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ARTICLE

Using Drift Video Transects and Maximum Likelihood Geostatistics for Quantifying and Monitoring Exploited Subpopulations of *Loxechinus albus* at a Mesoscale

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Abstract

Population monitoring of benthic species has been complicated by difficulties in defining appropriate spatial units for making observations that are relevant to the management of these fisheries. In many cases, this has led to the application of indirect models of stock evaluation using catch and effort information for estimating global quotas, in spite of the fact that experience suggests that limit reference points should be used. The aim of this study was to research the spatial distribution pattern of the Chilean sea urchin *Loxechinus albus* at sites identified as fishing beds through direct evaluation. Thus, video transect recordings were used and geostatistical methods were applied to determine the presence of significant spatial units. We found significant spatial structures represented by beds of 1–120 ha with 20,000–2,300,000 sea urchins that revealed fragmentation of the exploited *L. albus* populations within the study area. Smaller beds were observed close to the landing ports, suggesting that, in the extreme north of the study area, the beds were "in transition" toward becoming unoccupied habitats. This fragmentation is influenced by first-order (habitat availability) and second-order (principally fishing and recruitment) effects. The bed structure observed can be classified as mesoscale, where contraction and expansion dynamics operate. This can

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lead to persistence, extinction, and reemergence over periods that span more than a generation. Observing the expansion–contraction of these beds can, therefore, be extremely informative when interpreting population and large-scale fishery dynamics. Our hypothesis is that the structure of beds observed is associated with local fishery indicators and therefore can be used to monitor and improve management on a regional scale. This would be particularly useful for such complex regions as the Chilean inland sea, although the methodology requires further adjustment.

The Chilean sea urchin *Loxechinus albus* fishery along the Chilean coast produces more than 50% of world sea urchin landings (FAO 2014). After experiencing rapid growth toward the end of the last century, the landings in this fishery have declined progressively from the beginning of the present century, reflecting the global pattern of sea urchin fisheries (e.g., Andrew et al. 2002; Botsford et al. 2004; Orensanz et al. 2005).

The fishery unit between 41º30′S and 46º30′S, which has been regulated since 2005 by the first management plan implemented in Chile (the Plan de Manejo de la Zona Contigua; hereafter, PMZC; see Moreno et al. 2007) accounts for around 50% of sea urchin landings in Chile. A global catch quota was established in 2011 based on an evaluation of the stock that only considers catch, effort, and landings information obtained directly from the fishing process (Jolly and Hampton 1990; Roa-Ureta et al. 2015). Nevertheless, the experience of North American sea urchin fisheries suggests that using population information independent of the fishery, together with the definition of limit reference points, would result in more efficient management measures (Botsford et al. 2004).

One fundamental step toward improving the monitoring and management processes of benthic coastal fisheries is defining appropriate spatial scales (Bruckmeier and Neuman 2005; Lorenzen et al. 2010). In view of this, extending the management unit of the *L. albus* fishery under the PMZC to close to 560 km (around 67,200 km^2; Figure 1) seems inadequate for monitoring and management purposes. This is because adult sea urchins are sedentary animals with a high degree of spatial segregation that can lead to considerable local variations in abundance and mortality (Orensanz et al. 2005; Moreno et al. 2011).

Orensanz and Jamieson (1998) and Orensanz et al. (2006) discussed five scales for analyzing the spatial dimension of population processes: the megascale (thousands of kilometers), the macroscale (hundreds of kilometers), the mesoscale (kilometers), the microscale (centimeters to meters), and the nanoscale (millimeters). They stress that the macroscale, mesoscale, and microscale are the most important scales for evaluating stock and management. The expansion–contraction processes in subpopulations or demographic units would operate more directly in the mesoscale (Luck et al. 2003). As a result, this would also be the most appropriate scale for applying traditional fishing models oriented toward predicting the effects of a fishery on the size, structure, and abundance of these units. In benthic invertebrates the expansion–contraction processes are mediated primarily by larval advection and habitat availability (Orensanz and Jamieson 1998; Orensanz et al. 2005) as well as the secondary, often very localized, effects of fishery activity. Thus, these subpopulations tend to be relatively small and may persist, become extinct, and/or reemerge over periods of time that span more than a generation; furthermore, changes can be slow and erratic.

The PMZC management unit is a macroscale unit in which 439 "fishery origins" have been identified. These origins correspond to the destinations of the fishing fleet conducting this fishery activity (Molinet et al. 2011; Figure 1). They represent one or more beds in close proximity, where shellfish divers obtain between 500 and 4,000 kg of *L. albus* daily per fishing vessel.

Although the precise surface area of the beds exploited in each origin is unknown, it seems reasonable to classify them within the mesoscale proposed by Orensanz and Jamieson (1998). As one approaches the microscale in each bed, one or more sea urchin patches can be found that were formed as a result of habitat heterogeneity, diver activity, or both (Molinet et al. 2014).

Direct evaluation of the sea urchin and other benthic resources is affected by two significant operational and methodological difficulties. First, wave action and the proximity to the coast hinder the implementation of rigorous sample designs and, as a result, complicate the application of design-based analysis methodologies. Second, given the large expanses and considerable number of beds included in the study area, the relationship between sample precision and effort must be maximized. However, conventional methods (e.g., Jolly and Hampton 1990) tend to underestimate the precision of estimates based on continuous observation, which are averaged within each transect. To alleviate both problems without violating the basic assumption of independence, design-based geostatistical methods can be applied, such as those proposed by Roa-Ureta and Niklitschek (2007). Nevertheless, uncertainty remains as to whether they are adequate for modeling the spatial structure of the microscale patches present in the sea urchin beds or that of other benthic resources with a similar distribution.

The main aim of this study was to characterize the abundance, spatial structure, and size of exploited *L. albus* beds and to explore the relationship between these factors and the recently recorded catches for their corresponding fishery origins, with a view to improving the management of this benthic fishery. Simultaneously, we attempted to evaluate the precision

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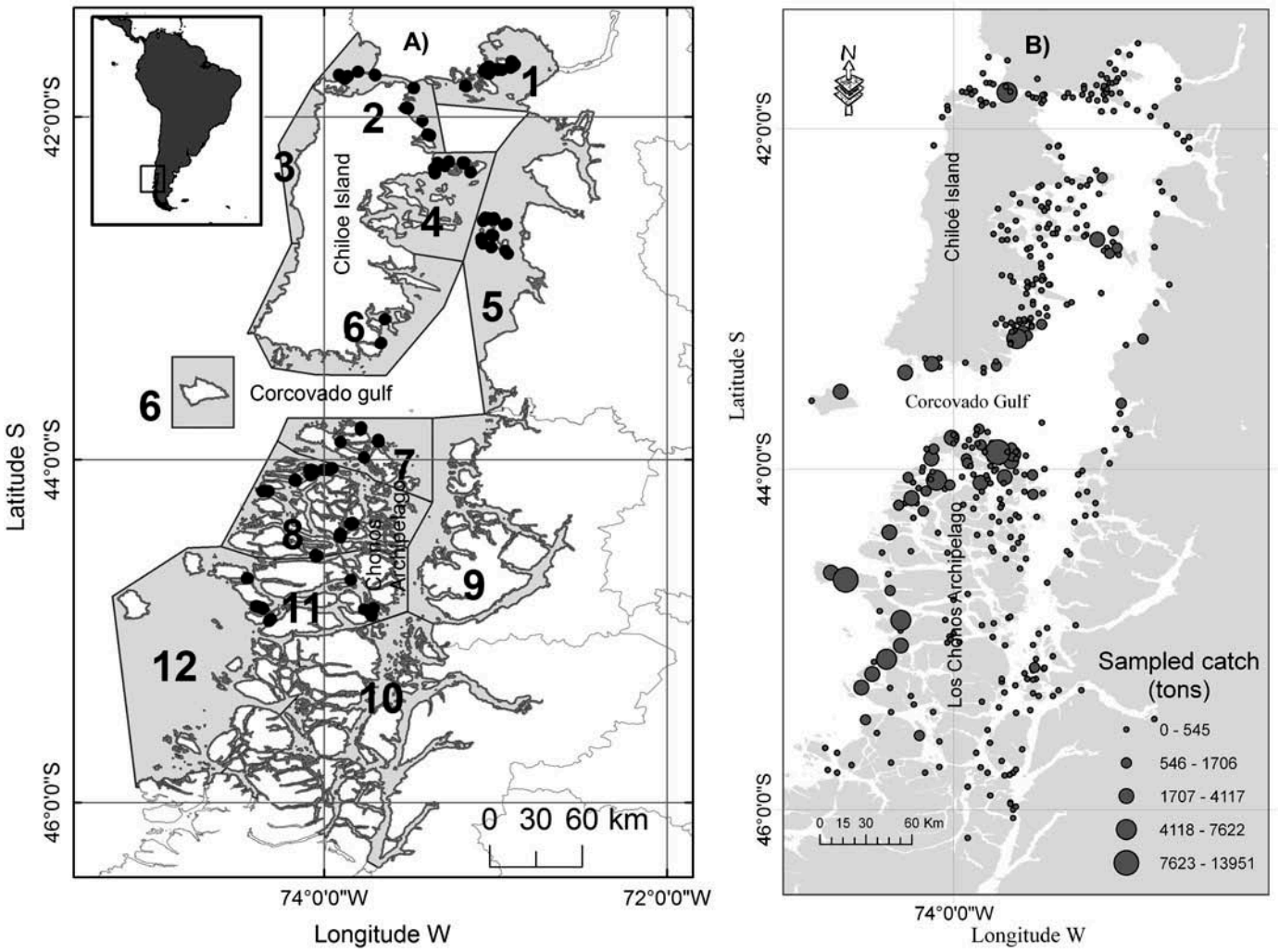


FIGURE 1. Panel **(A)** shows the division of the study area into 12 fishing zones, as proposed by Molinet et al. (2009, 2011). The black dots show the locations of the sites visited during the present study. Panel **(B)** shows the fishing origins identified by the IFOP benthic monitoring program. The sizes of the circles indicate the magnitudes of the landings sampled.

levels obtained by combining sampling through video transects and geostatistical methods for the analysis of sea urchin distribution and abundance in the mesoscale (beds).

STUDY AREA

The study area, located between 41°40′00″S and 45°44′30″ S, has heterogeneous characteristics determined primarily by geological processes on a regional scale that followed the retreat of the last Pleistocene glaciation and, to a lesser extent, tectonic processes (Quiroz and Duhart 2006). Secondly, climate characteristics and the effect of river discharge create a north–south temperature gradient and an east–west salinity gradient in the water column (see Molinet et al. 2011). A third element is the spatial distribution of the fishing effort, which responds to the spatial distribution of the species exploited. This heterogeneity of the study area was analyzed and synthesized in the identification of 12 fishing areas (from 439 fishing origins) (Molinet et al. 2011; Figure 1A, B), of which zones 2, 4, 6, 7, 8, and 12 account for more than 80% of the landings monitored by the Instituto de Fomento Pesquero (Fisheries Development Institute [IFOP]).

An expedition in the RV *Dr. Jürgen Winter* was undertaken in this area to identify and study specific beds representative of the main fishing zones. Sample sites were determined by considering the magnitude of the landings sampled by IFOP in each origin between 1996 and 2012 (Figure 1B), as well as information contributed by local divers and the experience of the team working in the study area.

A total of 40 origins were visited, distributed between the Calbuco Archipelago (41°50′S) and Tres Dedos Island (44°55′S), including 9 of the 12 fishing zones, recording 218 drift video transects (Table 1; Figure 1A). Due to adverse meteorological conditions, samples could not be

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TABLE 1. Numbers of sites at which drift video transect (DVT) recordings were made and Chilean sea urchins were collected, along with the numbers of DVT recordings showing the presence and absence of sea urchins.

	Sampled sites		Number of DVTs showing sea urchin		
Fishing zone	DVT recording	Sea urchin collection	Presence	Absence	Percent showing presence
1	5	4	22	8	73.3
2	6	1	26	10	72.2
4	3	2	12	4	75.0
5	6	1	16	5	76.2
6	3	2	12	4	75.0
7	4	2	12	7	63.2
8	7	5	36	14	72.0
11	2	2	12	1	92.3
12	4	4	15	5	75.0
Total			163	58	

obtained from the most exposed origins of fishing zone 12, where the largest sea urchin catches have been recorded.

METHODS

Drift video transect sampling.—The sampling process was divided into two stages: (1) identification of beds with sea urchin presence based on exploratory video transects in each area of interest and (2) evaluation of sites where the presence of sea urchins was confirmed.

For the identification stage, a Seaviewer Sea Drop Model 960 camera was deployed from a fishing boat, drifting 1–3 times through each selected site. Based on these recordings, we established the approximate limits of the bed according to presence/absence criteria, bathymetric range, and substrate type; this information was recorded in a logbook. The camera was maintained 0.5–1 m from the bottom and was connected by a 150-m cable to an observation and recording control panel. During the evaluation stage, 4–12 relatively parallel additional video transects were obtained while drifting at between 0.2 and 1 m/s (at speeds of over 1 m/s, specimens of *L. albus* cannot be identified with certainty from the video frame). This enabled the boat to drift from the same number of starting points distributed along the bed identified.

Bathymetrically, the video transects extended up to the limit of sea urchin presence or to a depth of 100 m (at sites where the sea urchin bed extended beyond this depth). The distance covered and the depth of the bottom were recorded every second, using a Garmin GPS Map Sounder Model 420 s, with an antenna installed in the vessel, at the same point where the camera was deployed. An error of 10–30 m was calculated, considering the difference in position between the camera and the boat, based on the angle of inclination of the cable with respect to the vertical axis.

In the shallow beds with greater sea urchin abundance, a diver collected sea urchin samples in 3 transects 40–50 m long, following a sled towed from the vessel. The bottom of each transect was filmed using a camera mounted on the sled, its visual field of 28 cm being delimited by lateral metallic rods to guide the diver. While the sled was towed at a speed of around 10 cm/s, the diver collected all the sea urchins found between the lateral rods of the sled; these samples were subsequently counted and measured on board. Later, video processing enabled us to compare the samples obtained by the diver with the number of sea urchins counted in the video.

Processing video transect data.—In the case of the drift video transects, the sample comprises a frame approximately 30–70 cm wide, identified by the recording time (hours–minutes–seconds); the image was frozen to classify substrate type, establish taxonomic identity, and count all the individuals that intersected the lower edge of each image. Two other species of sea urchin, *Arbacia dufresnii* and *Pseudechinus magellanicus*, observed in the videos were differentiated from *L. albus* based on their morphological characteristics (Häuseermann and Försterra 2009). The geographic coordinates of each frame sampled were obtained by aligning the video recording time and the GPS recording time. For the sled video transects, the sample comprised a frame identified by the recording time; the image was frozen to identify and count all the *L. albus* specimens.

Data analysis.—The substrate defined as suitable for *L. albus* corresponded to the rock, gravel, and boulders category (Molinet et al. 2009). This enabled us to compare surface areas with a suitable substrate with those with effective sea urchin presence.

To analyze the spatial structure and estimate the distribution and abundance of *L. albus* using the drift video transects, a geostatistical approach was taken. This approach was based on maximum likelihood (Roa-Ureta and Niklitschek 2007) using 5-m × 5-m cells. Two separate models were developed for each bed for the drift video transects: (1) a binomial model for the distribution analysis (presence/absence) of the stock and (2) a Gaussian model for the conditional density analysis

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(only positive data). In the latter case, to comply with the assumption of normality, the conditional density observations were transformed by means of a Box–Cox function. A Matérn (1986) spatial correlation function was used both in the geostatistical distribution model and the conditional density model. Subsequently, each spatial model was compared with its equivalent nonspatial model ("pure nugget") using the Akaike information criterion (Burnham and Anderson 2004). When the distribution spatial model was more informative than the nonspatial model, the existence of one or more patches in the bed was assumed.

The average density of each bed was estimated following Roa-Ureta and Niklitschek (2007). The surface of the bed was calculated as the product of the cell size (25 m^2) and the number of cells containing positive information in an aggregation. The average size of the patches within the bed was estimated as the product of the cell size (25 m^2) and the number of cells with positive observations in a patch (Molinet et al. 2010).

In 19 cases the bed visited could be associated with a fishing origin as determined by IFOP benthic monitoring during 2013 (some of the origins visited were not fished during 2013, and origins representing extensive areas are difficult to assign to one bed; see Figures 1, 2). In these 19 cases, correlations between estimated abundance and the catch per effort unit obtained from fishery monitoring were evaluated. This permitted an initial approximation with which to link the results of population monitoring with those of fishery monitoring.

The size distribution of each site sampled was recorded on a card (see the Supplement available online) that provided the baseline characterization of the bed. The size distribution per fishing zone was also represented. All statistical procedures were performed using R 3.0.2 (R Development Core Team 2013).

RESULTS

No *L. albus* were recorded in 2 of the 40 sites visited. The proportion of drift video transects with sea urchin presence varied between 63% and 92% in the 221 transects undertaken (Table 1).

The surface area with substrate suitable for *L. albus* varied between 48% and 89% and was greater than the surface area with sea urchin presence. This indicated the availability of suitable substrate in 36 of the 40 beds sampled (Table 2).

Characterization of Beds and Patches

At 31 of the 38 sites with sea urchins present, a geostatistical model could be applied to analyze the structure of the spatial correlation and estimate the distribution and abundance parameters. The geostatistical models proved to be more informative than the nonspatial models in 29 (93%) and 21 (68%) of the beds analyzed for distribution and conditional abundance, respectively.

TABLE 2. Estimated surface area of the total number of sites visited in selected fishing zones, percentage of this surface area at which Chilean sea urchins were present, percentage of substrate suitable for sea urchins, and average bed depth.

Fishing zone	Estimated surface area (ha)	Surface area with sea urchins (%)	Suitable substrate (%)	Average bed depth (m)
1	127.0	27.4	48.2	5
2	178.1	43.6	78.9	30[a]
4	111.5	72.2	77.8	7
5	120.3	23.3	84.9	5
6	59.0	51.3	82.4	15[b]
7	106.0	23.5	88.9	7
8	314.0	39.6	70.4	10[b]
11	18.4	44.5	77.1	7
12	80.4	52.0	81.2	7

[a]Two beds recorded deeper than 30 m.
[b]One bed recorded deeper than 30 m.

In 20 of the 31 beds (65%), both the spatial model of distribution and the spatial model of conditional density were more informative that their respective nonspatial alternatives. This provided evidence of spatial structures on a microscale (patches) within most of the beds. The geostatistical range of the distribution models presented great variability between beds (3–587 m), with an overall average of 162 m (Figure 2A). The geostatistical range of the conditional density models evidenced less variability between beds (10–117 m), with an average of 40 m (Figure 2B).

The surface area with effective sea urchin presence varied among fishing zones. The smallest expanses were observed in fishing zones 1 and 12 and the largest in zones 2, 4, and 7 (Figure 3A). The largest surface area was estimated for the Peligroso Islets (a group of small, exposed islands adjacent to Corcovado Gulf) at around 200 ha.Corcovado

North of Corcovado Gulf (fishing zones 1–6), beds with sea urchin presence occupied 2–63 ha in shallow, mainly gravel, subtidal habitat (3–10 m deep) (Figure 3A; Tables 1, 3). In fishing zone 2, two beds were identified, extending to a depth of approximately 60 m (Corona and Pulelo). One bed, previously discovered by Moreno et al. (2011) and extending to a depth of 110 m, was also considered (Table 3).

South of Corcovado Gulf (fishing zones 8–12), sites with sea urchin presence were characterized mainly by rocky and gravel habitats in the intertidal and subtidal zones. They reached a depth of around 15 m, occupying between 1 and 120 ha per bed (Figure 3A; Tables 1, 3). In exposed sites, narrow belts of sea urchins were observed along the coast between the intertidal and subtidal zones, coinciding with the descriptions of Moreno et al. (2011). In fishing zone 8, one bed was identified that extended to a depth of approximately 80 m. Greater numbers of patches were recorded in fishing zones 2 and 4 (between 2 and 25 patches per bed; Figure 3B). The estimated mean size of the patches was 164 m, varying

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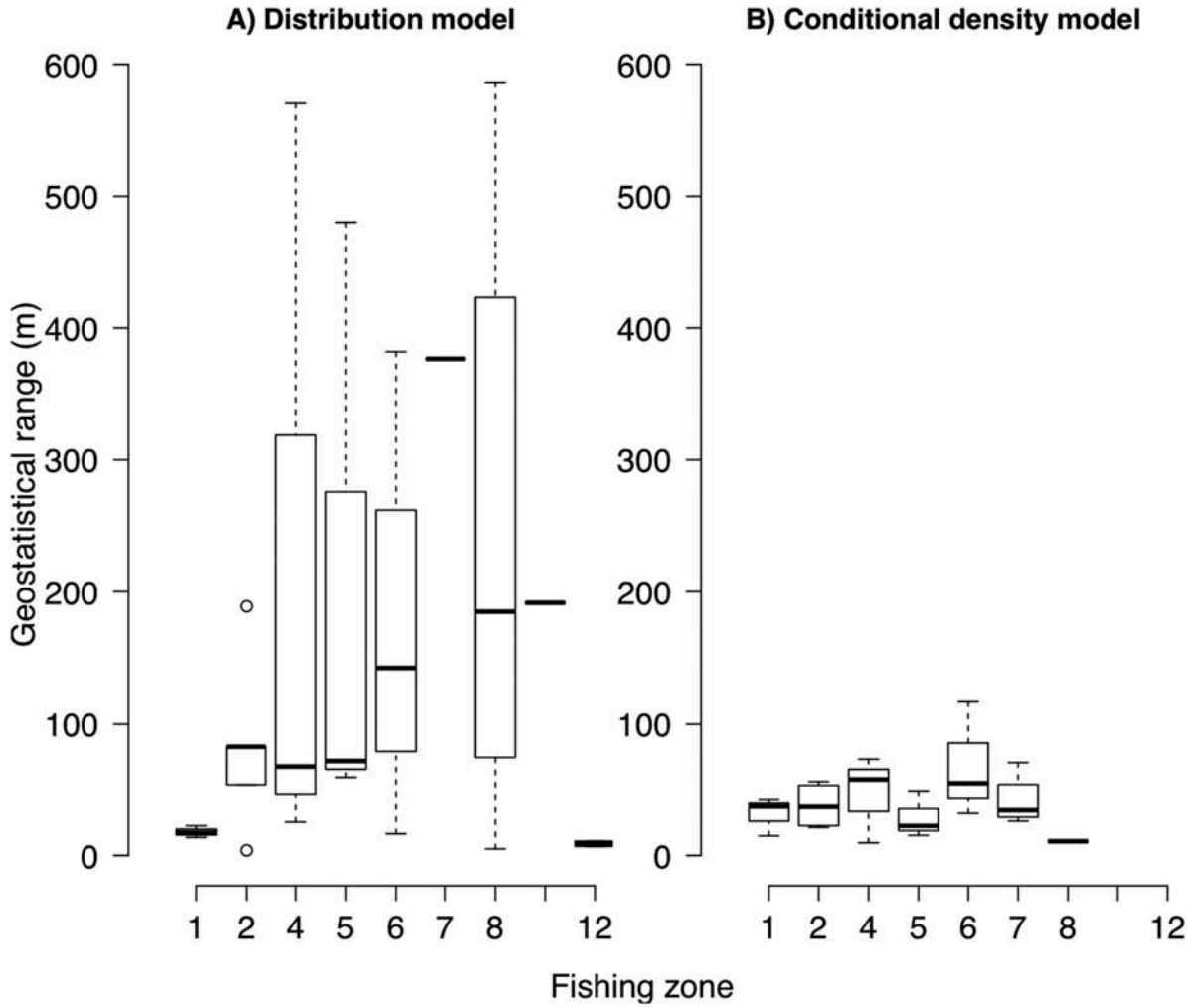


FIGURE 2. Geostatistical ranges of the **(A)** distribution (binomial) and **(B)** conditional density (Gaussian) models for the beds where the spatial models were more informative, following the methodology proposed by Roa-Ureta and Niklitschek (2007). The lower and upper boundaries of the boxes indicate the 25th and 75th percentiles of the observations, respectively. Lines within the boxes indicate the median, circles indicate outliers, and dashed whiskers indicate minimum and maximum values.

between 10 and 846 m, according to the bed sampled (Figure 3C), with smaller patches at the sites in fishing zones 1 and 12.

Sea urchin density at the sites visited varied between 0.44 and 10.7 individuals/m^2 (Table 3; Figure 3D). The highest sea urchin densities were observed in fishing zone 2 (associated with the presence of small sea urchins and ones in deep waters), as well as in fishing zones 8 and 12 (associated with the presence of sea urchins >60 mm test diameter (TD) in rocky belts).

Estimated sea urchin abundance varied between 11,604 and 2,546,000 per bed. The standard error of these estimates varied between 8% and 55% of the abundance calculated for each site (Table 3). The abundance observed in 19 of the beds visited correlated positively with the CPUE recorded for the respective origin that could be identified ($r = 0.53$, $P = 0.021$; Figure 4).

Test Diameter

A total of 4,712 *L. albus* were collected by divers, whereas 11,695 were counted from the video transect sled, with an average of 1.48 times more urchins being counted in the videos than collected by divers (Table 4).

The size distribution of the sea urchins obtained by direct sampling (diving) revealed a north–south gradient (Figure 5), with smaller urchins to the north of Corcovado Gulf (fishing zones 1–6) where, with the exception of fishing zone 2, sea urchins <60 mm TD predominated. To the south of Corcovado Gulf, a larger size range of 20 –80 mm TD was observed.

DISCUSSION

Our results indicate that *L. albus* has a discrete spatial distribution, with beds comprising variably sized patches characterized by differences in density, abundance, and individual

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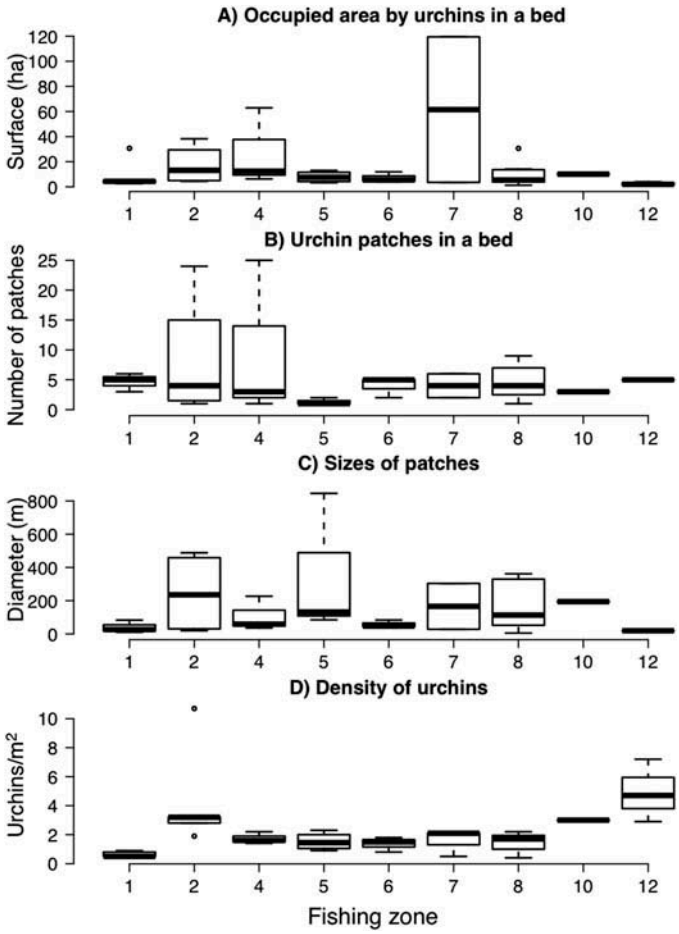


FIGURE 3. **(A)** Surface area occupied by Chilean sea urchins per bed, **(B)** number of patches per bed, **(C)** estimated sizes of the patches, and **(D)** sea urchin density per bed, all by fishing zone.

size structure. At the scale of our analysis, beds can be related to local CPUE obtained through fishery monitoring; this relationship could be used as a tool to improve the management of this resource.

The geostatistical range of the distribution model and the conditional density of the *L. albus* patches identified in this study coincides with that of the patches reported by Molinet et al. (2010) (75 and 118 m, respectively) for the management and exploitation areas for benthic resources (a kind of territorial use rights for fisheries implemented in Chile). These areas are defined and requested by artisanal benthic fishermen. This patch structure is modulated by both first-order effects, such as habitat availability (Caddy 2004), and second-order effects, such as recruitment and fishing.

In beds in which the level of exploitation exceeds recruitment, structures tend to be more irregular, with increasingly smaller patches. This would appear to be the case in fishing zone 1, where the size of the beds and the *L. albus* patches, together with the density and size distribution of the sea urchins, suggest a state of transition toward unoccupied habitats (i.e., extinction; Hanski and Ovaskainen 2003). Furthermore, this fishing zone had the lowest proportion of suitable substrate for *L. albus*, which could affect the probability of successful recruitment by this species and thus that of its being present in that location.

The sea urchin beds to the north of Corcovado Gulf (where sea urchin fisheries began in the 1970s) are more frequently visited, as evidenced by the trend in the size distribution for sea urchins in these fishing zones. The processes underlying the contraction of the patches in zone 1 require more in-depth study in order to develop management strategies to prevent extinction in this and other fishing zones.

In general, *L. albus* habitat consists of shallow bottoms (intertidal to 15 m deep) composed of gravel, boulders, and rock (Vásquez 2001), although on rare occasions beds have been recorded deeper than 40 m (Moreno et al. 2011; Moreno and Molinet 2013). The Chilean sea urchin's preference for shallow habitats is influenced by its herbivorous diet, which is highly dependent on drifting algae (Vásquez et al. 1984; Branch and Moreno 1989). In view of this, the presence of patches at depths of up to 110 m can only be explained by bathymetric traits that affect circulation and promote the transport of algae and particulate material to these depths (Cáceres et al. 2008; Molinet et al. 2012).

In this study, we identified three sea urchin beds that extended to depths of around 60–70 m, two in the Chacao channel (fishing zone 2), an extremely dynamic environment with high recirculation and friction (Cáceres et al. 2003), and one at Midhurst Island (fishing zone 8), for which no oceanographic information is available.

Our hypothesis is that deeper patches are located in a sink habitat (Pulliam 1988) and that they are maintained by dispersion from beds or patches in a source habitat. Thus, annual observation of the bathymetric expansion/contraction of these beds could provide more informative results about population tendencies. In this respect, Holt (1985) stated that the passive dispersion between source habitat and sink habitat in seasonally constant environments may be selectively disadvantageous, implying that the patches in sink habitats are transient.

The *L. albus* beds analyzed in this study can be classified at the mesoscale proposed by Orensanz and Jamieson (1998). That is, they form the basic unit of stock dynamics, in which persistence, extinction, and reemergence can be observed over periods of time spanning more than a generation. Bed dynamics can also be contrasted with the CPUE of the same beds obtained from fishery monitoring, which, as suggested by Orensanz et al. (2005), appears to be a good indicator of relative abundance on this scale for this type of fishery.

The values of and trends in the indicators identified in this study could be used to calibrate stock assessment models for *L. albus* and/or to formulate decision rules, either in the context of a formal management strategy evaluation (Holland 2010) or as part of an empirical management approach (Punt et al. 2012). For example, the size distribution and density indicators of two beds in fishing zone 2 point to a declining

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TABLE 3. Distribution and abundance of Chilean sea urchins observed in the drift video transects and areas sampled at each site. Alpha is the surface area where a sea urchin stock was found according to the geostatistical distribution model. Highlighting in dark gray indicates sites at which the spatial model was more informative than the binomial and Gaussian models. Highlighting in light gray indicates sites at which a binomial or Gaussian model was more informative. Bold italics denote sites at which sea urchins were recorded at depths of up to 80 m.

Fishing zone	Site	Sea urchins sampled	Density (no./ m^2)	SE of density	Alpha (ha)	SE of Alpha	Abundance	SE of abundance
1	Quenu Island	814	0.87	0.13	4.36	0.05	38,862	5,946
1	Southeast Guar Island	630	0.45	0.12	2.53	0.13	11,604	3,172
1	East Guar Island	286	0.54	0.20	4.94	0.22	26,928	10,478
1	Tautil Strait	148	0.82	0.24	3.40	0.16	27,924	8,404
2	Point Picuta	3,304	2.80	0.44	4.33	0.00	123,698	19,472
2	Linao Shoal	1,776	3.22	0.33	0.11	0.01	32,632	3,356
2	***Point Corona***	***1,524***	***3.24***	***0.13***	***38.24***	***0.88***	***1,240,012***	***127,200***
2	Lobos Headland	4,316	1.87	0.26	20.67	0.11	387,408	55,768
2	***Pulelo***[a]	***13,030***	***10.66***	***2.71***	***5.59***	***0.04***	***597,132***	***152,346***
4	Chincui Shoal	6,518	1.36	0.40	6.25	0.04	85,250	23,198
4	Tenaun Island	1,356	1.61	0.92	12.47	0.28	201,324	36,874
4	Point Pájaros	4,668	2.16	0.38	62.92	0.85	1,366,262	237,396
5	Chulin Island	1,378	2.28	0.52	10.11	0.40	231,262	53,226
5	Nayahue Island	928	1.66	0.42	13.01	0.11	217,374	54,574
5	Point Nayahue	640	0.94	0.34	5.08	0.09	47,654	18,268
6	Yatac[b]	650	1.81	0.98	5.45	0.18	98,808	54,428
6	Point Paula	780	0.76	0.29	3.82	0.10	29,164	11,058
6	San Pedro Island	720	1.48	0.56	11.91	0.71	178,124	67,456
7	Peligroso Islets	570	2.12	0.14	119.46	2.64	2,547,006	198,206
7	Cuatro Canales	171	2.1	0.19	3.55	0.00	74,550	7,400
8	Sierra Island	44	0.54	0.18	3.00	0.20	16,652	5,392
8	Erizo Island	656	2.08	1.26	1.22	0.07	25,620	14,684
8	Llanos Island	1,260	2.16	1.68	30.61	0.43	667,150	551,516
8	Canave Island	292	0.44	0.07	4.67	0.12	21,348	3,444
8	***Midhurst Island***	***4,440***	***1.68***	***0.32***	***14.12***	***0.12***	***237,636***	***44,816***
8	North Skorpios Channel	1,552	1.76	0.68	5.45	0.20	95,970	37,926
8	South Skorpios Channel	918	1.54	0.34	13.13	0.26	201,904	46,332
11	Tahuenahu Island	950	3.00	0.52	10.08	0.10	231,876	52,740
12	Goñi Channel	2,792	4.74	1.20	3.99	0.12	189,468	48,464
12	Northeast Rowlett Island	646	7.20	2.12	1.98	0.14	143,142	43,072
12	Southeast Stockes Island	156	2.86	0.60	1.33	0.09	38,246	8,604

[a] Also known as Point Tres Cruces.
[b] Also known as Blanco Islet.

trend for this population since 2011, which was reflected in the severe decline in landings in 2014 (Barahona et al. 2014).

In the future, bed monitoring should enable us to identify which beds best represent variability per fishing zone, as an aid to assessing the relationship between indicator trends (fragmentation, density, and size distribution), conservation status, and the state of the fishery in each zone. Once this calibration is made, it will be possible to recommend limit reference points (possibly per fishing zone) by which to avoid further overfishing.

Our data show that it is possible to consider spatial correlation at the microscale (patches) in each bed, and that this is

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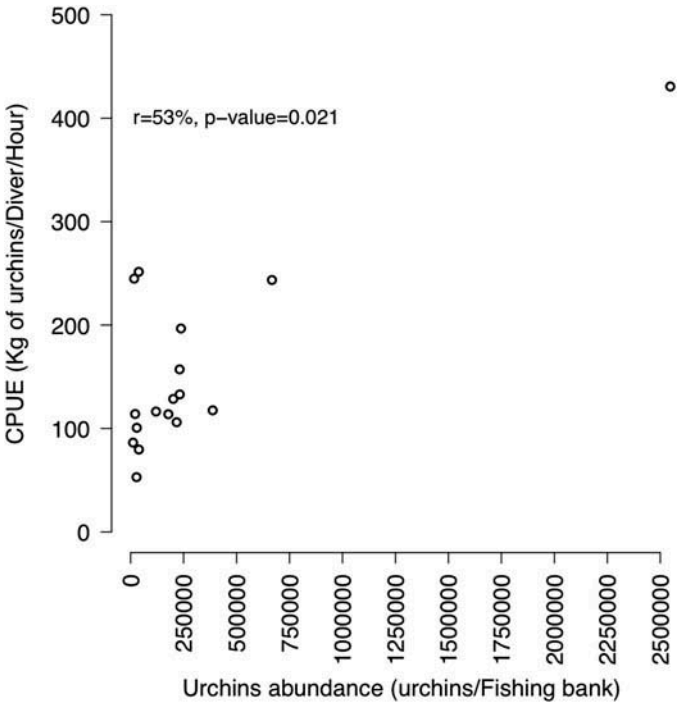


FIGURE 4. Relationship between the abundance of Chilean sea urchins in the beds sampled and catch per unit of effort for the respective origin as determined by IFOP fishery monitoring in 2013.

TABLE 4. Comparison of the number of sea urchins collected by diving and the number recorded by video transects. The sample mismatch is the percentage by which the latter number exceeds the former number.

Fishing zone	Number collected by diving	Number counted from video transects	Sample mismatch (%)
1	776	862	11.1
2	411	402	–3.2
4	622	2,628	322.5
5	330	353	7.0
6	654	1,569	139.9
7	739	1,095	48.2
8	709	3,147	343.9
10	143	423	195.8
12	328	1,216	270.7
Total	4,712	11,695	148.2

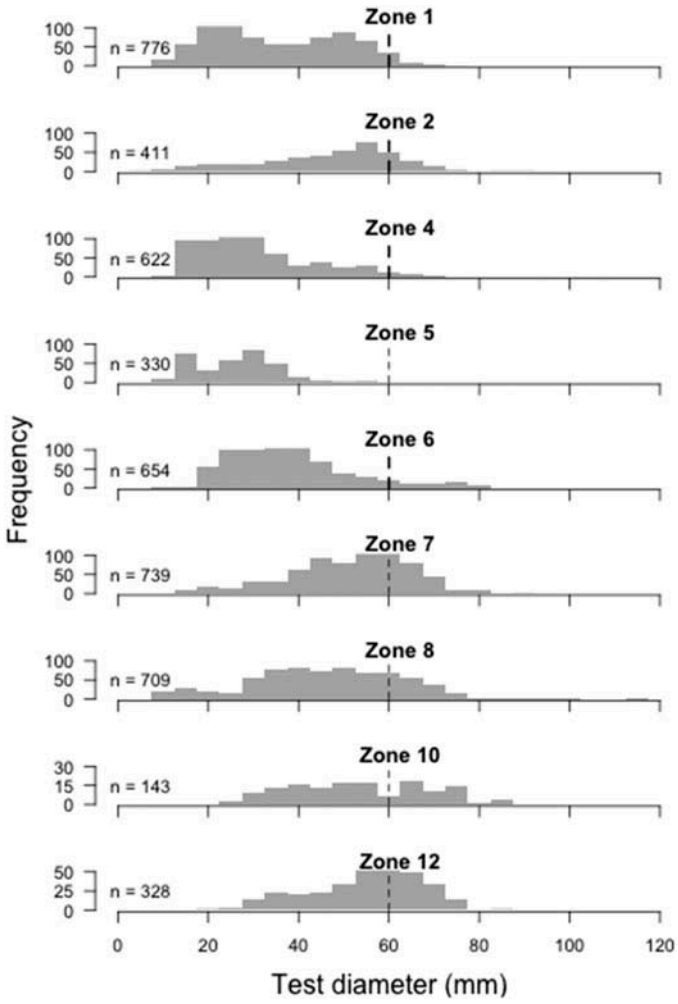


FIGURE 5. Distributions of the test diameters of Chilean sea urchins collected by diving in nine fishing zones; *n* indicates the number of sea urchins collected. Note that there are some differences in scale among the various distributions.

necessary to provide information with regard to fragmentation. It was found that the levels of precision in estimating the abundance of stock at the mesoscale were unacceptable in some of the beds evaluated, with standard errors of up to 55%. Consequently, the design requires improvement, increasing the number and optimizing the distribution of transects to evaluate the highly variable microscale between beds. We believe that it would then be possible to reduce the standard errors to conventionally accepted values of around 10% of the estimated abundance.

Blanchard et al. (2008) suggest that the best monitoring design for observing tendencies in population dynamics is one that is stratified and comprised of fixed stations. Our proposal suggests stratification by fishing zones following Molinet et al. (2011) and identification of beds that are representative of the study area, including those with a greater bathymetric range. Additionally, geostatistical methods for the evaluation of local stocks of the purple sea urchin *Paracentrotus lividus* have been applied to management at local scales (Addis et al. 2009). Methods devised by Addis et al. (2009) could be used to study *L. albus* populations.

The differences in the numbers of sea urchins collected by divers and those derived from video transect sled recordings can be explained by the complexity of sample collection at very

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dynamic sites. For example, wind and tide action negatively affect the divers' performance, preventing collection of all of the sea urchins passing under the sled. This aspect of the methodology must also be improved to ensure a more precise representation of the number of sea urchins and their size distribution.

Image collection for monitoring benthic communities has permitted the accurate characterization of benthic species' patches, where natural variability in habitat structure can be observed. This, in turn, facilitates the detection of ecological changes (Smale et al. 2012). Results for the communities associated with the *L. albus* beds are available in the Supplement. The full database and video transects can be requested from the first author of this study; for more in-depth analysis, see Molinet et al. (2014).

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