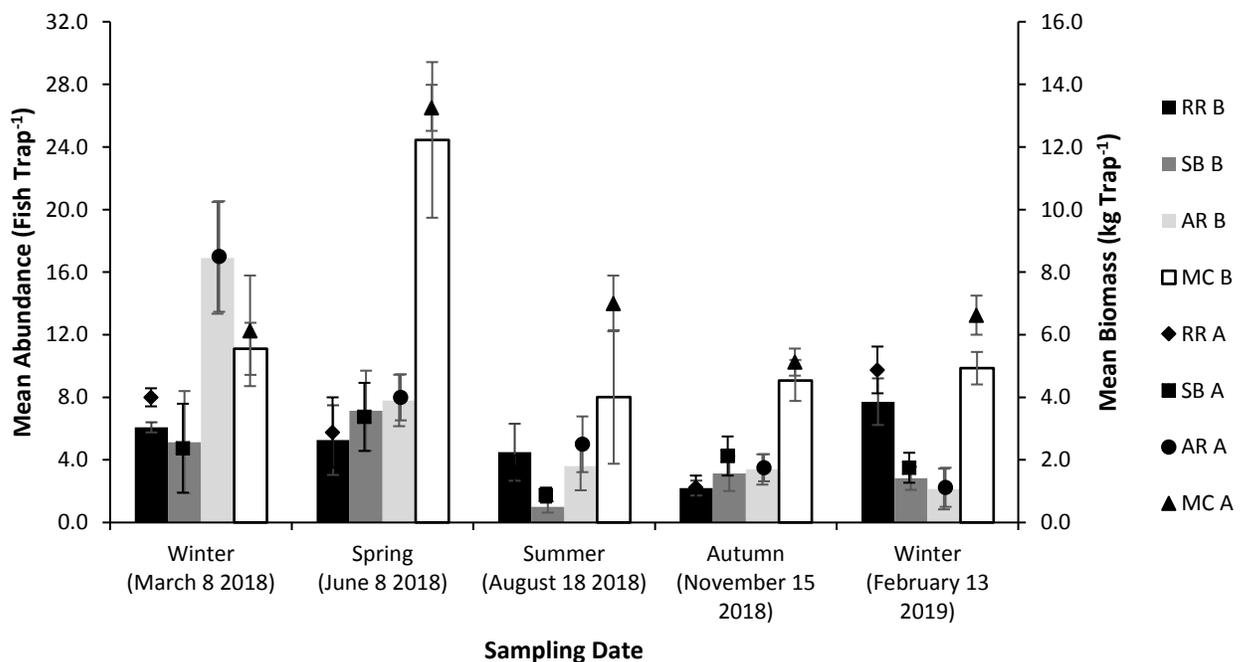


Figure 13. Abundance and biomass per site and season for the trap capture data. Seasonal means of abundance (A; fish trap⁻¹) and biomass (B; kg trap⁻¹) for the trap capture at the study sites: rocky reef (RR), soft bottom (SB), artificial reefs (AR), and main channel (MC). The bars represent the standard error.

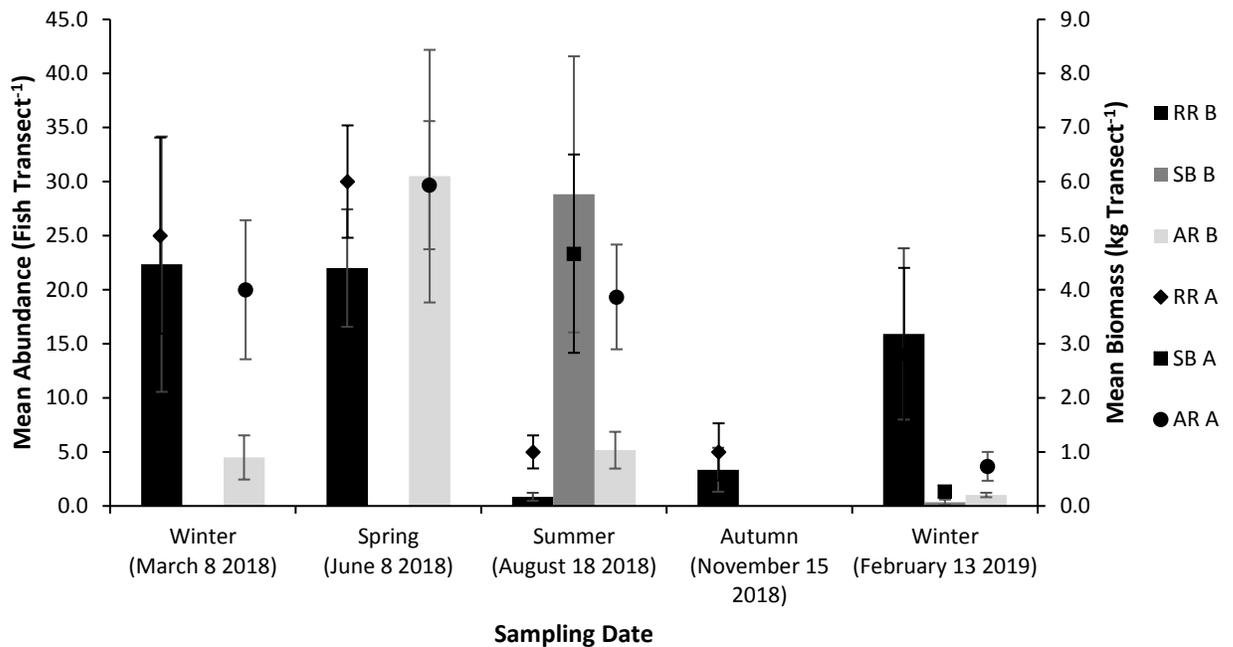


Figure 14. Abundance and biomass per site and season for the SCUBA census data. Seasonal means of abundance (A; fish transect⁻¹) and biomass (B; kg transect⁻¹) for the SCUBA census for all study sites: rocky reef (RR), soft bottom (SB), and artificial reefs (AR). The bars represent the standard error.

Biomass for the SCUBA census at this site had a mean of 2.9 ± 1.7 kg transect⁻¹ for the two seasons (summer 2018 and winter 2019) with summer having the highest mean biomass of 5.8 ± 2.6 kg transect⁻¹, and winter 2019 having the lowest mean biomass of 0.1 ± 0.0 kg transect⁻¹ (Figure 14). A t-test was performed between the log[biomass] of summer 2018 and winter 2019 with significant differences ($p=0.003$) between both seasons.

Artificial reef (El Conchalito)

This site showed a mean abundance with the trap capture data of 7.2 ± 1.5 fish trap⁻¹ for the entire study. The maximum mean abundance was winter 2018 with 17.0 ± 3.5 fish trap⁻¹, and minimum mean abundance of 2.3 ± 1.3 fish trap⁻¹ in winter 2019 (Figure 13). There were significant differences (K-W; $p=0.009$) in the trap abundance data between the seasons. Dunn's *post hoc* test showed differences between winter 2018 and winter 2019 ($p=0.008$).

Biomass for the AR site presented a mean of 3.4 ± 0.7 kg trap⁻¹ for the entire study period, with a maximum in winter 2018 of 8.5 ± 1.8 kg trap⁻¹, and a minimum of 1.1 ± 0.7 kg trap⁻¹ in the winter of 2019 (Figure 13).

Our data showed a significant difference (K-W; $p=0.009$) in the mean biomass per trap for the AR site between seasons. Biomass per trap was different between the winter 2018 and winter 2019 ($p=0.012$).

The abundance for SCUBA census at this site had a mean of 18.2 ± 3.5 fish transect⁻¹ for the complete study, with a maximum of 29.7 ± 5.9 fish transect⁻¹ in the spring of 2018 and a minimum of 3.7 ± 1.3 fish transect⁻¹ in winter 2019 (Figure 14). There were no significant differences (K-W; $p=0.074$) between seasonal means of fish per transect.

Biomass for the SCUBA census had a mean of 2.1 ± 0.9 kg transect⁻¹ for the whole study, with a maximum in spring 2018 of 6.1 ± 2.3 kg transect⁻¹, and a minimum in winter 2019 of 0.2 ± 0.0 kg transect⁻¹ (Figure 14). The log[biomass] for the SCUBA census was significantly different (ANOVA; $p=0.005$) between seasons. Tukey's *post hoc* test showed the difference was between spring 2018 and winter 2019 ($p=0.003$).

Main channel entrance (Carros Viejos)

This site was only sampled with traps. The abundance at Carros Viejos showed a general mean of 15.3 ± 1.8 fish trap⁻¹, with a maximum 26.5 ± 4.2 fish trap⁻¹ in the spring of 2018, and a minimum of 10.3 ± 1.8 fish trap⁻¹ in autumn 2018 (Figure 13). A significant difference between the means of fish traps at Carros Viejos (ANOVA, $p=0.022$) was found. Tukey's *post hoc* test showed a significant difference between spring 2018 and autumn 2018 ($p=0.015$).

Biomass at this site presented a mean for the entire study of 6.7 ± 0.9 kg trap⁻¹, with a maximum of 12.2 ± 2.5 kg trap⁻¹ in the spring of 2018 and a minimum of 4.0 ± 2.1 kg trap⁻¹ in the summer 2018 (Figure 13). There was a significant difference (ANOVA; $p=0.021$) in log[biomass] between seasons at this site. Tukey's *post hoc* test showed the differences in log[biomass] were between spring 2018 and both autumn 2018 ($p=0.021$) and winter 2019 ($p=0.042$).

3.3.4 Between sites within each season

Winter 2018

The maximum mean abundance per trap was at the AR site, and the minimum at the SB site (Table 4). There was a significant difference (K-W; $p=0.038$) between sites in our data. However, Dunn's *post hoc* test shows no difference in abundance between the sites. The AR site had the higher mean biomass per trap, while the lowest mean biomass per trap was at the SB site (Table 4). There was a significant difference between the sites (K-W; $p=0.035$) in mean biomass per trap. However, the *post hoc* test showed there were no significant differences in mean biomass per trap between the sites.

For the SCUBA census, abundance had a maximum mean at the RR site and a minimum at the AR site (Table 4). There was no significant difference between the RR and AR (t-test; $p=0.338$) sites. Biomass for the SCUBA census had a maximum mean at the RR and minimum at the AR site (Table 4). A t-test showed there were no significant differences between the sites ($p=0.105$).

Spring 2018

Abundance per trap during spring 2018 had a maximum at the MC site and a minimum at the RR site (Table 4). There was a significant difference (K-W; $p=0.030$) in abundance per trap between the sites. Dunn's *post hoc* test showed a significant difference ($p=0.044$) between the RR and MC sites. The biomass per trap had the highest mean at the MC site, and lowest mean at the RR site (Table 4). There was a significant difference (ANOVA; $p=0.003$) in the biomass per trap between sites. According to Tukey's *post hoc* test, there were significant differences between the MC site and all other sites: RR ($p=0.004$), AR ($p=0.012$), and SB ($p=0.009$).

Abundance per transect had a maximum at the RR site and minimum at the AR site (Table 4). There was no significant difference (t-test; $p=0.484$) between both the AR and RR sites this sampling period. The RR site had the lesser mean biomass per transect, and the AR site had a higher mean biomass per transect (Table 4). There was no significant difference (t-test; $p=0.273$) in biomass per transect between the AR and RR sites.

Summer 2018

During summer 2018, abundance in trap capture was highest at the MC site and lowest at the SB site (Table 4). There were significant differences (ANOVA; $p=0.012$) in abundance per trap between sites. Tukey's *post hoc* test showed there was a significant difference in $\log[\text{abundance}]$ between the MC and SB ($p=0.009$) sites. Mean biomass per trap had a maximum at the MC and a minimum at the SB site (Table 4). A One-Way ANOVA showed significant differences ($p=0.015$) between sites for $\log[\text{biomass}]$. Tukey's *post hoc* test shows significant differences in $\log[\text{biomass}]$ between the MC and SB ($p=0.009$) sites.

For the SCUBA census, abundance had a maximum at the SB site and a minimum at the RR site (Table 4). There were no significant differences in abundance per transect between the AR, RR, and SB (ANOVA; $p=0.159$) sites. Finally, SCUBA census biomass per transect had a maximum at the SB site and minimum at the RR site (Table 4). There were significant differences (ANOVA; $p=0.013$) in $\log[\text{biomass}]$ between the sites. Tukey's *post hoc* test showed the differences in $\log[\text{biomass}]$ between RR and SB ($p=0.011$) sites.

Autumn 2018

Trap capture data for abundance had a mean that was highest at the MC site and lowest at the RR site (Table 4). One-Way ANOVA showed there were significant differences ($p=0.003$) in abundances per trap between the sites. Tukey's *post hoc* test showed that the differences in abundance were between the MC site and all other sites: RR ($p=0.003$), AR ($p=0.010$), and SB ($p=0.022$).

On the other hand, biomass per trap had a maximum mean at the MC site, and a minimum mean at the RR site (Table 4). There was a significant difference (ANOVA; $p=0.023$) in $\log[\text{biomass}]$ per trap between sites. Biomass was different between the MC and both the RR ($p=0.024$), and SB ($p=0.049$) sites, as per Tukey's *post hoc* test.

Winter 2019

Mean abundance per trap was highest at the MC site and lowest at the AR site (Table 4). There was a significant difference (K-W; $p=0.006$) between sites in mean abundance per trap. Dunn's *post hoc* test showed this difference to be between the MC and both AR ($p=0.014$), and SB ($p=0.047$) sites. Mean biomass per trap was highest at the MC site and lowest at the AR site (Table 4). There was a significant difference (K-W; $p=0.015$) in mean biomass per trap. However, after performing Dunn's *post hoc* test, no

significant difference between sites was evident. Two sites (AR and SB) were close to being significantly different from the MC ($p=0.056$ for both) site.

In the SCUBA census, mean abundance per transect had a maximum at the RR and a minimum at the SB site (Table 4). There was no significant difference (K-W; $p=0.066$) in mean abundance per transect. Biomass per transect had a maximum mean at the RR, and a minimum mean at the SB site (Table 4). There was a significant difference (ANOVA; $p=0.037$) in $\log[\text{biomass}]$ per transect between the sites. Tukey's *post hoc* test showed the differences in $\log[\text{biomass}]$ per transect were between the SB and RR ($p=0.031$) sites.

Table 4. Summary of mean abundance and biomass. Summary table of the mean abundances and biomass per site (RR: natural rocky reef, SB: soft bottom, AR: artificial reefs, MC: main channel entrance) with each sampling method (trap capture and SCUBA) in each season from winter 2018 to winter 2019.

Season	Site	Abundance		Biomass	
		Trap (Fish Trap ⁻¹)	SCUBA (Fish Transect ⁻¹)	Trap (kg Trap ⁻¹)	SCUBA (kg Transect ⁻¹)
Winter 2018	RR	8.0±0.6	25.0±9.1	3.0±0.2	4.5±2.4
	SB	4.8±2.8		2.6±1.6	
	AR	17.0±3.5	20.0±6.4	8.5±1.8	0.9±0.4
	MC	12.3±1.5		5.6±0.8	
Spring 2018	RR	5.8±2.3	30.0±5.2	2.6±1.1	4.4±1.1
	SB	6.8±2.2		3.6±1.3	
	AR	8.0±1.5	29.7±5.9	3.9±0.8	6.1±2.3
	MC	26.5±4.2		12.2±2.5	
Summer 2018	RR	3.3±1.0	5.0±1.5	2.2±0.9	0.2±0.1
	SB	1.8±0.5	23.3±9.2	0.5±0.2	5.8±2.6
	AR	5.0±1.8	19.3±4.8	1.8±0.8	1.0±0.3
	MC	14.0±4.1		4.0±2.1	
Autumn 2018	RR	2.3±0.8	5.0±2.6	1.1±0.2	0.7±0.4
	SB	4.3±1.3		1.6±0.6	
	AR	3.5±0.9		1.7±0.5	
	MC	10.3±1.8		4.5±0.7	
Winter 2019	RR	9.8±1.5	14.0±8.0	3.9±0.7	3.2±1.6
	SB	3.5±1.0	1.3±0.0	1.4±0.4	0.1±0.0
	AR	2.3±1.3	3.7±1.3	1.1±0.7	0.2±0.0
	MC	13.3±1.3		4.9±0.5	

3.4 Community indexes

Two community indexes were calculated each season using the data from both sample methods. These community indexes are the index of community importance (ICI; Eq. 1) and the index of relative importance (IRI; Eq. 2).

3.4.1 Index of community importance (ICI)

One species held the most important value during every sampling season by ICI, the spotted sandbass, *P. maculatofasciatus* (Table 5). However, during summer 2018, the highest value was shared among *P. maculatofasciatus* and the sandbass, *P. nebulifer*. This last species was the second most important species during all other seasons. Consistently the third most important species under ICI was the pacific porgy, *C. brachysomus*. All three of these species are of subtropical distribution as seen in the species listings (Table 5). Following these three species is *P. clathratus*.

3.4.2 Index of relative importance (IRI)

The highest values of IRI come from mainly two species: *P. maculatofasciatus* –which has the highest values in winter 2018, spring 2018, and winter 2019– and *P. nebulifer* –which has the highest values in summer and autumn 2018 (Table 5). Third, after these previously mentioned species is once again, *C. brachysomus* (Table 5). Fourth place was occupied by the kelpbass, *Paralabrax clathratus*.

Table 5. Species ICI and IRI values. Index of Community Importance (ICI) and Index of Relative Importance (IRI) for the fish species identified in SCUBA transects or trap capture during the sampling seasons. In bold are the most important values for each index each season.

Species	Sampling Date									
	Winter 2018 (March 8)		Spring 2018 (June 8)		Summer 2018 (August 18)		Autumn 2018 (November 15)		Winter 2019 (February 13)	
	ICI	IRI	ICI	IRI	ICI	IRI	ICI	IRI	ICI	IRI
<i>Paralabrax maculatofasciatus</i>	2	10076.1	3	4990.3	3	2739.8	2	3986	2	8051.2
<i>Paralabrax nebulifer</i>	4	4076.9	4	4952.5	3	4831.6	4	4382.6	4	1587.5
<i>Paralabrax clathratus</i>	8	157.3	9	190.9	10	147.8	8	269.8	9	279.2
<i>Calamus brachysomus</i>	6	1694.7	5	4070.2	7	844.6	6	1705.5	6.5	656.1
<i>Anisotremus interruptus</i>	--	--	18.5	2.4	--	--	10	10.8	--	--
<i>Halichoeres semicinctus</i>	10	52.5	9	131.8	12.5	40	--	--	8.5	212.2
<i>Hypsypops rubicundus</i>	13.5	8.2	--	--	--	--	--	--	--	--
<i>Semicossyphus pulcher</i>	12.5	7.5	18.5	6.9	--	--	--	--	--	--
<i>Heterodontus francisci</i>	--	--	12	92.8	21.5	17.1	--	--	--	--
<i>Sphoeroides annulatus</i>	--	--	18.5	2.6	16.5	13.9	--	--	12	18.7
<i>Anisotremus davidsonii</i>	--	--	16	2.8	11	227.7	--	--	--	--
<i>Pseudobatos productus</i>	--	--	18.5	21.2	--	--	--	--	14	26.5
<i>Balistes polylepis</i>	--	--	--	--	23	5.1	--	--	--	--
<i>Eucinostomus currani</i>	--	--	--	--	10.5	52.2	--	--	--	--
<i>Mycteroperca xenarcha</i>	--	--	--	--	16.5	82.7	--	--	--	--
<i>Urolophus halleri</i>	--	--	--	--	21.5	5.8	--	--	--	--

3.5 Correlation between environmental and biological factors

Spearman's rank correlation was used to determine if there was any correlation between the abundance and biomass data with the three environmental variables (seawater temperature, salinity, and seawater temperature).

Trap Capture

Abundance had a low negative correlation ($r=-0.24$, $p=0.005$) with temperature (Figure 15; Table 6). Biomass had a similar low negative correlation ($r=-0.2$, $p=0.02$) with seawater temperature (Figure 16; Table 6).

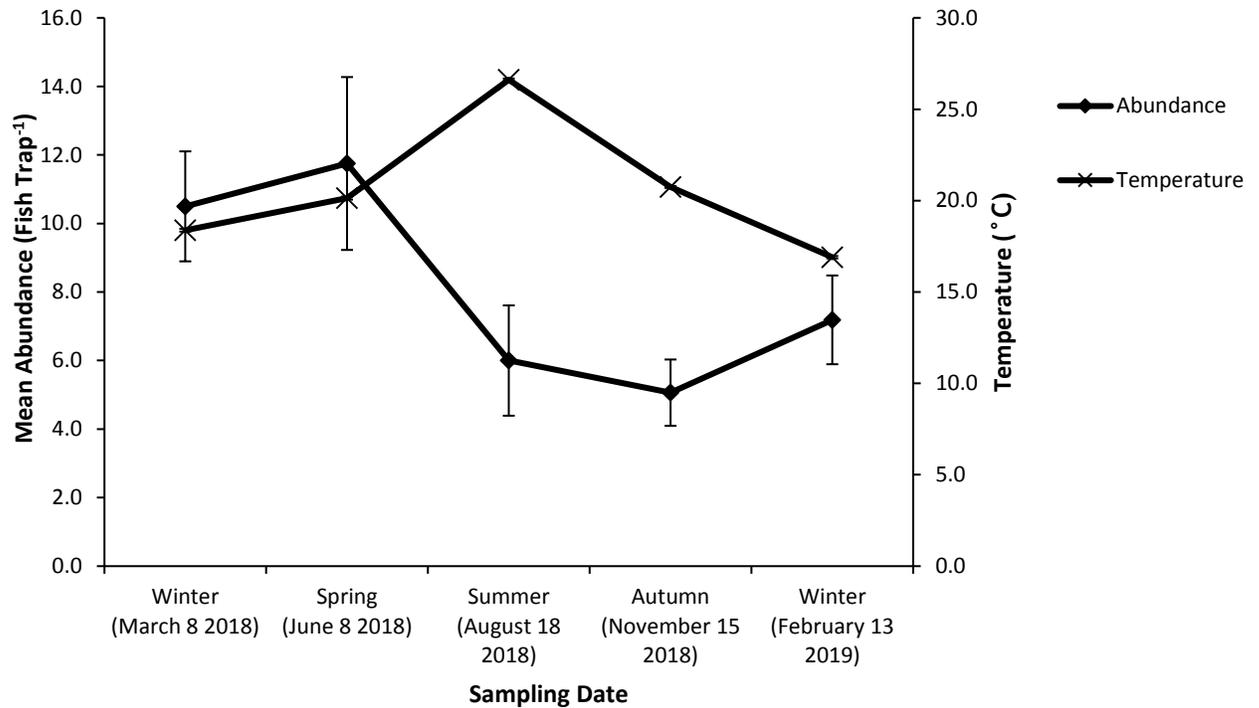


Figure 15. Abundance and temperature by season. Seasonal means of abundance (fish trap⁻¹) captured with traps and seawater temperature (°C) for the whole study that shows the negative correlation between abundance and temperature. The bars represent the standard error.

SCUBA Census

Abundance had a low negative correlation value ($r=-0.36$, $p=0.002$) with dissolved oxygen (Table 6) and a low positive correlation ($r=0.33$, $p=0.004$) with seawater temperature (Table 6). Also, abundance had a moderate positive correlation ($r=0.6$, $p<0.001$) with salinity (Table 6).

Biomass had a moderate positive correlation ($r=0.64$, $p<0.001$) with salinity (Table 6), and a low positive correlation ($r=0.32$, $p=0.006$) with seawater temperature (Table 6).

Table 6. Correlations between abundance and biomass with physical parameters. Spearman's correlation coefficient (r_s) with their significance p-value in parenthesis for all three environmental variables: dissolved oxygen, salinity, and seawater temperature, with the abundance and biomass with each sampling method. In bold are the significant values.

	<i>Trap Capture</i>		<i>SCUBA</i>	
	Abundance	Biomass	Abundance	Biomass
<i>Dissolved Oxygen</i>	0.11 (0.203)	0.01 (0.914)	-0.36 (0.002)	-0.18 (0.12)
<i>Salinity</i>	0.12 (0.16)	-0.16 (0.07)	0.6 (<0.001)	0.64 (<0.001)
<i>Seawater temperature</i>	-0.24 (0.005)	-0.2 (0.02)	0.33 (0.004)	0.32 (0.006)

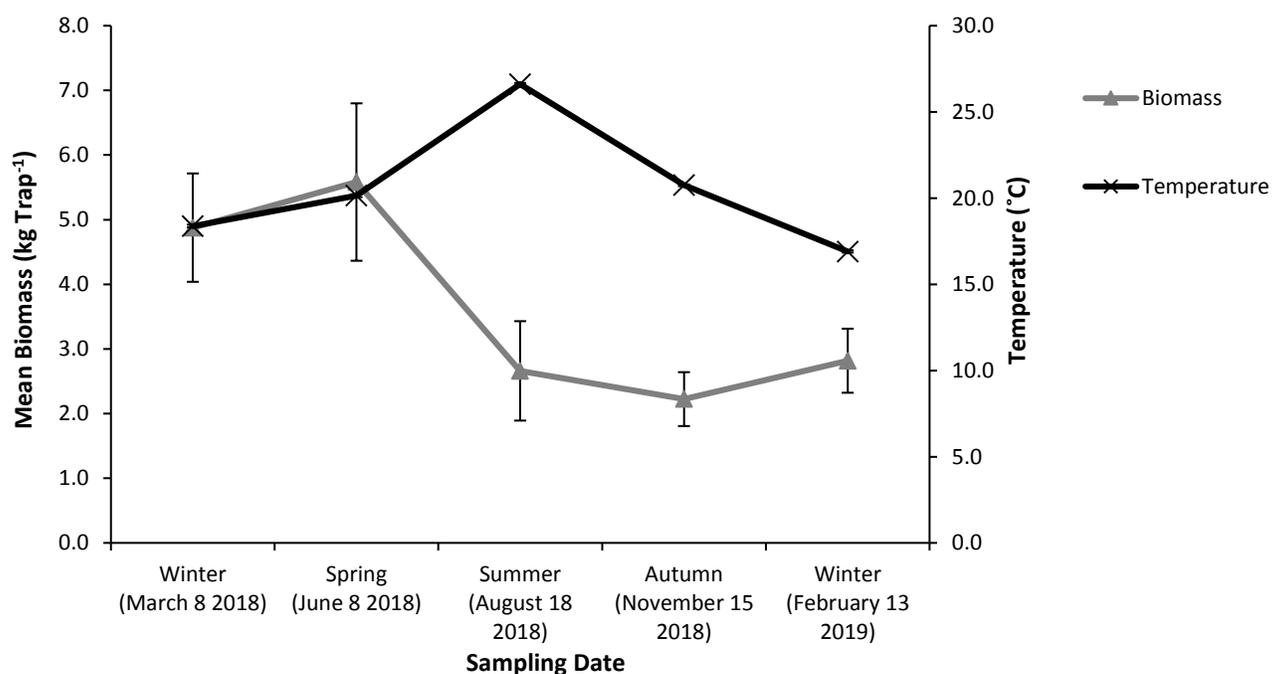


Figure 16. Biomass and temperature by season. Seasonal means of biomass captured with traps (kg trap⁻¹) and seawater temperature (°C) for all study sites. The bars represent the standard error.

3.6 Diversity

For the visual transects data, a SHE analysis was conducted to determine if the species frequency was sampled from the same statistical distribution (community) or from a different distribution. Species richness (S), Buzas-Gibson's evenness (E), and the diversity index (H) were then calculated for each

statistical distribution obtained. The Shannon-Weiner diversity index (H') and Pielou's equitability (J') were calculated for the trap capture data.

3.6.1 SHE analysis for the SCUBA census

SHE analysis for community structure (SHECSI)

There was the required linear tendency of the $\ln E$ (Buzas-Gibson Evenness) versus $\ln N$ (number of individuals) regression at the RR site for a single community during all the survey periods of this study (Figure 17). A characteristic of a log series distribution is that the slopes of the equations for $\ln E$ and $\ln S$ (number of species) should be equal in magnitude but with opposite signs. The slopes of the regression lines of $\ln S$ versus $\ln N$ and $\ln E$ versus $\ln N$ at the RR site are not precisely equal in magnitude (0.32, and -0.22 respectively, important to note these are tendencies without statistical tests) but are opposite in sign (Figure 17).

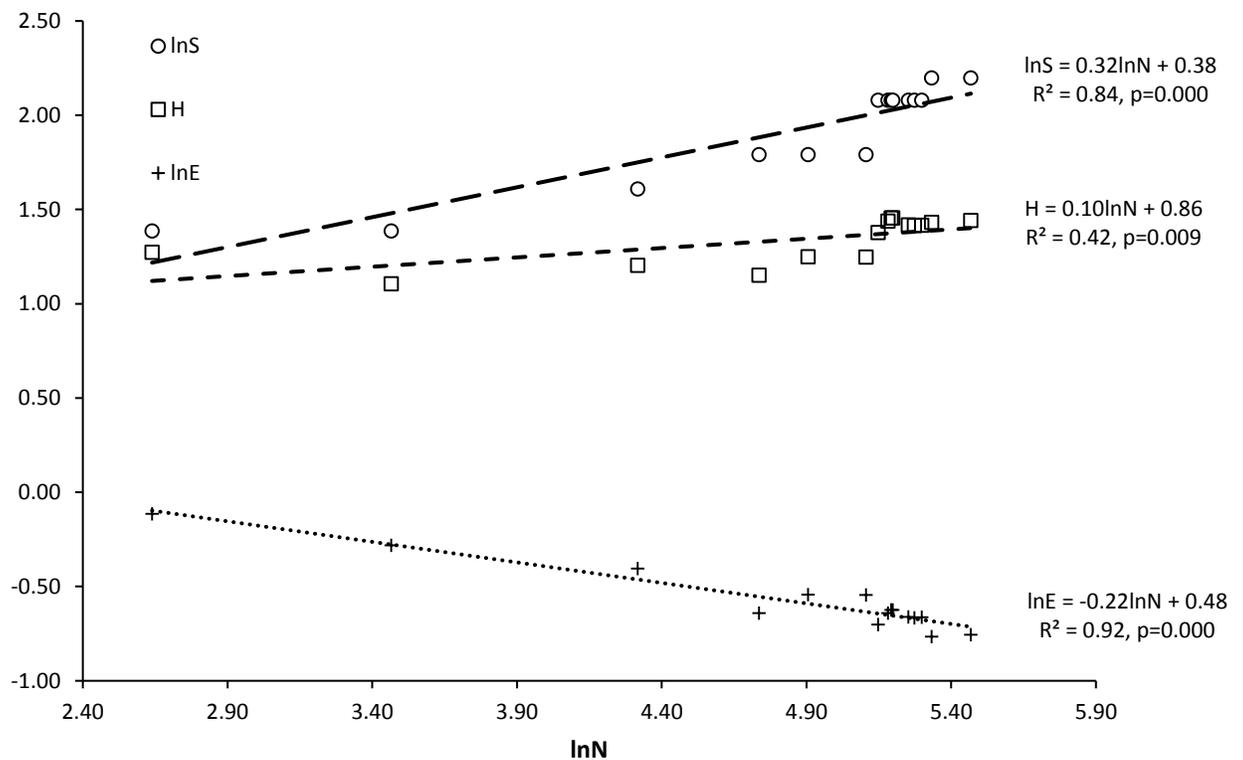


Figure 17. Rocky reef (RR) SHE analysis. SHE analysis of the RR site with SCUBA census. Linear regressions for $\ln S$ (S : species richness), H (diversity index), and $\ln E$ (E : Buzas-Gibson evenness) versus $\ln N$ (N : number of fishes) with their corresponding R^2 and p values.

The slope of graph $\ln S$ versus $\ln E$ should be -1 for a perfect log series distribution yet at the RR site it is -1.43 (Figure 18). The intercept of the regression of $\ln S$ versus $\ln E$ (Figure 18) is 1.09 , while our mean H is 1.33 . Evenness (E) and equitability (J') have a decreasing pattern, while Fisher's α and Simpson's $1/\lambda$ have what appears to be a possible constant pattern (Figure 39). At the RR site, the community is not from a log normal distribution, yet not a perfect log series, the results are between a log series and log normal distribution.

The AR site had a break in the linear tendency of $\ln E$ versus $\ln N$ after the third point (Figure 19). Between the third and fourth point is the switch between sampling seasons, the third point representing the final sample of winter 2018, and the fourth point representing the first sample of spring 2018.

The slopes of $\ln S$ and $\ln E$ are almost perfect negatives reciprocals (0.37 , -0.30 respectively) for the samples of winter 2018 at the AR site (Figure 20). While the graph of $\ln S$ versus $\ln E$ gives us a negative -1.24 slope, which is near to -1 , the intercept of 0.00 is near the mean H of the samples of 0.09 (Figure 21). The E index has a decreasing trend, while J' has an increasing trend, α and $1/\lambda$ appear to be constant (Figure 40). The AR site winter 2018 community has an imperfect log series distribution.

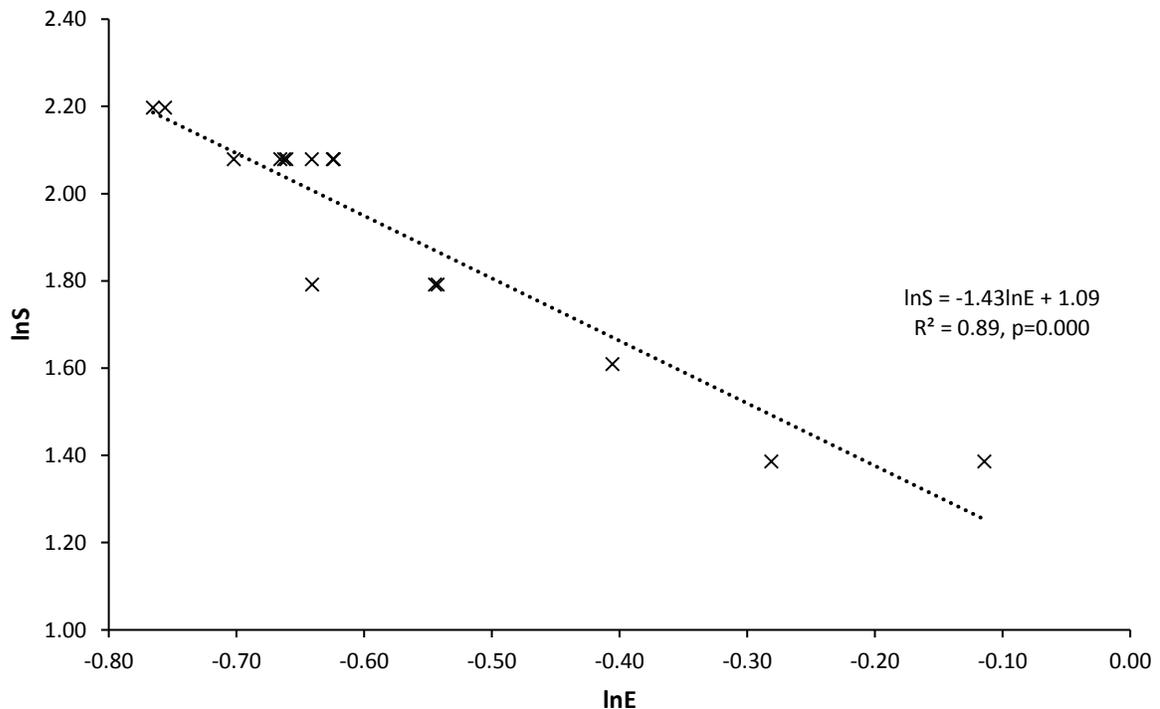


Figure 18. $\ln S$ versus $\ln E$ regression at the rocky reef (RR) site. Linear regression of $\ln S$ (S : species richness) versus $\ln E$ (E : Buzas-Gibson's evenness) at the RR site from the SCUBA census with the corresponding R^2 and p values.

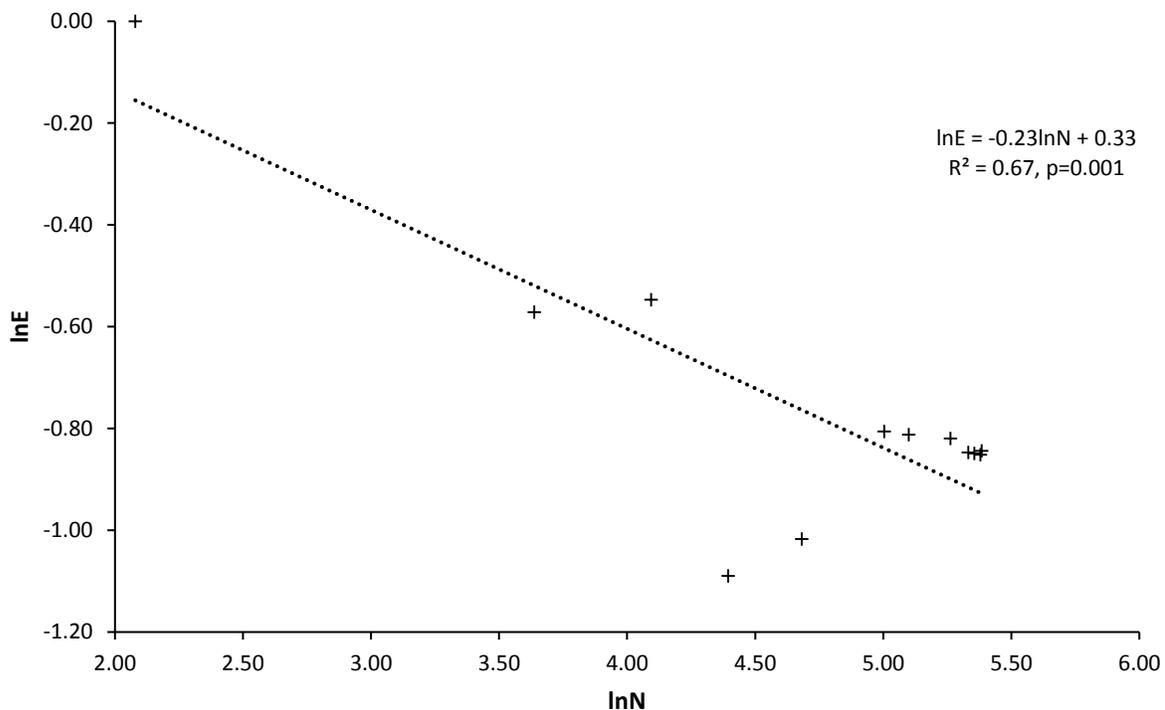


Figure 19. Regression of lnE versus lnN for the artificial reef (AR) site. A regression of the lnE (E: Buzas-Gibson's evenness) versus lnN (N: number of fishes) of the AR site, which shows a break in the linear trend at the third point. The regression equation with its corresponding R^2 and p values is shown.

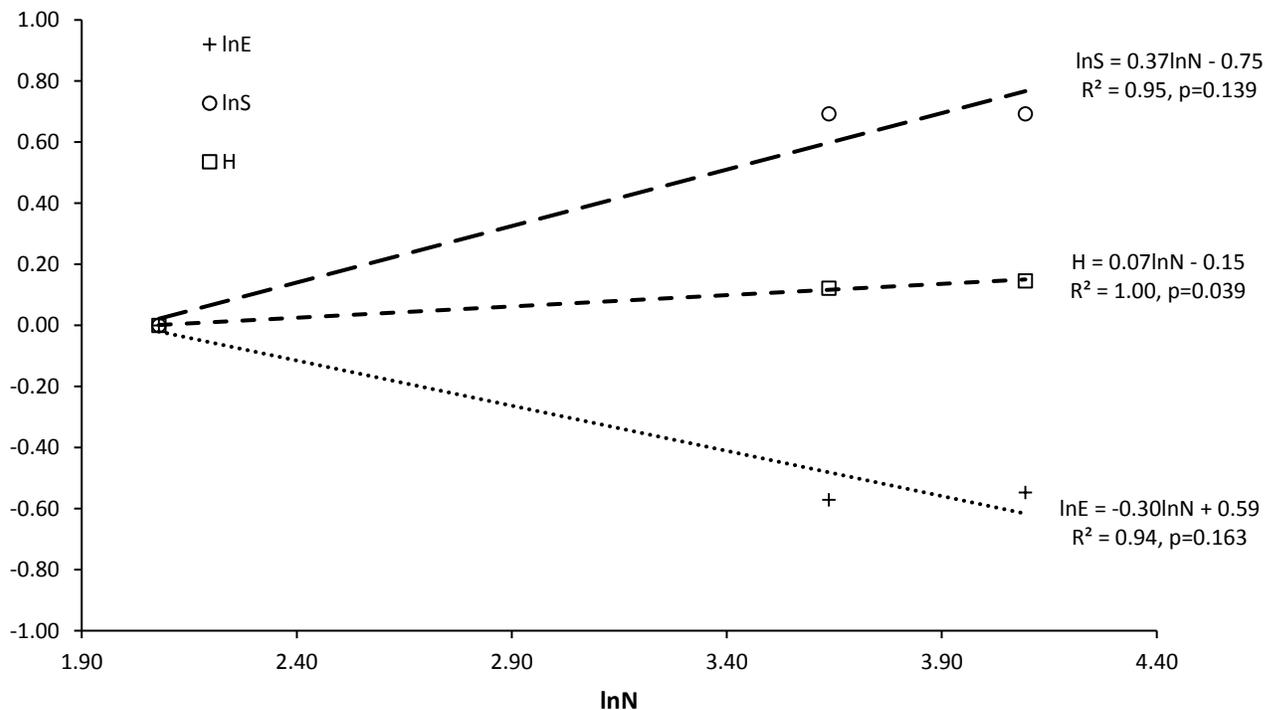


Figure 20. Winter 2018 SHE analysis of the artificial reef site (AR). SHE analysis for the AR site with only the data from winter 2018. The linear regressions of lnS (S: species richness), H (diversity index), lnE (Buzas-Gibson's evenness) versus lnN (N: number of fishes) with their corresponding equations, R^2 and p values.

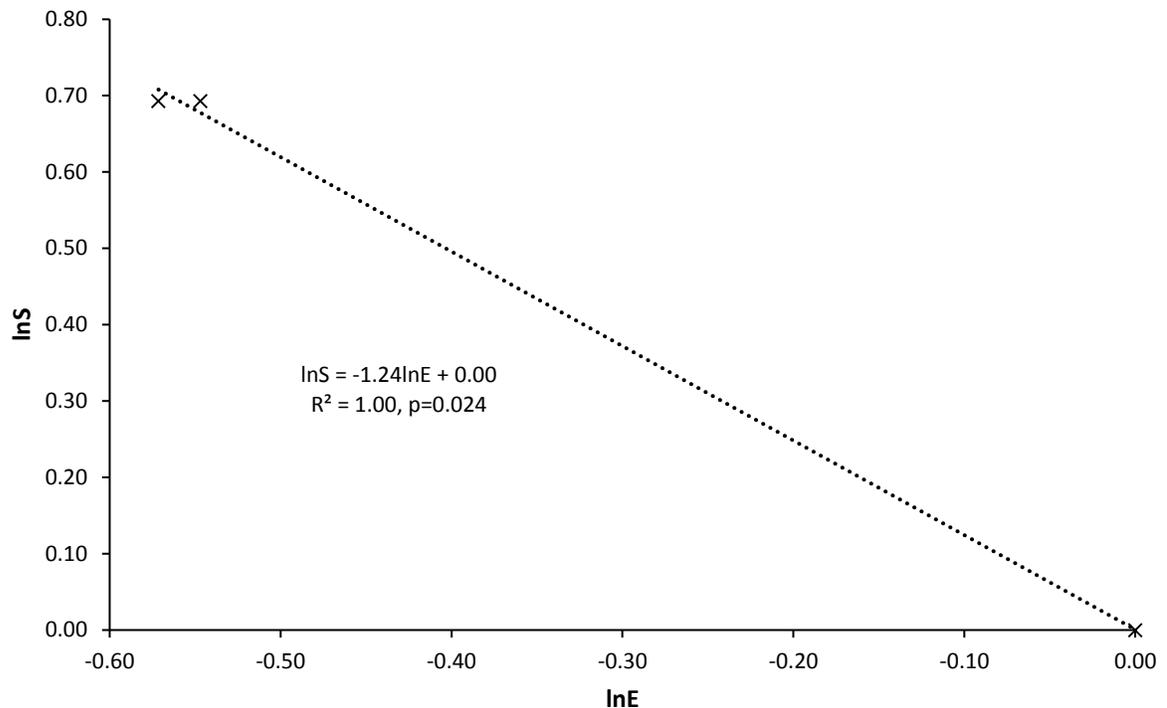


Figure 21. lnS versus lnE regression at the artificial reef (AR) site. A regression with its linear equation for lnS (S: species richness) versus lnE (E: Buzas-Gibson's evenness) at the AR site in the winter 2018 sampling. R^2 and p values for the linear regression equation are shown.

After the reanalysis of the data post-winter 2018, a linear trend in the regression line for lnE shows a singular community at the AR site (Figure 22). The slopes for the equation of lnS and lnE are near similar in magnitude (0.09 and -0.16, respectively) and of opposite sign (Figure 22). The E and J' indexes have a slightly decreasing trend, while $1/\lambda$ and α have a more pronounced decreasing trend (Figure 41). The slope of lnS versus lnE is -0.53, slightly different from the required -1 for a perfect log series (Figure 23). H is statistically constant at 1.49 (Figure 22). The mean H for all samples is 1.20, while the intercept of lnS versus lnE is 1.47 (Figure 23). The statistical distribution for all sampled seasons except winter 2018 at the AR site is an imperfect log series.

A SHE analysis of winter 2018 between both the RR and AR site revealed a linear trend in the lnE versus lnN regression (Figure 24), indicating a singular community. The slopes of the regression line for lnS (0.12) and lnE (-0.20), are similar in magnitude and have opposite signs (Figure 24). The slope of lnS versus lnE is -0.56 (Figure 25), and the regression has an intercept of 1.31, which is slightly different from the mean H of 1.14. Both E and J' have a decreasing trend as well as α and $1/\lambda$ (Figure 42). The winter 2018 community between both the RR and AR sites has a distribution of an imperfect log series.

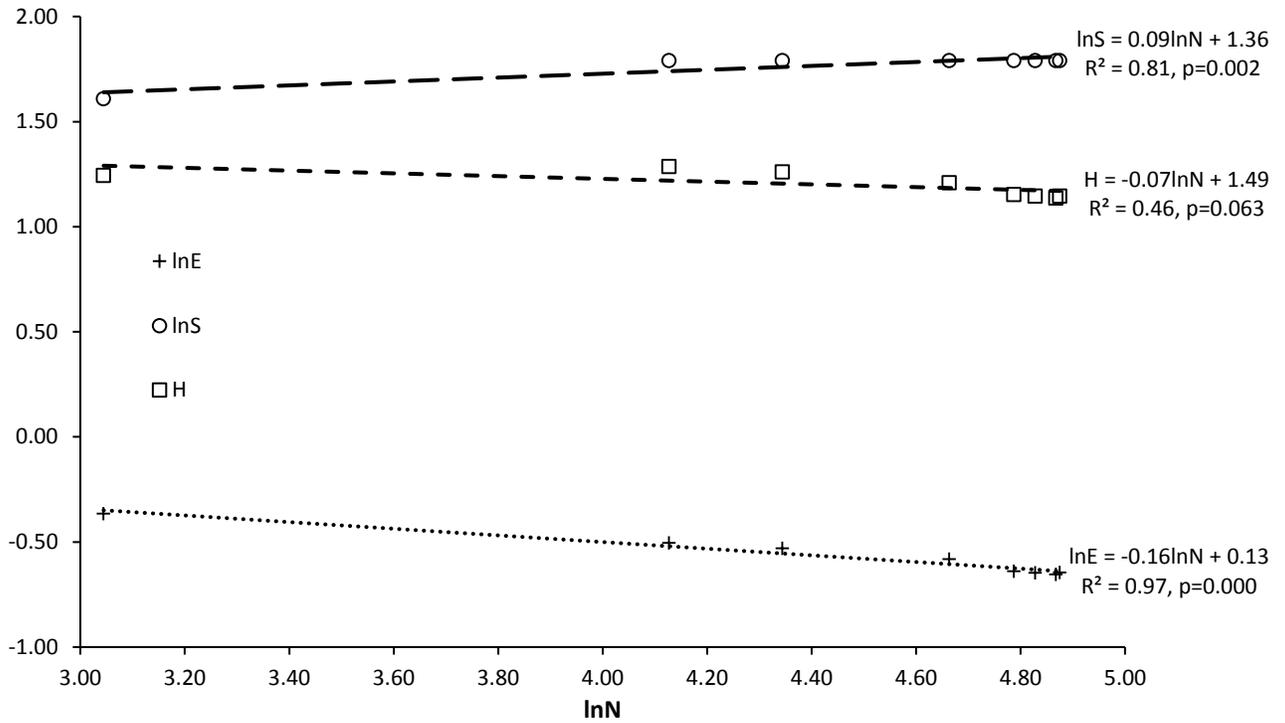


Figure 22. SHE analysis of the artificial reef site (AR) from spring 2018 to winter 2019. SHE analysis for the AR site of spring and summer of 2018, and winter 2019. Linear regressions of lnS (S: species richness), H (diversity index), lnE (E: Buzas-Gibson’s evenness) versus lnN (N: number of fishes) with their corresponding equations, R² and p values.

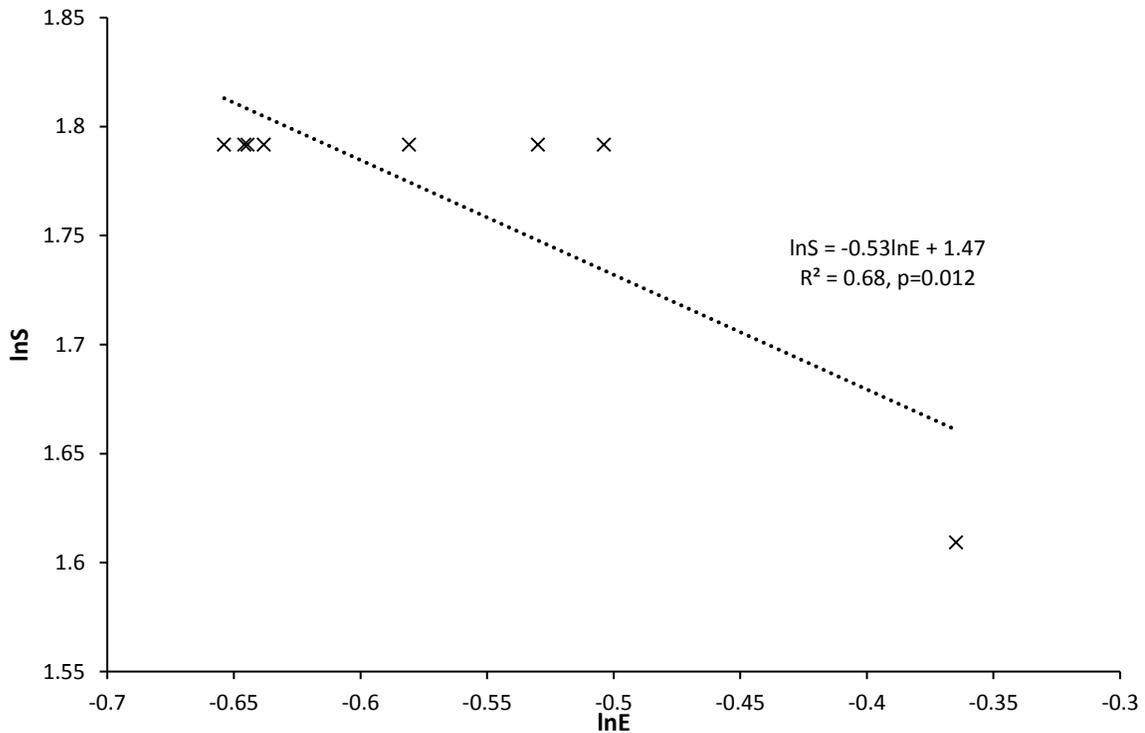


Figure 23. lnS versus lnE regression at the artificial reef (AR) site for spring 2018 thru winter 2019. A regression with its linear equation for lnS (S: species richness) versus lnE (E: Buzas-Gibson’s evenness) at the AR site for the spring and summer 2018, and winter 2019 sampling. R² and p values for the linear regression equation are shown.

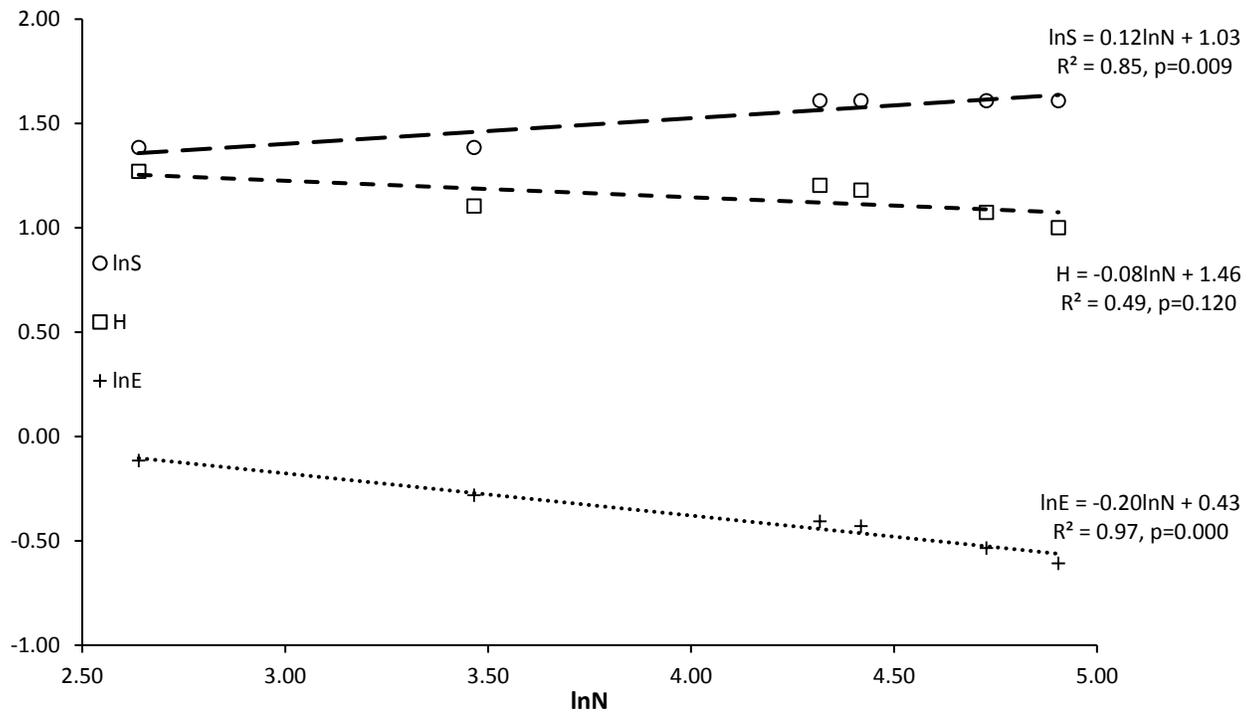


Figure 24. Winter 2018 SHE analysis for the artificial reef (AR) and rocky reef (RR) sites. SHE analysis for both the AR and RR site in the winter of 2018. Linear regressions of $\ln S$ (S: species richness), H (diversity index), $\ln E$ (E: Buzas-Gibson's evenness) versus $\ln N$ (N: number of fishes) with their corresponding equations, R^2 and p values are shown.

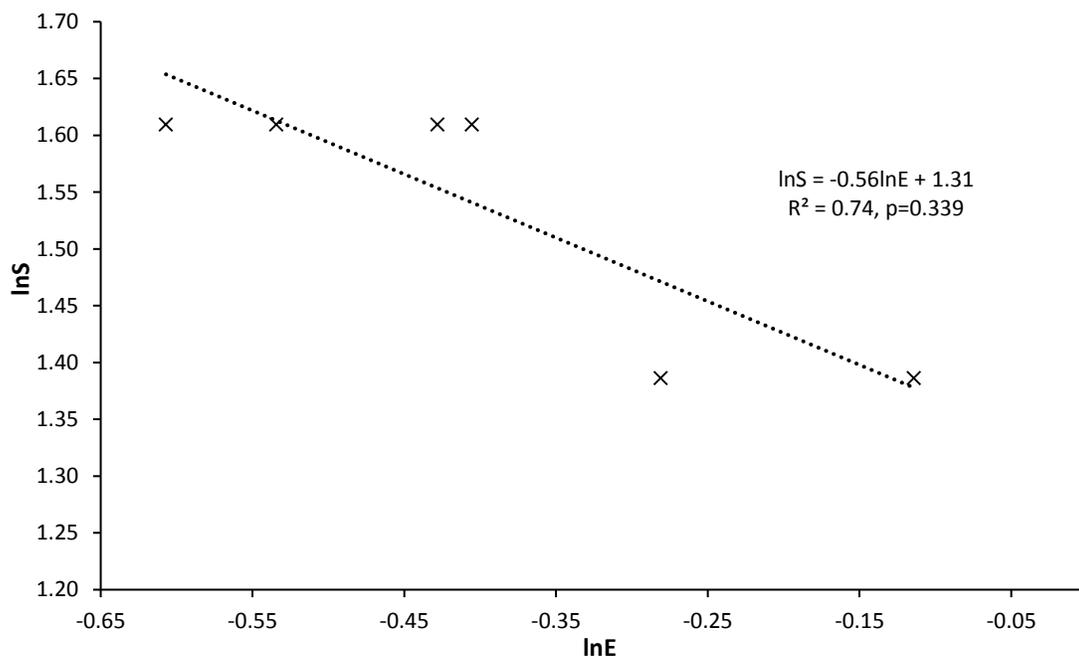


Figure 25. Winter 2018 regression of $\ln S$ versus $\ln E$ for the artificial reef (AR) site and rocky reef (RR) site. A regression with its linear equation for $\ln S$ (S: species richness) versus $\ln E$ (E: Buzas-Gibson evenness) for both the RR and AR sites in the winter 2018 sampling. R^2 and p values for the linear regression equation are shown.

Only taking into consideration the data for the spring of 2018, which is for the RR and AR site, the linear trend of $\ln E$ versus $\ln N$ stops after the third point (Figure 26). From the third point to the fourth point is the change between the RR and AR site, which reconfirms the observations from Figures 19, 20, 22, and 24 of a change between the AR and RR community occurring between winter and spring 2018.

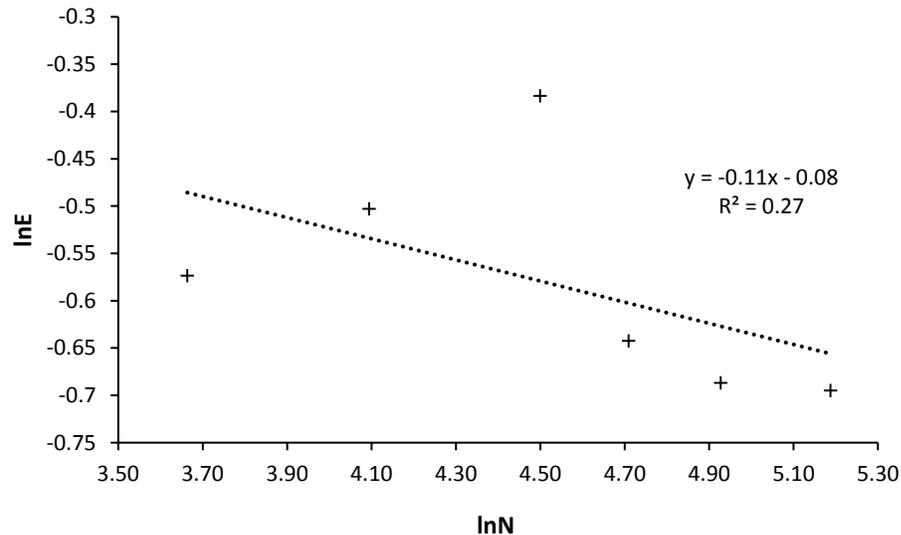


Figure 26. Regression of $\ln E$ versus $\ln N$ for spring 2018. Regression line of $\ln E$ (E: Buzas-Gibson's evenness) versus $\ln N$ (N: number of fishes) for the spring 2018 survey. The linear trend is seen broken after the third point in the data set, confirming the change in the artificial reef site community.

The line for $\ln E$ versus $\ln N$ for the summer 2018 sampling period followed a linear trend, indicating a single community during this sampling season (Figure 27). Buzas-Gibson's E had a decreasing trend while Pielou's J' remained relatively constant, α , $1/\lambda$ (Figure 43), and H increase (Figure 27) with $\ln N$. The rate at which H increased can be deduced by the difference in the $\ln S$ versus $\ln E$, as the slope for this graph is -2.51, instead of the -1 required for a constant H (Figure 28). The statistical distribution that fits these parameters is the log normal.

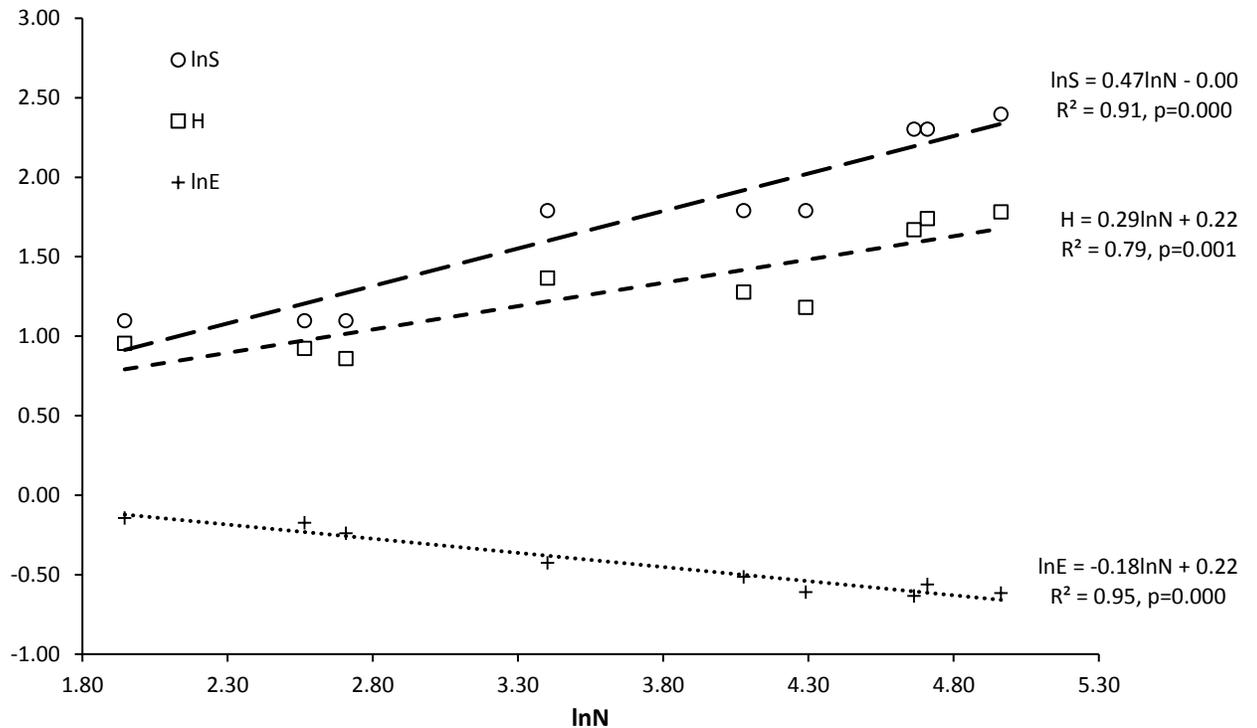


Figure 27. SHE analysis of the artificial reefs (AR), rocky reef (RR), and soft bottom (SB) sites during summer 2018. SHE analysis for the AR, RR, and SB site in the summer of 2018. Linear regressions of lnS (S: species richness), H (diversity index), lnE (E: Buzas-Gibson's evenness) versus lnN (N: number of fishes) with their corresponding equations, R^2 and p values are shown.

The SHE analysis for autumn 2018 was not done due to only the RR site being sampled. During winter 2019, the linear trend of lnE versus lnN was broken after the third sample, between the third and fourth sample was the change between the RR and AR sites (Figure 29). After reanalyzing from the fourth point on, there was a break in the linear trend after the third point (Figure 30), this represents the change between the AR and SB sites. All three sites had their own communities during this sampling season. From previous SHE analysis, we know that both the RR and AR communities are different log series communities from spring 2018 onward in our samples. The SB was sampled in only two seasons, summer 2018 and winter 2019. There is no linear pattern in the samples from summer 2018 and winter 2019 (Figure 31). However, SHE analysis of just winter 2019 of the SB site shows a linear trend in lnE versus lnN (Figure 32) suggesting a single community. The slope of lnS versus lnE is -5.03 (Figure 33), making it too high for a log series community.

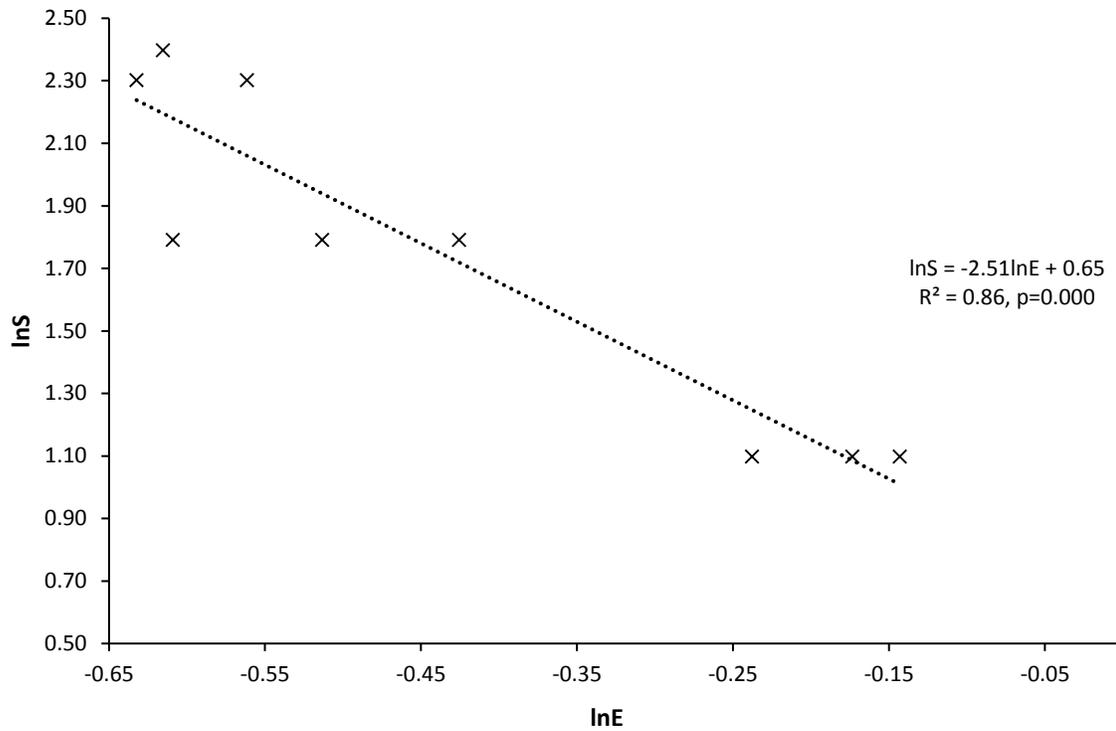


Figure 28. Regression of ln S versus ln E for summer 2018. A regression with its linear equation for lnS (S: species richness) versus lnE (E: Buzas-Gibson's evenness) for the rocky reef (RR), artificial reef (AR), and soft bottom (SB) sites in the summer 2018 sampling. R² and p values for the linear regression equation are shown.

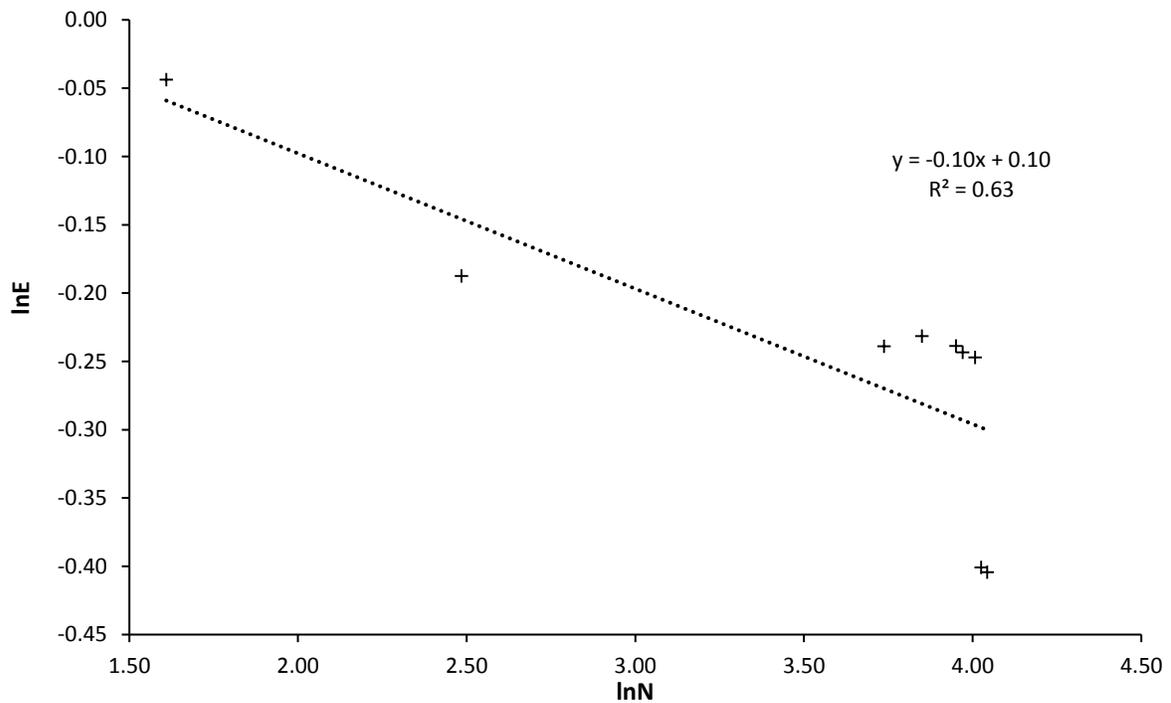


Figure 29. Winter 2019 regression of ln E versus ln N for all sites. The regression of lnE (E: Buzas-Gibson's evenness) versus lnN (N: number of fishes) for the three sites, rocky reef, artificial reef, and soft bottom, for the winter of 2019 in which the linear pattern is broken between the third and fourth point.

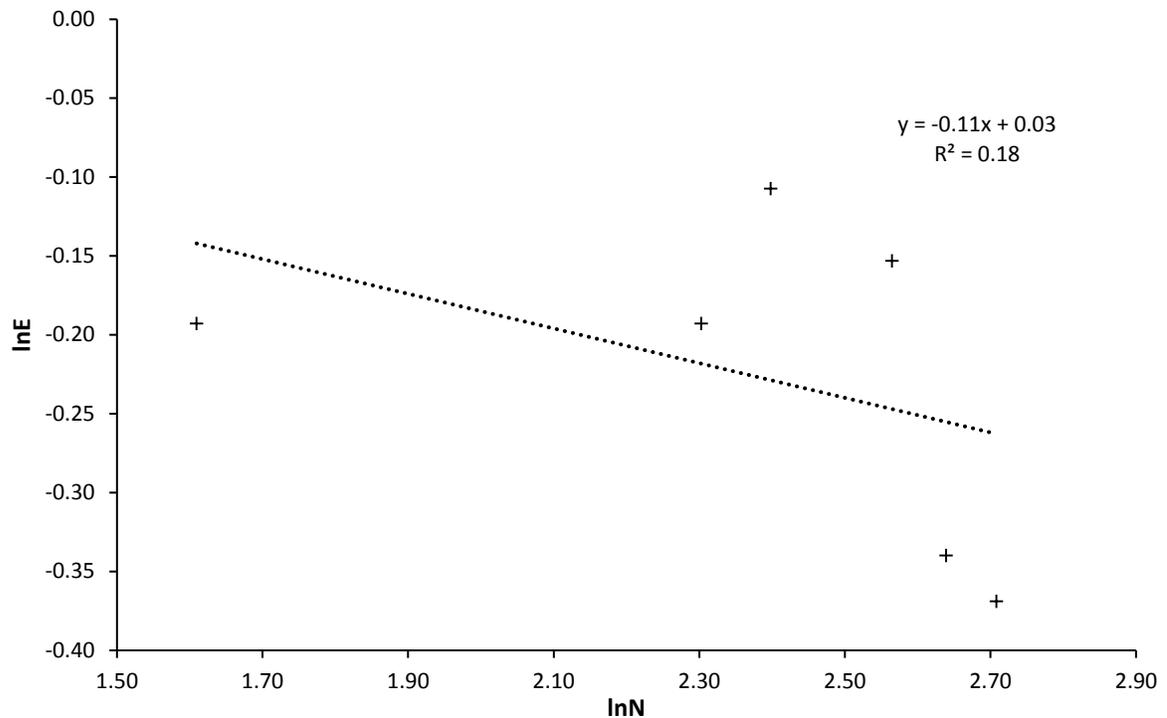


Figure 30. Winter 2019 regression of lnE versus lnN for the artificial reef (AR) and soft bottom (SB) sites. The regression of lnE (E: Buzas-Gibson's evenness) versus lnN (N: number of fishes) for two sites, AR and SB, for the winter of 2019 in which the linear pattern is broken between the third and fourth point.

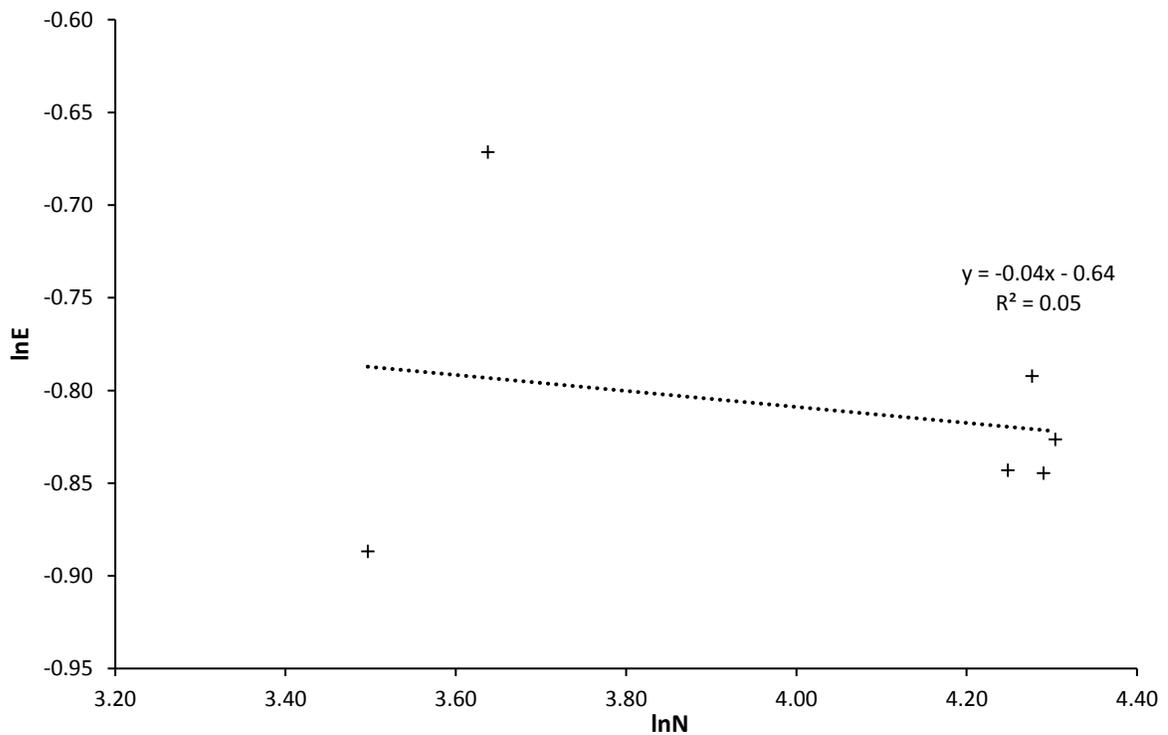


Figure 31. Regression of lnE versus lnN of the soft bottom (SB) site. Regression line of lnE (E: Buzas-Gibson's evenness) versus lnN (N: number of fishes) of the SB site during both its surveys, summer 2018 and winter 2019. No clear linear trend between three points is discernable.

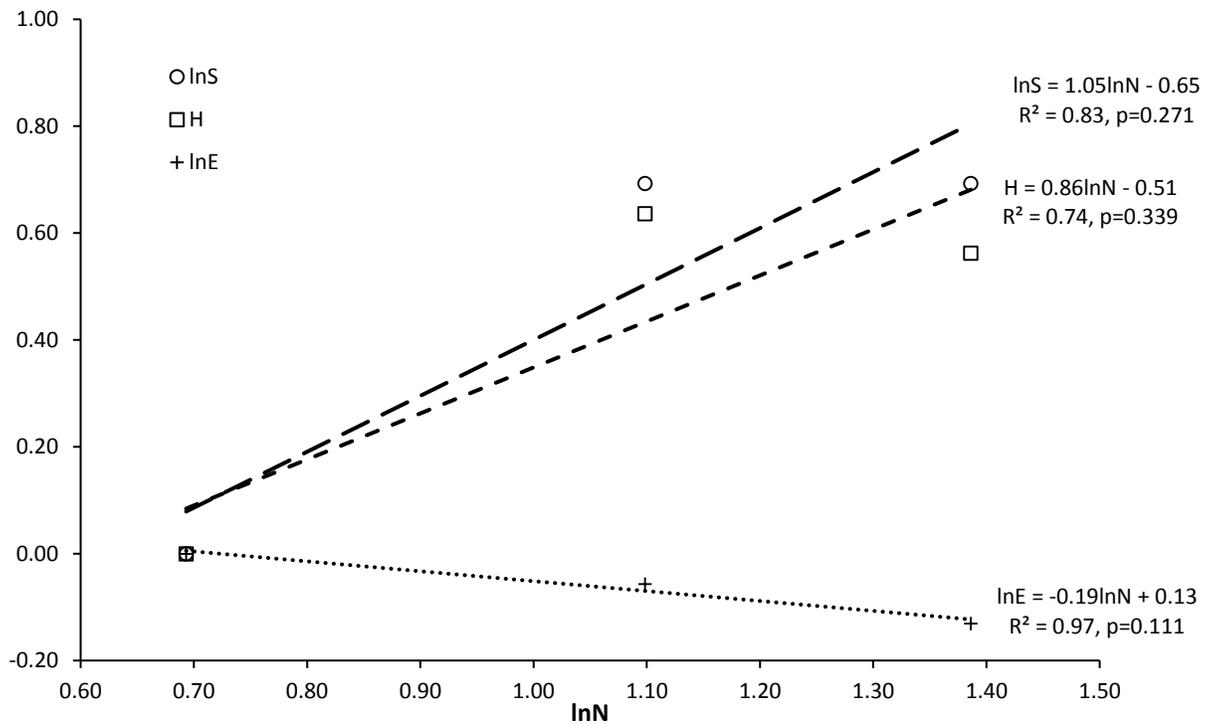


Figure 32. SHE analysis of the soft bottom (SB) site during winter 2019. Linear regressions of lnS (S: species richness), H (diversity index), lnE (E: Buzas-Gibson's evenness) versus lnN (N: number of fishes) with their corresponding equations, R² and p values are shown.

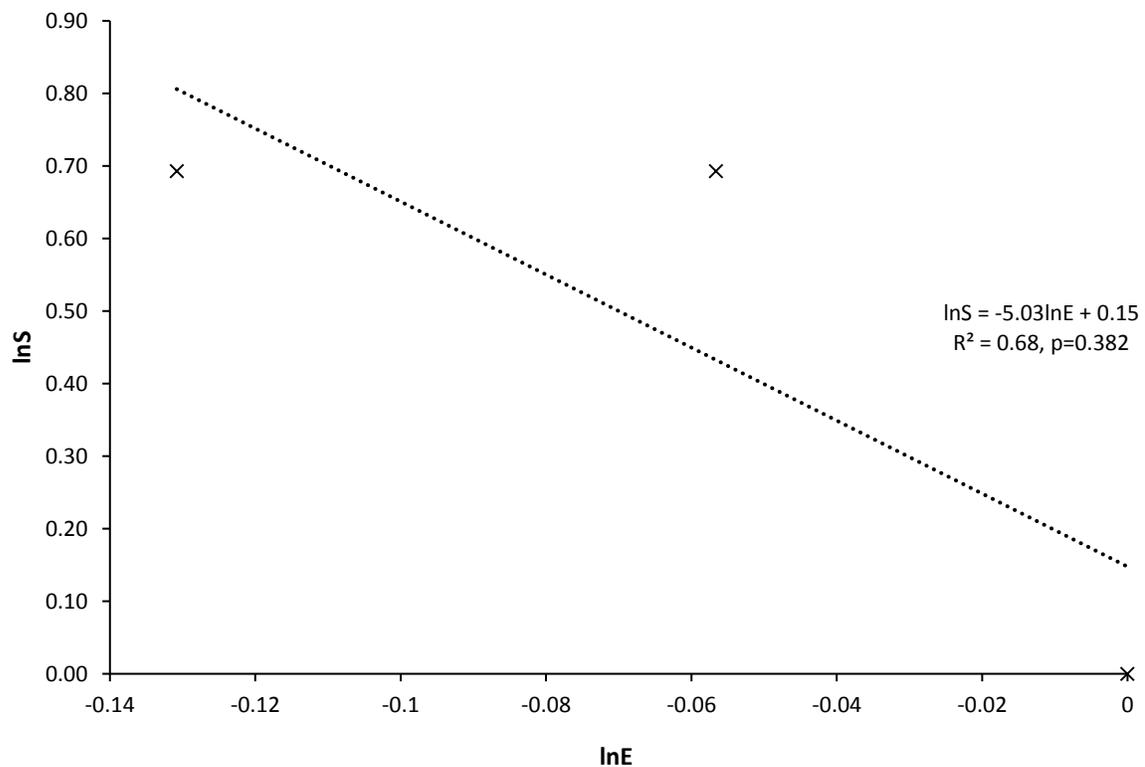


Figure 33. Regression of lnS versus lnE of the soft bottom (SB) site during winter 2019. A regression with its linear equation for lnS (S: species richness) versus lnE (E: Buzas-Gibson's evenness) for the SB site for the winter 2019 sampling. R² and p values for the linear regression equation are shown.

In the SB community of winter 2019, the E index remains constant, while H (Figure 32), J' , α , $1/\lambda$ (Figure 44) all have an increasing trend. These are the characteristics of a broken stick distribution.

Comparing between communities

Estimates based on equal sample sizes on the RR community, the AR community from spring onward, the Winter 2018 community, the Summer 2018 community, and the Winter 2019 SB community show that species richness was highest in the summer community, followed by the RR community, the lowest species richness was in the winter 2018 community, the AR community had the second-lowest species richness (Table 7). Comparing the decomposition equation of H between communities, and estimated values of $\ln S$ and $\ln E$ from the regressions, we find that both the winter 2018 and RR communities have the highest evenness followed by the AR reef community (Table 7). The summer 2018 community has the highest diversity primarily due to its species richness since it has low evenness. The winter 2019 community had the lowest evenness.

Table 7. Community values of species richness (S), Buzas-Gibson's evenness (E), and the decomposition H. Values of S at equal sample sizes ($\ln N = 2, 3, 4$) using the equivalent alpha method for abundifying and rarefying, and the ratio of log relationships method for each of the communities. Evenness estimates are given in \ln of Buzas-Gibson's E from the regression equations for the sample sizes used to estimate species richness. The decomposition equation for H with its estimates of $\ln S$ and $\ln E$ from the regression equations from the sample sizes for species richness. Underlined are the values that fall within the range of the sample number of the regression or is closest to it.

	RR community (Log series) $\alpha=1.8$	AR Spring 2018 and onward community (Log series) $\alpha=1.3$	Winter 2018 community (Log series) $\alpha=1$	Summer 2018 community (Log normal) $11/\ln(143)$	Winter 2019 SB community (Broken Stick) $2/\ln(4)$
Species Richness (S)	$S_2=2.93$ $S_3=4.50$ $S_4=6.20$	$S_2=2.47$ $S_3=3.64$ $S_4=4.89$	$S_2=2.12$ $S_3=3.05$ $S_4=4.02$	$S_2=4.43$ $S_3=6.65$ $S_4=8.86$	$S_2=2.88$ $S_3=4.33$ $S_4=5.77$
Evenness ($\ln E$)	$\ln E_2 = -0.04$ <u>$\ln E_3 = -0.18$</u> <u>$\ln E_4 = -0.40$</u>	$\ln E_2 = -0.19$ $\ln E_3 = -0.35$ <u>$\ln E_4 = -0.51$</u>	$\ln E_2 = 0$ <u>$\ln E_3 = -0.17$</u> <u>$\ln E_4 = -0.37$</u>	<u>$\ln E_2 = -0.14$</u> <u>$\ln E_3 = -0.32$</u> <u>$\ln E_4 = -0.5$</u>	<u>$\ln E_2 = -0.25$</u> $\ln E_3 = -0.44$ $\ln E_4 = -0.63$
Decomposition equation $H = \ln S + \ln E$	$H_2 = 1.02 + (-0.04) =$ 0.98 <u>$H_3 = 1.34 + (-0.18) =$</u> <u>1.16</u> <u>$H_4 = 1.66 + (-0.40) =$</u> <u>1.26</u>	$H_2 = 1.54 + (-0.19) =$ 1.35 $H_3 = 1.63 + (-0.35) =$ 1.28 <u>$H_4 = 1.72 + (-0.51) =$</u> <u>1.21</u>	$H_2 = 1.27 + (0) =$ 1.27 <u>$H_3 = 1.39 + (-0.17) =$</u> <u>1.22</u> <u>$H_4 = 1.51 + (-0.37) =$</u> <u>1.14</u>	<u>$H_2 = 0.94 + (-0.14) =$</u> <u>0.80</u> <u>$H_3 = 1.41 + (-0.32) =$</u> <u>1.09</u> <u>$H_4 = 1.88 + (-0.5) =$</u> <u>1.38</u>	<u>$H_2 = 1.45 + (-0.25) =$</u> <u>1.2</u> $H_3 = 2.5 + (-0.44) =$ 2.06 $H_4 = 3.55 + (-0.63) =$ 2.92

3.6.2 Shannon–Weiner diversity index (H') and Pielou's evenness index (J') for trap capture samples

Both autumn and spring 2018 had the highest Shannon-Weiner diversity (H') for all sampling seasons, with values of 1.30, and 1.28, respectively (Figure 34). Summer 2018 had the lowest diversity of 1.06. Both winters also had low diversity indexes (H') of 1.10 for winter 2018, and 1.09 for winter 2019 (Figure 34). Equitativity or evenness measured by Pielou's (J') index was highest in autumn 2018 with a value of 0.81 and lowest in spring and summer of 2018 with values of 0.62 and 0.66 respectively (Figure 34).

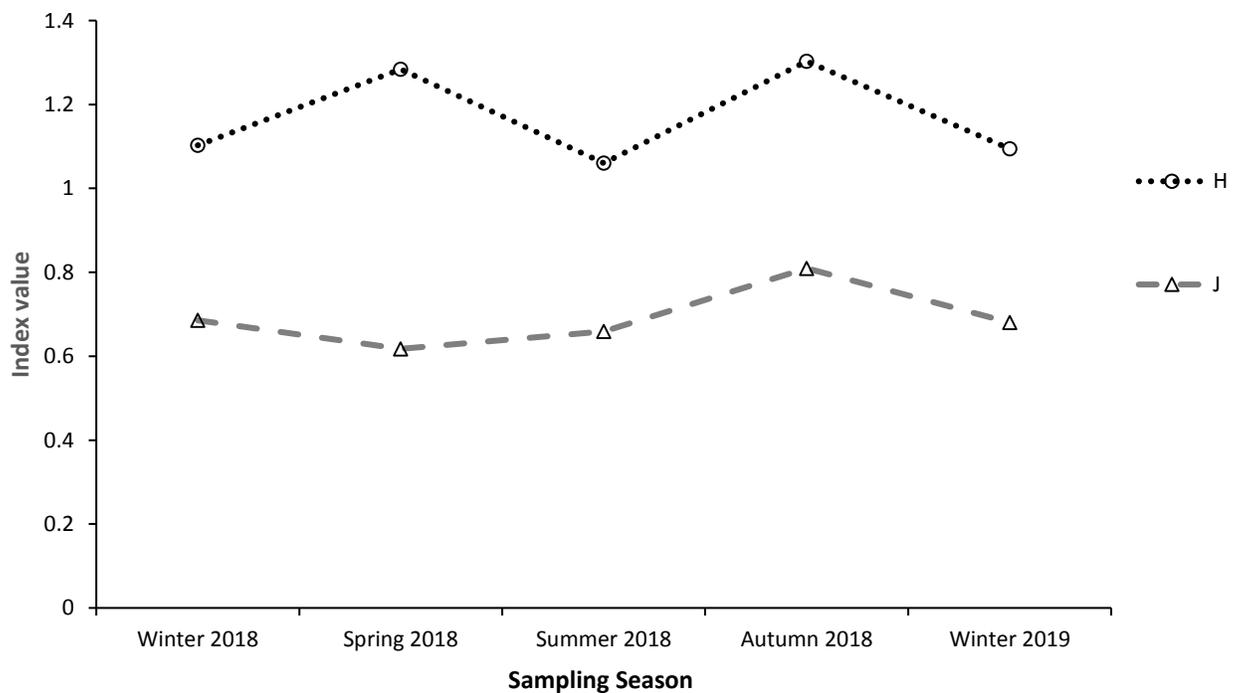


Figure 34. Trap capture Shannon-Weiner index (H') and Pielou's equitability index (J'). H' and J' for the trap capture data during each sampling season.

As for the sites with trap capture data, H' and J' were highest at the SB site with values of 1.41 and 0.79 respectively, while the lowest H' was at the AR reef site (1.07) and the lowest J' with a value of 0.56 was at the MC site (Figure 35). The MC site had the second-highest H' even though it has the lowest J' . The RR and AR sites had similar values of both H' and J' , with the RR site having H' of 1.15 and J' of 0.71, while the AR had an H' of 1.07 and a J' of 0.66 (Figure 35).

The sites themselves follow the pattern seen in Figure 34 in terms of H' and J' each sampling season, with the SB site having the highest H' (1.41) and J' (0.79; Figure 36 and 37). The SB site had the least variation in J' , while all other sites had a drop in J' in winter 2019 (Figure 37).

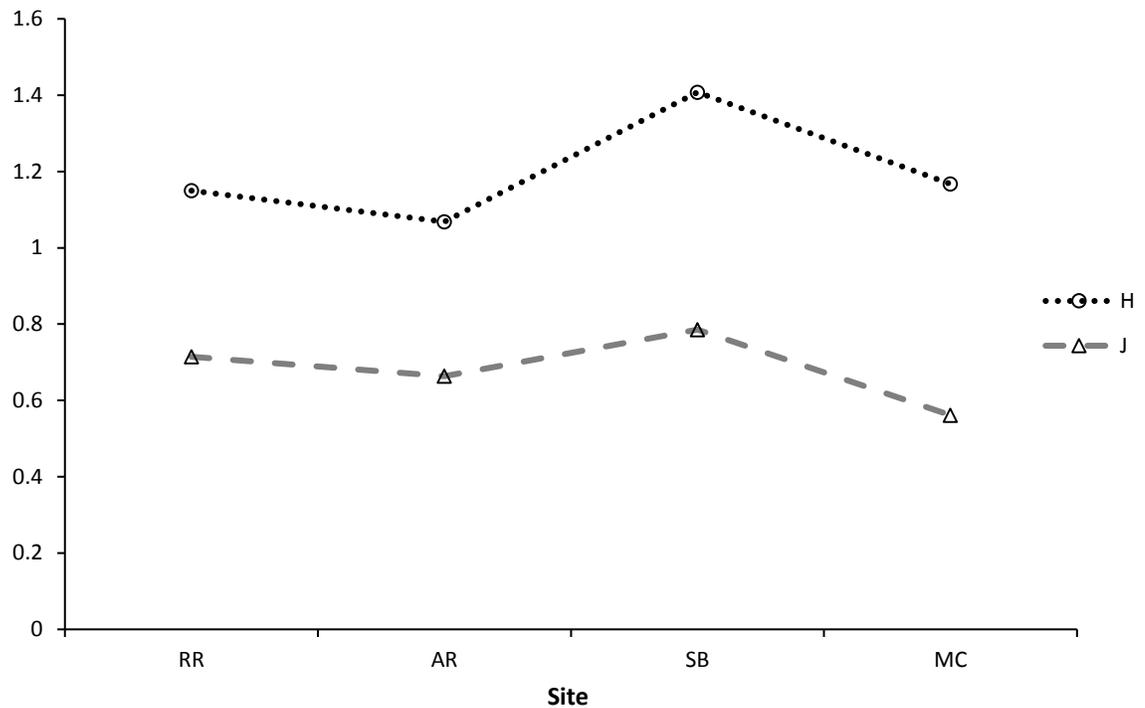


Figure 35. Shannon-Weiner's diversity index (H') and Pielou's equitability index (J') between sites for the trap capture value. H' and J' for the trap capture data between the study sites: rocky reef (RR), soft bottom (SB), artificial reefs (AR), and main channel (MC) for all the samples taken in this study.

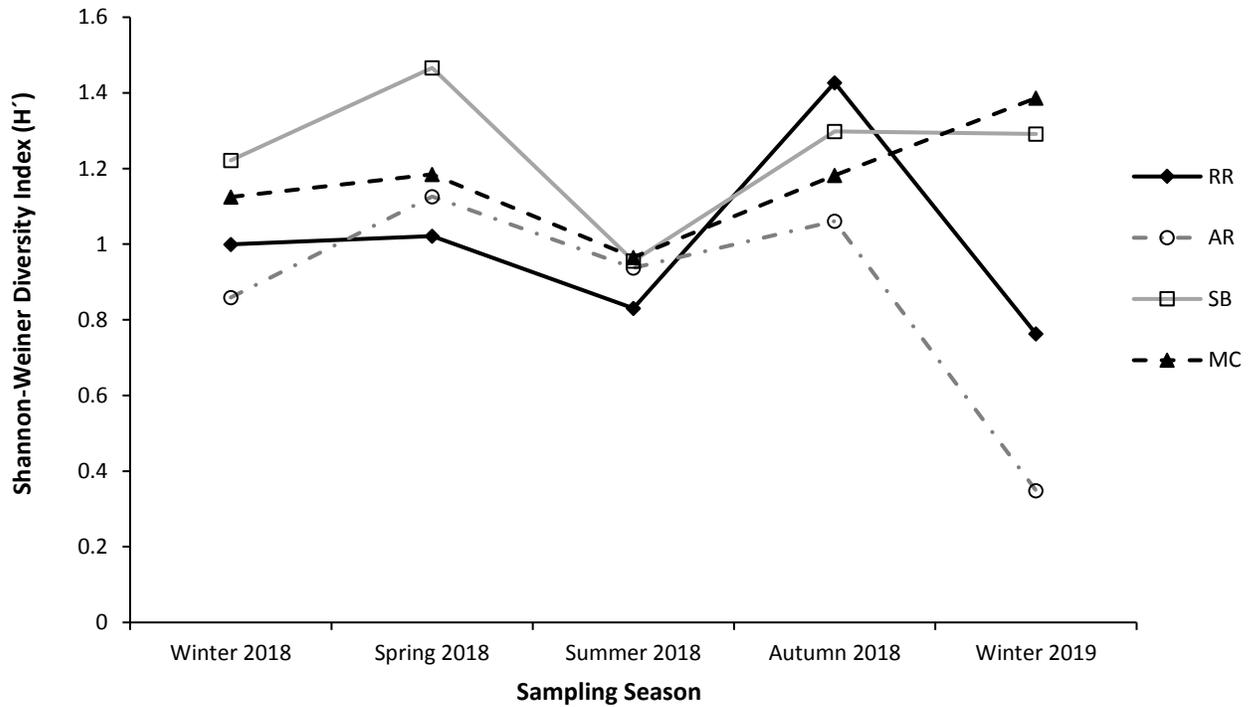


Figure 36. Per site Shannon-Weiner diversity index. Shannon-Weiner diversity index (H') per site (RR: rocky reef, AR: artificial reef, SB: soft bottom, MC: main channel entrance) for each sampling season for the trap capture data.

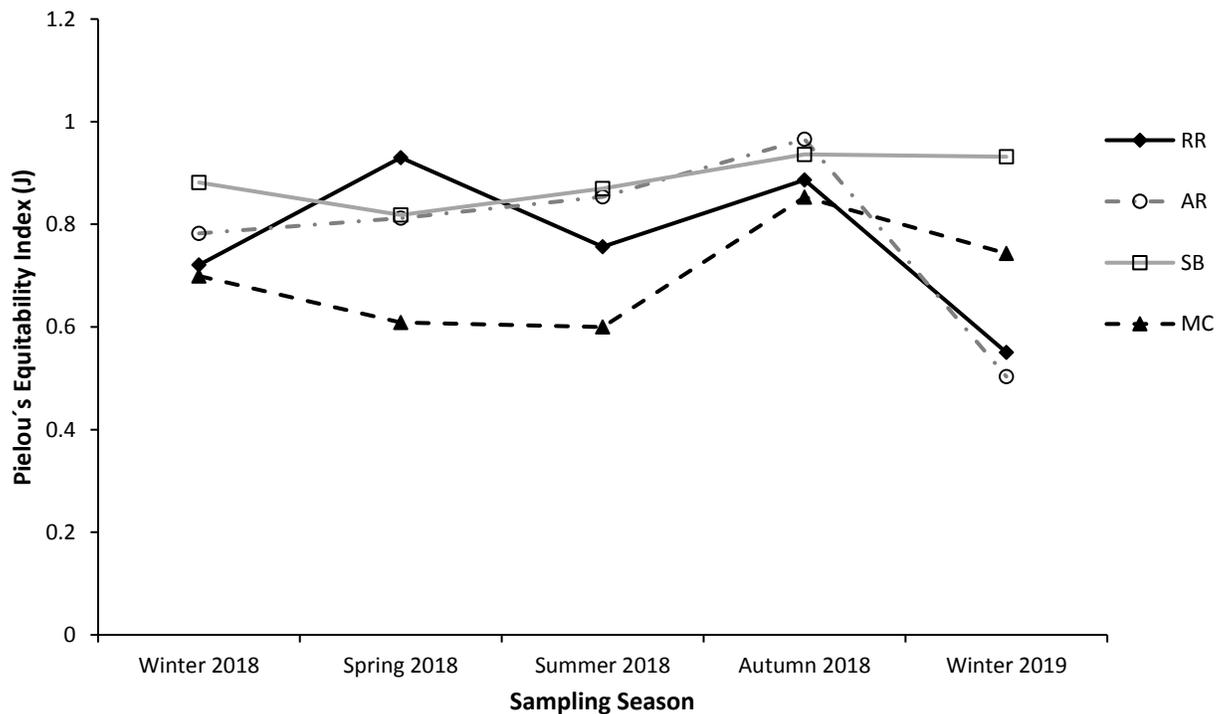


Figure 37. Pielou's equitability index per site. Pielou's equitability index (J) per site (RR: rocky reef, AR: artificial reef, SB: soft bottom, MC: main channel entrance) for each sampling season for the trap capture data.

Chapter 4. Discussion

4.1 Environmental parameters

The seawater temperature in Laguna Ojo de Liebre during the study period showed a seasonality with the highest temperatures during summer and the lowest in winter, however, the lagoon was influenced by temperature anomalies in the Eastern Pacific of $-0.6\text{ }^{\circ}\text{C}$ (March 2018), $+0.1\text{ }^{\circ}\text{C}$ (June 2018), $+0.2\text{ }^{\circ}\text{C}$ (August 2018), $+0.9\text{ }^{\circ}\text{C}$ (November 2018) and $+0.8\text{ }^{\circ}\text{C}$ in February 2019 (NOAA, n.d.). Similar seawater temperature variations were observed in OLL by Alvarez Borrego and Granados Guzman (1992). Phleger and Ewing (1962) reported that seawater temperature in the lagoon was mostly constant near the mouth, then increased about a degree Celsius in the middle lagoon, and increased another degree in the inner lagoon.

Salinity was significantly different between spring and autumn 2018, with it being highest in spring. It is well known that in coastal lagoons with no freshwater input, tides and evaporation rates have a significant effect on salinity, and OLL being an anti-estuarine lagoon with higher values in salinity in the inner lagoon (Phleger and Ewing, 1962; N. P. Smith, 1994). We observed a seasonal tendency in salinity which peaks in or near the summer months and lower salinities towards the winter. Winter 2019 showed a drop in the mean salinity but had high variability due to one site having near-zero salinity. Cívico Collados (2016) also saw a similar decline in salinity levels to near zero in the same month and location (SB), which seems to rule out instrument issues, possibly, there is groundwater seepage happening in this site. There is no reported freshwater source into the lagoon that has been officially documented but the site located near the southwestern section of the lagoon borders the Sierra San José de Castro that can have runoff during the winter rainy season (Chávez López, 2010; Programa de Manejo Reserva de la Biosfera El Vizcaíno, 2000).

The site with the AR had higher salinities than the other three sites. A substantial effect on salinity comes from both the tide phase and current velocities due to salinity being affected by advection and diffusion (N. P. Smith, 1994). Advection is achieved through the tidal currents in the lagoon, which are strongest in the channels; two sites sit near or on the main channel, these are MC and SB, while RR is relatively near the inlet of the lagoon and on a minor channel, the AR site is in a shallow area away from the main channel (annex Figure 38; Alvarez Borrego and Granados Guzman, 1992).

We monitored at or near the low tide due to the lower currents making it safer to SCUBA dive; during low tide the high salinities from the inner lagoon are moved closer to the mouth of the lagoon, which could

have been the situation during the spring of 2018 and why its salinities were higher than during summer 2018. According to Inman *et al.* (1966), in Guerrero Negro and the lagoon, a spring-summer (March to October) pattern with the highest wind speeds is registered, respect to the winter (November-February) pattern; the summer wind speeds elevate the evaporation rates, and possibly this explains the high salinities recorded in the lagoon during spring and summer 2018. Seasonally, differences in salinity occur between sites near the mouth (MC: Carros viejos and RR: El Borbollon) and from farthest sites (AR: artificial reefs and SB: La Ventana); these salinity differences were recorded during all monitored seasons.

The concentration of dissolved oxygen in the sea is a function of salinity, temperature and some biological factors (García and Gordon, 1992; Hull *et al.*, 2008). In OLL, dissolved oxygen had the highest levels during winter 2019 and at the same time lower salinities and lower temperatures, while the lowest oxygen values were recorded in summer 2018, coinciding with the highest water temperatures; this is due to the fact that oxygen solubility and temperature have an inverse relationship (N. P. Smith, 1994). Similar patterns of an inverse relationship between dissolved oxygen and water temperature have been observed in hypersaline lagoons such as OLL (Cívico Collados, 2016; Valdez-Holguín, 1994). In the present study, dissolved oxygen was lower at the MC site than at the SB site during winter 2018, as opposed to what was found by Cívico Collados (2016) during winter.

In terms of temperature, the MC site had lower seawater temperatures than the AR and SB site in winter 2018. Considering that the temperature and oxygen solubility is inversely related, the differences in dissolved oxygen recorded herein cannot be explained by temperature. Possibly incorporating primary productivity can help answer these differences, since chlorophyll concentrations in March have proven high around the RR and SB sites (Hull *et al.*, 2008; Millán Núñez *et al.*, 1987).

The limitations of the present study is that the data are discrete in time and space and the variability associated with tides, humidity, upwelling, rainfall, and biological productivity, were not taken into account among other factors that affect the seawater temperature, salinity and, dissolved oxygen (Hull *et al.*, 2008; Martínez-Fuentes *et al.*, 2016; Phleger and Ewing, 1962).

4.2 Species listing

Almost all species identified in the present study at OLL are from a tropical or subtropical distribution (OLL is in the subtropical region); the Burrito grunt (*Anisotremus interruptus*), the Shovelnose guitarfish (*Pseudobatus productus*), the Gungo highhat (*Pareques viola*), the Giant electric ray (*Narcine entemedor*), the Pacific flagfin mojarra (*Eucinostomus. Currani*), and the Largemouth blenny (*Labrisomus xanti*), all have a tropical affinity (R. Froese and Pauly, 2016); fish species identified in the present study have been reported previously (Acevedo Cervantes, 1997; Cívico Collados, 2016; De La Cruz-Agüero *et al.*, 1996), with the exception of the Largemouth blenny, *Labrisomus xanti* which is registered for the first time in OLL, one individual was observed and recorded on camera during the summer 2018 sampling date. Until recently, the furthest northern limit of *L. xanti* was the adjacent Bay of Sebastian Vizcaino, and due to the northward migration of its northern limit it has increased its abundance at this boundary site (Cavole *et al.*, 2016), and in California it was recently added to its list of marine ichthyofauna (Love *et al.*, 2016).

Summer 2018 had the most species observed, most of which were seen by SCUBA census. Species that were first identified by Cívico Collados (2016) at OLL were seen again during this survey, and these include *P. viola* and *B. polylepis*. As expected, due to the warmer waters of the summer months, more tropical species were seen in this survey, with four having a tropical distribution, these being *N. entemedor*, *P. viola*, *E. currani*, and *L. xanti*. These four mentioned species have their main distributions south of Sebastian Vizcaino Bay and might be more common to find them at their northern limits during summer months since there also appears to be a positive correlation of abundance of fish with a tropical distribution with temperature (R. Froese and Pauly, 2016; Harrison and Whitfield, 2006).

For the monitoring of summer 2018, an individual Big Skate, *Beringraja binocolata* was identified, this is a temperate distribution species whose southern limit is the Cedros island (R. Froese and Pauly, 2016), and it was the only recorded species of temperate affinity throughout the study period, however it was important to find it in the lagoon and during summer when seawater temperatures are higher than those of Sebastian Vizcaino Bay.

During the autumn of 2018 monitoring eight species were seen, a diving problem caused us to cancel the SCUBA census after only monitoring the RR site. Although a weak ENSO event had started (NOAA, n.d.), it was surprising to find only one species of tropical distribution, the Burrito grunt, *A. interruptus*, the other seven species were from a subtropical distribution.

Winter 2019 showed another low number of species with only one from tropical distribution, *P. productus*, while the others are from a subtropical distribution; although *P. productus* is considered a tropical species, their distribution range goes to the temperate waters of San Francisco Bay (R. Froese and Pauly, 2016), thus not being uncommon for this species to be seen during winter at a subtropical location.

A similar number of fish species (25 species in 18 families) were observed during our study with respect to the 27 species in 19 families reported by Cívico Collados (2016) using the same methodology in OLL, although with a larger number of sites which could explain the difference in the number of identified species.

4.3 Abundance and biomass

In the present study, the lowest trap captures were during the summer and autumn seasons, similar to that found by Cívico-Collados (2018). During the summer season, both *Paralabrax nebulifer* and *P. maculatofasciatus* reproduce, and while the first species migrates for reproductive aggregation an average distance of 17 km (Jarvis *et al.*, 2010), the second inhabits protected areas such as OLL, and it aggregates in the mouths and its movements are smaller (L. G. Allen *et al.*, 1995; Hovey and Allen, 2000). These species tend to aggregate in specific locations to reproduce and have different-time-of-day reproduction between species (Baca-Hovey *et al.*, 2002; Hovey and Allen, 2000; Jiménez-Rosenberg *et al.*, 2007). These movements possibly reduced their abundance in the monitoring sites during the summer monitoring.

The higher water temperatures in shallow areas of the lagoon can also be causing seasonal migration to the deeper waters of Sebastian Vizcaino bay; such migration is not unheard of as has been reported in AR in the Adriatic Sea (Fabi and Sala, 2002).

A similar pattern in trap captures and the SCUBA census was observed, with greater abundance and biomass in the winter-spring seasons compared to autumn-summer, this was also reported by the recent study at Ojo de Liebre lagoon by Cívico Collados (2016). The higher abundances in winter 2018 compared to winter 2019 could be associated with the ending of a cold period (La Niña) that ended in April of 2018 and the beginning of a warm ENSO (El Niño Southern Oscillation) period which began on September 2018 (NOAA, n.d.). Previous studies have shown that fish larvae and zooplankton biomass are reduced during warm ENSO periods; this could have reduced the recruitment of YOYs (young-of-the-year) of the most

abundant species of *Paralabrax sp.* during the winter months of 2019 (Franco-Gordo *et al.*, 2004; Lehodey *et al.*, 2006; Love *et al.*, 1996).

The site closest to the mouth of the lagoon (MC) presented an abundance and biomass higher than the other three sites studied, possibly environmental conditions such as salinity influences the fish since it generally has a negative correlation with their abundance, in addition to the immigration of transient fish from Bahia Sebastian Vizcaino, increasing the species density and richness (Sosa-López *et al.*, 2007; Vega-Cendejas and Hernández De Santillana, 2004).

Our two most abundant species, *P. maculatofasciatus* and *P. nebulifer*, are closely related and have a habitat preference in the ecotone between natural rocky reefs and sandy bottom; this is one possible explanation to us not finding a difference in abundance and biomass between the AR, SB, and RR sites (Love, 2011; Mason and Lowe, 2010) as all three sites resemble their preferred habitat.

The AR used in this study were designed to provide lobsters with shelter and were set up at varying distances from each other in such a way that per transect, one to two AR units were surveyed, this amounts to 1.7 to 3.4 m² of the 60 m² of each transect, or 3-6% of the area surveyed. This low AR area could have reduced the effectiveness in the increase of fish abundance and biomass as previous studies have shown that more extensive reefs, in terms of area, increase abundance and species richness (Jordan *et al.*, 2005). Vertical relief (0.25 m) is another aspect which reduced the effectiveness of the AR at OLL for fish abundance and biomass, as has been shown for fish associated with coral reefs, the higher relief an AR has, the more abundance and species richness (Rilov and Benayahu, 2000).

Structural complexity also plays an important role that should be considered when designing AR to increase abundance and species richness of fish. There were some cases in our surveys where two AR units were stacked, increasing both vertical relief and complexity, these stacked units, which were rare, appeared to have a higher density of fish surrounding them than just single units (Hunter and Sayer, 2009; Rilov and Benayahu, 2000; Sherman *et al.*, 2002). Though, not particularly taken into consideration in this study, increasing habitat complexity by stacking AR units can have a more positive impact on increasing species richness and abundance.

Also, the substrata on AR tend to develop different biota than that on natural substrata, primarily because of cement's high pH and presence of toxic compounds; this different biota consequently affects the fish communities (Carr and Hixon, 1997; Perkol-Finkel and Benayahu, 2007; Perkol-Finkel *et al.*, 2006; Sella and

Perkol-Finkel, 2015; Svane and Petersen, 2001). In addition, Sella and Perkol-Finkel (2015) have produced and patented what they term EConcrete® Antifers, whose substrata develops biota resembling natural reef communities as well as fish communities with higher abundance, richness, and diversity, compared to antifers made of the heavily used on coastal and sea structures Portland cement. Unfortunately, the specifications on the cement used on the AR at OLL were not provided for this study.

Though not significant, the AR site had the largest abundance of fish with the trap capture data in the winter of 2018. During the SCUBA census, we found a large number of juvenile *Paralabrax spp.* at the AR site. The presence of so many juvenile fish at the AR site could have attracted larger fish of the same genus whose diet incorporates smaller fish as they get larger; such events have been studied previously in AR sites at or near lagoons due to them functioning as nurseries for many fish species (Herrera *et al.*, 2002; Leitão *et al.*, 2008; Love, 2011).

Summer 2018 had a larger presence of tropical species, particularly in two sites, the RR site and the SB site. There appeared to be a reduction of the common *Paralabrax spp.* at the RR site and replaced with the tropical, and much smaller, *E. currani*. While, at the SB site, there was the presence of a large tropical fish, *N. entemedor*, and the small *L. xanti*, and *P. viola*, which may have attracted larger piscivorous fishes, like *M. xenarcha*. These less common species were the leading cause of the significant differences in biomass found in the SCUBA census between the SB site and the RR site. There have been reports of the northern extension in the distribution of *L. xanti* due to warming water (Cavole *et al.*, 2016; Love *et al.*, 2016), as well as a possible northern migration of *N. entemedor* during the summer months, a behavior seen in a related Atlantic species, *Narcine brasiliensis* (Coles, 1913).

There were no comparisons between sites of SCUBA census during the autumn of 2018 due to only the RR site being surveyed. *P. maculatofasciatus* and *C. brachysomus* were the two main species observed, which is a return to the “normal” condition of the site according to common fish species.

In the winter of 2019, there were no significant differences in abundance between all three sites surveyed, RR, AR, and SB sites; however biomass does show significant differences between the RR and SB site, being the opposite of that seen during summer 2018, with the higher biomass being the RR site. SB habitats are unique compared to other habitat types, as indicated by a massive study done in California on habitat types and fish (L. G. Allen, 1985), as thus, we expected to find significant differences in this habitat to all other habitats. Surprisingly, in the present study, differences in abundance were not found, but there were

differences in biomass, suggesting different community composition between habitats, as found by Allen (1985).

4.4 Community indexes

4.4.1 Index of community importance (ICI)

Similarities exist between the results of ICI in this study and that of Cívico Collados (2016) done in the same sites of the lagoon. For one, the top three species were the same: *P. maculatofasciatus*, *P. nebulifer*, and *C. brachysomus*. However, the order in which they are ranked is not the same. In Cívico Collados (2016) study, *P. nebulifer*, and *C. brachysomus* both occupy the top ranking, followed by *P. maculatofasciatus* in third; *P. clathratus* was the fourth-ranked species in both studies. These differences may be due to *C. brachysomus* having had more favorable conditions during the stronger El Niño event during Cívico Collados' study (NOAA, n.d.) and due to *C. brachysomus* having a higher affinity to warmer temperatures than *P. nebulifer* and *P. maculatofasciatus* (R. Froese and Pauly, 2016).

Species of the genus *Paralabrax* are an important component of the fish communities in coastal bodies of water in the northern Baja California peninsula, however, *C. brachysomus* ranking drops dramatically or is absent in the northern peninsula suggesting OLL is near its northern limit in the eastern Pacific (R. Froese and Pauly, 2016; Rosales-Casián, 1997, 2004).

The Barred sand bass (*P. nebulifer*) move out of coastal bodies of water such as lagoons, bays, and estuaries towards their preferred open coast habitats for reproduction and as they grow into adulthood (Love, 2011; McKinzie *et al.*, 2014; Oda *et al.*, 1993). The rank in ICI of *P. nebulifer* increased in the summer months, this is consistent with reproductive behavior since this species aggregates for reproduction during the summer months and can be using the lagoon as an aggregating area (Love, 2011; Oda *et al.*, 1993).

4.4.2 Index of relative importance (IRI)

Once biomass is taken into consideration with IRI, similar results to ICI with a slight change are seen, as *P. nebulifer*, has the highest IRI values during summer and autumn of 2018 while *P. maculatofasciatus* is the highest in the other seasons. As mentioned previously, the increase of values of ICI and IRI for *P. nebulifer* could be associated with reproductive behavior (Jarvis *et al.*, 2010)

The three members of the *Paralabrax* genus (species *clathratus*, *maculatofasciatus*, and *nebulifer*) are important coastal fish along southern California and the Baja California peninsula; high values of IRI were seen in San Diego Bay for these species (L. Allen *et al.*, 2002). In the study by L. Allen *et al.* (2002), in which IRI is called ICI, *P. maculatofasciatus* and *P. nebulifer* are the fifth and sixth-ranked species respectively, while *P. clathratus* is ranked thirty-second out of fifty. Differences in results with IRI and that of L. Allen *et al.* (2002) could be explained by the different sampling methods used in the two studies.

Cívico Collados' (2016) values of IRI at OLL are consistent with her values of ICI, with *C. brachysomus* being the species with the highest value, followed by *P. nebulifer* and *P. maculatofasciatus*, respectively, she also had higher values for *P. nebulifer* in summer and autumn months.

4.5 Correlation between environmental and biological factors

This study found correlations with all three environmental factors depending on the sampling method. With trap capture data, only seawater temperature was correlated with both abundance and biomass showing weak negative values (-0.24 and -0.2 respectively). Cívico Collados (2016) also found a negative relationship with seawater temperature as well as salinity in both biomass and abundance; however, her correlations were moderate with values ranging from -0.47 to -0.55 depending on the census method.

For the SCUBA census, positive correlations with seawater temperature and salinity in both abundance and biomass were found. Abundance also had a weak negative correlation (-0.36) with dissolved oxygen. Relationships between abundances and biomass with environmental factors can be quite complex. L. Allen *et al.* (2002) found that seawater temperature, salinity, depth, and distance to the mouth of San Diego Bay was correlated with abundances using the canonical correlation. A study in various South African estuaries found that seawater temperature and salinity were the most important abiotic factors influencing

abundances of various species (Harrison and Whitfield, 2006); this study in South Africa was exceptional due to the differences in physical parameters on their three coasts, in the eastern coast tropical conditions abound, while the southern coast is subtropical, and the western coast is temperate. For tropical species, Harrison and Whitfield (2006) found that seawater temperature had a positive correlation with abundance for tropical fish species, while it is opposite for temperate species. They also found that resident estuarine fish had a negative correlation with salinity; however, they also found that the relationship with salinity was closely related to whether the fish species was euryhaline or stenohaline. This helps possibly explain why seawater temperature had a positive correlation in the SCUBA census with abundance and biomass as various tropical species were seen through the SCUBA census. A more detailed examination can better help elucidate the relationships of abundance and biomass with the physical parameters within each species in this study.

Salinity has also been shown to affect fish density on hypersaline lagoons. Vega-Cendejas and Hernández De Santillana (2004) found that in a hypersaline lagoon in Mexico's Yucatan peninsula, fish density declined as salinity increased. This relationship is consistent with the results from Cívico Collados (2016) in OLL yet different from our results from the SCUBA census. These results can be skewed by the few fish sampled at the SB site during the winter of 2019 when salinities dropped to near zero.

4.6 Diversity

SHE analysis for the SCUBA census

SHE analysis found four central communities of our SCUBA census, this is one of the main advantages of the SHE analysis, the ability to distinguish between communities based on their statistical distributions. The RR community (which includes the winter 2018 AR community), the AR community from spring 2018 onward, the summer 2018 community, and the SB winter 2019 community. There are a couple weaknesses to note on these results, the main one being the low sample size of the SB winter 2019 community. The other weakness is the grouping of distinct communities into a single community when changing the parameter from temporal to spatial (summer's AR and RR communities).

It appears that shortly after the installation of the AR in OLL, colonization of this new habitat may have happened from the RR site, or that both the AR and RR sites community was a general lagoon community

during winter 2018. This rapid colonization of AR from nearby sites has been observed in various studies (Buckley and Hueckel, 1985; Herrera *et al.*, 2002; Solonsky, 1985), making it probable that a similar situation happened during this study.

Second, the change in the community of fishes between the AR and RR in the spring of 2018 could be evidence of succession occurring in the AR habitat or the effects of the arrival of predators after initial colonization by smaller fish, or due to the physical differences in the structure between the RR and AR. In spring 2018, the abundance at the AR site seen with traps (larger, piscivorous fish) was higher than in other sites, the presence of a large number of predators could have reduced recruitment of juveniles due to the priority effect (Herrera *et al.*, 2002; Shulman *et al.*, 1983). Perkol-Finkel *et al.* (2006) noted that communities of invertebrates differ between artificial and natural reefs based on the structural similarity of the reefs. Successional changes and their markers, such as decelerating increases in species richness, abundance, mean individual size, and species loss rates with time and decelerating decreases of species gain and turnover rates have been observed on AR (Santos *et al.*, 2011). The successional markers mentioned were not the focus of this study; however, we did observe a decrease in abundance and a change in the community structure that suggest succession could have taken place.

The SB site was only surveyed on two occasions, summer 2018 and winter 2019. These two were found to be from different communities. The winter 2019 SB community was its own community; however, the number of fish and sample size was low, and more samples are needed to see a better trend in the data.

Community comparisons at equal sample sizes show a higher species richness in the RR community compared to the AR community (spring and onward), with more dominance occurring at the AR community, the SB winter 2019 community was intermediate between the two in species richness but had the most dominance. Of the RR and AR communities, the RR community had the highest diversity (H). A similar study comparing sites also found higher dominance, or less equitability, at an AR site compared to both a natural reef site and a sandy-mud bottom, with equitability being more important to species diversity than species richness (Fujita *et al.*, 1996). Unfortunately, a pattern to species richness and diversity when comparing AR to natural reefs has not been clear as some studies have found higher diversity and species richness in AR than natural reefs, while others have found less or similar diversity and species richness (Granneman and Steele, 2015; Rilov and Benayahu, 2000; Rooker *et al.*, 1997). In this study, the high abundance of juvenile *P. maculatofasciatus* at the AR site contributed to its lower equitability or evenness as compared to the RR site.

Differences between summer and winter 2018 communities shows that summer 2018 had the highest species richness, there is more dominance in the summer community which reduces diversity, however, the larger the samples, the quicker the diversity index increases in the summer 2018 community compared to winter 2018. There appears to be a trend in higher species richness and diversity during the warmer seasons in subtropical habitats, as fish from tropical distributions migrate poleward (Amezcuca and Amezcuca-Linares, 2014; Heithaus, 2004; Lin and Shao, 1999; Nakamura *et al.*, 2013). The SB site attracted three species of tropical distribution during the summer census; this increased the species richness.

The subtropical region, being an intermediate between tropical and temperate regions, can have species from those distributions because of its intermediate status. It is not unheard of that tropical species migrate north during the summer months when water seawater temperatures are similar to those found in tropical distributions (Coles, 1913; Love *et al.*, 2016). Recalling Rapoport's rule of species richness related to latitude (Stevens, 1996; Willig *et al.*, 2003), one would expect that a northern migration of tropical species would increase the species richness of a subtropical or even temperate location as has been reported not only seasonally, but due to climate change (Chaudhary *et al.*, 2016; Hiddink and ter Hofstede, 2008).

Trap Capture

The SB site had the highest Shannon-Weiner diversity index with trap capture. One of the main contributors to the SB site having the highest diversity index was its evenness (J'). High evenness of a SB habitat was reported in an analysis of habitat types of southern California by L. Allen (1985). In fact, in his analysis of the different habitat types, the highest diversities were found in kelp beds, soft bottoms, and rocky reefs, however, he did consider the soft bottom habitats diversity as possibly inflated due to the various depths strata included.

The MC site had one of the highest species richness in trap capture data; similar to a study of San Diego Bay, which found the inlet to the bay having the highest species richness (L. Allen *et al.*, 2002). This is usually due to habitat homogeneity around the inlet and because of the immigration of fish (Vega-Cendejas and Hernández De Santillana, 2004). The same study by L. Allen *et al.* (2002) found that evenness increases as the distance from the mouth of the bay increased, similar results in evenness (J') were seen in this study. The sampling method in the present study was done using traps baited with sardines; this reduced our samples to piscivores. Had the possibility of SCUBA existed we could have found an even higher species richness at the MC site.

L. Allen *et al.* (2002), also reported higher diversity in terms of species richness in spring and summer, while ours had the highest in spring and fall for trap capture and lowest in summer. The previous study in OLL had the highest species richness and diversity in spring and lowest in summer (Cívico Collados, 2016) which was similar to the present study.

As a final note, various studies already mentioned here have either concluded or seen the presence of many smaller fish, mainly juveniles concentrated around AR. This present study also saw the presence of many juveniles of sand bass, both Spotted sand bass (*P. maculatofasciatus*), and Barred sand bass (*P. nebulifer*) around the AR. The shelter these AR provide these fishes may help increase survival at this important life stage. My suggestion would be to increase monitoring and testing for such a hypothesis, as well as using new techniques (Biomass flux; J. A. Smith *et al.*, 2016) to help answer the production versus aggregation debate on these particular artificial reefs as well as suggest future AR at the lagoon be placed in a manner that increases the number of AR per surface area as well as increased complexity.

Chapter 5. Conclusions

- The physical parameters of salinity, seawater temperature, and dissolved oxygen at OLL vary greatly depending on distinct factors. These factors could be physical, such as evaporation rates due to wind stress, irradiation, solubility, and seasonality, freshwater seepage, and tide phase. Biological factors also affect these physical parameters, among which are primary productivity and respiration.
- In general, seasonally, dissolved oxygen appeared to have an inverse relationship with seawater temperature due to it affecting the solubility in water. As per site, dissolved oxygen seemed to have an inverse relationship with salinity, possibly due to saturation maximums. Salinity had the lowest mean salinity at the SB site probably due to freshwater seepage and had the highest mean dissolved oxygen. Salinity, for the most part, was lower at the MC site near the inlet, as expected in a hypersaline lagoon.
- The fish species were similar at all sites, except during summer where there were influenced by the presence of tropical species, especially at the SB site. The AR site had a large number of juvenile fishes from the genus *Paralabrax* during the winter sampling dates.
- Abundance and biomass were similar between the RR, AR, and SB sites using both methods of surveying, the MC site had significantly greater abundance and biomass than all other sites.
- Three species made up 92% of the total abundance considering both sample methods, *P. maculatofasciatus*, *P. nebulifer*, and *C. brachysomus*, these species were the most important species measured with both indexes used, ICI and IRI.
- All three environmental variables measured were correlated to abundance and biomass depending on the sampling method. Seawater temperature was the only environmental variable correlated with abundance and biomass when using trap capture as a sampling method, and it was negatively correlated with both abundance and biomass. On the other hand, with the SCUBA census, abundance was negatively correlated with dissolved oxygen and positively correlated with both seawater temperature and salinity. Biomass was positively correlated with seawater temperature and salinity.

- SHE analysis showed the presence of three spatial communities, the RR site, the AR site from spring 2018 onward, and the SB (seen only in winter 2019). There were also two temporal communities, the winter 2018 and summer 2019 communities. The AR site had the least species richness and the SB site the least evenness. In terms of the temporal communities, summer had a higher species richness and lower evenness than the winter 2018 community.
- With trap capture, the SB site had the highest diversity (H') and equitability (J'), the AR site had the lowest diversity (H') and the MC site the lowest equitability (J'), the seasons with the highest diversity were spring and autumn (H'), summer had the lowest diversity. This is the same seasonal pattern seen in the study at OLL by Cívico Collados (2016) with the trap capture.

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Annexes

Table 8. Length-weight equations used for the different fish species . Length (cm)-weight (gr) relationship equations used in the calculation of biomass for the estimated lengths of the SCUBA census with their length range (cm), r² value, and literature source. The length-weight relationships with source Perrin were done by the author of this thesis with the trap capture data following the statistical assumptions for a robust linear model.

Species	Length (LT)- weight (W) equation	Length range (cm)	r²	Source
<i>P. nebulifer</i>	logW=- 1.71+2.89logLT	21-51	0.92	Perrin
<i>Paralabrax spp.</i> (YOY-minnows)	logW=- 1.61+2.783logLT	4-20	0.97	Perrin
<i>P. clathratus</i>	logW=- 1.95+3.041logLT	24-49	0.93	Perrin
<i>P. maculatofasciatus</i>	logW=- 1.19+2.54logLT	24-41	0.43	Perrin
<i>C. brachysomus</i>	logW=- 1.12+2.53logLT	21-40	0.87	Perrin
<i>A. interruptus</i>	W=0.05690LT ^{2.91}	na	0.933	R. Froese and Pauly, 2016
<i>S. annulatus</i>	W=0.018LT ^{3.050}	8.6-33	0.996	R. Froese and Pauly, 2016
<i>H. francisci</i>	W=0.000009LT ^{2.9948}	11-75	0.98	E. F. Miller <i>et al.</i> , 2008
<i>P. productus</i>	W=0.00450LT ^{2.920}	17-54	0.993	R. Froese and Pauly, 2016
<i>M. xenarcha</i>	W=0.01838LT ^{3.00}	--	--	R. Froese and Pauly, 2016
<i>A. davidsonii</i>	W=0.02225SL ^{3.0457}	4.1-32.7	0.99	E. F. Miller <i>et al.</i> , 2008
<i>U. halleri</i>	W=0.00734LT ^{3.00}	--	--	R. Froese and Pauly, 2016
<i>Halichoeres spp.</i>	W=0.0160LT ^{2.987}	3.1-19.3	0.980	Kulbicki <i>et al.</i> , 2005
<i>H. rubicundus</i>	W=0.02239LT ^{2.99}	--	--	Rainer Froese <i>et al.</i> , 2014
<i>E. currani</i>	W=0.00800LT ^{3.140}	1.7-16.9	0.97	R. Froese and Pauly, 2016



Figure 38. Bathymetry of Ojo de Liebre lagoon at the survey sites. The main channel is the large one coming in from through the inlet and moving down near the western coastline. Obtained from Gutiérrez de Velasco (2000).

Table 9. Values of abundance, relative abundance and cumulative percent for the entire study. Fish species abundance (Ab.), relative abundance (%Rel), and cumulative percent (%Cu), between both sample methods, trap and SCUBA, at Ojo de Liebre lagoon in five surveys between winter 2018 and winter 2019.

Species	Trap			SCUBA			Trap + SCUBA		
	Ab.	%Rel	%Cu	Ab.	%Rel	%Cu	Ab.	%Rel	%Cu
<i>Paralabrax maculatofasciatus</i>	286	44.13	44.13	215	40.72	40.72	501	42.60	42.60
<i>Paralabrax nebulifer</i>	239	36.88	81.01	23	4.36	45.08	262	22.28	64.88
<i>Calamus brachysomus</i>	77	11.88	92.89	156	29.54	74.62	233	19.81	84.69
<i>Halichoeres semicinctus</i>	0	0	92.89	58	10.98	85.60	58	4.93	89.62
<i>Paralabrax clathrathus</i>	36	5.56	98.45	4	0.76	86.36	40	3.40	93.02
<i>Anisotremus davidsonii</i>	0	0	98.45	47	8.90	95.26	47	4.00	97.02
<i>Eucinostomus currani</i>	0	0	98.45	10	1.89	97.15	10	0.85	97.87
<i>Heterodontus francisci</i>	2	0.31	98.76	3	0.57	97.72	5	0.42	98.29
<i>Sphoeroides annulatus</i>	2	0.31	99.07	4	0.76	98.48	6	0.52	99.81
<i>Semicossyphus pulcher</i>	3	0.47	99.54	0	0	98.48	3	0.25	99.06
<i>Mycteroperca xenarcha</i>	0	0	99.54	3	0.57	99.05	3	0.25	99.31
<i>Anisotremus interruptus</i>	2	0.31	99.85	0	0	99.05	2	0.17	99.48
<i>Pseudobatos productus</i>	0	0	99.85	2	0.38	99.43	2	0.17	99.65
<i>Urolophus halleri</i>	0	0	99.85	2	0.38	99.81	2	0.17	99.82
<i>Hypsypops rubicundis</i>	0	0	99.85	1	0.19	100	1	0.09	99.91
<i>Balistes polylepis</i>	1	0.15	100	0	0	100	1	0.09	100
Total	648			528			1176		

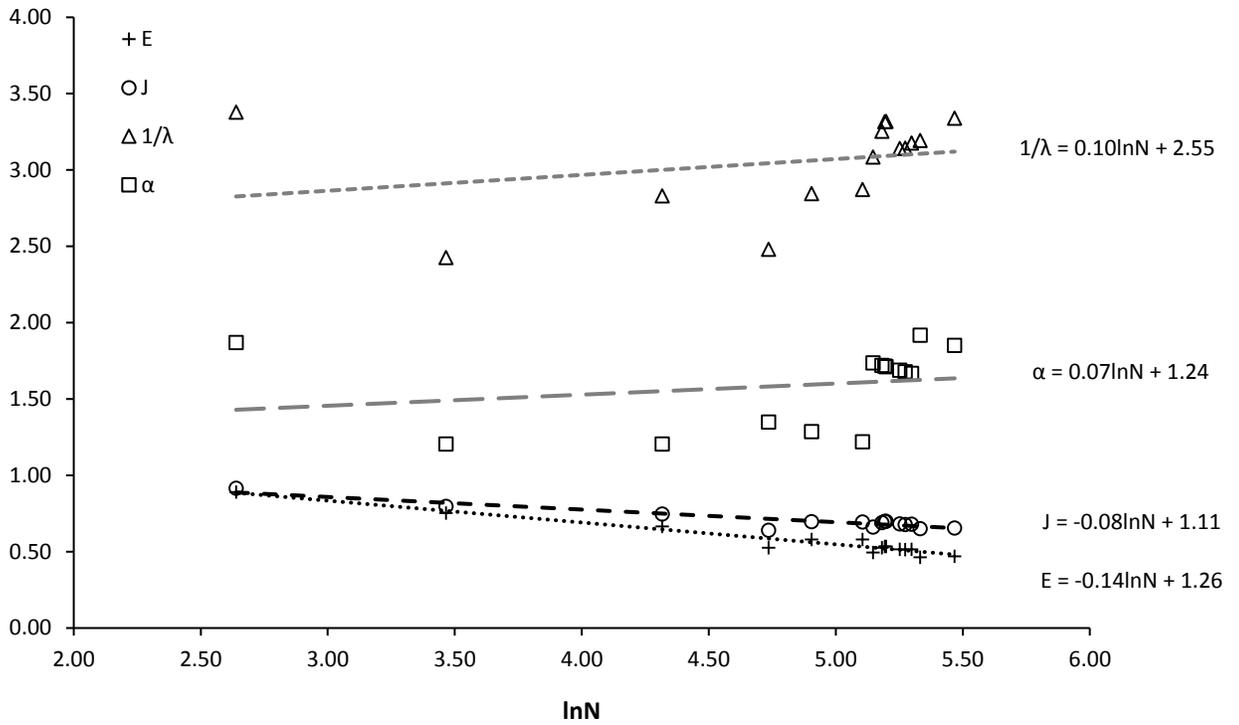


Figure 39. Distribution determination trend lines for the rocky reef (RR) community. Trend lines with their equations for Fisher’s α, Simpson’s 1/λ, Buzas-Gibson’s E, and Pielou’s J’ for the RR community in all sampling seasons.

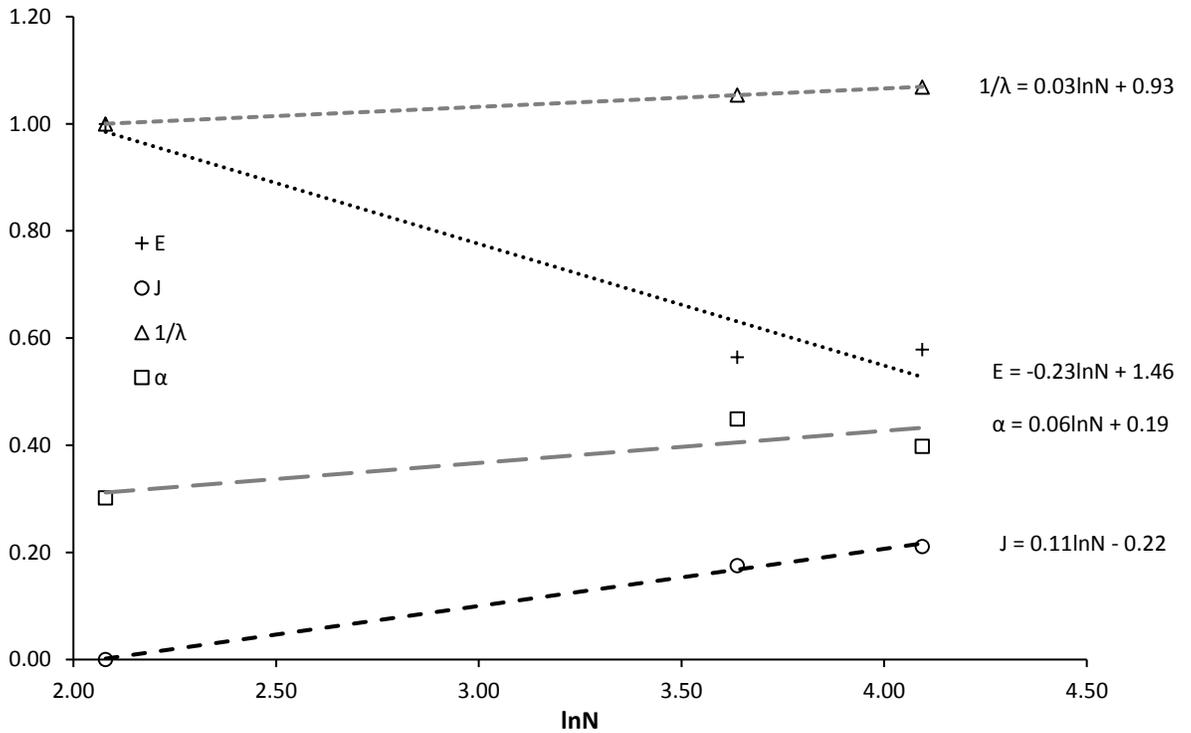


Figure 40. Distribution determination trend lines for the winter 2018 artificial reef (AR) community. Trend lines with their equations for Fisher’s α, Simpson’s 1/λ, Buzas-Gibson’s E, and Pielou’s J’ for the AR community in the winter of 2018.

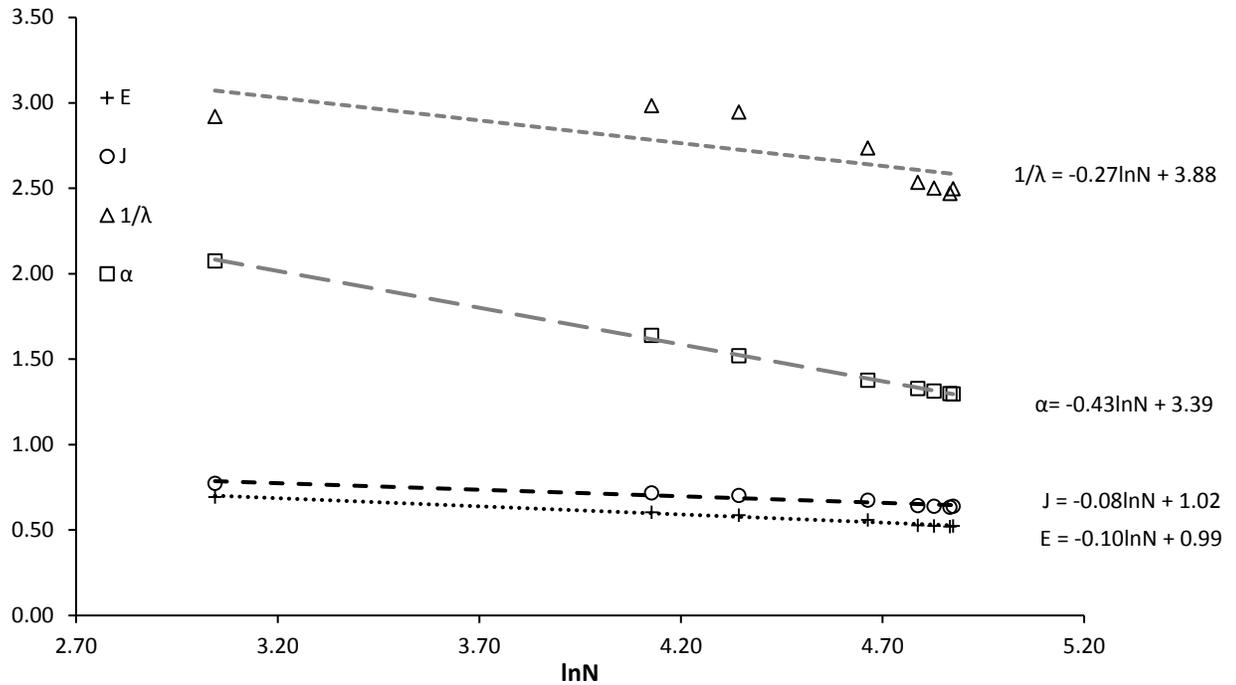


Figure 41. Distribution determination trend lines for the artificial reef (AR) community from spring 2018 thru winter 2019. Trend lines with their equations for Fisher’s α, Simpson’s 1/λ, Buzas-Gibson’s E, and Pielou’s J’ for the AR community from spring and summer 2018, and winter 2019.

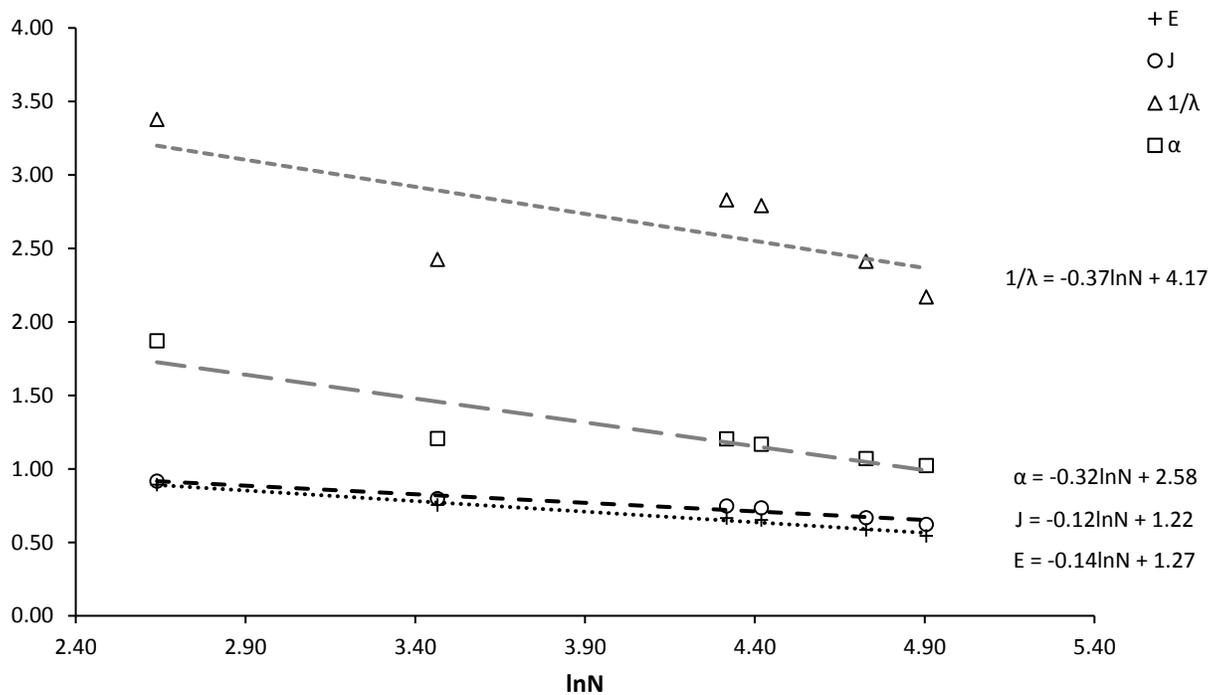


Figure 42. Distribution determination trend lines for the winter 2018 artificial reef (AR) and rocky reef (RR) community. Trend lines with their equations for Fisher’s α, Simpson’s 1/λ, Buzas-Gibson’s E, and Pielou’s J’ for both the AR and RR sites in the winter of 2018.

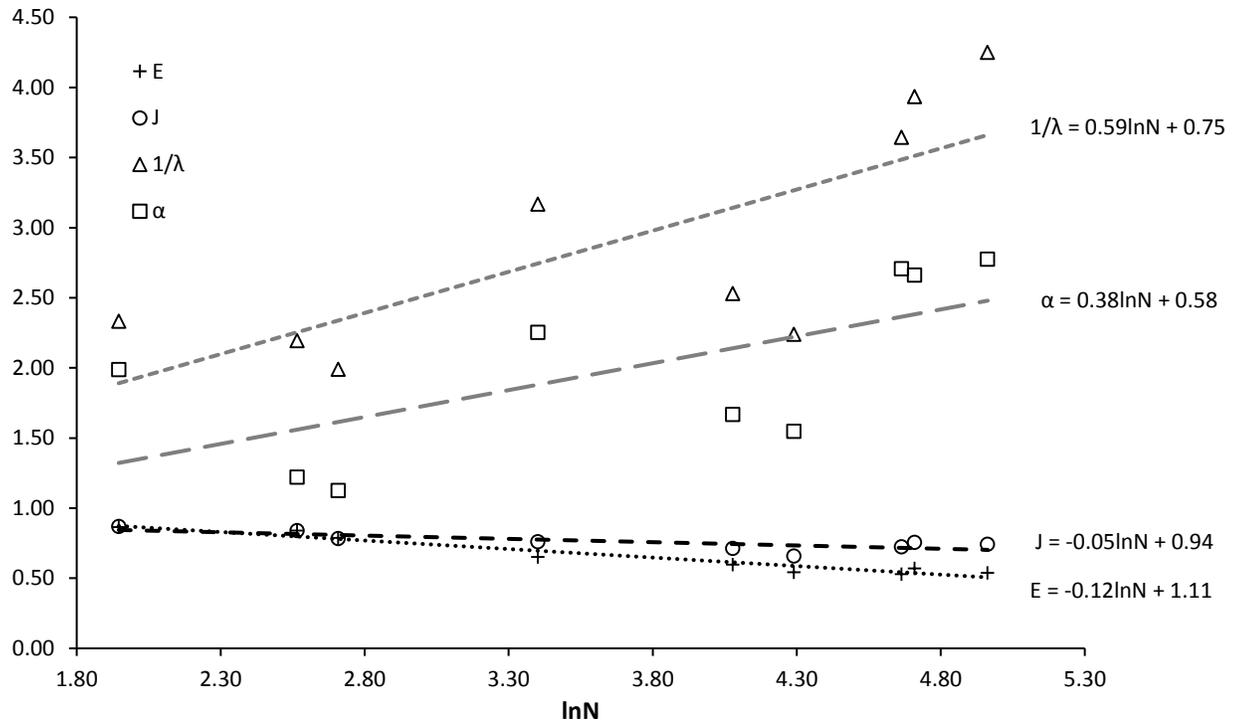


Figure 43. Distribution determination trend lines for the summer 2018 community. Trend lines with their equations for Fisher’s α , Simpson’s $1/\lambda$, Buzas-Gibson’s E, and Pielou’s J’ for the three sites, RR, AR, and SB in the summer of 2018.

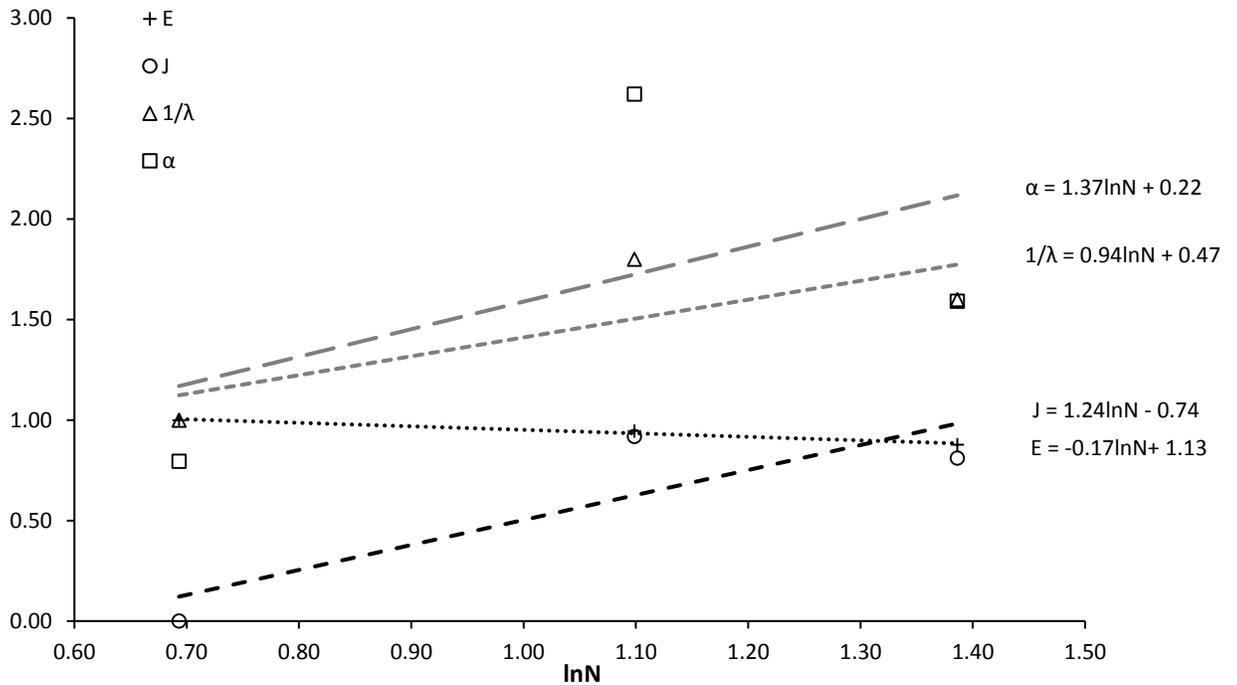


Figure 44. Distribution determination trend lines for the soft bottom (SB) winter 2019 community. Trend lines with their equations for Fisher’s α , Simpson’s $1/\lambda$, Buzas-Gibson’s E, and Pielou’s J’ for the SB community during the winter of 2019.