UNIVERSIDAD AUTONOMA DE BAJA CALIFORNIA

FACULTAD DE CIENCIAS MARINAS

DESARROLLO DE UN MODELO PREDICTIVO QUE INTEGRE LOS FACTORES AMBIENTALES Y BIOLÓGICOS PARA DETERMINAR LA TRAYECTORIA POBLACIONAL DEL ERIZO ROJO, BAJO DIFERENTES ESCENARIOS DE VARIABILIDAD AMBIENTAL Y PESQUERA.

TESIS

QUE PARA CUBRIR PARCIALMENTE LOS REQUISITOS NECESARIOS PARA
OBTENER EL GRADO DE

DOCTOR EN CIENCIAS EN OCEANOGRAFÍA COSTERA

PRESENTA

ALFONSO MEDELLÍN ORTIZ

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FACULTAD DE CIENCIAS MARINAS
INSTITUTO DE INVESTIGACIONES OCEANOLÓGICAS
POSGRADO EN OCEANOGRAFÍA COSTERA

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RESUMEN de la tesis de **ALFONSO MEDELLÍN ORTIZ**, presentada como requisito parcial para la obtención del grado de **DOCTOR EN CIENCIAS** en **OCEANOGRAFÍA COSTERA**. Ensenada, Baja California, diciembre 2020.

“Desarrollo de un modelo predictivo que integre los factores ambientales y biológicos para determinar la trayectoria poblacional del erizo rojo, bajo diferentes escenarios de variabilidad ambiental y pesquera”

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En Baja California, el erizo rojo (*Mesocentrotus franciscanus*) se captura de manera comercial desde 1972, desde Tijuana hasta Isla de Cedros y genera cerca de 2,000 empleos directos e indirectos durante cada temporada de pesca. La pesquería de erizo rojo cuenta con una larga historia y está regulada por la NOM-007-PESC-1993 (ahora NOM-007-SAG/PESC-2015; DOF 25-Jun-2015). El presente trabajo tuvo como objetivo desarrollar un modelo que integra variables oceanográficas y biológicas relacionadas con la población del erizo rojo, con la finalidad de pronosticar trayectorias futuras en diferentes escenarios de variabilidad ambiental, biológica y pesquera. Para esto, fue necesario reevaluar las tendencias poblacionales, esfuerzo pesquero y capturas de erizo rojo, incorporando toda la información disponible de diversas fuentes en los últimos 19 años (2000 – 2018). La información fue agrupada y analizada en 14 sitios de desembarque ubicados a lo largo de la costa oeste de Baja California. Para estimar las tasas de captura, mortalidad por pesca y densidad por sitio, se implementó un análisis virtual de población basado en tallas (LVPA, por sus siglas en ingles). Las capturas de erizo rojo tuvieron amplias variaciones en una misma área y entre áreas. En la gran mayoría de los sitios, la densidad poblacional de erizo rojo fue menor a 1 erizo m^{-2}, y se conformó por densidades mayores de reclutas y juveniles que pudieran mitigar parcialmente los efectos de la captura. Los resultados de esta reevaluación permitieron identificar sitios específicos en donde los atributos poblacionales (biomasa y densidades), información pesquera (captura y esfuerzo) y la combinación de ambos (diagramas de Kobe), sugieren que las poblaciones de erizo rojo requieren de mayor atención. Además, se resalta la necesidad de adoptar nuevas medidas de manejo, tales como una talla máxima legal de 110mm, mejorar la recolección de información mediante las bitácoras de captura y dar seguimiento permanente a la población, independiente de la información de la pesquería. Estas medidas permitirán observar los cambios en la población que no son tan aparentes cuando se observan únicamente las capturas/biomasas. Además, es importante reforzar la talla mínima de captura ya que los resultados sugieren que los sitios con altas abundancias de erizos pequeños tienen el potencial de soportar mayores capturas. Una vez hecha la reevaluación del estado de las poblaciones de erizo rojo, nos enfocamos en comprender como la variabilidad ambiental local, regional y oceánica puede afectar a las poblaciones de erizo rojo. Con este fin, se desarrollaron modelos generales lineales (GLM) para diferentes condiciones termales observadas: 1) Condiciones normales, 2) durante una onda cálida y 3) post onda cálida. Además, se identificaron los sitios cuyas características termales durante el periodo analizado (2000 – 2018) fueron superiores, inferiores o promedio y se desarrollaron modelos generales lineales para estos grupos de sitios. Las variables utilizadas fueron: capturas de erizo rojo, pez vieja, cabrilla sargacera y langosta roja, biomasa de sargazo gigante, temperatura superficial del mar, potencia del oleaje, índice de surgencias, índice multivariado de “El Niño” y el índice de la oscilación del giro del Pacifico Norte. Nuestros resultados mostraron que los sitios con mayor densidad de erizo rojo fueron Isla Todos Santos, Punta Banda y Punta Baja; las capturas de pez vieja fueron mayores en Punta Banda y Punta Baja, mientras que las capturas de cabrilla sargacera fueron mayores en Canoas. La captura de langosta fue mayor en la región de Punta Baja – Isla San Jeronimo – Arrecife Sacramento. Observamos que la variabilidad ambiental local, así como la disponibilidad de alimento y abundancia de depredadores fue de mayor importancia para los cambios en la densidad de erizo rojo, comparado con las variables de escala regional (índice de surgencias) o de escala oceánica (MEI y NPGO). También observamos que la importancia de estas variables cambia dependiendo de la escala espacio – temporal considerada, lo que significa que las condiciones “normales o promedio” están reguladas por variables distintas, comparadas con condiciones ambientales extremas como El Niño o La Mancha. Además, nuestros resultados sugieren que en ausencia de los principales depredadores de erizo rojo en la región, los buzos se han convertido en el principal control poblacional, por lo que mejorar el manejo de la pesquería puede traer beneficios directos a las poblaciones de erizo rojo y las comunidades costeras que dependen de este recurso y beneficios indirectos al ecosistema asociado a los mantos de *Macrocystis pyrifera* en México.

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Contenido

Capitulo I. Introducción General

Conocer las variaciones espaciales y temporales de una población es vital para proponer estrategias de manejo que permitan el aprovechamiento sostenible de los recursos. Factores físicos como surgencias, temperatura del agua, sedimentación, acción del oleaje, además de la pesca, pueden ser determinantes en la abundancia de las especies marinas (Hereu et al., 2012). El aprovechamiento sostenible de las especies con valor comercial requiere de estimaciones precisas de parametros poblaciones y un entendimiento profundo de las causas de la variabilidad espacio – temporal asociada a esas estimaciones (O'Malley, 2009).

El erizo rojo *Mesocentrotus franciscanus*, es uno de los equinodermos más estudiados y conocidos en el mundo. Es una especie dominante de la comunidad de arrecifes rocosos en la costa oeste de Norte América, capaz de estructurar las comunidades algales submareales y determinar indirectamente la diversidad de la comunidad (Rogers – Bennet, 2007). Esta especie se distribuye desde Alaska hasta la Isla de Cedros en Baja California y habita en la zona submareal hasta los 90 m de profundidad (Ebert et al., 1999). Son organismos dioicos con fertilización externa y larvas planctónicas que se dispersan a lo largo de la costa en función de la intensidad de los giros, turbulencia y velocidad de las corrientes de cada sitio y cuyo asentamiento depende de las señales químicas particulares del sustrato de cada sitio (Ebert et al., 1994; Botsford, 2001; Largier, 2003; Gaylord et al., 2013). Las poblaciones de erizo rojo son susceptibles a variaciones naturales como el aumento de la temperatura del océano (debido a eventos El Niño, la "Mancha", la oscilación decadal del Pacifico, entre otros), la cual puede tener efectos directos en sus poblaciones como el aumento de enfermedades (Gilles y Pearse, 1986; Scheibling y Lauzon-Guay, 2010; Scheibling et al., 2010), así como disminuir el éxito reproductivo (disminuyendo la longevidad de los gametos) y la calcificación (Byrne et al., 2009; Rahman et al., 2009). También se han observado efectos indirectos a través de la cadena trófica, ya que las temperaturas altas promueven el reclutamiento de la langosta roja (*Panulirus interruptus*) y el pez vieja (*Semicossyphus pulcher*), principales depredadores de juveniles del erizo rojo, generando una alta mortalidad post asentamiento (Tegner y Levin, 1983; Cowen, 1983; Hamilton y Caselle, 2015). El Niño del 97–98 destruyó completamente los mantos de sargazo de *Macrocystis pyrifera,* principal fuente de alimento de los erizos, debido a la presencia de aguas cálidas pobres en nutrientes y oleaje por arriba del promedio. Después de este evento, las poblaciones de *M. pyrifera* tardaron en recuperarse más de dos años (Edwards, 2004). Sin embargo, durante el Niño 97–98, el aumento de temperatura superficial fue minimizado por una intensificación en las surgencias costeras (Gonzalez et al., 1998) y, como consecuencia, no se observaron mortalidades masivas de invertebrados costeros. Durante el Niño 82–83, se observaron bajos niveles de asentamiento y reclutamiento de erizos juveniles, consecuencia de un bajo rendimiento gonadal de los adultos y una producción limitada de larvas, así como larvas planctónicas famélicas por las bajas concentraciones de nutrientes en el agua y patrones de circulación modificados que alejaron a las larvas competentes de los sitios de asentamiento (Tegner y Dayton, 1991). La deforestación de los mantos de sargazo no sólo implica la disminución de la disponibilidad de alimento para los erizos, sino que también puede provocar una disminución de la abundancia de larvas y de la supervivencia post asentamiento (Watanabe y Harrold, 1991).

En escalas interanuales, las fluctuaciones dinámicas y biogeoquímicas del Pacifico Norte están dominadas por dos tipos de variabilidad: la Oscilación Decadal del Pacifico y la Oscilación del Giro del Pacifico Norte (PGO y NPGO, respectivamente por sus siglas en ingles). Se ha observado una relación estadística entre el NPGO y la variabilidad del viento costero (que favorece surgencias) a lo largo de la Corriente de California, el cual es más intenso en invierno (diciembre a marzo). Las fluctuaciones del NPGO están asociadas con un cambio de 1 a 2 meses en el establecimiento de la temporada de surgencias (Chenillat et al., 2012), lo que modifica la disponibilidad de nutrientes en la zona costera. Las surgencias pueden jugar un papel importante en el asentamiento de larvas y la diversidad genética de las poblaciones de erizo a lo largo de la costa de Baja California. Por otro lado, se sabe que existen procesos oceanográficos locales que pueden generar estructuras genéticas particulares (Olivares – Bañuelos et al., 2008). Sin embargo, no se conocen los efectos sinérgicos de eventos oceanográficos de escalas regionales y las variaciones poblacionales de erizo rojo a lo largo de la costa de Baja California.

La explotación pesquera también puede afectar a las poblaciones del erizo. En la costa este de Canadá y el Golfo de Maine, se ha señalado a la pesquería de erizo verde como un depredador tope que puede tener consecuencias profundas sobre sus poblaciones, así como en las comunidades bentónicas asociadas (Scheibling y Hatcher, 2007). En California, después del auge de la pesquería en los años 70 fue necesario modificar las estrategias de manejo, ya que hasta antes de 1987 se trataba de maximizar la captura de erizo para aumentar la abundancia de sargazo gigante, llevando así a una disminución considerable de las poblaciones de erizo rojo al norte y sur del Estado (California Department of Fish and Wildlife, 2003).

En Baja California, el erizo rojo se captura de manera comercial desde 1972, desde Tijuana hasta Isla de Cedros (Jurado–Molina et al., 2009). Esta pesquería genera cerca de 2,000 empleos directos e indirectos durante cada temporada de pesca. La pesquería de erizo rojo cuenta con una larga historia (Fig. 1) y está regulada por la NOM-007-PESC-1993 (ahora NOM-007-SAG/PESC-2015; DOF 25-Jun-2015); sin embargo, Jurado–Molina et al. (2009) reportaron que la biomasa del stock disminuyó drásticamente a partir de 1985 y desde el 2004 representa únicamente el 17% de la biomasa virgen. Evaluaciones independientes de las poblaciones de erizo rojo realizadas en 10 sitios de Baja California, reportan que la densidad de las poblaciones estudiadas ha disminuido de más de 8 erizos por m^{-2} en 2003 a menos de 4 erizos por m^{-2} en 2008, así como una disminución en las tallas de 80 mm a menos de 60 mm (Palleiro–Nayar et al. 2012). Estos autores atribuyen estos cambios a la intensidad de pesca y reclutamiento esporádico.

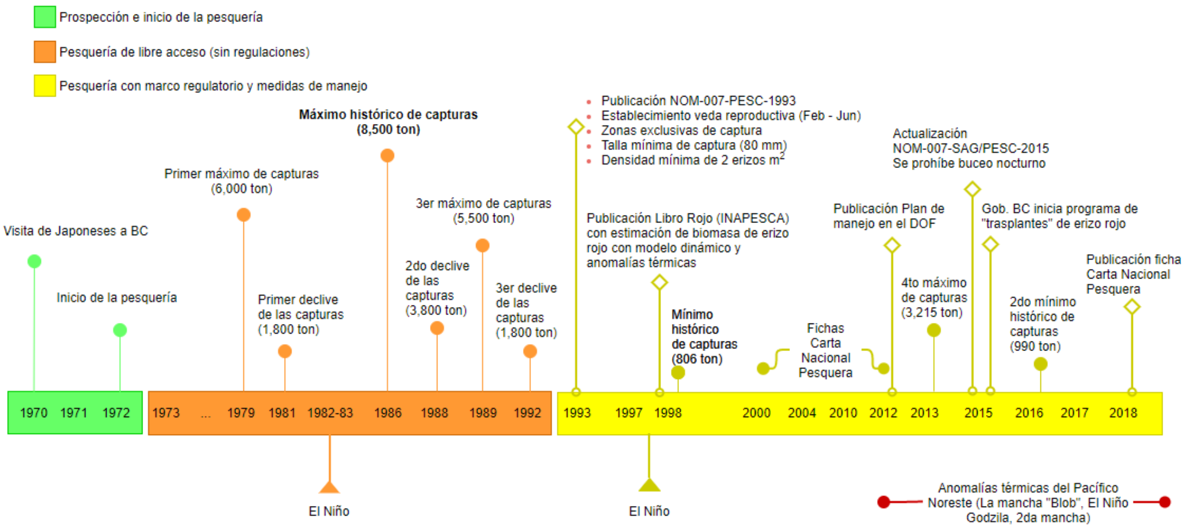

Figura 1. Breve historia de la pesquería de erizo rojo en México. Los círculos representan puntos importantes en las capturas; los rombos representan puntos cardinales en el manejo de la pesquería; los triángulos representan eventos climáticos relevantes. Elaboración propia con información de fuentes oficiales.

La sobrepesca y los cambios ambientales imponen un gran riesgo al bienestar de los stocks de recursos pesqueros en el mundo. El manejo pesquero procura disminuir los riesgos asociados a la extracción en las poblaciones aprovechadas, desafortunadamente, existen fuentes de incertidumbre asociados a la información y a los métodos estadísticos para la evaluación de stocks, situación que incrementa la incertidumbre de los resultados obtenidos. El reto se vuelve mayor cuando los stocks con más necesidad de ser evaluados carecen de suficiente información y los recursos para recabar nueva información son escasos (Pulkkinen, 2015). La evaluación de stocks pesqueros incluye diversas fuentes de incertidumbre, especialmente cuando el conocimiento de la abundancia, éxito reproductivo, patrones de migración o comportamiento de las especies marinas es muy limitado. Muchos factores pueden determinar el estado de los stocks, incluyendo la disponibilidad de alimento, niveles de depredación, factores ambientales y la influencia humana por la pesca, eutrofización y contaminación. La información disponible sobre el estado de los stocks y los muchos factores que los afectan es escasa, sesgada y contradictoria (Pulkkinen, 2015). A pesar de estos retos, existe la necesidad de contar con estimaciones lo más certeras posibles acerca de los stocks pesqueros para así poder generar medidas de manejo encaminadas a su aprovechamiento sostenible (Hilborn y Walters, 1992).

En Baja California, Jurado–Molina et al (2009), desarrollaron un marco bayesiano para la valoración del stock del erizo rojo, así como un análisis de decisión de la pesquería en Baja California, proyectando diferentes escenarios a futuro. Estos autores utilizaron información de la captura y la Captura por Unidad de Esfuerzo (CPUE) para estimar la distribución de probabilidad de los parámetros del modelo mediante simulaciones Monte Carlo de cadenas de Markov; sin embargo, todas las estimaciones se basan en información pesquera y muestreos independientes, sin incluir parámetros ambientales.

En el presente trabajo se construye y aplica el primer modelo que toma en cuenta la complejidad y fuentes de variación que afectan a las poblaciones de erizo rojo en Baja California. Debido a la relación tan estrecha entre las poblaciones de erizo rojo, los mantos de sargazo, sus depredadores (langosta y pez vieja), así como los efectos de variables y fenómenos ambientales relevantes (temperatura, oleaje, surgencias, el Niño y NPGO), resulta de vital importancia incluir todos estos factores para modelar y predecir con mayor precisión las variaciones en las poblaciones de erizo rojo.

Hipótesis

Existe un efecto diferenciado de las variables y fenómenos ambientales y biológicas sobre la pesquería de erizo rojo que puede ser integrado en modelos predictivos para generar escenarios que informen las estrategias de manejo de la pesquería.

Objetivo general

Desarrollar un modelo que integre variables oceanográficas y biológicas relacionadas con la población de erizo rojo, con la finalidad de pronosticar trayectorias futuras en diferentes escenarios de variabilidad ambiental, biológica y pesquera.

Objetivos particulares

1.1. Evaluar el estado poblacional y el de la pesquería del erizo rojo *Mesocentrotus franciscanus* en Baja California

1.2. Determinar la importancia de las variables y fenómenos oceanográficos (, Temperatura superficial, El Niño, la Oscilación decadal del giro del Pacifico Norte y el Índice de surgencias), así como de las biológicas (abundancia de la langosta roja, pez vieja, pez cabrilla y *M. pyrifera*) en las fluctuaciones observadas en la pesquería y población del erizo rojo.

1.3. Elaborar un modelo que integre los componentes oceanográficos y biológicos antes mencionados, para pronosticar las variaciones del erizo rojo bajo diferentes escenarios de cambio climático y pesquero.

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Capitulo II. Re planteando la historia de la pesquería de erizo rojo en México

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Resumen

La pesquería de erizo rojo tiene una larga historia de aprovechamiento y manejo en la costa noreste del Pacifico. En México, el erizo rojo se aprovecha comercialmente desde 1972 y, a pesar de ser una de las pesquerías más importantes de Baja California, los esfuerzos para evaluar la condición y la dinámica de los stocks aprovechables han sido únicamente sobre algunas áreas específicas y con escasa información independiente a la pesquería. El presente trabajo reevalúa las tendencias poblacionales, el esfuerzo pesquero y las capturas, incorporando toda la información disponible en los últimos 19 años (2000 – 2018). Para ello, se agrupó y analizó toda la información en 14 sitios de desembarque ubicados a lo largo de la costa oeste de Baja California. Se implementó un análisis virtual de población basado en tallas (LVPA, por sus siglas en ingles) para estimar las tasas de captura, mortalidad por pesca y densidad por sitio. Las capturas de erizo rojo tienen una gran variabilidad en una misma área y entre áreas. En la gran mayoría de los sitios, la densidad poblacional de erizo rojo fue menor a 1 erizo m^{-2}, y se conformó por densidades mayores de reclutas y juveniles que pudieran mitigar parcialmente los efectos de la captura. Los resultados del LVPA fueron el doble de las estimaciones de biomasa previas. Sugerimos que el modelo utilizando para

estimar la biomasa, así como la relación entre sus parámetros, puede no reflejar características biológicas clave del erizo rojo y no reproducir las tendencias poblacionales de manera adecuada. Los resultados de este trabajo permitieron identificar sitios específicos en donde los atributos poblacionales (biomasa y densidades), información pesquera (captura y esfuerzo) y la combinación de ambos (diagramas de Kobe), sugieren que las poblaciones de erizo rojo requieren de mayor atención. Es necesario adoptar nuevas medidas de manejo, tales como una talla máxima legal de 110mm, mejorar la recolección de información mediante las bitácoras de captura y seguimiento permanente e independiente de la pesquería para observar los cambios en la población que no son tan aparentes cuando se observan únicamente las capturas y biomasas calculadas, además de reforzar la talla mínima de captura ya que los resultados sugieren que los sitios con altas abundancias de erizos pequeños pueden soportar capturas mayores.

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Retelling the History of the Red Sea Urchin Fishery in Mexico

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The red sea urchin fishery has a long harvest and management history along the Northeastern Pacific coast. In Mexico, it has been commercially harvested since 1972, and although it is one of the most important fisheries in Baja California, efforts to assess the condition and dynamics of harvestable stocks have been focused on certain harvested areas with scarce fisheries independent data. Additionally, the analysis of yearly information for small geographic areas has obscured the actual status of harvested populations. This study aims to re-assess population trends, fishing effort, and catches, incorporating all available information from the last 19 years. Information was grouped based on 14 landing sites along Baja California's Pacific coast. Length based virtual population analysis (LVPA) was implemented to estimate site-specific catch rates and densities. Red sea urchin catches/landings varied widely within and between areas. Population density was below 1 urchin m^{-2} in most of the sites, and was composed of higher recruits and juvenile densities that may partially mitigate for fishery removals. LVPA produced biomass estimations that double previous estimates. We suggest that the model parameters used in previous estimations might not reflect key biological traits of the red sea urchin, failing to reproduce population trends accurately. Results from this study allowed identifying the specific sites where population attributes (biomass, densities), fishery data (catch, effort), and the combination of both (Kobe plots), suggest that urchin populations may need attention. New management measures must be adopted: maximum legal size of 110 mm, improvement on fishery logs and analysis, continuous fishery independent surveys to track changes in the population that might not be so apparent when observing only catch/biomass data. Reinforce the under legal size management strategy, since results suggest that sites with high abundances of small urchins can support higher catches.

Keywords: red sea urchin, VPAs, fisheries, population trends, harvest rate, *Mesocentrotus franciscanus*

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INTRODUCTION

Red sea urchins (*Mesocentrotus franciscanus*) are one of the must-studied echinoderms in the Northeaster Pacific coast, from Mexico to Alaska (Ebert et al., 2018). As a fishery, it has a long harvest and management history along the Pacific coast (Pfister and Bradbury, 1996; Andrew et al., 2002). According to historical records kept by the Mexican National Fisheries and Aquaculture

Institute (INAPESCA), this fishery began in 1970 as Japanese divers taught Baja Californians to harvest red sea urchin. When the fishery started there were no management strategies, nor processing plants for the roe (Palleiro-Nayar, 1982, 2013), and the lack of regulations lasted for over 20 years. The first official regulations emerged in 1993, and consisted on limited effort (one diver per vessel), fishing seasons based on reproductive peaks (closed from March to July each year), exclusive fishing areas, total allowable permits, fishing logs, minimum urchin density for fishing sites, minimum legal harvest size (80 mm and up) based on unpublished reproduction studies (Tapia, 1986; Ruiz et al., 1987), and even individual catch quotas. All these regulations, but the quotas, have been kept in place since, with little or no modifications. The fishery management plan was published in DOF (2012), and more recently the NOM-007-SAG/PESC-2015 (DOF, 2015) incorporates the prohibition of night diving and allows a maximum of 5% under legal size urchins; also, the purple sea urchin harvest was included. Catch quotas were abandoned around 1996 due to under reporting and difficulty to track individual quotas (Palleiro-Nayar, 2013). Although this is one of the most important fisheries in Baja California, with revenues that range from 1.2 to 4.2 million dollars a year (Palleiro-Nayar, 2013), efforts to assess the condition and dynamics of harvestable stocks have been focused on certain harvested areas, with infrequent or no fisheries independent data for all fishing areas (Palleiro-Nayar et al., 2011, 2012; Palleiro-Nayar, 2013); also, information on Baja California's red sea urchin population recruitment is lacking. Moreover, the information has been analyzed as yearly summarized data that represent small geographic areas that obscures the actual status of the different harvested populations. Although the fishery was officially considered as overfished (Jurado-Molina et al., 2009; Palleiro-Nayar et al., 2011, 2012), it lacks a stock assessment analysis to evaluate its current status. The present study incorporates all available information to re-analyze the last 19 years of the fishery, in order to assess total population trends, fishing effort, and catches. Main findings attempt to provide an integrated tool for better managing this important fishery.

MATERIALS AND METHODS

Catch/landing data were requested through Mexico's National Transparency Platform[1]. Database contained harvest date, whole red sea urchin catch in kg, landing site and number of vessels used for fishing. Number of vessels was considered equal to number of divers because regulations allow one diver per vessel only. An unstructured interview of all permit holders was conducted to gather information on latitude/longitude of fishing areas that were plotted and delimited to red sea urchin depth range and maximum dive depth reported by urchin divers (5–30 m). All information was grouped based on landing site (14 sites total) distributed along Baja California's Pacific coast (**Figure 1**). In order to better understand and describe fishing effort and avoid duplicating fishing days, effective fishing days were counted for each permit holder per site, so that in any given month the minimum and maximum number of days fished were between 1 and 31 (depending on each month) (**Supplementary Table S1**). Effort was then calculated using the reported number of vessels/divers and effective fishing days on each site. CPUE was calculated as tons of whole red sea urchin per diver per effective fishing day. All daily data were summarized in monthly and yearly data for each site.

Information on catch size structure and percentage of under legal catch was gathered from government public records (Sea Urchin Management plan; DOF-17-Abr-13), for the 2000–2010 fishing seasons, and was used to estimate the percentage of catch corresponding to legal (>80 mm) and sub legal (<80 mm) red sea urchins for seasons 2011–2018. Catches of sub legal sized urchins were estimated using average values based on official records of catch samplings at processing plants and permit holder plant surveys (**Supplementary Figure S1**).

Weight at length data of red sea urchins sampled at a processing plant was used to transform Catch/landing data to number of red sea urchins per ton:

$$N = {}^{1 \times 10^6}/_{w}$$

where N is the estimated number of red sea urchins in one ton of catch, and w is the average weight (in grams) for each size class.

Length information from plant surveys was used to perform site-specific LVPA (Jones, 1981, 1984; Punt et al., 2013) to calculate the numbers by length-class using the formula:

$$N_{L1} = N_{L2} \left(\frac{L_\infty - L1}{L_\infty - L2} \right)^{M/k} + C_{1-2} \left(\frac{L_\infty - L1}{L_\infty - L2} \right)^{M/k}$$

where N_{Li} is the number of urchins at length L_i, C_{1-2} is catch in numbers between lengths L_1 and L_2, L_8 and k are growth parameters for individuals in the population and M is the rate of natural mortality.

Natural mortality was estimated for each site using Pauly's (1980) equation:

$$\begin{aligned} \log(M) &= 0.0066 - 0.279 \left(\log\left(L_\infty\right)\right) + 0.6543 \left(\log\left(K\right)\right) \\ &\quad +0.4634 \left(\log\left(T\right)\right) \end{aligned}$$

where L_8 and K are growth parameters for individuals in the population and T is temperature (in°C). We used site-specific sea surface temperature data (**Figure 1B**; $T \geq 9.6 \leq 18.10 \geq 29.9°C$), based on monthly multisensory SST time series for 2000–2018 (4 km pixel, daylight at 11 μ; from sensors AVHRR, MODIS Terra, MODIS Aqua and VIIRS Suomi-NPP). Images were processed in SEDAS (7) following Kahru et al. (2015) and Kahru et al. (2012) criteria. L_8 and k growth parameters used in both estimations where based on Rogers-Bennett et al. (2003), and varied depending on urchin size ($L_\infty \geq 134.56 \leq 139.90$ mm, and $k \geq 0.033 \leq 04.38$). Fisheries independent information on size structure for 9 sites for 2003, 2005, and 2006 were taken from Palleiro-Nayar et al. (2012), and was used as template to group VPA size class data into four size classes: Recruits ($\geq 7 \leq 37$ mm), juveniles ($\geq 42 \leq 52$ mm), sub-adults ($\geq 57 \leq 77$ mm), and adults

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[1] https://www.infomex.org.mx/gobiernofederal/home.action

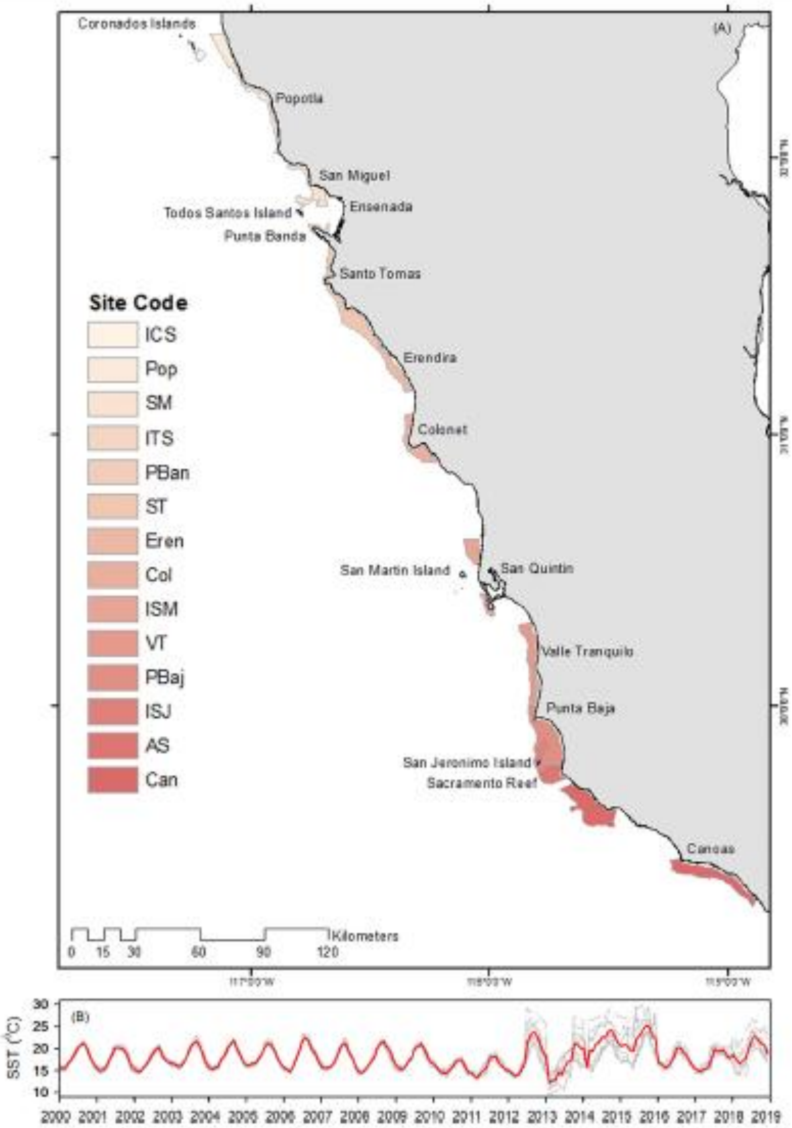

FIGURE 1 | Depth delimited authorized fishing areas for red sea urchin **(A)**, and SST time series profile **(B)** for each site (gray line) and average (red line). The color bars were used to separate sites only. Sea surface temperature was used to estimate natural mortality. Note the wide variations among sites in SST from 2012 to 2018.

($\geq$80 mm). N_{Li} was later transformed into biomass (B_{Li}):

$$B_{Li} = \sum \frac{N_{L1} - N_{L2}}{Z_{L1,L2} \times w}$$

where $Z_{\mathrm{L1,L2}}$ is the total mortality for each length interval calculated with:

$$Z_{Li} = F_{Li} + M_{Li}$$

where F_{Li} is fishing mortality for each length interval estimated using:

$$F_{Li} = M_{L1,L2} \times \left(\left[\frac{C_{L1,L2}}{N_{L1} - N_{L2}} \right] \Big/ 1 - \left[\frac{C_{L1,L2}}{N_{L1} - N_{L2}} \right] \right)$$

Sub adult and adult biomass estimations were considered the spawning stock biomass, whereas adult biomass was considered the harvestable biomass (HB), and the sum of each size class biomass for each year corresponded to the population biomass. Fisheries independent information was not included in the LVPA to avoid unrealistic catch of size classes that are not being taken by the fishery.

Biomass estimations for all sites and all size classes were used to calculate harvestable, spawning stock and population biomass proportion to maximum sustainable yield ratio (B/B_{MSY}); overall and site specific fishing mortality to fishing mortality at MSY ratio (F/F_{MSY}) was also calculated. To establish overall and site-specific harvestable, spawning stock and population biomass trends, B/B_{MSY} and F/F_{MSY} ratios were integrated into Kobe plots using KobePlot Ver (2019) software V5.

We calculated site-specific harvest rate (HR) as a percentage of the catch adult biomass estimates:

$$\mathrm{HR}_{\mathrm{site}} = \left(\frac{B_{80,\,\mathrm{site}}}{\mathrm{Ct}_{\mathrm{site}}}\right) \times 100$$

where $B_{80,\mathrm{site}}$ corresponds to biomass estimations for the >80 mm size class per site, and $\mathrm{Ct}_{\mathrm{site}}$ is the site specific catch.

Each site depth delimited fishing area was measured and red sea urchin preferred substrate area was calculated based on Palleiro-Nayar et al. (2011; **Supplementary Table S2**), coupled with site specific results to estimate sea urchin density:

$$\mathrm{Density} = \frac{N_{\mathrm{site,\,size}}}{dFa_{\mathrm{site}}}$$

where $N_{\mathrm{site},\ \mathrm{size}}$ is the number of urchins estimated for each size class and dFa_{site} is the depth/preferred substrate delimited fishing area in m^2.

All results where geo-referenced and coordinates were randomized within each fishing area; randomization of latitude was limited by northern and southern fishing area coordinates, while longitude randomization was limited by the 5 and 30 m isobaths.

RESULTS

Red sea urchin total annual catches at the beginning of the time series (2000) were 2,200 tons, and remained below 2,000 tons from 2001 to 2008. From 2009 on, catches increased around 2,700–3,000 tons, and peaked in 2013 with 3,215 tons. Catches dropped to 989 ton in 2016, the lowest catch recorded for the 19-year period, and remained at low levels (**Figure 2**). Catches of sublegal size urchins were very high from 2000 to 2002, accounting for almost 50% of the catch, they decreased gradually from 2003 onward, and increased again to 25–28% the last three years (**Figure 2**).

During the course of the last 19 years of the fishery, the number of permit holders averaged 36.8 (max 41), with a maximum of 380 divers, accounting for 2,058 fishing days. Not all sites reported catch for the 228 months analyzed, with a minimum of 63 months for Isla San Jeronimo (ISJ) and maximum of 176 months for Punta Banda (PBan) (**Supplementary Table S1**). Average catch for all sites oscillated from 100–300 tons, and showed a smooth increase along the 18 years. Catch was highly variable depending on site, with highest catches, above average most of the years, at ST, PBaj, AS, and PBan. These sites showed high variability in catches during the 18 years; in contrast, lower catches were common at Isla Coronado (ICS) and ISJ, with low variability during the same period. Average catch remained below 150 tons for 2017–2018 (**Figures 3A,B**). Average cumulative yearly effort oscillated from 0 to 1,000 (divers per effective fishing day), with a similar decreasing trend as catch in 2016–2018. PBaj displayed the highest average effort and variability of all sites, while ICS, Isla Todos Santos (ITS), and ISJ showed the lowest average effort (**Figures 3C,D**). Average CPUE displayed a trend similar to that observed in average catch, with a slight increase from 2008–2016, and a sudden drop in 2017–2018 (**Figure 3E**). All sites displayed average CPUE values between 0.1 and 0.45 ton/diver/effective fishing day, with Col being the site with lowest average CPUE, and Pop the highest (**Figure 3F**).

Site-specific HB estimates (urchins >80 mm) showed low values from 2000 to 2008, a subsequent period where average biomass levels increased (2009–2015), and a steep decline in 2016 at all sites; specially at PBaj where HB fell from 6,230 ton in 2015 to 115 ton in 2016 (**Figures 4A,B**). Two sites displayed the lowest HB values: ICS and ISJ; while four sites showed HB above 1,500 ton: PBaj, AS, ST and PBan (**Figure 4B**). Average HR was above 60% in 2000 and the average for the entire period analyzed was 20.43% (SE ± 0.97). Yearly HR was above 60% in almost all sites in 2000, excluding ITS and ISJ, and decreased to 20% and below throughout 2002–2015; however, all sites displayed HR above 40% for the 2016–2018 fishing seasons (excluding ICS). The lowest HR was calculated for ICS in 2006 (0.88%), while the highest HR was for Pop in 2000 (74.85%). None of the sites displayed constant HRs (**Figures 4C,D**).

Average densities were below 3 urchin m^{-2} most of the years; densities showed a smooth increase from 2003 to 2015, and sharply declined in 2016. Densities reached 7 urchin/m^2 on 2010 and 8 urchin/m^2 2015 (**Figure 5A**). ISJ and Can showed very low average densities (under 0.5), higher values were common at Pop, ST, Eren, PBaj, and AS, just above 2 urchin m^{-2}; only ITS and PBan presented average densities above 10 urchin m^{-2} (**Figure 5B**). A similar pattern among sites was observed with recruits (7–37 mm), juvenile (42–52 mm), and sub-adults (57–77 mm) average densities, although values among size classes differed. Recruits were more abundant (densities = 2.21 urchins m^{-2} ± 0.12); juveniles and sub-adult densities averaged 0.79 (±0.04) urchins m^{-2}; and adult densities were low, with only 0.26 (±0.01) urchins m^{-2}. We also observed a significant correlation between adult density and recruits in all sites ($0.97 = r^2 = 0.90$, $p < 0.001$; **Figure 6**).

Mapping catches/landings and population density revealed the geographic overlap between areas where fishing occurs with high yields, and low average population densities, with the exception of ITS and PBan where yields and population densities were high. With this representation of catch and population densities, it was evident that large areas of Baja California display low urchin population densities, even sites that have been traditionally branded as "overrun by urchins" such as PBaj, ISJ, and AS (**Figure 7**).

Population biomass estimations in this study showed a decreasing trend from 1,000 ton in 2000 to a minimum of 192 ton

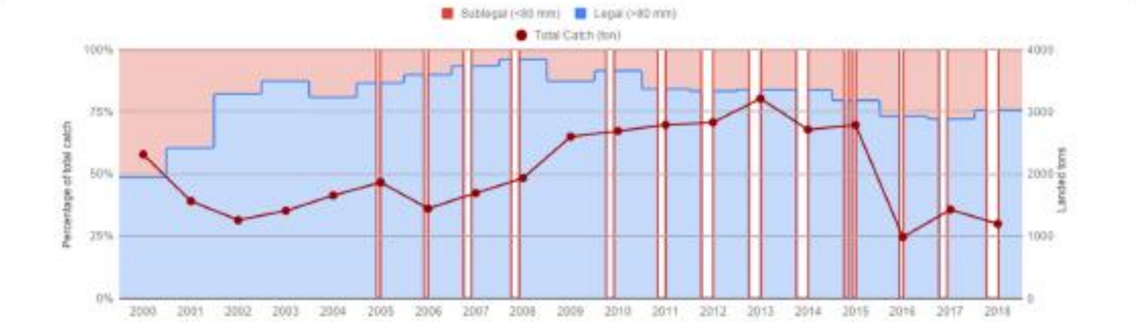

FIGURE 2 | Total catch/landings for the red sea urchin fishery from 2000–2018. Percentage of legal and sub legal red sea urchins represent monthly catches. Gaps represent closed seasons.

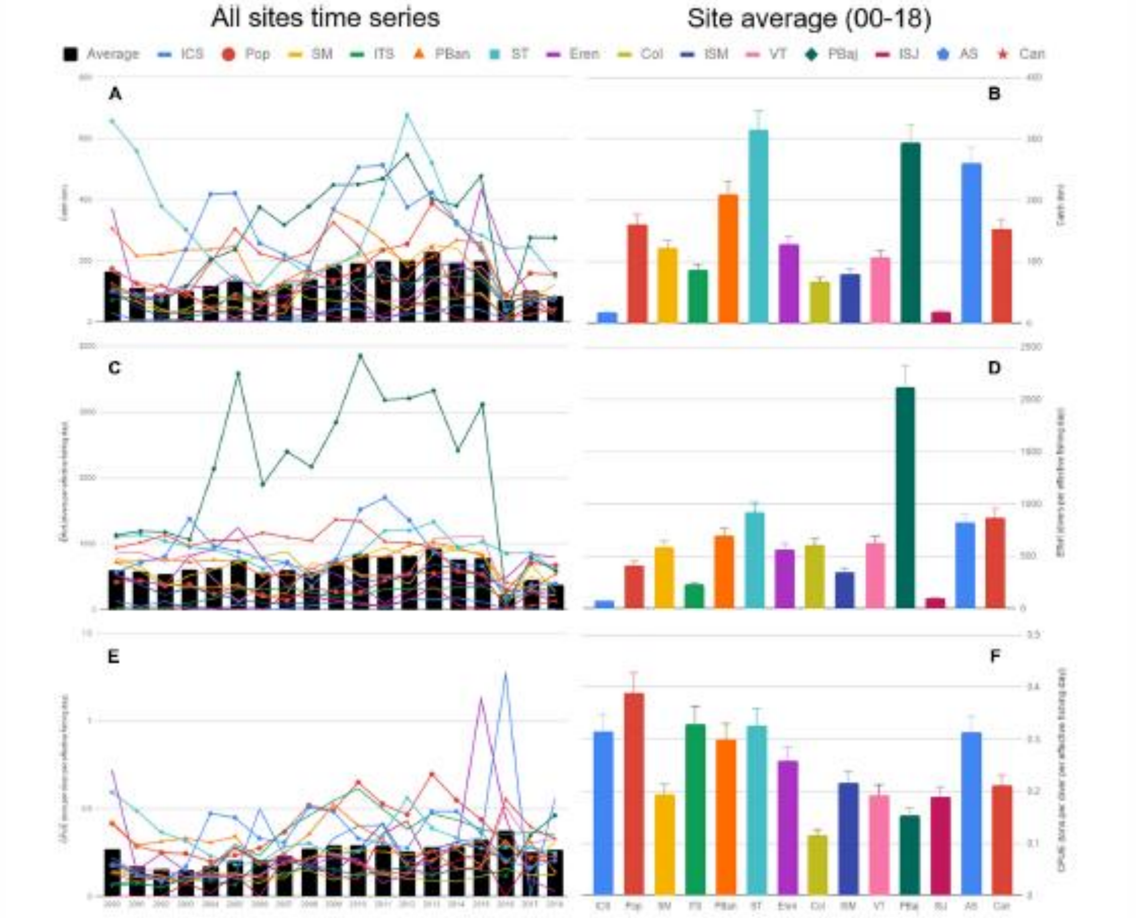

FIGURE 3 | Average catch, effort and CPUE time series **(A,C,E)**, and the corresponding average per site **(B,D,F)**. Bars represent SE.

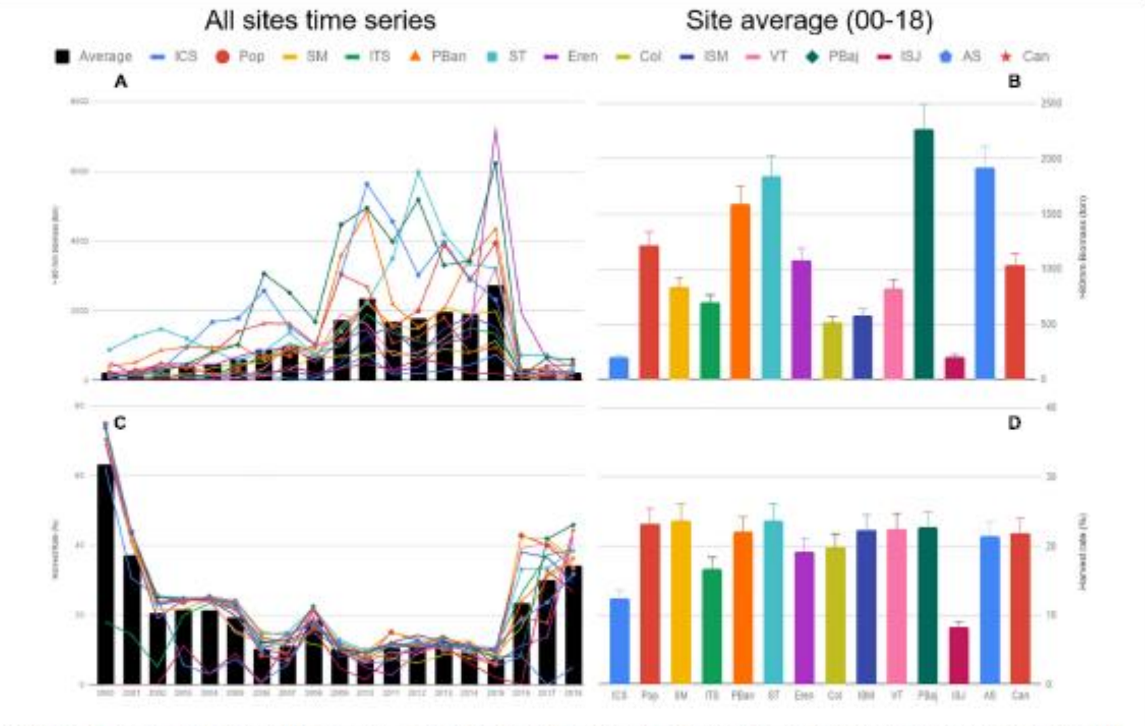

FIGURE 4 | Red sea urchin harvestable biomass (urchins >80 mm) **(A)**, and harvest rate (%) **(C)** from 2000 to 2018, and the corresponding average per site **(B,D)**. Bars represent SE.

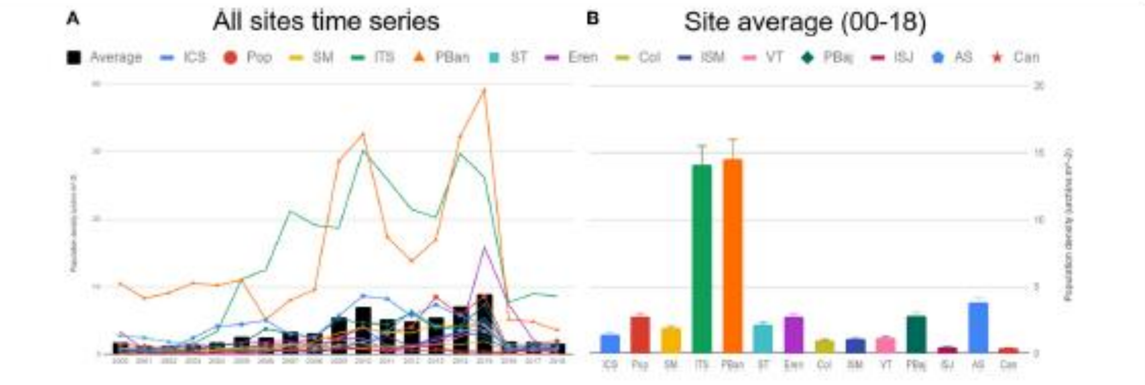

FIGURE 5 | Average population density estimates from 2000 to 2018 **(A)**, and 19-year site average **(B)**.

in 2010. Official biomass estimations were at least 1,000 ton below the reported catch for each year; data were available only to 2013. Current HB estimations ranged from 3,206.38 ton in 2000 to 3,269.58 ton in 2018, with a maximum in 2015 (38,338.99 ton) and a period of high HB from 2006 to 2015 (**Figure 8**). Biomass estimations from this study correspond with adult density values, in such a way that both density and biomass increased from 2000 to 2010, remained high until 2015, and sharply decreased in 2016 (**Figure 8**).

When analyzing the fishery trajectory and population status through Kobe plots, trends in overall harvestable, spawning stock and population biomass display similar patterns. Starting in 2000, F/F_{MSY} was below the "overfished" threshold ($F/F_{MSY} = 1$), HB levels falling within "recovery" ($B/B_{MSY} < 1$), while spawning

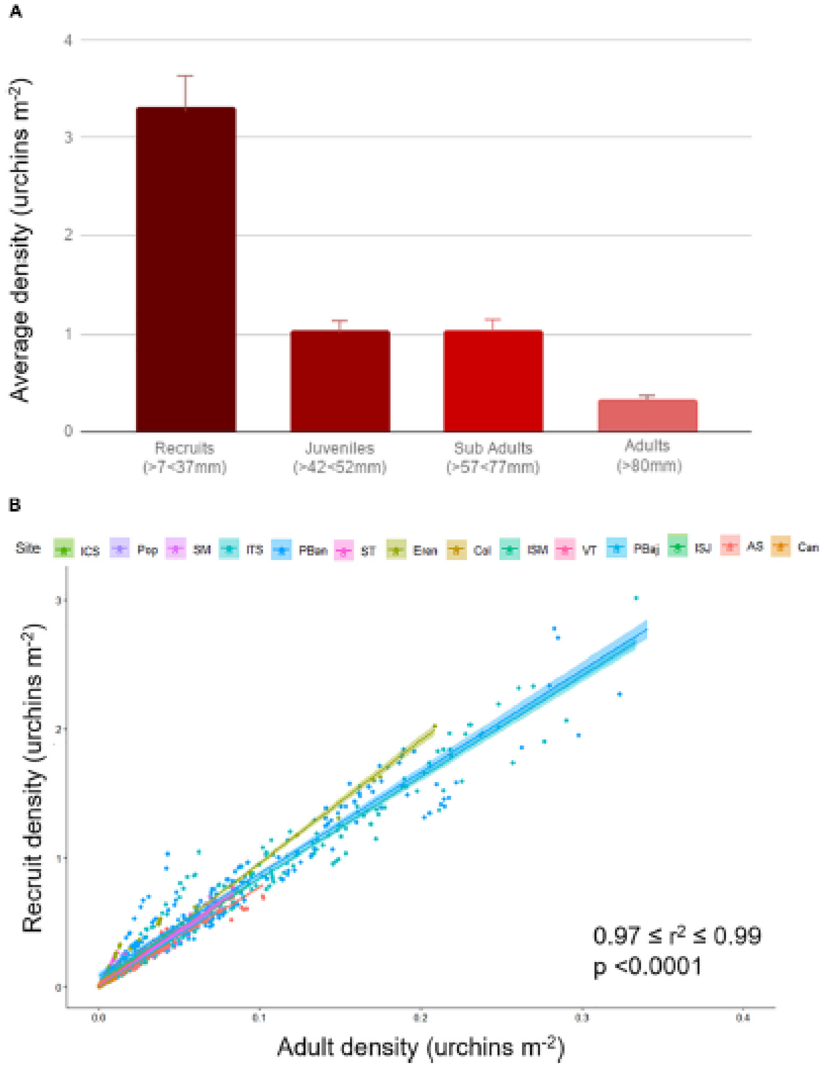

FIGURE 6 | Average red sea urchin density (±SE) estimated by size class, 19-year data pooled **(A)**. Recruits (7–37 mm), juvenile (42–52 mm), sub-adults (57–77 mm), and adults (> 80 mm). Red sea urchin adult and recruit density relation per site **(B)**. Data for each site contains all time series; color shade represents 95% confidence interval.

stock and population biomass fell within the "safe zone" ($B/B_{MSY} > 1$). The population remained in this status until 2009 when a slight increase in biomass was observed. High F/F_{MSY} and B/B_{MSY} ratios were observed in 2012–2013, which resulted both years above the "overfishing" threshold. After 2015, both F/F_{MSY} and B/B_{MSY} declined, and harvestable and spawning stock biomass returned to "recovery" levels (**Figure 9**).

Site-specific Kobe plots showed that each site displayed its own fishing mortality and biomass dynamics (**Figure 10**). Sites like SM, ST, and Isla San Martin (ISM) displayed F/F_{MSY} and B/B_{MSY} ratios that placed the fishery in overfished status for two years at SM and for several years at ISM; particularly the population at ISM remained either as overfishing or overfished most of the years, and stayed in the recovering zone for three years only (2006–2008). The population in ST displayed an overfishing status most of the years. Sites like CAN, PBan, PBaj, Pop, Ere, and VT, stayed at the safe zone several years; although these sites never reached an overfished condition, they spent several years as overfishing. ITS, ISJ, AS, and ICS remained within the recovering and the safe zone all years, and no overfishing occurred at these

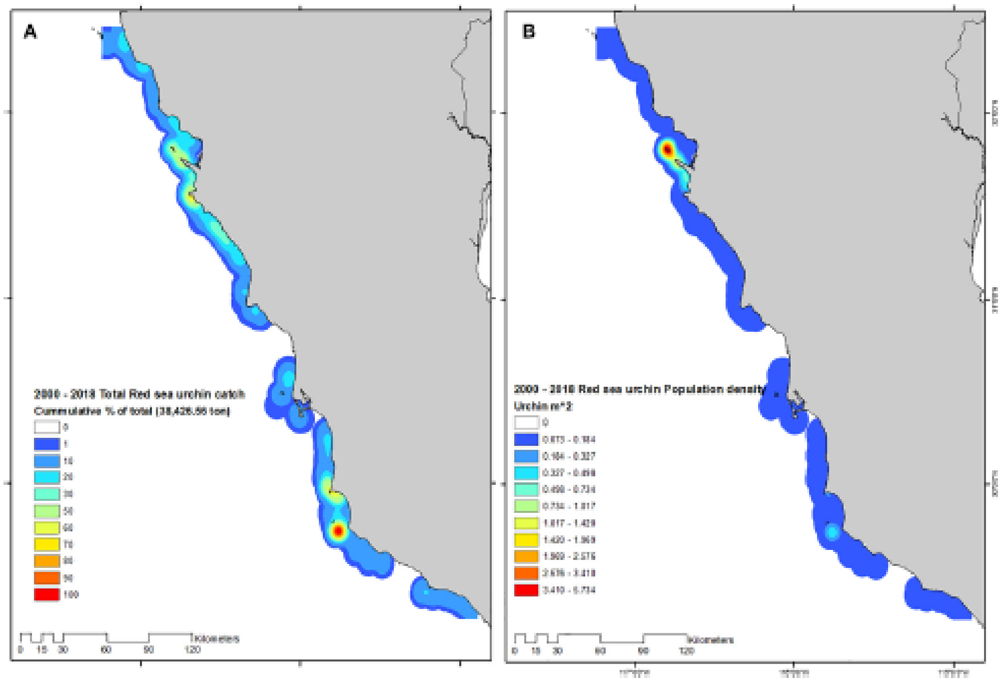

FIGURE 7 | Total catch **(A)**, and population density **(B)** estimated for the 2000–2018 red sea urchin fishing seasons.

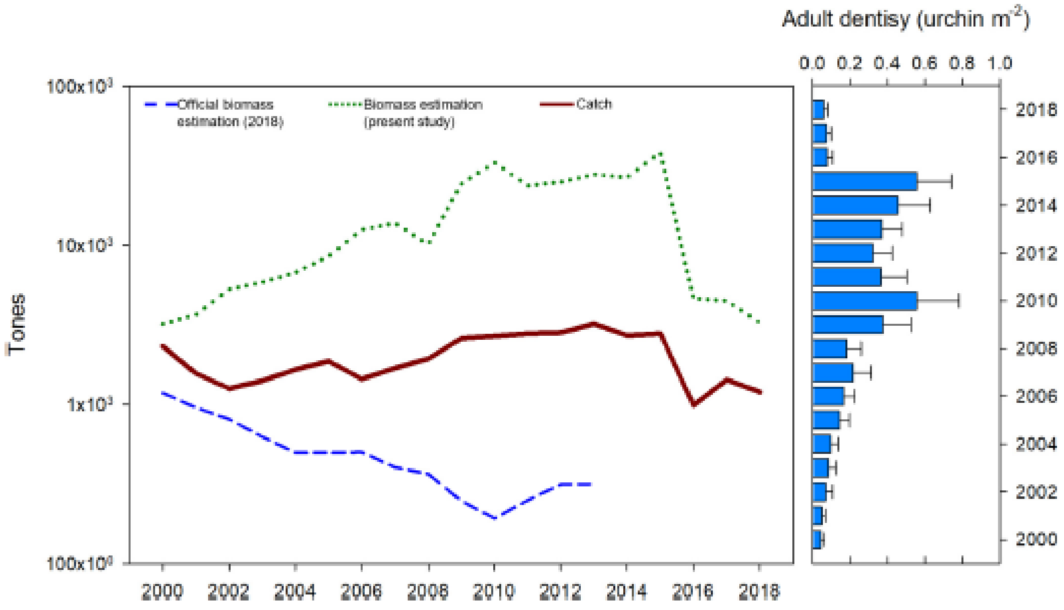

FIGURE 8 | Official red sea urchin estimations (DOF, 2018), total catch, harvestable biomass estimations and adult urchin average density from this study.

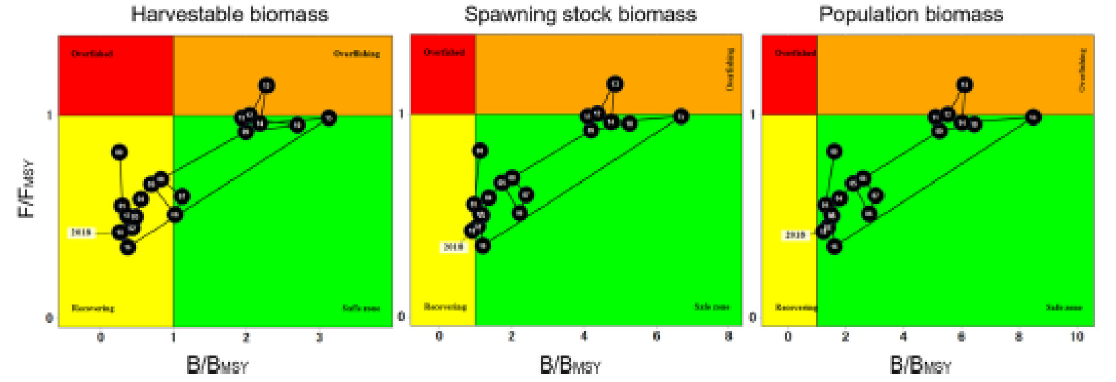

FIGURE 9 | Kobe plots representing 19-year trend of overall harvestable, spawning stock and population biomass of red sea urchin.

sites during the analyzed period, except ICS that experience overfishing one year only. The population at Col stayed in the recovering zone 18 years, and at the safe zone one year only. By 2018, most sites showed a recovering status, except SM and ISM that returned to the overfished zone (**Figure 10**).

DISCUSSION

Red sea urchin catches/landings for the 2000–2018 seasons varied widely within and between areas. Observed variability could be related not only to urchin availability but also to number of permit holders operating in each area, fishable days due to weather conditions, divers' willingness to work and endurance, co-op internal agreements, and market conditions. These variability patterns have also been observed in northern and southern California (Kato, 1972; Kato and Schroeter, 1985; Kalvass and Hendrix, 1997; Morgan et al., 2000; Kalvass and Rogers-Bennett, 2001; Andrew et al., 2002; Schroeter et al., 2009), as a result of a combination of intense fishing, a series of strong El Niño periods, and difficult marketing conditions (DeWees, 2004). Time span analyzed in this study included data from the 2014–2016 highly anomalous years produced by "The Blob" and El Niño (Bond et al., 2015; Schiermeier, 2015; Whitney, 2015). SST data suggest different responses to these events depending on site location; this source of site dependent environmental variability was included in the analysis when calculating red sea urchin natural mortality.

This is the first attempt to introduce "high resolution" CPUE estimates, compared to the ones used to manage Baja California's fishery. Historically, CPUE has been calculated as kg of urchin/day, since official records lack information about the number of dives performed by each diver on a working day, as well as the time spent diving. The lack of detail information accounted for an unreal effort measurement (e.g., official records report effort ≈1,700 days in 2000, which would require 4.6 years to achieve). The present study calculated 105 fished days for the same year (2000) because we included only the days that fishermen actually harvested; yet, a better effort measurement is still needed, such as diving hours per diver, as well as harvest location. This information is key to ensure a reliable CPUE as indicator to provide information on spatial distribution, fishing effort and success, as well as red sea urchin abundance and size (DFO, 2016). Internal agreements within and between permit holders, as well as market status, should also be considered given that in an exploited system the behavior of the harvesting sector may be equally important determinant to predict management options (Wilen et al., 2002), and biological outcomes.

Virtual Population Analysis are excellent for looking at the history of long lived organisms; length – based VPAs are often used to assess stocks of marine resources where age – related information is scarce or species are hard to age (Punt et al., 2013), with the assumption of a steady state and average conditions over time (Gulland and Rosenberg, 1992). However, the use of site specific natural mortality, intrinsic growth rate, and daily catch and effort values may provide some light in determining the impact of fishing mortality on site specific stocks, and thus separating the contributions of natural and fishing mortalities to the total mortality, by maximizing the possible variation in fishing mortality (Gulland, 1987). LVPA analysis implemented in this study produced biomass estimations that double previous estimates from other authors (Ramirez-Felix and Manzo-Monroy, 2004; Jurado-Molina et al., 2009) whose results are based on Schaefer's biomass static/dynamic model, official catch and CPUE data (as total annual values), and fixed intrinsic growth rate (r). We suggest that such biomass estimations and maximum carrying capacity (K) of 24,465 ton, are applicable to harvestable size sea urchins (>80 mm) only, attributable to the use of total annual catch and CPUE data. In addition, both parameters used in Shaefer's model have a specific relation that "force" the biological behavior of the assessed population to be short lived, fast growth or slow growth long lived species $\left(B_t = rB_t\left[1 - \frac{B_t}{K}\right]\right)$. This parameter relation may produce biomass estimates that do not necessarily reflect red sea urchin biological traits as a long lived, fast growing species with no decrease in reproductive potential with increasing age (no senescence; Ebert, 2008); thus, producing biomass estimations

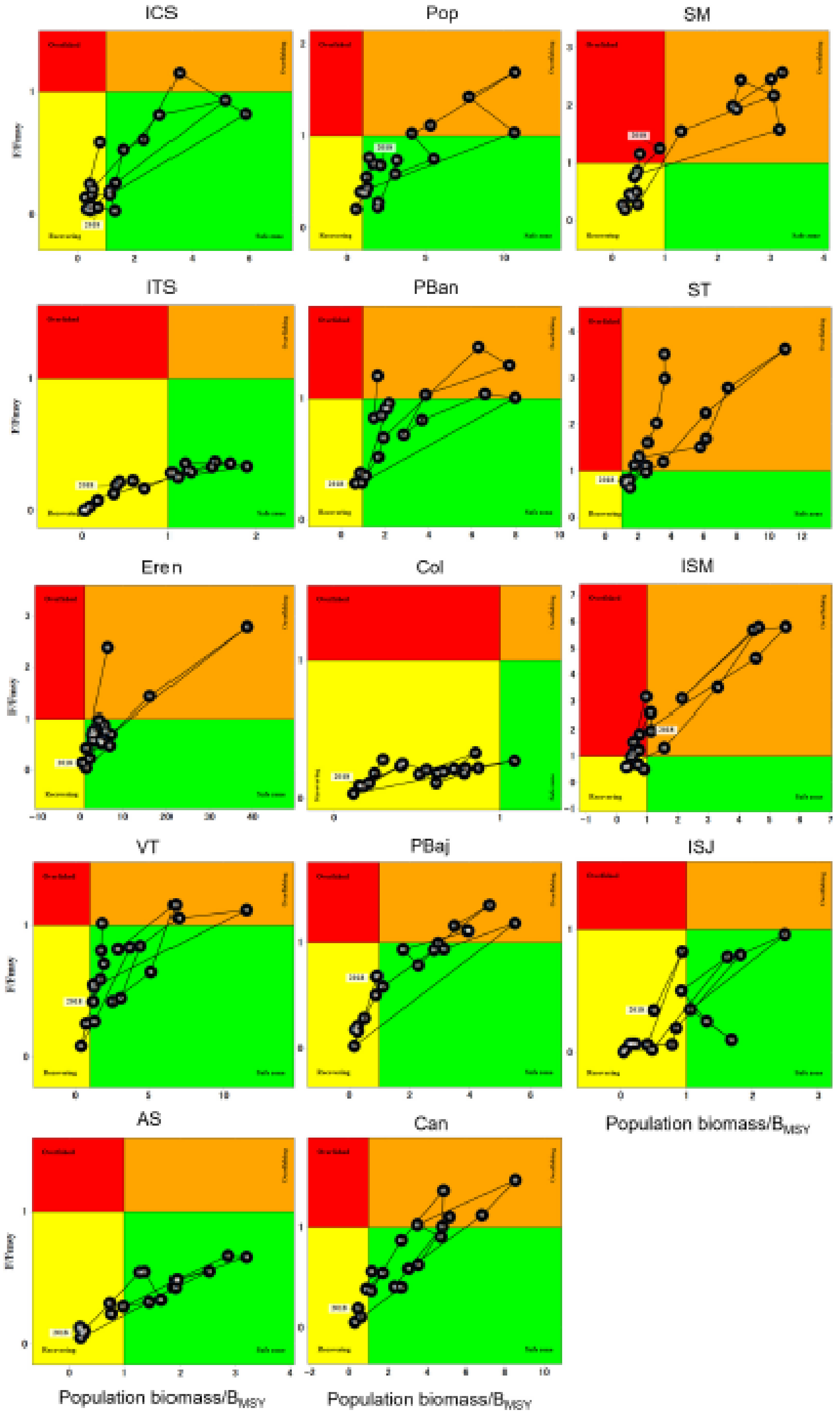

FIGURE 10 | Site-specific Kobe plots for red sea urchin population biomass.

that could not support reported catch levels. LVPA allowed the inclusion of fishery independent information on size class frequencies, urchin size and site dependent k, M and temperature values, that in turn resulted in estimates that better reflect overall population biomass, and not only HB. It also allowed us to estimate red sea urchin densities for the whole 19-year time series, even reflecting *in situ* measurements conducted by Palleiro-Nayar et al. (2012). Fisheries independent information was used as basis to select same size class intervals; however, data from the independent surveys was not included into the LVPA. The structure of the model integrates all data as "catch" and results reflect unrealistic catch of length classes that are off limits to the fishery (minimum legal size >80 mm). This method can be a powerful tool for assessing red sea urchin population and fishery status. Moreover, when revising the Beverton–Holt life-history invariants of our parameters (specifically $\frac{M}{k}$, where M is natural mortality and k is individual growth rate from growth models), estimates fall within what has been estimated for 123 marine species by Prince et al. (2015), with average values of $\frac{M}{k} = 1.89$ (SE $\pm$ 0.051). It is important to note that Prince et al. (2015) did not include echinoderms in their 123 species meta-analysis.

Results from this study allowed identifying the specific sites where population attributes (biomass, densities), fishery data (catch, effort), and the combination of both (Kobe plots), suggest that urchin populations may need attention. Sites such as SM, ST, and ISM have remained at an overfishing status for several years (>8), and overfished (SM and ISM). This condition can be due to the combination of high catches, high mortality rates, and high effort during several years, and the resulted decline of HB and densities. In contrast, AS, ISJ, ITS, and Col never showed signs of overfishing during the 19 years. Results also allowed identifying more productive sites from those with low productivity; for example, ST presented similar average catches, effort, and adult biomass than AS; yet, the population at ST has been overfishing for 14 years, whereas the one at AS has never been either overfishing nor overfished. The same contrast was evident with SM and VT.

Consistent settlement/recruitment may partially mitigate for fishery removals, as is the case for Southern California (DeWees, 2004). Intense harvest may temporary deplete local HB without an effect on population recruitment, since red sea urchins are broadcast spawners which larvae remain pelagic for a period of 2–4 months before settlement (Strathmann, 1978), so they can travel long distances (Mileikovsky, 1971; Strathmann, 1974). However, adult urchins provide protected shelter to juveniles, and this protection may greatly enhance juvenile survival rate (Tegner and Dayton, 1981; Palleiro-Nayar et al., 2011); nevertheless, an exceedingly high adult density may impose negative effect on recruitment due to competition for space and food (Zhang et al., 2011). Similar patterns of HR in different sites may reflect higher recruitment in such areas, but it is paramount to determine and measure recruitment in all sites, and to establish sites that serve as sources and sink for recruits. It is also important to mention that HR is based solely on biomass-catch ratio for legal size urchin (>80 mm, or HB), and is sensitive to changes in biomass; if the available HB is high and catches do not reach the same levels, HRs will be low (as seen in **Figure 4** from 2002 though 2015). High HRs are observable when HB is low and catch is high relative to that biomass (2006–2018, same **Figure 4**). However, for this latter period, fishing effort and fishing mortality drops, HB is also low but catches are high (relative to the HB), producing high HR. Our results show a high degree of selectivity over urchins bigger than >80 mm; however, not all sites displayed high densities of recruits (>7 <37 mm). Higher densities of recruits were only observed constantly at ITS and PBan (sites with also high adult densities). Management strategies should be implemented in all sites where juvenile, sub adult and adult urchin densities fall below 4 urchins m^{-2}; densities under 1–3 urchins m^{-2} could hinder red sea urchins recruitment and reproductive success (Levitan et al., 1992; CDFG, 2001), as well as limit settlement of new recruits to the population and fishery.

Site specific and overall densities for red sea urchin population showed to be heavily skewed toward size classes smaller than 80 mm due to the established legal minimum size limit. This pattern is also observed in areas of British Columbia where red sea urchin populations are subject to high sea otters (*Enhydran lutris*) predation, where a rapid non-linear decrease in urchin size with otter occupation time is observed (Stevenson et al., 2016). Baja California's red sea urchin divers are as effective in removing specific size classes of urchins as sea otters. In the absence of otters, spiny lobsters (*Panulirus interruptus*), California sheephead (*Semicossyphus pulcher*), sun stars (*Pycnopodia helianthoides*) and rock crabs (*Cancer* spp.), became red sea urchins predators (Kato and Schroeter, 1985). The extent, to which this predation may contribute to specific size class decrease in densities and biomass in Baja California, is not determined yet.

Information on geo-referenced catch, biomass and densities will contribute to a better understanding of the dynamic of the red sea urchin population and fishery in Baja California. Data generated with GPS tracking technology may allow determining changes in time and space of fishing hotspots, important to design "tailored" management strategies. This kind of technology has been applied in monitoring the sea urchin fishery of Galicia (Fernández-Boán et al., 2013), where CPUE did not exhibit a significant trend throughout the fishing season, but catch per area and area covered per unit of diving time were inversely related, indicating that fishers stay longer in high density patches, that may result in localized depletion.

Furthermore, permit holders have started translocation programs (Sepesca Bc, 2016) to "enhance recruitment" and roe yeild. Yet, these programs are far from addressing key ecological and population issues that could arise from trans locating red sea urchins.

Kobe plots have traditionally being used in the tuna fisheries managed by the International Commission for the Conservation of Atlantic Tunas (ICCAT), where scientific advice within the Regional Fisheries Management Organizations (tRFMOs) is based on the Kobe advice framework, requiring a stock assessment, estimates of reference points and projections. Advice is based on low risk of fishing mortality exceeding F_{MSY} and biomass falling below B_{MSY} (Kell, 2011, 2016). For the red sea urchin fishery, these phase plots allowed to identify not only the sites that have been overfished for several years, but also to discern the years when the decline in biomass might be linked to causes different from fishing (years where $F/F_{MSY} < 1$ and $B/B_{MSY} < 1$). We propose that the decline in 2016–2018 is the aftermath of the 2012–2016 highly variable period, where temperature variations increased due to the anomalous oceanographic conditions. Kobe plots for the red sea urchin fishery should be adopted in order to evaluate the consequences of different harvest scenarios, management strategies, or separate changes in biomass due to fishing from natural variation.

RECOMMENDATIONS

Populations that have been overfished for several years need management actions oriented to promote its recover. ICS, ITS, ISM, and ISJ were declared Biosphere Reserve in 2016, so the red sea urchin population in its adjacent marine area needs to be managed appropriately to ensure its recovery.

Based on our findings, we propose that although there might be an apparent decline in the overall population (e.g., ICS, ST, ISM, where average site densities are below 1 urchin m^{-2}), juvenile red sea urchins (>42 <52 mm) contribute with recruits to the fishery as they grow and larvae (recruits) to the population every year. We found that the most important size classes in the red sea urchin population of Baja California were juveniles (≥ 42 ≤ 52 mm) and sub adults (≥ 57 ≤ 77 mm), which should not be harvested until they reach >80 mm. This becomes more important for those sites where most of the population consists of these size intervals and few adults of harvestable size (all sites except for ITS and PBan). In addition, based on the relation between adult density and recruitment (for spawning and sheltering recruits) we propose establishing a maximum size limit of 110 mm TL; this measure would ensure the protection of larger spawners which contribute a disproportionately high proportion of recruits.

Translocation of urchins has not been properly evaluated. While our results might suggest that such translocations may not have an impact on a population level, removing red sea urchins may clear areas for purple sea urchin to colonize, reducing suitable areas for red sea urchins to settle and leaving juveniles unprotected from predation. In addition, impacts of hyper aggregation of red sea urchins on kelp forests in Baja California are yet to be assessed.

Although biomass estimations are important for the red sea urchin fishery, based on its biological and ecological characteristics, we recommend tracking changes in adult density rather than biomass levels. Tracking changes in biomass for this species may result in biased decision making linked to high or even low values of biomass; it also sets the mind of fisher folk and managers in regards to the availability of red sea urchin linked to a specific area. Estimating red sea urchin density per site is a key indicator of population health and should be considered above any other assessment method. To this end, we propose that changes in fisheries data reporting and processing are necessary; current fisheries logbook for red sea urchin does not allow detailed analysis and may obscure key information. This new fishing log must include precise geographic locations, time spent by divers allocated independently for each dive in a day, and number of urchins harvested per dive (rather than kilograms). In addition, fisheries independent surveys must continue, as means to track changes in the population that might not be so apparent when observing only catch data. All the suggested information and analysis won't put extra cost on permit holders or fishermen; these changes are thought to improve management, maintain an important fishery worth average two million dollars and preserve a key species of the kelp forest ecosystem.

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/**Supplementary Material**.

AUTHOR CONTRIBUTIONS

AM-O: project lead, data analysis, data base procurement and curation, result processing, and manuscript elaboration. GM-M: Ph.D. lead adviser, manuscript revisions, and data analysis. CA-F: Ph.D. adviser, fisheries model fine tuning, and manuscript revisions. ES-D-A: Ph.D. adviser, sea surface temperature data processing, and manuscript revisions.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2020.00167/full#supplementary-material

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Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Capitulo III. Descifrando la variabilidad ambiental y del ecosistema en las pesquerías asociadas a los bosques de sargazo en México. Understanding the impact of ecosystem and environmental variability on the sea urchin population inhabiting kelp forests in México.

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Resumen

La pesquería de erizo rojo es una de las más importantes en Baja California y la única pesquería de erizo en México. El presente estudio se enfoca en comprender como la variabilidad ambiental local, regional y oceánica puede afectar a las poblaciones de erizo rojo. Se desarrollaron modelos generales lineales (GLM) para diferentes condiciones termales observadas: 1) Condiciones normales, 2) onda cálida y 3) post onda cálida; además se desarrollaron modelos para los sitios cuyas características termales fueron superiores, inferiores o promedio. Las variables utilizadas para estimar la densidad de erizo rojo fueron: capturas de erizo rojo, pez vieja, cabrilla sargacera y langosta roja, biomasa de sargazo gigante, temperatura superficial del mar, potencia del oleaje, índice de surgencias, índice multivariado de “El Niño” y el índice de la oscilación del giro del Pacifico Norte. Resultados previos mostraron que los sitios con mayor densidad de erizo rojo fueron Isla Todos Santos, Punta Banda y Punta Baja; las capturas de pez vieja fueron mayores en Punta Banda y Punta Baja, mientras que las capturas de cabrilla sargacera fueron mayores en Canoas. La captura de langosta fue mayor en la región de Pinta Baja – Isla San Jeronimo – Arrecife Sacramento. Observamos que la variabilidad ambiental local, así como la disponibilidad de alimento y abundancia de depredadores fue de mayor importancia para los cambios en la densidad de erizo rojo, comparado con las variables de escala regional (índice de surgencias) o de escala oceánica (MEI y NPGO). También observamos que la importancia de estas variables cambia dependiendo de la escala espacio – temporal considerada, lo que significa que las condiciones “normales o promedio” están reguladas por variables distintas comparadas con condiciones ambientales extremas como El Niño o La Mancha.

Abstract

The red sea urchin fishery is one of the most important fisheries in Baja California and the only urchin fishery in México. This study focuses on understanding how local, regional and oceanic environmental variability may affect red sea urchin populations. General Linear Models were developed for different periods: Pre – heatwave, heat wave, and post – heatwave, as well as for sites above, below and average SST. Variables included in the models were kelp biomass, red sea urchin, sheep head, kelp seabass and spiny lobster catches, SST, wave power, upwelling index, MEI and NPGO. Locations with the highest values for biological variables were Isla Todos Santos, Punta Banda, Santo Tomas, Punta Baja, Isla San Jeronimo and Arrecife Sacramento. We observed that local environmental variability, food availability, and predator abundance are highly important factors on determining red sea urchin population changes, compared to regional (upwelling index) and oceanic scale variables (MEI and NPGO). We also found that the relative importance of these variables change depending on the spatial and temporal scale being considered, meaning that “normal or average” conditions are regulated by different variables compared to extreme environmental conditions such as El Niño or the Blob events. The intricate relation between urchin divers and urchin population density needs to be viewed differently; in light of the absence of sea otters in the region, and low levels of abundance of other important predators such as sheep head and spiny lobster, red sea urchin divers have become the most important red sea urchin population control. Improving management of the red sea urchin fishery might yield benefits for the entire kelp forest ecosystem, fishermen and red sea urchin populations in Mexico.

Keywords: red sea urchin, fisheries, environmental variability, kelp forests, marine heatwaves

Introduction

Knowledge of the variability in time and space of a population is vital to understand the underlying mechanisms responsible for maintaining or changing a natural community structure. Physical factors such as upwelling, water temperature, sedimentation, wave action or fishing can be determinant in the abundance of marine species (Hereu et al., 2012). Sustainable harvest of commercial marine species requires precise estimations of population trajectories as well as a deep understanding of the causes of variability (O’Malley, 2009).

Changes in the environment and overfishing impose great risk to fisheries stocks around the world. In fisheries management, strategies are focused on reducing the risks of harvested populations; when data are scarce, the statistical methods for stock assessment cannot provide accurate and unbiased estimates about the dynamics of the fishery, increasing the uncertainty in the results. The challenge becomes greater when the economic resources to acquire such information are also scarce (Pulkkinen, 2015). Despite these challenges, there is a need for accurate estimations of fished stocks towards sustainable fishing (Hilborn and Walters, 1992).

Kelp forests are highly dynamic and productive ecosystems, showing considerable temporal variation and resilient to small-scale disturbances that results in considerable stability on a time scale of years (Tegner and Dayton, 2000). Events such as El Niño (ENSO), storms, predation (grazing), and recently marine heatwaves, are among the most important sources of disturbances for kelp forests (Dayton, 1985a; Arafeh – Dalmau et al., 2019; Cavanaugh et al., 2019). El Niño has been documented to have adverse effects on red sea urchin populations in its northern distribution. Several works have been able to track the changes on kelp beds, urchin recruitment rates and density due to El Niño events in California. It has been shown that kelp beds are more likely to be affected by the conditions of oligotrophic warm water, halt on coastal upwelling, and intense winter storms brought by the event, causing patchy deforestation (Grove et al., 2002; Tegner and Dayton, 1987; Tegner et al., 1997; Cavanaugh et al., 2011); however, the effects on red sea urchin population density and recruitment dynamics goes from unchanged (Tegner and Dayton, 1991) to highly variable, being driven by daily temporal variability in upwelling winds and coastal topography (Botsford, 2001). Moreover, it has been documented that El Niño has affected the red sea urchin fishery, by decreasing roe yield as a result of low food availability and lower feeding activity (Kato and Schroeter, 1985; Tegner and Dayton, 1991). In Baja California, it has been suggested that El Niño events have detrimental effects on red sea urchin populations and fishery by increasing natural mortality (Jiménez – Quiroz et al., 2013).

The North Pacific Gyre Oscillation (NPGO) is another relevant source of environmental variability, being a strong indicator of fluctuations in mechanisms driving planktonic ecosystem dynamics and its role in forcing global scale decadal changes in marine ecosystems (Di Lorenzo et al., 2008) because it directly influences the California Current upwelling system (Chenillat et al., 2012). Specifically, kelp biomass has been correlated with NPGO in long periodic cycles (over ten years), and it has been shown that NPGO affects cycles of kelp recruitment and mortality of entire plants with a lag of 3 years; observing a large amount of heterogeneity in the response of kelp biomass to different environmental factors (Cavanaugh, et al., 2011, Bell et al., 2018). More recently, the phenomenon of Marine Heat Waves (MHW) has had detrimental effects on the community structure of kelp forests (Arafeh – Dalmau et al., 2019), which have geographical variation in response to these events, with canopy forming kelps being more sensitive to warming throughout their range (Beas – Luna et al., 2020).

Predator abundance also influences red sea urchin density, and it has been documented from manipulative experiments in California that sheep head (*Semicossyphus pulcher*) predate on urchins during the day and spiny lobster (*Panulirus interruptus*) during the night (Nichols et al. 2015). Kelp seabass (*Paralabrax clathratus*) is a generalist diurnally active carnivore with high site fidelity (Hobson et al., 1981; Lowe et al., 2003), a possible predator of both spiny lobster (Young, 1963) and the red sea urchin, and space competitor for sheep head (Claisse et al., 2012).

Kelp forests have significant effects on seawater chemistry, productivity, microbial assemblages (Pfister et al., 2019), and harbor a highly diverse community of micro and macro animals such as mollusks, crustaceans, echinoderms, fish and marine mammals (Steneck et al., 2002; Wilmers et al., 2012; Perez – Matus et al., 2012; Castorani et al., 2018). Some of these species are commercially harvested (red sea urchin, California sheep head, kelp seabass, abalone, spiny lobster). Efforts to understand the effects of environmental variability, extreme events and fishing on these ecosystems have been focused mainly on kelp (Tegner and Dayton, 2000; Perez – Matus et al., 2012; Castorani et al., 2018; Arafeh – Dalmau et al., 2019;

Cavanaugh et al., 2019), yet very few have focused on the effects of variability, extreme events and ecological interactions in fished species (e.g. Tegner and Dayton, 1991; Pfister and Bradbury, 1996; Free et al., 2019), such as the red sea urchin.

As one of the essential fisheries in Baja California, this study focuses on understanding how local, regional and oceanic scale variability may affect red sea urchin populations. We propose that local environmental variability (food availability, predator abundance, SST, and wave energy), are more critical in determining red sea urchin population changes, compared to regional (upwelling index) and oceanic scale variables such as the Multivariate El Niño Index (MEI) and the NPGO. We also propose that these local variables change in importance under different large-scale conditions, meaning that different variables may regulate "normal" conditions and extreme environmental conditions such as El Niño or the Marine Heatwaves.

Materials and Methods

Biological variables

Red sea urchin density (*Mesocentrotus franciscanus*, *rsu*) was estimated using a length based virtual population analysis (Jones, 1984 with modifications and data from Medellin – Ortiz et al., 2020), with delimited depth and available substrate in all fishing areas. Monthly number of urchins by length-class was calculated using the formula:

$$N_{L1} = N_{L2}\left(\frac{L_\infty - L_1}{L_\infty - L_2}\right)^{M/K} + C_{1-2}\left(\frac{L_\infty - L_1}{L_\infty - L_2}\right)^{M/K}$$

where N_{Li} is the number of urchins at length L_i, C_{1-2} is catch in numbers between lengths L_1 and L_2 , L_∞ and k are growth parameters for individuals in the population and M is the rate of natural mortality. Natural mortality was estimated for each site using Pauly's (1980) equation:

$$log(M) \ = 0.0066 - 0.279(\log(L_\infty)) + 0.6543(\log(K)) + 0.4634(\log(T))$$

where L_∞ and K are growth parameters for individuals in the population and T is temperature (in °C). We used site-specific sea surface temperature data ranging from 9.6 to 29.9° C (average 18.10° C); L_∞ and k growth parameters used in both estimations where based on Rogers – Bennett et al (2003), and were varied depending on urchin size ($134.56 \leq L_\infty \leq 139.90$ mm and $0.033 \leq k \leq 04.38$). Fisheries independent information on size structure for 9 sites for 2003, 2005 and 2006 produced by Palleiro – Nayar et al. (2012), was used as basis to pool VPA size class data into four test length (TL) size classes: Recruits ($7 \leq TL \leq 37$ mm), Juvenile ($42 \leq TL \leq 52$ mm), sub adult ($57 \leq TL \leq 77$ mm), and adult ($TL \geq 80$ mm).

Each site depth delimited fishing area was measured, red sea urchin preferred substrate area dFa_{site} was calculated based on Palleiro – Nayar et al. (2011), and used to estimate sea urchin density:

$$Density = \frac{N_{site,size}}{dFa_{site}}$$

where $N_{site,size}$ is the number of urchins estimated for each size.

All information was grouped based on landing site (14 sites total) distributed along Baja California's Pacific coast (Fig.1). Fisheries landings were geo-referenced using a randomizing function for coordinates within each fishing area; randomization of latitude was limited by northern and southern fishing area coordinates, while longitude randomization was limited by the 5 and 30 m isobaths. It is important to notice, the specific point locations generated do not account for actual fishing sites or actual hotspots within each area.

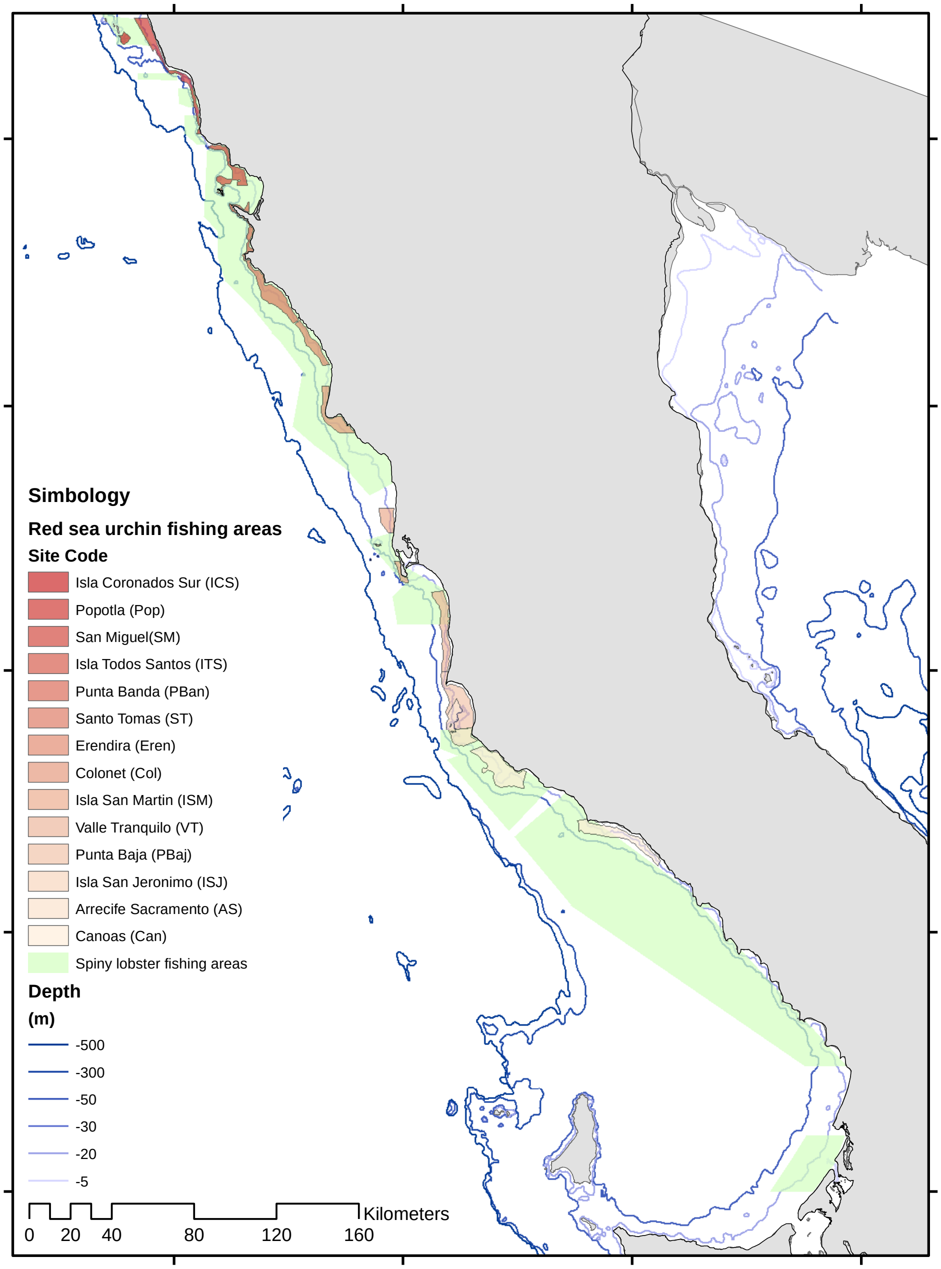

Figure 1. Red sea urchin and spiny lobster authorized fishing areas.

Kelp biomass (*Macrocystis pyrifera*) data was estimated monthly using Landsat satellite imagery to estimate canopy biomass across the study area at 30 m resolution, from 2000 to 2018 following methods described by Cavanaugh et al. (2011) and Bell et al. (2018). Kelp forest were delimited in two zones depending on biomass levels: Outer kelp forest, and Core kelp forest.

Daily catch records for **California sheep head** (*Semicossyphus pulcher*), **kelp seabass** (*Paralabrax clathratus*) and **spiny lobster** (*Panulirus interruptus*) were acquired through Mexico's National Transparency Platform[1]. Database includes: date, catch in kg, landing site and number of vessels used for fishing; California sheep head and kelp seabass are mainly caught by hook and line, while spiny lobster are caught with baited traps. California sheep head and kelp seabass are fished under "finfish" commercial permits, mainly with hook and line onboard artisanal vessels, with a small finfish mid-range fleet of 25 boats. Spiny lobster fishing can be done under commercial fishing permits or concessions only with authorized traps within the authorized fishing area. The three species are caught in all Baja California, including Guadalupe and Cedros Islands; but these two sites were excluded from the analysis since there is no red urchin catch reported for those sites.

Sheep head and kelp seabass catch data were geo located based assuming a higher abundance of sheep head in the outer zones of the kelp forests, while kelp seabass inhabits middle and outer zones of the forest (Claisse et al., 2012). Based on catch records landing sites and species, catch was assigned a random location either outer or in the core zones of the kelp forests surrounding landing sites. Spiny

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[1] www.infomex.org.mx

lobster catches were geo located based on each permit holder authorized fishing area; given that traps are not placed inside kelp forests nor dropped too deep, kelp covered areas and depths greater than 300 m were not included. Due to authorized landing site or fishing areas, it was necessary to group sheep head and spiny lobster data sites in the following groups: 1) Isla Coronados Sur and Popotla (ICS-Pop), 2) San Miguel - Isla Todos Santos – Punta Banda – Santo Tomas –Erendira - Colonet (SM-Col), 3) Punta Baja-Isla San Jeronimo –Arrecife Sacramento (PBaj-As).

Environmental variables

Local scale variables

Site-specific **sea surface temperature** (SST) data was gathered monthly from multisensory SST time series for 2000 to 2018 (4 km pixel, daylight at 11μ; from sensors AVHRR, MODIS Terra, MODIS Aqua and VIIRS Suomi-NPP). Images were processed in SEDAS (7) following Karu et al. (2015) and Karu et al. (2012) criteria.

Wave power (CgE) was obtained from the IOWAGA hindcast (Rascle & Ardhuin, 2013), produced by the French Research Institute for the Exploitation of the Sea, Ifremer. The hindcast is based on a global multigrid implantation of the Wave Watch 3 model, with ~18 km spatial resolution in the study area. Wave power is computed every 3hrs from wave directional spectra S(f,θ):

$$CgE = \rho g \iint Cg(f) S(f,\theta)\ df d\theta$$

where ρ is water density (≈1024), g is gravity acceleration (≈9.8 m s-1), and Cg is group velocity. For each site the closest model node was chosen to represent wave power at that site. All sites were closer than 8 km from the chosen node.

Regional scale variables

Upwelling index (upw) data was gathered from NOAA's Environmental Research Division (https://oceanview.pfeg.noaa.gov/products/upwelling/intro, "Accessed 20 Jan 2019"). We used the traditional Bakun (1973, 1975) upwelling index time series for two nodes: 33 and 30º N, as they were the closest stations to Baja California's kelp forest locations. This index is calculated with Eakman's theory of mass transport due to wind stress; positive values can be considered as the amount of water being upwelled, while negative values imply downwelling.

Oceanic scale variables

Multivariate El Niño Index (MEI) data was gathered from NOAA's Earth System Research Laboratory (http://www.esrl.noaa.gov/psd/enso/mei/, "Accessed 20 Jan 2019"). MEI was selected due to the importance of El Niño/Southern Oscillation (ENSO) as a phenomenon causing global climate variability on inter-annual timescales (Wolter and Timlin, 1993, 1998). MEI is based on the six main observed variables over the tropical Pacific: sea-level pressure, zonal and meridional components of the surface wind, sea surface temperature, surface air temperature, and total cloudiness fraction of the sky (Wolter and Timlin, 1993, 1998).

North Pacific Gyre Oscillation (NPGO) data were obtained from KNMI Climate Explorer[2] for the same time span as the other variables.

All data for each site were summarized monthly and yearly, and we calculated standardized anomalies for biological, local and regional variables:

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[2] http://climexp.knmi.nl/getindices.cgi?WMO=O3Ddata/npgo&STATION=NPGO&TYPE=i&id=someone@somewhere; "Accessed 20 Jan 2019"

$$Anomaly = \frac{value_{x_i} - \mu}{\sigma}$$

where the total time series average μ is subtracted from each value x_i and divided by the time series standard deviation. Indexes such as MEI and NPGO are already presented as standardized anomalies. All further analyzes were conducted using the standardized anomalies.

We built correlation matrices and correlation network plots between all variables using the packages Corrr (Kuhn et al., 2020) and Ggraph (Lin Pedersen, 2020) in R (Ver. 4, 2020).

We implemented general linear models (GLM), with all variables full time series of all variables, as:

$$\begin{aligned} rsu_{ijk} = {} & \beta_0 + \beta_1 kelp.biomass_{ijk} + \beta_2 Urchin.ct_{ijk} + \beta_3 Calsh.ct_{ijk} + \beta_4 Kelpsbs.ct_{ijk} \\ & + \beta_5 Lobst.ct_{ijk} + \beta_6 SST_{ijk} + \beta_7 CgE_{ijk} + \beta_8 Upw_{ijk} + \beta_9 MEI_{ijk} \\ & + \beta_{10} NPGO_{ijk} + \epsilon_{ijk} \end{aligned}$$

where *rsu* is red sea urchin population density for month *i*, year *j*, site *k*; ε_i is the random error under the effect of *Urchin.ct*, *Calsh.ct*, *Kelpsbs.ct*, *Lobst.ct*, *SST*, *CgE*, *Upw*, *MEI*, and *NPGO* for month i, year j, site k, respectively; $\beta_{0...10}$ are parameters to be estimated. All variables were entered as standardized anomalies.

We developed specific models based on SST variability that depict different climatological conditions: 1) Pre – heat wave (normal conditions), 2) Heat wave and, 3) Post – heat wave. During each condition, we identified sites that presented average SST, as well as sites that were above (+1 SD) and below average SST (-1

SD), during the different thermal events (Fig. 2). We implemented a GLM with the same structure used for the full time series.

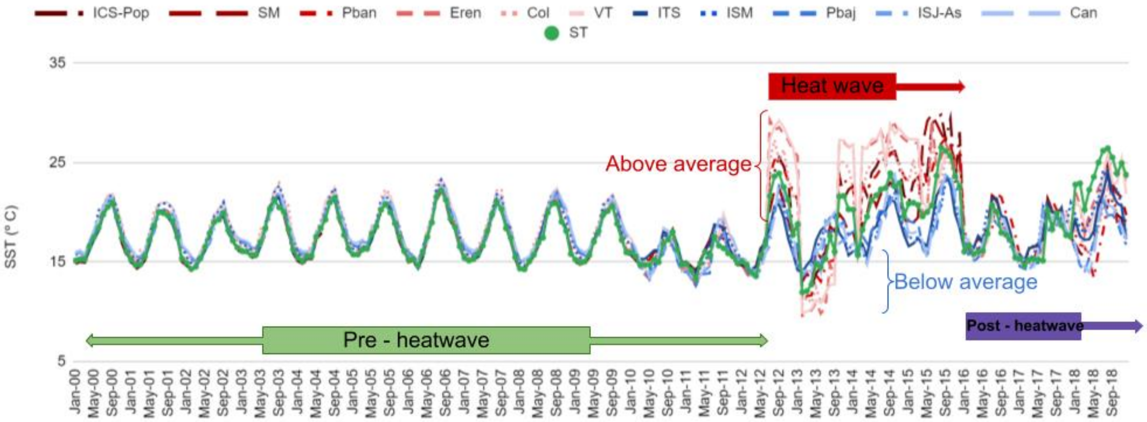

Figure 2. Sea surface temperature time series for each site. Thermal periods and conditions analyzed are indicated. Red tones represent above average SST sites; blue tones represent below average SST sites. Average SST not shown as values for site ST correspond to average SST values.

An Analysis of Residual Sum of Squares (ARSS) was employed in order to compare among models. This analysis tests whether two or more curves are statistically different (Zar, 1984).

Time lags between possible effects of the selected variables on *rsu* was explored using the cross-correlation functions in R (2020).

We examined causality of the variables which were determined to have an effect on *rsu* through the GLMs, using CausalImpact package (Brodersen, et al., 2015; Brodersen and Hauser, 2020) in R. This approach allows to infer the causal impact an “intervention” has exerted on an outcome metric over time, on the basis of a diffusion-regression state-space model that predicts the counter-factual response in a synthetic control that would have occurred without an intervention (Brodersen et al., 2015); it models the behavior the response variable (*rsu*) might have in the absence of the possible effects produced by the predictor variables. We considered the significant variables from GLMs to be the “intervention” to assess their impact on *rsu* as outcome metric over the full time series, as well as the pre-heatwave, heatwave and post-heatwave periods.

Results

Biological variables

Monthly average kelp biomass for the 2000 – 2018 period was 391 ton; yearly kelp biomass averaged 4,695 ton for the entire period. We observed higher kelp biomass values (> 7,500 ton) at ISM and PBan; intermediate values (3,000-5,000 tons) were common at ICS, Pop, VT, PBaj-ISJ, AS, and Can); the lowest biomass was observed at Col. Kelp forest delimitation was based on biomass levels for 2000-2018; areas with average kelp biomass from 270 to 680 ton/Ha were common on outer kelp forest, and ranged from 800 to 1,300 ton/Ha at the core. Red sea urchin population density was very high at PBan and ITS (averaging +4 urchins m^{-2}), compared to the rest of the sites that showed densities below two urchins m^{-2}. Sites like ISJ and Can showed very low densities (<1 urchin m^{-2}). Red urchin catches were high (above 200 ton) at ST, PBaj and AS, and low (below 20 tons) at ICS and ISJ. Sheep head catches were high (above 50 ton) at PBaj-AS and SM-COL, and low (below 5 ton) at ICS-Pop. Seabass catches were high (60 ton) in Can, about 10 ton in Col and Pbaj, and below 5 ton at the rest of the sites. Spiny lobster catches were higher (75 ton) in PBaj – ISJ – AS, about 50 ton at SM-COL and Can, and below 25 ton at ISM; the rest of the sites showed very low catches (Fig. 3).

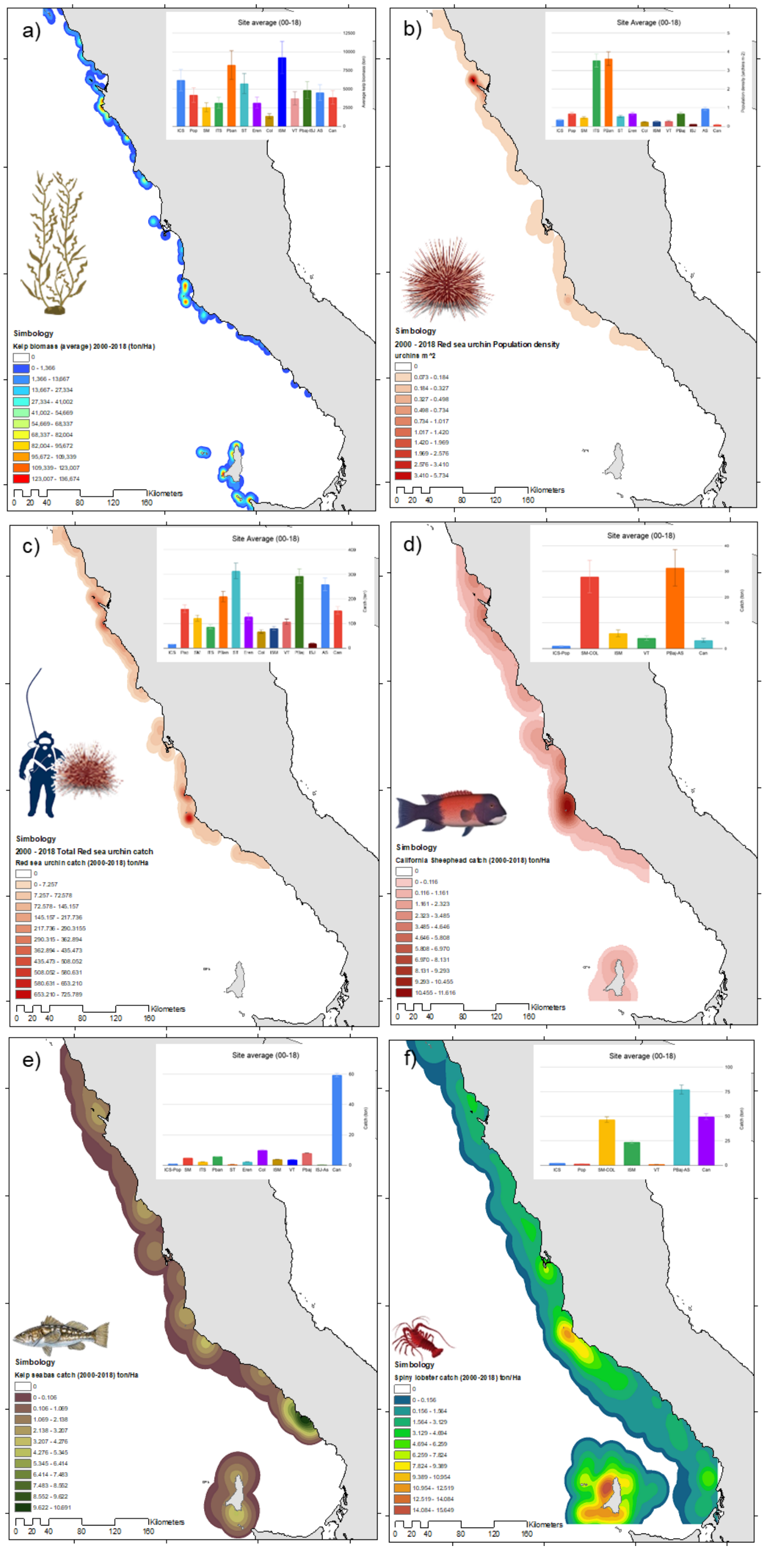

Figure 3. Distribution and density of: a) Kelp biomass, b) red sea urchin population density, and c) catch, d) California sheep head, e) kelp seabass and f) spiny lobster catch, along the Pacific coast of Baja California.
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Kelp biomass standardized anomalies were positive for 8 years, with no identifiable pattern of occurrence; positive anomalies were higher in 2005 and 2009, and high negative anomalies occurred in 2016. Sites with positive kelp biomass anomalies were ICS, PBan, ST and, ISM. Red sea urchin population density anomaly was negative from 2000 to 2008, and shifted to positive values from 2009 to 2015, after this year, anomalies returned to negative values. ITS, PBan and AS displayed positive anomalies for the entire period. No identifiable pattern was seen with sheep head and kelp seabass catch anomalies, both showed positive anomalies for 7 years, however, anomalies occurred in different years, further, anomalies from 2016 to 2018 were inverse for both species. SM-Col and PBaj-AS displayed positive sheep head catch anomalies, while mainly Can had positive kelp seabass catch anomalies during analyzed period. We observed negative anomalies in spiny lobster from 2000 to 2010, the trend shifted to positive catch anomalies from 2011 to 2018. SM-Col, PBaj-As and Can were the only sites with positive catch anomalies for the period (Fig. 4).

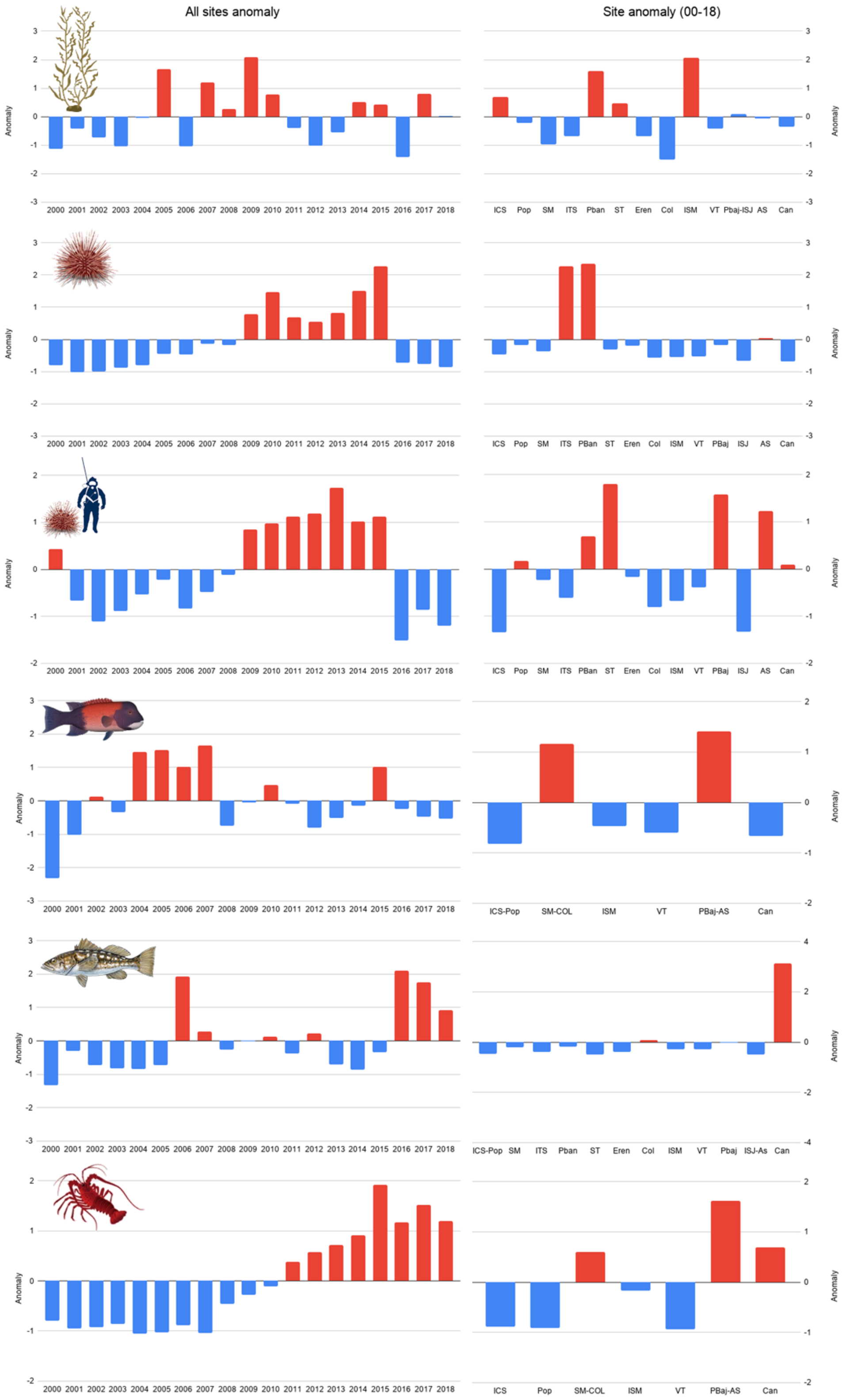


Figure 4. Kelp biomass, red sea urchin population density, California sheep head, kelp seabass and spiny lobster catch anomalies from 2000 to 2018 and 19 year site anomaly.

Environmental variables

We observed high variability in the environmental variables during the study period. Yearly average SST ranged from 16.7 to 22.4 ºC, with higher values during 2014, 2015 and 2018 (21.4, 22.4 and 19.8º C, respectively), years with anomalies higher than two. Sites with positive SST anomalies for the whole period were ICS-Pop, SM, PBan, Eren, Col and VT, being the last three sites the ones with higher anomalies. High negative anomalies were observed at ITS, PBaj and ISJ-AS. Average CgE for the whole period ranged from 9.4 to 14.3 kw m^{-1}, with positive anomalies from 2000 to 2006, except 2004 with a slightly negative anomaly. From 2007 – 2018, the anomalies were negative; however, 2010 and 2016 showed the greatest positive anomalies of the period, above 2 units. Positive anomalies were observed in 4 sites: PBan, ISM, PBaj and ISJ-AS, where the anomaly was almost 2 units. Average upwelling index ranged from 105.5 to 143.6 m^{-3} per 100 m coastline; positive anomalies were higher in 2007 and 2012, and high negative values were observed in 2004, 2014 and 2015. A regional pattern was evident, with northern sites being more exposed to positive upwelling index anomalies, and southern sites with negative values. Average yearly MEI values showed two periods of positive anomalies in 2002 – 2004 and 2015 – 2016; while NPGO positive anomalies were more common from 2000 – 2004 and 2007 – 2013; during this period (2007-2013) the upwelling index and NPGO showed inverse anomaly patterns (Fig. 5).

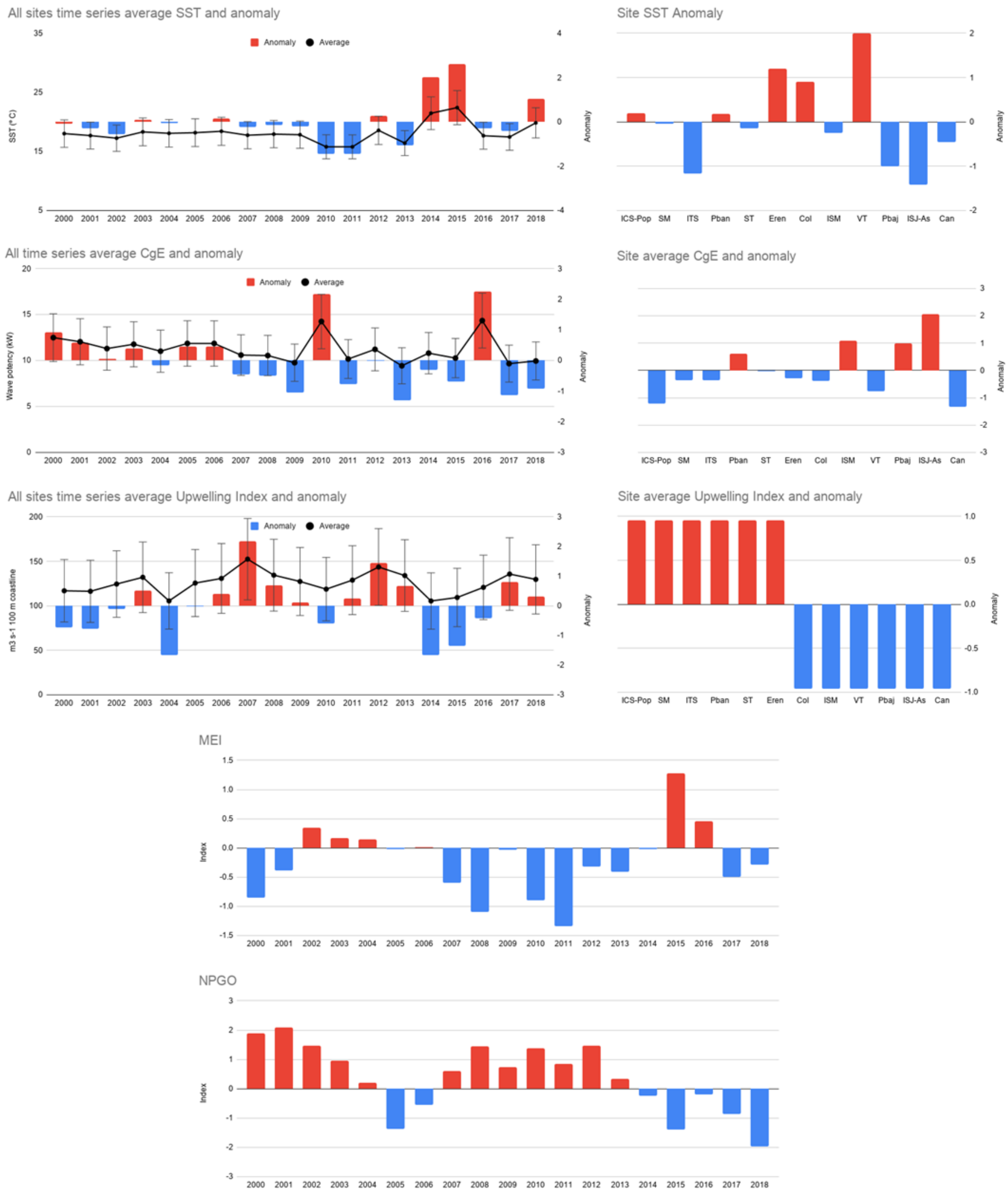
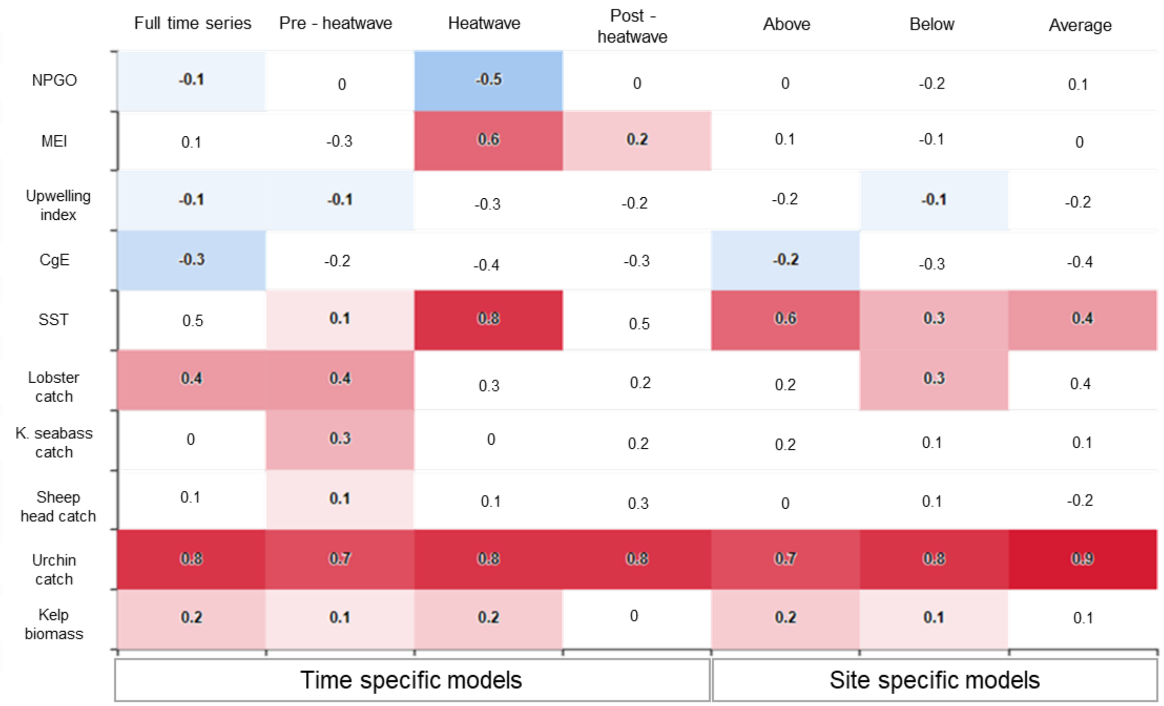


Figure 5. Local (Average SST, CgE), regional (Upwelling index), and oceanic (MEI, NPGO) variables values and anomalies from 2000 to 2018 and 19 year site anomaly.

Correlations

Patterns were evident when comparing correlations from all analyzed scenarios (Table I). Urchin catch was the variable with higher correlation (0.7-0.9) with red sea urchin density in all scenarios. Although the strength of the correlation with SST was variable, it was positive for all scenarios, and it was high (≥ 0.5) for the heatwave, above average, full time series, and post-heat wave scenarios. Kelp biomass correlation with urchin density was always positive, although low (0-0.2). Lobster catch correlation was always positive and ranged from 0.2 to 0.4. Seabass catch was always positively correlated, although low (0.1-0.3), and no correlation was observed for the full time series and heatwave scenarios. Sheep head catch was positive and low (0.1-0.3) for most scenarios, except under the average conditions when it was negative (-0.2). The correlation with NPGO was negative under the full time series (-0.1), heatwave (-0.5) and below average scenarios (-0.2), and positive under average conditions (0.1); no correlation was found for the rest of the scenarios. The correlation with MEI was positive and low (0.1-0.3) in most of the scenarios, and high (0.6) under heatwave conditions. The correlation with upwelling index and CgE were negative and low (0.1-0.4) for all scenarios (Table I). Other variables that where positively and negatively correlated were sheep head – kelp seabass (0.3), sheep head – spiny lobster catch (-0.4), and kelp seabass – spiny lobster (-0.3), SST and spiny lobster (0.4), kelp biomass and the upwelling index (0.4), urchin catch and upwelling index (-0.4), SST and MEI (0.4); MEI and NPGO were negatively correlated (-0.3). These correlations changed depending on the time frame or location for each model (Fig. 6).

Table I. Summary of significant variables (colored) and correlations (numbers) for each model under time specific and location dependent thermal conditions. Intense color indicates variables that were more significant (red, positive and blue, negative) and pale colors indicate less significant variables. Variables with no color were not significant.



	Full time series	Pre - heatwave	Heatwave	Post - heatwave	Above	Below	Average
NPGO	**-0.1**	0	**-0.5**	0	0	-0.2	0.1
MEI	0.1	-0.3	**0.6**	**0.2**	0.1	-0.1	0
Upwelling index	**-0.1**	**-0.1**	-0.3	-0.2	-0.2	**-0.1**	-0.2
CgE	**-0.3**	-0.2	-0.4	-0.3	**-0.2**	-0.3	-0.4
SST	0.5	**0.1**	**0.8**	0.5	**0.6**	**0.3**	**0.4**
Lobster catch	**0.4**	**0.4**	0.3	0.2	0.2	**0.3**	0.4
K. seabass catch	0	**0.3**	0	0.2	0.2	0.1	0.1
Sheep head catch	0.1	**0.1**	0.1	0.3	0	0.1	-0.2
Urchin catch	**0.8**	**0.7**	**0.8**	**0.8**	**0.7**	**0.8**	**0.9**
Kelp biomass	**0.2**	**0.1**	**0.2**	0	**0.2**	**0.1**	0.1
	Time specific models				Site specific models		

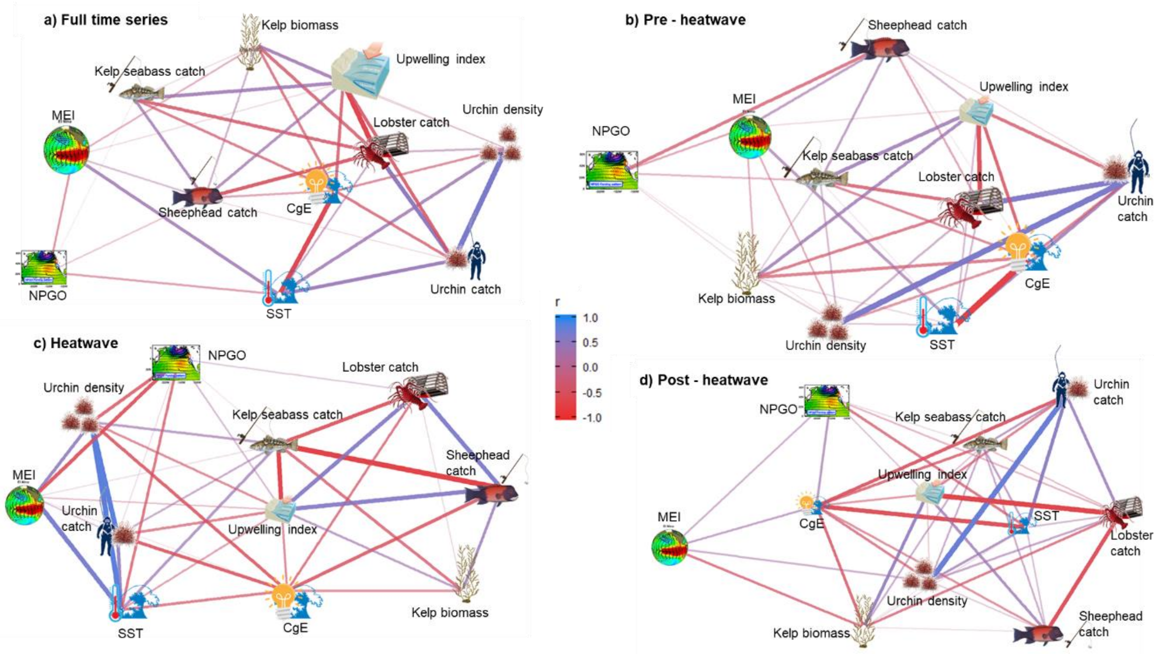


Figure 6. Correlogram of variables potentially influencing red sea urchin abundance along the Pacific coast of Baja California. Correlations were analyzed under different

conditions: a) Full time series (2000 – 2018); b) Pre – heatwave (2000 – 2011); c) Heatwave (2012 – 2015); d) Post – heatwave (2016 – 2018).

Time Specific Analysis

a) General linear models (GLM's) and relative importance

Full time series model (GLM$_{00\text{-}18}$) significant variables were kelp biomass, urchin catch, lobster catch, CgE, upwelling index and NPGO:

$$rsu_{ijk} = 0.149_{\pm 0.045} kelp.biomass_{ijk} + 0.848_{\pm 0.055} Urchin.ct_{ijk} + 0.134_{\pm 0.056} Lobst.ct_{ijk} + 0.181_{\pm 0.057} CgE_{ijk} + 0.258_{\pm 0.056} Upw_{ijk} - 0.087_{\pm 0.034} NPGO_{ijk}$$

Pre heatwave model (GLM$_{00\text{-}11}$) significant variables were kelp biomass, urchin catch, sheep head, kelp seabass and lobster catch, SST and upwelling index:

$$rsu_{ijk} = -0.204_{\pm 0.048} + 0.132_{\pm 0.401} kelp.biomass_{ijk} + 0.770_{\pm 0.064} Urchin.ct_{ijk} + 0.091_{\pm 0.039} Calsh.ct_{ijk} + 0.186_{\pm 0.045} Kelpsbs.ct_{ijk} + 0.222_{\pm 0.073} Lobst.ct_{ijk} - 0.277_{\pm 0.067} SST_{ijk} + 0.192_{\pm 0.053} Upw_{ijk}$$

Heatwave model (GLM$_{12\text{-}15}$) significant variables were kelp biomass, urchin catch, SST, MEI and NPGO:

$$rsu_{ijk} = 0.634_{\pm 0.087} + 0.132_{\pm 0.061} Kelp.biomass.a_{ijk} + 0.583_{\pm 0.067} Urchin.ct_{ijk} + 0.209_{\pm 0.07} SST_{ijk} + 0.247_{\pm 0.106} MEI_{ijk} - 0.302_{\pm 0.078} NPGO_{ijk}$$

Post heatwave model (GLM$_{12\text{-}15}$) significant variables were only urchin catch and MEI:

$$rsu_{ijk} = -0.413_{\pm 0.050} + 0.474_{\pm 0.074} Urchin.ct_{ijk} + 0.184_{\pm 0.036} MEI_{ijk}$$

The variables that were significant for the full time series account for 65.75% of the observed variability. Red sea urchin catch accounted for most of the variability, with spiny lobster, kelp biomass, and upwelling index accounting for less than 20% of model's R^2. Significant variables in Pre – heatwave model account for 73.25% of the observed variability. Red sea urchin catch continued to be one of the most important variables in the model, followed by lobster catch, SST, kelp seabass, and sheep head. During the heatwave, significant variables accounted for 90.03 % of the observed variability; red urchin catch continued to be the most important variable, followed by SST, NPGO and MEI. Significant variables during the post – heatwave period account for 82.17% of the observed variability, with red sea urchin catch being the most important variable, followed by MEI (Fig. 7).

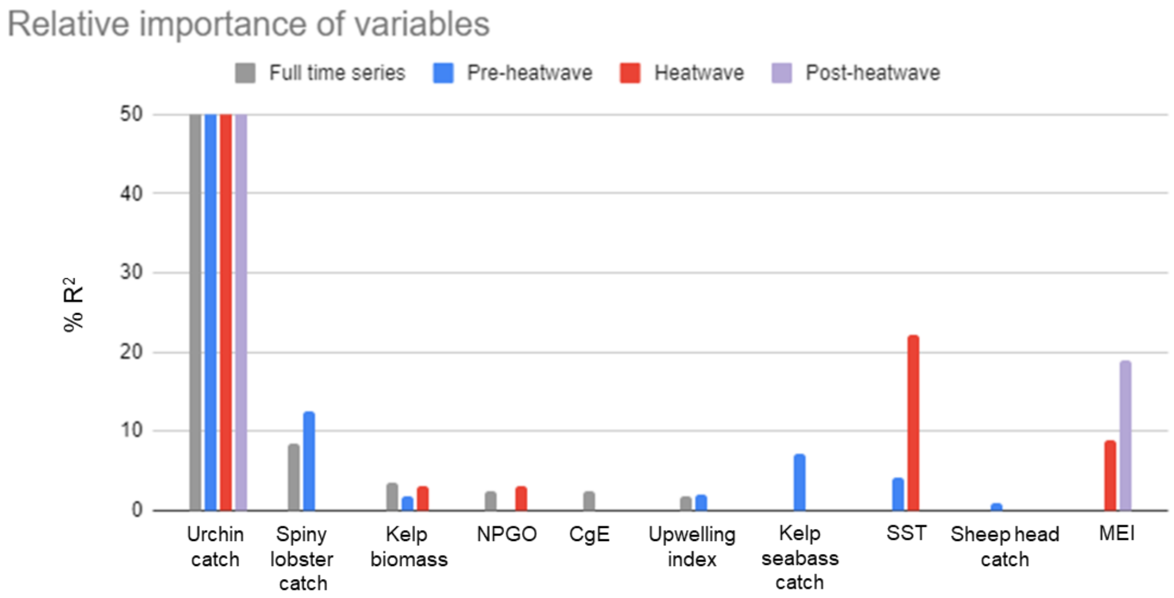


Figure 7. Share of R^2 (as percentage) of significant variables for all models. Metrics are normalized to sum 100% for each scenario. Urchin catch share of R^2 was >50% in all scenarios.

Time lag of significant variables

Significant variables of the full time series model showed different time lags in their effects on *rsu*, kelp biomass anomalies have a positive effect on red urchin with a time lag of 0-2 months to 1.2 years. Positive and negative urchin and lobster catch anomalies showed a lag from 0 to 1 year, and positive upwelling index anomalies showed positive effects on *rsu* after 1.4 years. Negative wave power anomalies had positive effects on *rsu* after 5 months, while positive wave power anomalies had negative effects on *rsu* between 1.5 -1.7 years after the anomaly occurred. Despite being a statistically significant variable for the full time series model, NPGO did not display any time lag in its effect on *rsu*. During the pre-heatwave period, positive and negative values of significant variables displayed different time lags in their effects over *rsu*. Kelp biomass anomaly had positive effects on *rsu* that could be seen from 2 months to over 1.5 years after the event, while positive California sheep head catch anomalies yielded positive effects on *rsu* after 3 to 4 months. Urchin, kelp seabass and lobster catch anomalies displayed the same pattern whether negative, neutral or positive values had positive effects on *rsu*, with lags ranging from 0 – 8 months to 1 year. Sea surface temperature anomaly had negative effects on *rsu*, with time lags from 5 months to 1.5 years. Upwelling index anomaly did not display any significant time lag on its effect on *rsu*. During the heat wave, significant variables had lagged effects on *rsu* that ranged from 0 months to 1 year. Kelp biomass anomaly had positive effects on *rsu* with a lag of 0 – 3 months to 1.1 years, while urchin catch anomaly had a time lag of 3 – 4 months on its effect on *rsu*. SST anomaly, MEI and NPGO had similar time lags from 0 to 4 months to have positive (SST and MEI) or negative (NPGO) effects on *rsu*. Post heat wave significant variables showed time lags shorter than 6 months on their effect on *rsu*. Urchin catch anomaly had significant negative effects on red sea urchin density can be observed around 4 months after a positive catch anomaly. Same pattern was observed with MEI (S1).

b) *Causal Impact*

Causal impact analysis on full time series GLM_{00-18} revealed that significant variables as a whole had a positive impact on red sea urchin density anomaly (*rsu*), with an average value of 0.011; in the absence of the effects of these variables, the expected value was -0.64. The average causal effect of these variables over *rsu* was 0.65 ($-0.24 \leq \mu \leq 1.47$). Analysis on Pre – heatwave conditions GLM_{00-11} revealed that significant variables as a whole had a positive impact on *rsu*, with an average value of -0.16; in the absence of the effects of these variables, the expected value was -0.84. The average causal effect of these variables over *rsu* was 0.68 ($-0.54 \leq \mu \leq 1.36$). During the heatwave GLM_{12-15} the analysis revealed that significant variables as a whole had a positive impact on *rsu*, with an average value of 1.13; in the absence of this effect, *rsu* average *rsu* expected value was 0.60. Average causal effect of significant variables on *rsu* was 0.53 ($0.25 \leq \mu \leq 0.83$). The analysis on Post – heatwave conditions GLM_{16-18} revealed that as a whole, significant variables had negative impact on *rsu*, with an average value of -0.62; in the absence of this effect, the average expected value of *rsu* was 1.04. Average causal effect of these variables on *rsu* was -1.66 ($-2.20 \leq \mu \leq -1.19$). The probability of obtaining these effects by chance is very small, which suggests the effects can be considered significant (Fig. 8).

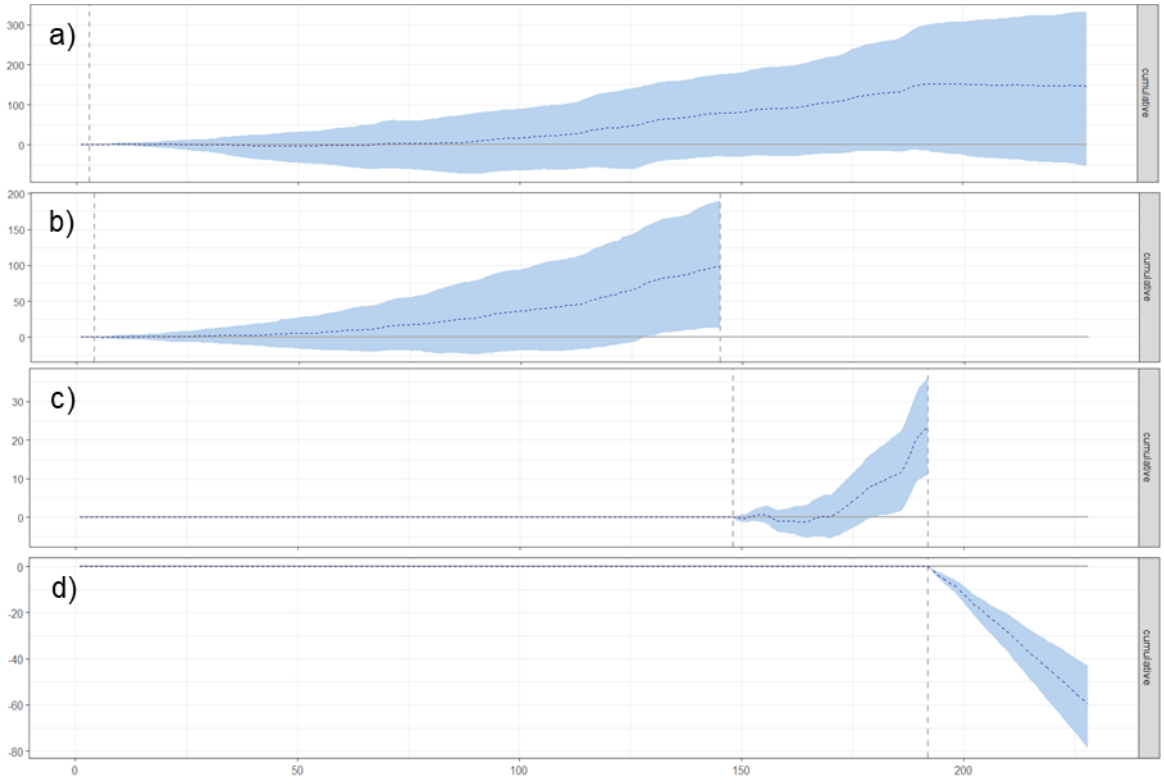


Figure 8. Cumulative results of causal impact on red sea urchin population density anomaly (dotted) and confidence intervals at 95% (shaded) for a) full time series, b) Pre – heatwave, c) Heatwave, and d) Post – heatwave models and significant variables; horizontal axis are number of months from Jan-2000 (0) to December-2018 (229). Dotted vertical line indicates where the causal effect was tested.

c) Comparison between models

Having determined the relative importance of the variables on each time frame, we modeled *rsu* using specific important variables for the full time series and for heatwave conditions. Comparison between predicted values from full (GLM_{00-18}) and condition specific ($GLM_{00-11}|GLM_{12-15}|GLM_{16-18}$) models suggested that these approaches were significantly different (ARSS, $F_{2,0.05} = 34.04$, $p<0.001$), with the condition-specific model showing better fit ($R^2 = 0.78$; Fig. 9).

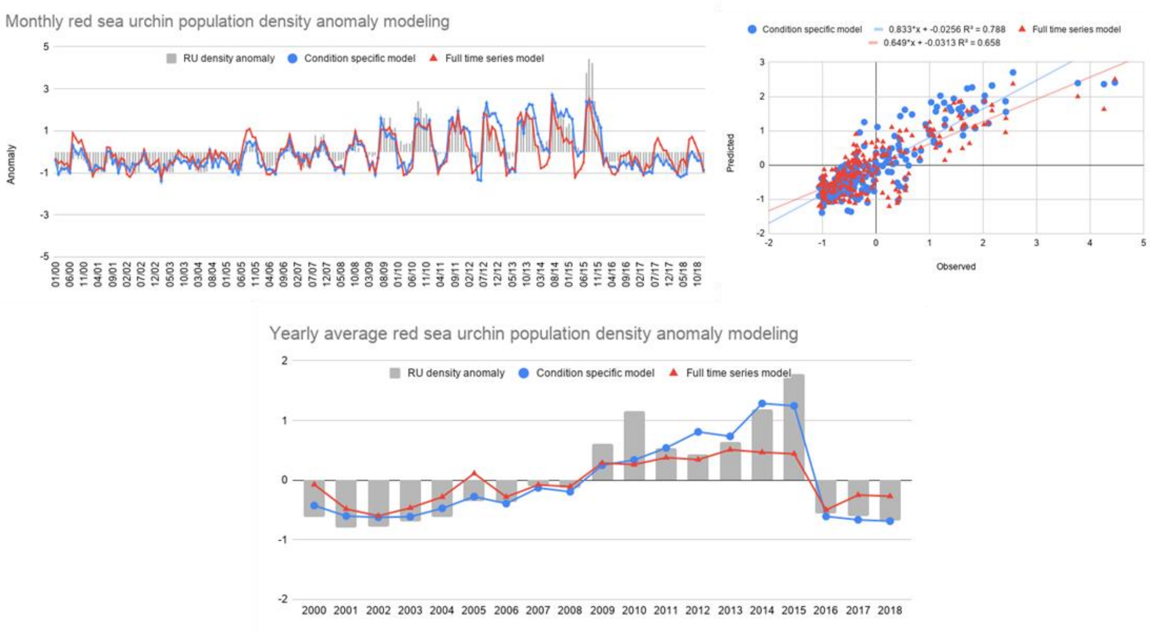


Figure 9. Comparison between observed *rsu* values (bars), and predicted values from full time series (▲) and Condition-specific (●) models.

Site Specific Scenarios

Sites identified as above average SST were: ICS, Pop, SM, PBan, Eren, Col and VT.

General linear models (GLM's) and relative importance

Above average SST sites model (GLM$_{ab}$) significant variables were kelp biomass, urchin catch, SST and CgE:

$$rsu_{ijk} = 0.386_{\pm 0.082} kelp.biomass_{ijk} + 0.721_{\pm 0.051} Urchin.ct_{ijk} + 0.241_{\pm 0.058} SST_{ijk} + 0.172_{\pm 0.060} CgE_{ijk}$$

Significant variables for below average SST sites (GLM$_{be}$) were kelp biomass, red sea urchin and lobster catch, SST, and upwelling index:

$$rsu_{ijk} = 0.126_{\pm 0.056} Kelp.biomass_{ijk} + 0.874_{\pm 0.045} Urchin.ct_{ijk} + 0.228_{\pm 0.046} Lobst.ct_{ijk} - 0.117_{\pm 0.054} SST_{ijk} + 0.276_{\pm 0.053} Upw_{ijk}$$

Average SST site was a coastal location within one of the biggest kelp forests in the region; significant variables for the model (GLM_{μ}) were red urchin catch and SST anomalies:

$$rsu_{ijk} = 0.213_{\pm 0.044} + 0.844_{\pm 0.040} Urchin.ct_{ijk} + 0.135_{\pm 0.046} SST_{ijk}$$

Significant variables in above average sites explain 65.42% of the observed variability. Red sea urchin catch was the most important variable, followed by SST, kelp biomass and CgE. Red sea urchin catch anomaly was the most important variable in below average sites, followed by lobster catch anomaly and upwelling index. Significant variables explain 71.21% of the observed variability. Red sea urchin catch anomaly was the most important variable in the average site, followed by SST; these variables explain 78.68% of the observed variability (Fig. 10).

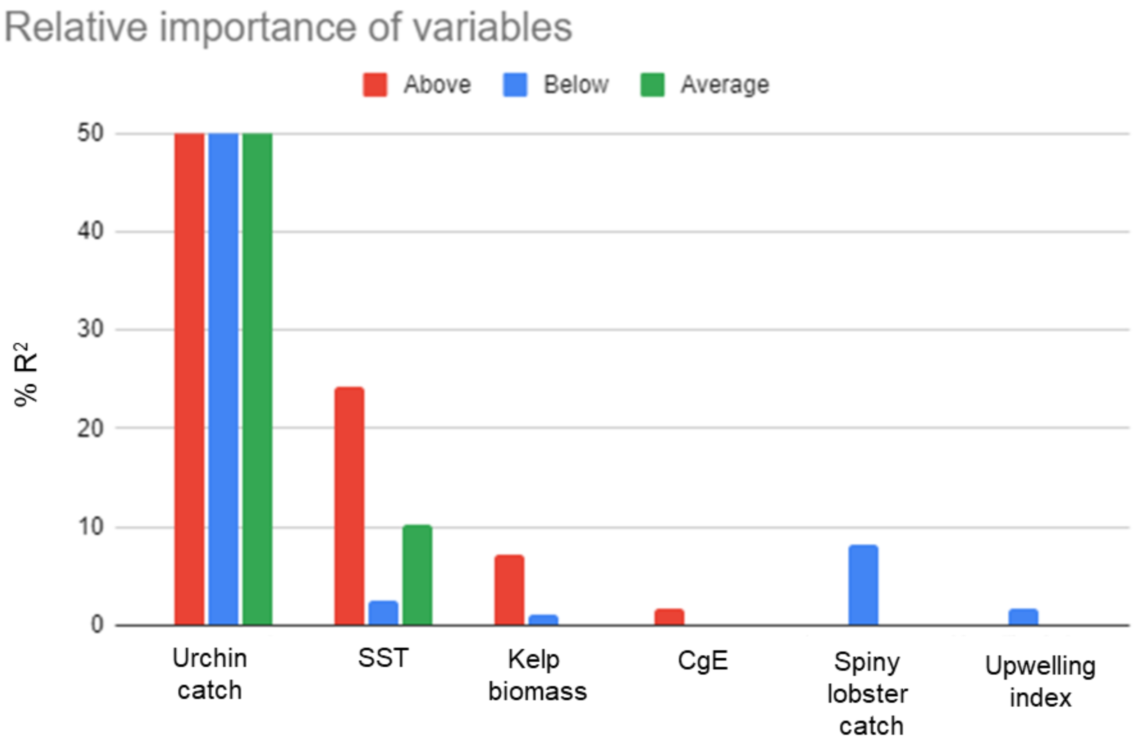


Figure 10. Share of R^2 (as percentage) of significant variables for above GLM_{ab}, below GLM_{be}, and average GLM_{μ} models. Metrics are normalized to sum 100%. Urchin catch % of R^2 is >50% in all scenarios.

With these results, we modeled *rsu* using above, below and average site important variables. All three models displayed good fits, with model GLM_{μ} performance being higher (R^2 =0.78; Fig. 11).

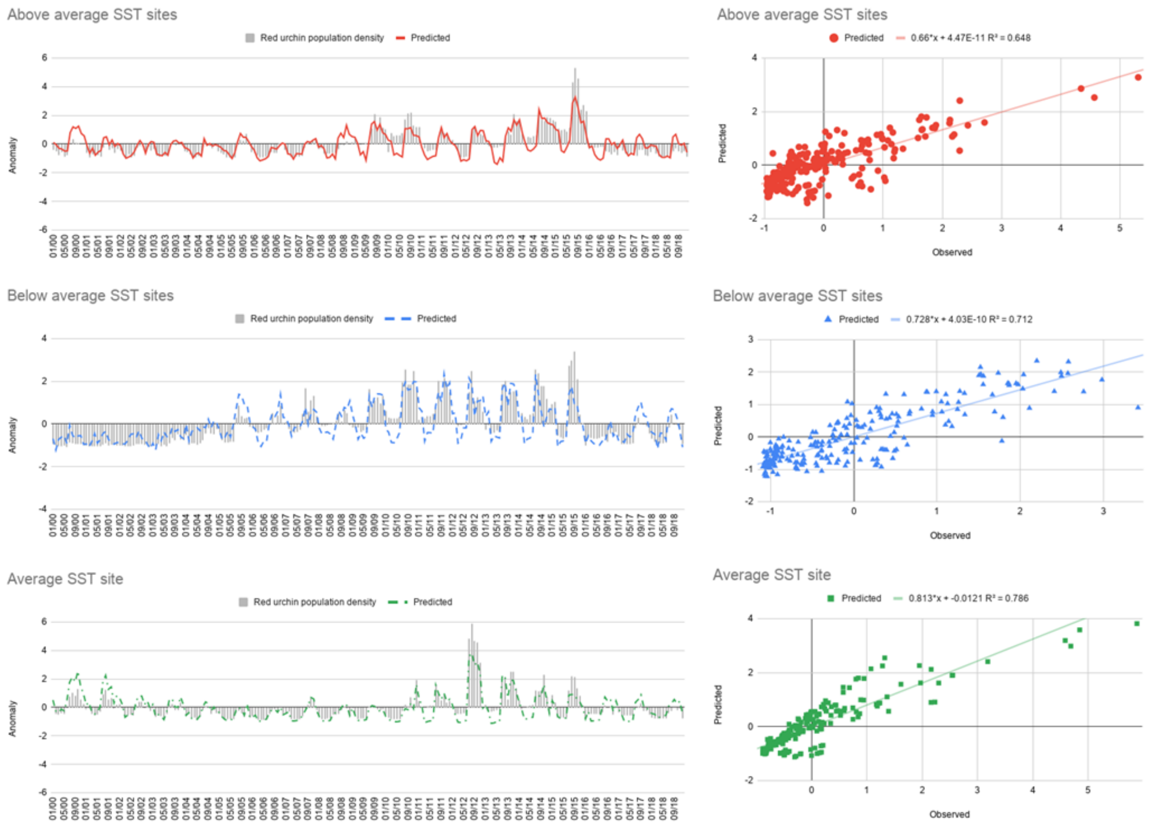


Figure 11. Comparison between observed *rsu* values (bars), and predicted values from full time series for above (solid line), below (long dash) and average SST conditions (dash-dot) models.

Discussion and conclusions

Our results show that red sea urchin population density in Baja California is subject to different environmental, biological, and anthropogenic sources variability, in the form of changing thermal conditions, wave energy exposure, upwelling, variable food availability (kelp), different levels of predator abundance and fishing pressure. Results suggest that under all scenarios analyzed, ***fishing is the most crucial factor affecting red sea urchins***. Harvest has been observed to reduce sea urchin densities, where sites under experimental harvest showed a reduction between 93.7 and 96.3% red sea urchin densities, and affecting size distribution significantly, with a reduction in the density of legal-size urchins in the population in northern Washington (Carter and VanBlaricom, 2002). Effects of harvest on red sea urchins have also been reported for northern and southern California, with legal size urchins decreasing between 15 and 73% (Tegner and Dayton, 1981; Pfiester and Bradbury, 1996). There have also been comparisons in red sea urchin density between protected (MPA) and unprotected sites, observing significant differences in terms of gonadosomatic index (GSI) and adult size of urchins inside MPAs (Teck et al., 2017). Evidence suggests that fishing pressure is a key driver in red sea urchin densities across the geographic range it's fished.

The fishery

Harvest depends not only on urchin density but roe yield and quality, fishable days due to weather, and market conditions (Kato, 1972; Kato and Schroeter, 1985; Kalvass and Hendrix, 1997; Kalvass and Rogers-Bennett, 2001; Andrew et al., 2002; Schroeter et al., 2009; Medellín – Ortiz et al., 2020). Roe yield depends entirely on food availability (kelp) and is determined by the diver in a pre – harvest sampling at the beginning of the dive: if the 3 to 5 randomly selected urchins of the patch have orange-yellow gonads that fill over 80% of the urchin, then the patch is harvested; otherwise, the patch is left as it is and the diver moves to another area. This behavior might result in low catches and high “skinny” urchin density areas. Inter-annual differences in urchin density have been reported in different areas in the region for 2003, 2005, 2006 and 2008, relating such differences to fishing intensity and recruitment rates in each site (Palleiro – Nayar et al., 2012). However, there is no assessment of recruitment rates, frequency or patterns for red sea urchin in Baja California. It has been proposed that the observed catch trends for the period being analyzed here might be related to consistent settlement and recruitment (Medellín – Ortiz et al., 2020); consistent with Southern California’s recruitment patterns observed by Ebert et al., 2004), with over harvesting depleting temporarily legal size urchins without impairing population recruitment (Medellín – Ortiz et al., 2020). This effect has also been described for red sea urchins in Southern California (DeWees, 2004).

Kelp

Kelp biomass was highly variable among sites and through time. It has been documented that kelp biomass in the region suffered negative impacts from heatwaves; however, its resilience was spatially variable and not significantly related to SST metrics indicating that local-scale environmental and biotic processes might play a larger role in kelp recovery (Cavanaugh et al., 2019). Our results show that kelp biomass variability was not highly correlated to red sea urchin density, and it was observed as a significant variable under the heatwave period and in the above average temperature site group only. This might be related to red sea urchins' ability to feed as an active grazer as well as to consume hidden drifting algae, so it is possible that satellite-based kelp biomass estimation and variability do not reflect changes in food availability from drift kelp fronds that might be a significant food source, and an indicator of the level of nourishment of the urchins (Harrold and Reed, 1985). Harrold and Reed (1985) describe that sea urchins in kelp dominated areas are well nourished, move little, occupy cracks and crevices and probably feed on drift algae, compared to those urchins found in barrens, where drift algae are sparse, and poorly nourished sea urchins actively graze the exposed substratum (Watanabe and Harrold, 1991), and subsisting on coralline algae in the absence of kelp (Harrold and Reed, 1985). Moreover, high density urchin aggregations have been shown to be a prerequisite to destructive grazing by red sea urchins (Dean et al., 1984). High densities of red sea urchins have been documented to cause cavitation damage on the holdfast of kelp, increasing kelp mortality due to storms or even intense wave action, causing kelp to dislodge and produce urchin “barrens” (Tegner et al., 1995). However, grazing activity of red sea urchins in Baja California has not been assessed, and since the reproductive potential of individual sea urchins is closely tied to food availability (Rogers-Bennett et al., 1995), it becomes necessary to assess grazing behavior by red sea urchins as well as the amount, seasonality and importance of drift kelp in the diet of red sea urchins in Baja California.

Predators

California sheep head and kelp seabass catch information were curated in order to remove misplaced dates stored as catch or as the number of vessels, or highly unlikely catch values (e.g., one vessel with average max. capacity of 1.2 ton reporting 5 ton in one single day trip); this was done by cross-referencing with permit holder data bases[3] and catch records. It is important to note that due to how landings are reported, it is very difficult to discern whether it is one artisanal vessel landing many times or many artisanal vessels landing one time each. Reports on coastal fishing of Baja California are non-existent, and very few reports on artisanal fishing have been conducted, focused only on schooling fish or expensive invertebrates (Rosales – Casián and Gonzalez – Camacho, 2003). According to Rosales – Casián and Gonzalez – Camacho (2003), California sheep head and kelp seabass accounted for 25.9 and 10.6% of the finfish caught in Baja California, respectively, and ranked 3.5 based on their Index of Community Importance (ICI) values. They also describe San Quintin – San Martin Island and Punta Baja as important finfish fishing sites. Our results show that Punta Baja (PBaj) and Canoas (Can) areas are important fishing sites for California sheep head and kelp seabass, respectively. However, it has been suggested that low catch levels in other sites could be attributable to high levels of fishing (Alonzo et al., 2004) and the loss of kelp forests due to climatic events such as El Niño and the Blob, resulting in the loss of cold-water species (Arafeh – Dalmau et al., 2019). It has been observed that fishing of

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[3] Publicly available at https://datos.gob.mx/busca/dataset/permisos-y-concesiones-de-pesca-comercial-para-embarcaciones-mayores-y-menores, and requested through Mexico's National Transparency Institute: https://www.infomex.org.mx/gobiernofederal/home.action#

sheep head has affected size-structured interactions with sea urchin prey in Southern California, reducing sizes and altering life histories of sheep head that lead to increased urchin densities (Hamilton and Caselle, 2015). High red sea urchin densities in sites like ITS, PBan and PBaj could be related to the high levels of sheep head catch observed in those sites. The case for spiny lobster fisheries data was no different; extensive cross-referencing and data curation was also needed. Spiny lobster catch data reflect higher values in the same locations red urchin catches are also high (namely PBaj, AS, and ISJ). It has been proposed that spiny lobster fishing triggers a trophic cascade leading to increased urchin densities and decreased macroalgal biomass (Guenther et al., 2012). This has been observed in Bahia Asuncion (Baja California Sur), where higher densities of spiny lobsters led to increased abundances of kelp (Jenkinson et al., 2020). This phenomenon could explain the observed correlation between spiny lobster catch and red urchin densities in our study, with sites like PBaj, AS and ISJ sustaining intense lobster catch that also promote high urchin densities and the corresponding high urchin catch and high kelp biomass.

Environmental factors

The relative importance and significance of variables were different depending on the time frame or site analyzed. These changes in significance and relative importance might be related to the variability in time and space of each variable. The range of variables that have been reported to affect kelp forest ecosystems include significant wave height (winter storms), warming events (El Niños and MHW), nutrient input by terrestrial runoff, upwelling, urchin grazing, lobster fishing (Botsford, 2001; Halpern et al., 2006; Guenther et al., 2012; Arafeh – Dalmau et al., 2019, Cavanaugh et al., 2019), all operating in varying spatial and temporal scales, with complex multifactorial interactions causing long term changes in these systems that are difficult to demonstrate and predict (Tegner et al., 1996a). However, studies have focused on kelp as a foundation species and main source of food for red sea urchin populations, and other sources of variability over red sea urchin have been limited to effects of El Niño, the abundance of predators, and fishing pressure (Tegner and Dayton, 1991; Estes and Duggins, 1995; Pfister and Bradbury, 1996; Morgan et al., 2000; Teck et al., 2017), analyzed independently from one another. Understanding how this complex ecosystem, its components and interactions respond to different conditions over time and space as a whole is key to the continued existence of the ecosystem and of those who depend on it.

Model choices

While aggregated models are widely applied to understand, predict, and manage populations and marine ecosystems, they fall short if the underlying system exhibits non-linear dynamics (Ye and Sugihara, 2016). Also, the model output may underestimate variability and overestimate stability (Griffith, 2020). Our condition-specific model was better at reproducing estimated red sea urchin density; this might be related to its ability to capture condition-specific variability (normal conditions vs. heatwave, post-heatwave conditions) by using only the variables that were significant and relatively important in such conditions, instead of averaging out variability and considering that the effects of all variables remain the same through time (assumption made in the full time series model). Despite the fact that some variables had low correlations, the effect exerted by the different combinations of variables on red sea urchin densities was considered statistically significant through the causal impact analysis. While this type of analysis has been mainly used in marketing, the characteristics of marketing data match those of biological survey data: often observational, rarely following the ideal of randomized design, subject to multiple seasonal variations, and are often affected by unobserved variables and their interactions. While correlations have been one of the most commonly used statistical techniques in biological studies (see Yadav, 2018), the similarity between biological and marketing data make the causal impact analysis suitable for biological data, and its use in our field should be further explored.

Temporal effects

We observed a lagged effect on red sea urchin population density from important variables in time and thermal condition-specific models; these lagged effects took from two months to two years depending on the scale of the variable, where local variables effects were noticeable after months (e.g., SST), while regional and oceanic variables took longer to produce an effect on red sea urchin densities (e.g., upwelling index, MEI). Many factors may intervene for El Niño to have an effect on our region, mainly depending on the geographic extent of the Kelvin wave that may or may not travel northward from the Equator (Wang, 2003). Our region has also experienced the recent warming event known as “the Blob” (Marine heatwave, MHW), which traveled from north to south along the California Current System (Bond et al., 2015; Oliver et al., 2018). These two events and the delay in which they arrive at our region might have differential effects on northern and southerly distributed urchins; the later might be affected first by the northward movement of Kelvin waves; on the contrary, with the southward movement of the MHW, northern communities are affected first by increases in SST. In the presence of events of such magnitude, our results suggest that local effects become more important than oceanic events, especially in locations where even during these extreme events, SST remained below-average values. SST showed to be important during extreme conditions (heatwave) and sites with above average SST. Temperature has been shown to have direct effects on red sea urchin metabolic rates, with increasing oxygen consumption as temperature increases, suggesting a temperature-dependent metabolism (Ulbricht and Pritchard, 1972). It has also been suggested that red sea urchins may display different temperature preferences between day and night, moving away from unfavorable physiological temperatures (Salas et al., 2012). These thermal effects have also been observed on red sea urchin embryos, where high temperatures increased the body size of prism stage embryos, suggesting that moderate warming may improve the growth and thermal tolerance of red sea urchins (Wong and Hofmann, 2020). Although it has been stated that increased temperatures resulting from El Niño or MHW have negative impacts on red sea urchins, our results suggest that red sea urchins might benefit from increased temperatures, with increased metabolic rates and increased size “heat adapted” larvae that would result in the increasing urchin densities observed in our study. However, temperatures above 28º C for a prolonged period of time might have adverse effects on red sea urchins, causing reduced movement from tube feet and increase movement through spines, to stop all movement with all spines and tube feet relaxed and death (Hernández et al., 2004).

Conclusion

Management and protection of natural ecosystems requires of deep understanding of ecological interactions and how they shape the composition and structure of communities in time and space. It becomes more important in the face of increasing threats from anthropogenic disturbances and socio – economic pressures (Guenther et al., 2012). Long-term studies have demonstrated that biotic components in a kelp forest ecosystem, are more impacted by the frequency of disturbances, such as marine heatwaves, compared to the severity of the disturbance. The effects of these events on kelp forest community structure increase or decrease, depending on their dependence on physical, trophic, and habitat resources mediated by kelp (Castorani et al., 2018). Trophic redundancy in California kelp forests was evident when the dominant urchin predator (sunflower sea star) was functionally extirpated by disease and the remaining predators like California sheep head or spiny lobster controlled urchin densities (Eisaguirre et al., 2020). Understanding how ecological and fisheries systems behave and interact becomes even more critical in the face of environmental changing conditions. Separating environmental effects from fishing on natural populations is key to increase the protection of natural ecosystems and enhancing fisheries management. Based on extensive work done in kelp forests in other regions (e.g., Cowen, 1983; Tegner and Levin, 1983; Steneck 1998; Beas – Luna et al., 2020), it has been proposed that predators are the most important control for sea urchin populations. Despite the overall urchin population density trend, site-specific biological and environmental information displayed high variability within and among sites. This has also been observed in other kelp ecosystems where environmental variation at fine (depth within sites), local (within locations), and regional scales (among locations) will influence where urchins can affect macroalgae assemblages, regardless of predator density and fishing intensity (Shears et al., 2008). According to Jenkinson et al. (2020), the community structure of rocky reefs in this region varies substantially among sites, being controlled in some places by predation; however, other factors that obscure the importance of trophic interactions. The intricate relation between urchin divers and urchin population density needs a different interpretation in light of the absence of sea otters in the region (Gallo – Reynoso, 1997), and low levels of abundance of other important predators such as California sheep head and spiny lobster (Alonzo et al., 2004; present study). In the presence of sea otters, it has been observed that sea urchin density and abundance decreases dramatically, as well as median size of sea urchins due to selective size predation by sea otters (Stevenson et al., 2016). Eliminating the functional role of sea otters as apex predators from kelp dominated rocky reef ecosystems has caused dramatic shifts from kelp forests to urchin barrens, devoid of large macroalgae (Estes and Palmisano, 1974).

This study suggests that in Baja California, red sea urchin harvest has become the most important red sea urchin population control; hence, improving red sea urchin fisheries management might yield benefits for the entire kelp forest ecosystem, fishers, and the red sea urchin population in Mexico.

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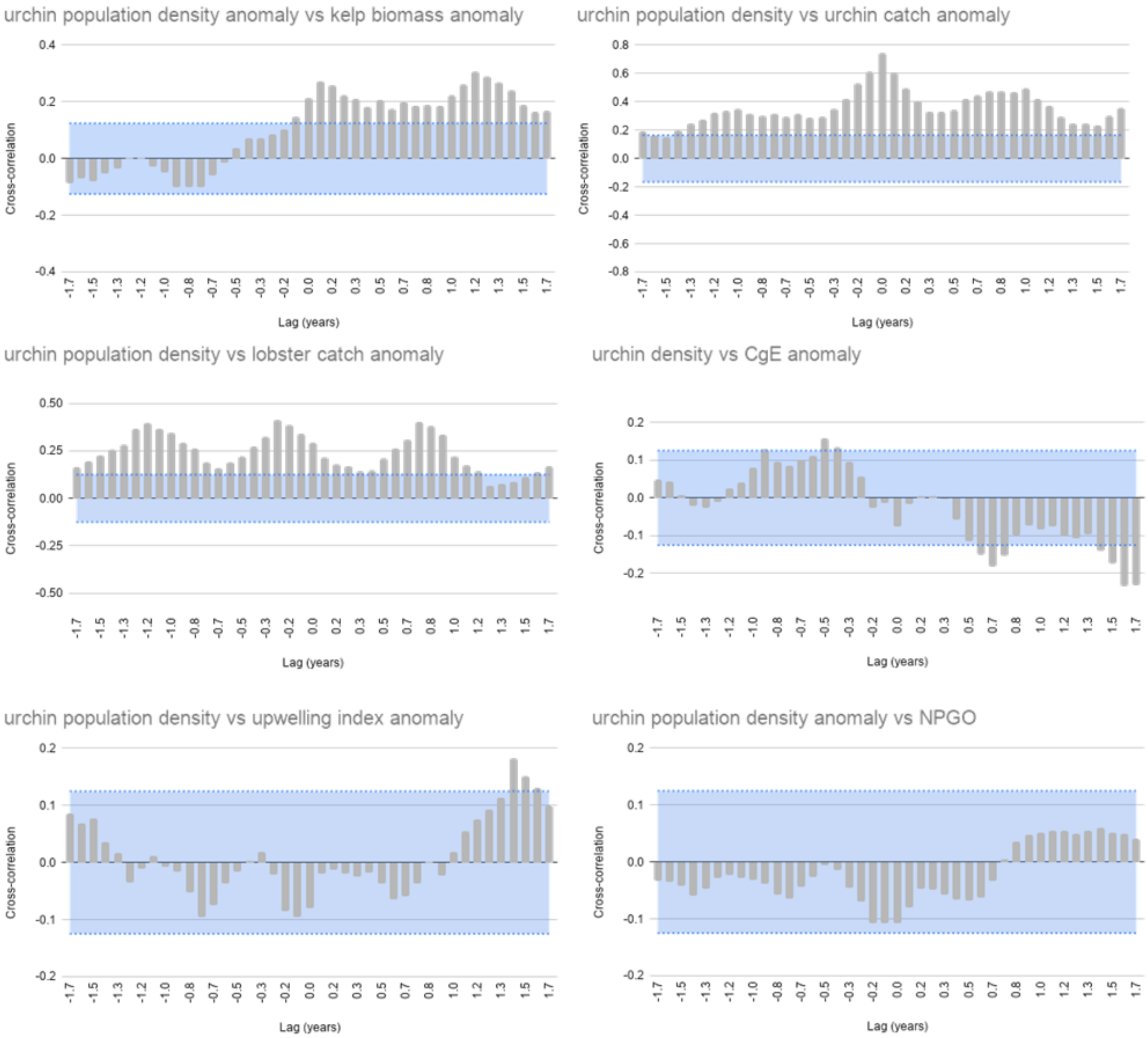
S1. Supplementary material 1. Time lag of significant variables.


urchin population density anomaly vs kelp biomass anomaly

urchin population density vs urchin catch anomaly

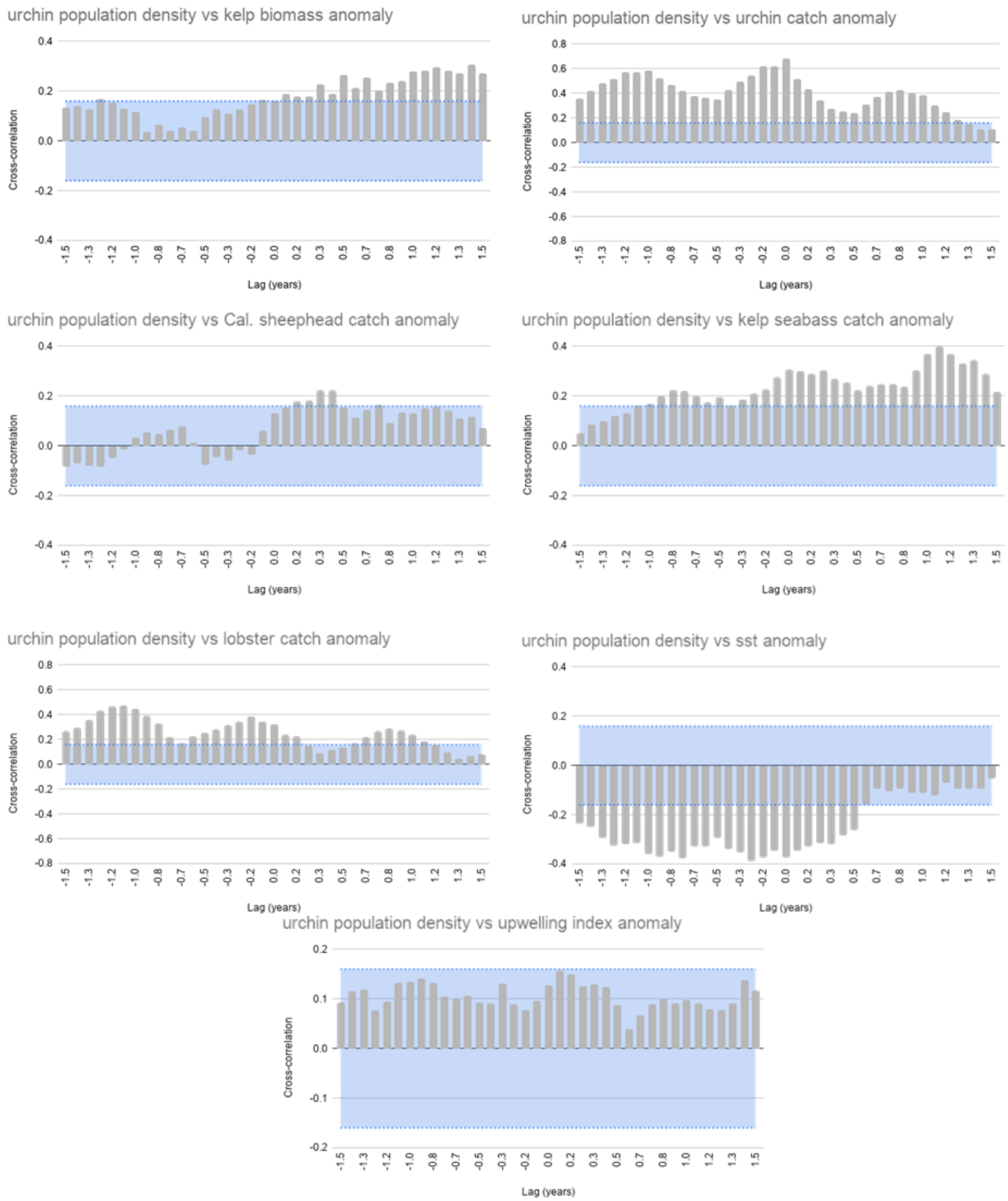
urchin population density vs lobster catch anomaly

urchin density vs CgE anomaly

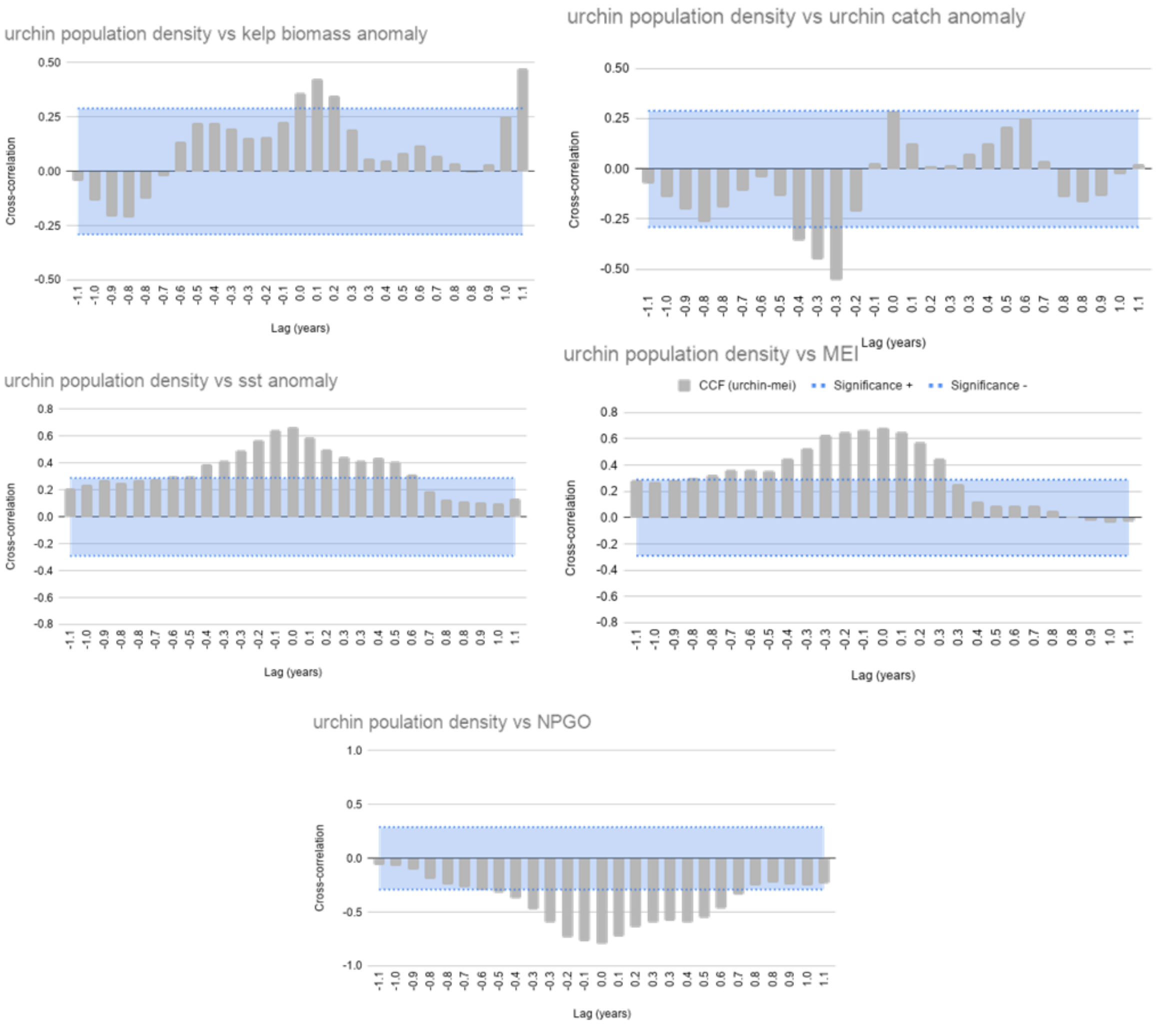
urchin population density vs upwelling index anomaly

urchin population density anomaly vs NPGO

S1. 1. Cross-correlation functions for significant variables for model $GLM_{00\text{-}18}$ (Full time series).



S1. 2. Cross-correlation functions for significant variables for model $GLM_{00\text{-}11}$ (Pre – heatwave).



urchin population density vs kelp biomass anomaly

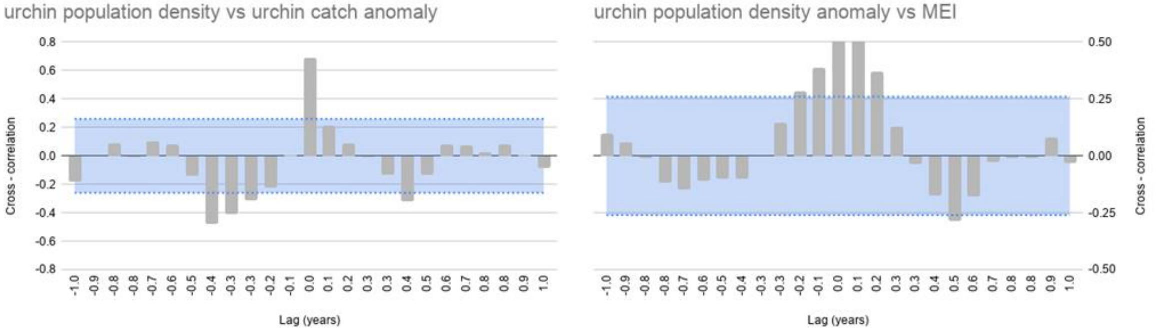
urchin population density vs urchin catch anomaly

urchin population density vs sst anomaly

urchin population density vs MEI

urchin poulation density vs NPGO

S1. 3. Cross-correlation functions for significant variables for model GLM12-15 (Heatwave).



urchin population density vs urchin catch anomaly

urchin population density anomaly vs MEI

S1. 4. Cross-correlation functions for significant variables for model GLM16-18 (Post – heatwave).

Capitulo IV. Conclusiones generales

La variabilidad observada en las capturas puede ser resultado de una combinación de la disponibilidad del recurso, periodos de fenómenos ambientales severos (El Niño, La Mancha) e inclusive condiciones difíciles de mercado.

El LVPA utilizado, que incluye la variabilidad biológica del erizo además de la variabilidad ambiental asociada a cada sitio de captura puede proporcionar mejor información sobre el impacto de la mortalidad por pesca en un sitio específico. Así mismo, las estimaciones de biomasa de erizo rojo con el modelo dinámico de Schaefer “forzan” el comportamiento biológico del modelo a ser un organismo de vida corta y reproducción temprana o un organismo muy longevo de reproducción tardía, por la relación entre los parámetros *K* (capacidad de carga) y *r* (tasa intrínseca de crecimiento). Esta relación entre los parámetros puede generar estimaciones de biomasa que no necesariamente corresponden al erizo rojo, cuya biología es de un organismo longevo, de reproducción temprana, sin decremento del potencial reproductivo con la edad; por esta razón, las estimaciones de biomasa con este modelo pudieran no estar de acuerdo con los niveles de captura.

Los diagramas de Kobe muestran que la trayectoria de las biomasas poblacional, reproductiva y aprovechable del erizo rojo se encuentran dentro de la zona segura y en recuperación, con la disminución de biomasa de 2015 a 2016 por razones diferentes a la mortalidad por pesca ($F/F_{MSY}<0.5$). Continuar con el uso de estos diagramas en la pesquería de erizo rojo puede permitir evaluar diferentes escenarios de aprovechamiento, estrategias de manejo o inclusive separar los casos donde la disminución de la biomasa este dada por factores diferentes a la mortalidad por pesca.

En general, la variabilidad ambiental local, así como la disponibilidad de alimento y abundancia de depredadores fue más importante, para los cambios en la densidad de erizo rojo, que las variables de escala regional (índice de surgencias) o de escala oceánica (MEI y NPGO). Sin embargo, la importancia de estas variables cambia dependiendo de la escala espacio–temporal, lo que significa que la densidad de erizo rojo está regulada por variables distintas cuando las condiciones ambientales son "normales o promedio", en contraste con las condiciones ambientales extremas como El Niño o La Mancha. Bajo condiciones "normales o promedio", la densidad del erizo rojo se regula principalmente por la captura de erizo rojo, y en menor grado por variables biológicas como la biomasa de sargazo gigante, la abundancia de depredadores (pez vieja, cabrilla y langosta), la temperatura superficial local y las surgencias. Bajo condiciones extremas, la densidad se regula también por la captura de erizo rojo, principalmente y por la biomasa de sargazo gigante, pero también por variables físicas locales (SST), así como variables oceanicas como el Niño y el NPGO.

Este trabajo sugiere que los análisis de impacto causal son una herramienta apropiada para datos biológicos y su utilización en nuestro campo debería ser explorada a profundidad.

Capítulo V. Recomendaciones

El modelo dinámico de biomasa utilizado históricamente puede no ser adecuado para las estimaciones cuando las especies son fecundas, longevas y de crecimiento rápido como el erizo rojo.

Es necesario dar seguimiento de la densidad poblacional por sitio de captura de manera independiente de la pesquería.

Debido a la relación observada entre densidad de adultos y reclutas, se propone la implementación de una Talla de escape para que no se capturen organismos mayores de 110 mm, lo que permitiría al menos mantener el número de reclutas por sitio.

Es necesario realizar cambios en el sistema de registro-reporte de las capturas, con la finalidad de contar con un estimador más fino de la abundancia relativa del recurso (número de erizos por hora de buceo). Este detalle de información permitirá evaluar cada sitio de captura de mejor manera.

Debido a la variabilidad natural asociada a cada sitio de captura, se propone la implementación de medidas de manejo diferenciado por zona permisionada.

La reubicación de erizos por cualquier motivo no ha sido evaluada apropiadamente. Si bien esta actividad puede no tener un efecto en la población, la remoción de erizos rojos de sitios donde no hay alimento, deja disponible espacios para ser colonizados por el erizo morado. Esta situación puede dejar a juveniles de erizo rojo sin protección contra depredadores y puede resultar en la disminución de hábitat disponible para el reclutamiento del erizo rojo. Además, los impactos del aumento en la densidad y agregación de erizos en mantos de sargazo gigante no han sido evaluados, por lo tanto se recomienda que los “trasplantes” y programas de reubicación no continúen.

En la ausencia de nutrias en la región y los bajos niveles de abundancia de otros depredadores importantes como el pez vieja o la langosta roja, los buzos se han convertido en el control poblacional más importante para el erizo rojo en la región, por lo que cualquier medida de manejo o conservación de las especies asociadas a los mantos de sargazo gigante debería incluirlos como parte importante del sistema.