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**Quebec Region**

# **Assessment of Green Sea Urchin (*Strongylocentrotus droebachiensis*) Stocks in Quebec's Coastal Waters in 2024**

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## Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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## ABSTRACT

This document describes the data and methods used to analyze the status of green sea urchin (*Strongylocentrotus droebachiensis*) stocks in sea urchin fishing areas in Quebec's coastal waters in 2024, in order to support the assessment of these stocks in the Estuary and northern Gulf of St. Lawrence. The status of sea urchin populations was assessed using the known data on landings and fishing effort from 2002 to 2024, information from a fishery-independent survey in the mouth of the Saguenay Fjord (subarea 9-1), and size measurements.

The relatively short history of the sea urchin fishery, spatiotemporal inconsistencies in the data on the distribution and intensity of fishing effort, and the uneven quality of fishery data make it difficult to analyze the effects of fishing. Owing to the lack of information on the dynamics of this population, its productivity and resilience to harvesting cannot be determined. The status of the green sea urchin stocks in Quebec's coastal waters is uncertain under the Fishery Decision-Making Framework Incorporating the Precautionary Approach, due to the absence of reference points. Defining an appropriate spatial scale for the management of the sea urchin fishery is recommended.

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## INTRODUCTION

The green sea urchin (*Strongylocentrotus droebachiensis*, hereafter referred to as sea urchin, Figure 1) is a widely distributed and dominant species in the marine ecosystems of the Estuary and Gulf of St. Lawrence (EGSL). This omnivorous species plays an important ecological role by controlling the growth of kelp beds and by structuring benthic communities in the subtidal zone (Vadas and Elner 1992; Scheibling et al. 2020). The first sea urchin landings in Quebec were made in 1991. This document describes the data and methods used to analyze the status of stocks in the sea urchin fishing areas (Figures 2 and 3) in Quebec's coastal waters in 2024. Information on sea urchin biology and ecology, as well as a description of the fishery in Quebec's coastal waters, is summarized in order to support the assessment of Quebec stocks. This stock assessment only targets sea urchin fishing areas 8 to 11—those that are actively fished. A peer review of the information described in this document took place on April 1 and 2, 2025 ([DFO 2025](#)). The documents associated with previous assessments are published on the [Fisheries and Oceans Canada \(DFO\) Canadian Science Advisory Secretariat \(CSAS\)](#) website.

## DISTRIBUTION AND HABITAT

The green sea urchin has a circumpolar distribution, occurring in most boreal and Arctic marine regions, including the EGSL (Himmelman et al. 1983; Scheibling et al. 2020; DFO 2025). In general, sea urchins can be found at depths of 0–300 m, but are more common at those of 0–50 m in the shallow subtidal zone. The species is associated with hard substrates such as bedrock, boulders, rocks, gravel and red coralline algae beds (rhodoliths), but can also be found on soft sandy bottoms. The shallowest depths at which sea urchins occur vary depending on the season, wave action and the presence of winter ice scour, which can dislodge individuals and limit their ability to feed (Scheibling et al. 2020; DFO 2022).

Throughout its range, the green sea urchin is strongly associated with macroalgae. Urchin densities vary greatly as a function of the quantity of macroalgae present, with very high densities found in kelp beds. At high densities, sea urchins graze destructively on macroalgal beds, transforming them into vast and impoverished urchin barrens. Sea urchins can persist indefinitely in these environments, preventing macroalgae from becoming re-established. Although coastal ecosystems can shift back and forth between domination by sea urchins and macroalgae, the EGSL is considered a stable urchin barren (Ling et al. 2015). Below the lower limit of the subtidal zone and the limit of macroalgae bed distribution, sea urchin density generally declines as depth increases. Aside from these factors, sea urchin distribution can also be limited by temperature and salinity; however, the species is able to adapt to local conditions to some extent, due to the plasticity of several of its life history traits (Ling et al. 2015; Scheibling et al. 2020; MPO 2022).

## POPULATION STRUCTURE

The pale sea urchin (*S. pallidus*) is similar to the green sea urchin and shares its habitat, including that in the EGSL. Hybridization between the two species also occurs. Morphologically, the two species are similar and difficult to differentiate without examining individuals' finer morphological structures. Differences in colour, size, and in the shape and number of various structures may occur that vary by habitat, but these traits overlap in the two species (Vasseur 1951; Strathmann 1981; Vader et al. 1986). The extent and magnitude of the overlap in the distribution of, and the hybridization between, the green sea urchin and the pale sea urchin in the EGSL are unknown (Swan 1962; Strathmann 1981; Addison and Hart 2005; Addison and Kim 2018; Scheibling et al. 2020; Addison and Kim 2022). Owing to the difficulties

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of distinguishing between the two species, all DFO research surveys in the EGSL identify sea urchins to genus only (i.e., *Strongylocentrotus* spp.).

Several recent studies have focused on genetic polymorphism and population structure in the green sea urchin in order to elucidate dispersal dynamics, regional genetic differentiation and interactions with the pale sea urchin throughout its distribution. Despite marked genetic differences between green sea urchins in the northeast and northwest Atlantic and Pacific oceans, populations in the northwest Atlantic, including those in Canadian waters, are genetically homogeneous. The results suggest that the population is panmictic and has a vast distribution, which is probably facilitated by the species' extensive larval dispersal capacities; larval dispersal can occur over a period of one to four months. This high degree of genetic homogeneity is consistent with a high level of gene flow and a low rate of genetic drift, and indicates that these populations form one large panmictic unit (Addison and Hart 2004, 2005; Addison and Kim 2018; Scheibling et al. 2020; Addison and Kim 2022).

### **Definition of a stock**

According to DFO (2023), "The term "stock" can be defined based on the management unit, assessment unit, and/or biological unit. A "stock" can be considered as a semi-discrete group of aquatic animals (fish, invertebrate, marine mammals) with some definable attributes in common that are of interest to managers and can be assessed as a unit. Therefore, a stock may be defined with reference to units for the purpose of fisheries management. Alternatively, a biological stock is a population of a given species that forms a reproductive unit and breeds little if at all with other units. The spatial and temporal domains for data collection, analysis and assessment may not be correctly aligned with the spatiotemporal distribution of the biological stock, although the extent of this misalignment may be uncertain. For this reason, the term "stock" is often used in reference to an assessment / management unit even if that unit does not fully encompass a biological stock or if it encompasses more than one biological stock."

In the case of the species considered in this document, a stock is defined in terms of units for fisheries management purposes, and therefore by fishing area and subarea. Although green sea urchins are widely distributed in the EGSL, the fishery—like the available data—is concentrated in small, highly localized areas containing commercially valuable aggregations of the species. With the data currently available, it is impossible to determine the status of all green sea urchins in the EGSL, or even to identify a possible substructure in the biological unit.

This definition of the term "stock" poses certain risks in fisheries management and the interpretation of the indicators described in this document (Marentette et al. 2021; DFO 2023). Although adult movements are known to be relatively limited (generally between 1 and 5 m/day, but up to 0.8 m/hour; Dumont et al. 2006), the duration of the planktonic larval phase (1–4 months) and genetic evidence indicate that all green sea urchins in the northwest Atlantic form a large, panmictic population (Scheibling et al. 2020). A scale mismatch—i.e., a poor overlap in space or time between the management unit and the biological unit—is therefore likely. More specifically, mismatches in spatial resolution are likely to occur (i.e., when a stock is assumed to be one biological unit when it actually contains several units with different demographics and dynamics, e.g., sea urchins that do or do not have commercially valuable gonads), along with mismatches in spatial extent (i.e., when a management unit contains only a portion of a biological unit) (Marentette et al. 2021; DFO 2023). These scale mismatches can serve as significant sources of bias in indices, undermine the effectiveness of the management measures in place, increase the risk of depletion of population sub-components, and jeopardize our ability to even detect changes in populations (Marentette et al. 2021; DFO 2023). Defining an appropriate spatial scale for managing sea urchin harvesting is therefore essential (Miller and Nolan 2000; Chen and Hunter 2003; DFO 2009; Johnson et al. 2012).

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## BIOLOGY

The green sea urchin has a moderately long lifespan (around 20–25 years) and grows slowly, to a great degree in response to environmental conditions. Its growth and development show strong phenotypic plasticity, with growth rates and gonadal investment varying according to food availability and quality. Marked differences in growth rates are found not only between populations, but also within the same area, reflecting the importance of food resources and the physical environment (Sainte-Marie and Paille 2020; Scheibling et al. 2020).

Juvenile sea urchins settling on the seabed have a test diameter (i.e., exoskeleton) of around 0.5 mm. At one year of age, test diameter can range between 6 and 33 mm; at two years, between 6 and 50 mm. Therefore, an individual measuring 50 mm in diameter can be anywhere between 1 and 25 years of age, demonstrating the lack of a straightforward correlation between size and age and underlining the high inter-individual variability in the species. The minimum catch size (50 mm) could correspond to a minimum age of 5–7 years. Under favourable conditions, sexual maturity is reached at a size of around 25–30 mm and about three years of age (Sainte-Marie et al. 2012; Sainte-Marie et Paille 2020; Scheibling et al. 2020).

Under favourable environmental conditions, sea urchins' gonads develop over the summer and fall, reaching maximum size in late winter. Spawning is synchronized and occurs in April or May, depending on the region. Reproduction takes place when males and females release gametes into the water column, with fertilization occurring externally. The embryonic phase is followed by a pelagic planktonic larval phase, which generally lasts from one to four months, allowing larvae to be transported for potentially significant distances in surface waters. Dispersal and recruitment are therefore highly variable in space and time. Once fully developed, the larvae settle on the bottom and metamorphose into juveniles, marking the transition to the benthic stage (Sainte-Marie et al. 2012; Sainte-Marie and Paille 2020; Scheibling et al. 2020).

The sea urchin is an omnivorous species that occupies a central position in subtidal benthic communities. Owing to its ability to exploit a wide range of food resources, combined with its population dynamics, which are strongly influenced by environmental conditions, it is a dominant structuring species of benthic habitats in coastal areas of the northwest Atlantic and the EGSL. Its ability to modify the algal cover influences species composition and the trophic structure in benthic habitats (Vadas and Elner 1992; Ling et al. 2015; Scheibling et al. 2020).

Sea urchins show a marked preference for certain genera of brown macroalgae, such as *Alaria*, *Chordaria*, *Saccharina*, *Laminaria* and *Petalonia*, as well as for the green alga *Ulvaria*. However, in the absence of its preferred food, it can survive by consuming less nutritious or drifting algae and a wide variety of other foods, including microalgae, live or dead animal prey, and even other sea urchins. Three feeding strategies have been described in the species:

1. passive detritivory, characterized by small, stationary aggregations of sea urchins feeding on drifting algae;
2. dispersed grazing, with small, mobile individuals occurring at low densities in urchin barrens, consuming drifting algae and other foods to survive; and
3. aggressive herbivory, corresponding to large aggregations at the margins of kelp beds, which they intensively graze, denuding the substrate and leading to the formation of urchin barrens, characterized by sparse algal cover and a high density of individuals.

An individual may adopt different strategies at different times in its life cycle. For example, when large sea urchins at the grazing front are removed by the fishery, they may be replaced by small individuals already present that grow into large individuals, or by the immigration and growth of

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urchins from less favourable peripheral areas (Dumont et al. 2006; Lauzon-Guay and Scheibling 2007; Scheibling et al. 2020).

In the EGSL, the absence of any major predator identified as a limiting factor on the species reinforces the species' structuring role in subtidal communities. In the EGSL, the main predators of sea urchins include lobster, crabs, large sea stars, wolffishes (*Anarhichas* spp.) and seabirds. However, predator-prey interactions and fishing pressure can locally modulate densities and feeding strategies, with additional impacts on the algal cover and the dynamics of other species in the community (Vadas and Elner 1992; Scheibling et al. 2020; Smale 2020). Aside from predation, fishing, and environmental conditions, disease can affect sea urchin mortality. Mass mortality associated with the pathogenic amoeba *Paramoeba invadens* and with periods of warmer water has been described in Nova Scotia (Miller and Nolan 2000; Scheibling et al. 2020). In the early 1980s, this pathogen reportedly caused the death of 270 kilotonnes of sea urchins in Nova Scotia, or 10 to 100 times the biomass removed by the fishery in this region during the 1995–2000 period (Ling et al. 2015; Scheibling et al. 2020). Changes in macroalgal productivity and distribution can also affect sea urchin mortality. Climate change and warming waters are altering the distribution of some macroalgae and diseases; for example, the recent invasion of the bryozoan *Membranipora membranacea* is threatening the kelp beds on which sea urchins feed in some regions, with suggested indirect effects on food and population dynamics (Smale 2020; Scheibling et al. 2020).

## **ECOSYSTEM AND CLIMATE CHANGE CONSIDERATIONS**

The green sea urchin is one of the most important structuring factors in the coastal ecosystem of the northwest Atlantic. It plays a central role in the distribution and abundance of aquatic vegetation, including macroalgae. Intensive grazing on macroalgae by sea urchins influences ecosystem productivity and the availability of critical habitat for a number of marine species (Ling et al. 2015; Scheibling et al. 2020).

The species exhibits a certain amount of ability to adapt to local and seasonal environmental conditions (temperature, salinity, food preferences). However, rising water temperatures due to climate change could lead to shifts in sea urchin distribution, a decrease in larval survival rates, and an increase in the rate of predation by crustaceans (Scheibling et al. 2020). The productivity and distribution of macroalgae, a key item in sea urchins' diet, are also influenced by warming waters (Smale 2020). Sea urchin distribution and productivity can also be influenced by rapid salinity changes, hypoxic conditions, and wave- and ice-induced coastal erosion dynamics (Scheibling et al. 2020).

The bottom waters of the EGSL are experiencing rising water temperatures, a reduction in winter ice volume, an increase in hypoxia in some places, and an increase in the abundance of crustacean predators such as American lobster (*Homarus americanus*) and Atlantic rock crab (*Cancer irroratus*; Bernier et al. 2023; Blais et al. 2023; Galbraith et al. 2024). However, the effects of these phenomena on the productivity of sea urchin stocks are unknown.

Mass mortality events following infections caused by the pathogenic amoeba *Paramoeba invadens* have been reported in regions surrounding the EGSL. The vulnerability of sea urchins in the EGSL to this amoeba could change, given the potential effects of climate change on the distribution of this pathogen (Miller and Nolan 2000; Ling et al. 2015; Scheibling 2020).

## **DESCRIPTION OF THE FISHERY**

Sea urchins are currently harvested solely for their gonads, which are considered a delicacy. The exploitation of this species is therefore dependent on its reproductive cycle. Although the sexes are separate in sea urchins, there is no apparent sexual dimorphism. Consequently, both

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male and female sea urchins are harvested. The gonads of sea urchins in the St. Lawrence Estuary increase in size and weight in autumn, reaching their maximum in late winter (up to 25% of an individual's weight). Spawning takes place in April or May, depending on location, and gonad weight generally remains below 10% of body weight throughout the summer. Consequently, harvesting green sea urchins in the EGSL is most lucrative between the months of September/October and April/May (Sainte-Marie et al. 2012; Sainte-Marie and Paille 2020).

There are 14 sea urchin fishing areas along Quebec's coastline (Figures 2 and 3). However, only areas 8 to 11 are actively fished, with most fishing taking place in areas 8 and 9 (Table 1; Figure 4). Areas 8 and 9 are located in the Estuary, area 8 in the southern portion and area 9 in the northern one. In area 8, the sea urchin fishery is concentrated in two subareas created in 2012: 8D around Île Verte and 8E around Île-aux-Lièvres and Île Blanche. In area 9, most fishing takes place at the mouth of the Saguenay River (subarea 9-1), which was separated from the rest of area 9 in 2010. This subarea encompasses Batture aux Alouettes, Baie Sainte-Catherine, Batture de la Pointe aux Vaches, and parts of the Saguenay–St. Lawrence Marine Park. Areas 10 and 11 are located along the North Shore and are less heavily exploited. Area 10 stretches between Pointe-des-Monts and the Sheldrake River in the northeastern part of the EGSL. Area 11 encompasses the Mingan Archipelago and is bounded by the Sheldrake River and Natashquan Point.

Two fishing methods are authorized in the sea urchin fishery in the Quebec Region: scuba diving and whelk traps. Trap fishing, although allowed, has never been widespread and was mainly practiced in area 9; however, it has not occurred there since 2007. Only two licence holders (commercial since 2024) are active in area 8, each in separate sectors (8D and 8E). Area 9 has seven licences, five of which have been commercial since 2009, but their use varies from year to year. Permanent licence holders have exclusive rights to fish in subarea 9-1. In area 10, there is only one exploratory licence holder. No fishing activities were recorded in area 10 in 2022, and only one and three fishing trips in 2023 and 2024, respectively. In area 11, there are three exploratory licence holders, but only a single licence has been active there in recent years (in 2022 and 2023, but not in 2024).

Management measures in effect for the sea urchin fishery in the Quebec Region vary by subarea, but have certain elements in common, including a 50 mm diameter minimum test size. Limits are also placed on vessel size and the number of active divers on a fishing trip. In area 8, the fishing season runs from August 1 of the current year to May 31 of the following year. An annual total allowable catch (TAC) is in effect in each subarea (8D: 126 t and 8E: 110.7 t). Vessels must hail in for fishing trips, and must also be equipped with a Vessel Monitoring System (VMS) and an Electronic Monitoring System (EMS). In area 9, the fishing season runs from March 20 to December 31. A daily individual maximum catch limit of 4,200 kg has been in effect in subarea 9-1 since 2018; it is 6,000 kg in the rest of area 9 (excluding subarea 9-1). Fishing effort is controlled by limiting the number of fishing days. A total of 80 fishing days are allowed in subarea 9-1. In the rest of area 9, 12 fishing days are allowed inside the marine park boundaries and 12 fishing days outside the marine park. Fishing effort in areas 10 and 11 was initially limited to 40 fishing days, but has been increased to 60 fishing days since 2019 for all licence holders.

## **METHODS**

### **FISHERY-DEPENDENT DATA**

Landings data (1985–2024) were acquired from logbooks and purchase slips and recorded in zonal interchange file format (ZIFF) format. Data on sea urchin landings in the Quebec Region

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have been available since 1991. These data are largely collected, validated and managed by the data management team and technical support staff at the Maurice Lamontagne Institute, including DFO's network of dockside samplers. The data on landings, fishing effort and catch per unit effort (CPUE) presented in this assessment were obtained from harvesters' logbooks and purchase slips. DFO's Regional Statistical Services compiles these data to produce the ZIFF files. The data were then validated by comparing the data in the ZIFF files with those from the original purchase slips and logbooks; corrections were made when necessary and missing values were filled in when possible.

Logbooks are completed daily by harvesters and provide the information on, among others, licence holder identification, vessel identification, landing date, catch date, fishing location, fishing area or subarea, landed weight, and duration and number of dives per diver. Fishing effort is expressed in diver-hours (dh), or the time one diver spends underwater harvesting sea urchins during a single dive. The recorded effort does not include the time in the boat required to find aggregations of sea urchins of commercial interest and does not take account of whether these aggregations were spotted during previous fishing trips. For the reasons discussed below, the daily totals for catches and effort during a single fishing trip, all divers and dives combined, are used to calculate CPUE. Consequently, the fishing effort per trip should take account of not only the time spent collecting sea urchins, but also the number of divers present on a trip, the number of dives made by each diver, the number of oxygen tanks allocated to each diver, the volume of the oxygen cylinders (each typically allowing roughly one hour of diving time) and divers' experience. Under ideal conditions, values of around 16 hours of total diving time could be expected during a fishing trip with a team of four divers, each equipped with four oxygen cylinders, enabling them to make four dives of one hour each.

Data on sea urchin size structures (based on test diameter to the nearest mm) were obtained from landings sampled in DFO's dockside monitoring program (1996–2024). Assigning samples to subareas 8D, 8E and 9-1 before they were created and delineated was not possible with the information available. However, the main areas fished in areas 8 and 9 remained relatively constant over the entire time series. Between 2022 and 2024, 18, 19 and 24 sea urchin samples comprising 3265, 2969 and 3594 individuals, respectively, were measured.

## **Analyses**

An initial validation of the landings data was carried out by excluding or filling in missing or implausible data for the essential variables (catch, effort, catch date, fishing location, and fishing area or subarea). The data for the last five years are generally considered preliminary because the data from some logbooks may not have been entered by the time of analysis. Landings in subareas 8D, 8E, and 9-1 prior to their establishment were reconstructed by using georeferenced data and other related information.

Daily landings and total daily effort were calculated by taking the sum of these values for each diver and dive during a fishing trip. Annual landings in each area and subarea were also calculated. CPUE values were obtained by calculating the ratio between the sum of landings (kg) and effort (diver-hours) for all divers and dives during a fishing trip. Annual averages and 95% confidence intervals (95% CI) (bootstrap  $n = 1000$ ) were then calculated for each subarea. Mean and 95% CI values were also calculated for the size structure data obtained from DFO's dockside sampling program.

## **FISHERY-INDEPENDENT DATA**

Agence Mamu Innu Kaikusseht (AMIK) carried out research surveys in 2016, 2019, and 2024 in subarea 9-1, at the mouth of the Saguenay River. The 2016, 2019, and 2024 surveys followed

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the same sampling protocol used by DFO in the 2008 and 2010 research surveys (Sainte-Marie et al. 2012; Bélanger 2020; DFO 2022). The objective of the surveys was to evaluate the demography of sea urchins at the mouth of the Saguenay River following the increased fishing effort in this sector since 2004. The sampling plan specifically targeted the “batture aux Alouettes” and the “batture de la pointe aux Vaches”, in locations near dense sea urchin aggregations around kelp beds and at deeper sites further away from the kelp beds. The survey was carried out by a team of commercial divers, who sampled underwater transects by scuba diving from a boat.

Sampling of the underwater transects was carried out along a 100 m long weighted rope, marked every metre and positioned roughly perpendicular to the contour of the tidal flat or coastline. The start and end positions of the transects were recorded using GPS. Along each transect, the diver had to take 6 samples at 20 m intervals, i.e., at 0, 20, 40, 60, 80, and 100 m, with the 0 m mark always positioned closest to the contour of the tidal flat or the coast and the 100 m mark towards the open sea. Each sample consisted of the sea urchins present in one to four adjacent quadrats of 0.25 m<sup>2</sup> each. If the first quadrat had fewer than 50 sea urchins, additional quadrats were sampled until at least 50 individuals were obtained. This method made it possible to characterize the size structure of sea urchins but, owing to the variable sampling effort, also introduces a bias if the resulting data are used to calculate sea urchin density.

All sea urchins visible in each quadrat were harvested manually and placed in a diving net with a mesh size of around 15 mm, which could allow the smallest individuals to escape. On board the boat, the sea urchins in each sample were counted, the entire sample weighed, and the test diameters of the sea urchins in at least every other sample were measured. The average density (individuals/m<sup>2</sup>) in each transect was calculated by combining the data from each sample, without taking into account the number of quadrats sampled. The number of sea urchins per individual quadrat was not recorded. The proportion of commercial-size sea urchins ( $\geq 50$  mm) in each transect was calculated in the same way. These two values were multiplied to calculate the density of commercial-size sea urchins. The overall annual mean was then calculated, along with the 95% confidence intervals (95% CIs), by bootstrapping with 1,000 iterations.

The sampling plan and protocol for taking the size structure measurements of the subsamples were not always followed during the surveys. Weather conditions and currents also prevented the complete coverage of the survey area at times. In addition, the positioning of transects was not always as systematic as desired, and was partly dependent on the captain’s knowledge and on winds and currents. The addition or elimination of certain transects over the years of the survey also means that the sampling effort was not temporally and spatially consistent, and the sampling of fixed standard transects was not always followed (Sainte-Marie et al. 2012). The sampling effort along all transects, from transect to transect and from year to year, was inconsistent. The increased effort expended (i.e., the addition of up to three quadrats per sample spaced 20 m apart) when fewer than 50 sea urchins were obtained in the first quadrat biases the density calculations, as these differences in sampling effort are not taken into account. Unfortunately, information was not recorded on the number of specimens counted per 0.25 m<sup>2</sup> quadrat. In addition, the omission of small individuals in the counts, due to the possibility of these individuals falling through the mesh of the diving nets, means that calculations of the density of sublegal size sea urchins are unreliable. For these reasons, only the global mean density of sea urchins above the minimum legal size is presented here, along with the value of the corresponding 95% CI, without regard for the size classes considered in previous advisories (DFO 2022). All statistical analyses were performed with R software, version 4.4.1 (R Core Team 2024).

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## RESULTS AND DISCUSSION

### FISHERY-DEPENDENT DATA

#### Landings and effort

Between 1991 and 2005, annual sea urchin landings in Quebec were low, ranging from 7 t to 188 t (Table 1; Figure 4). Since 2010, annual landings have remained above 400 t, with a peak of 804 t recorded in 2016. Although landings have been recorded in areas 3, 7, 10, 11, 12 and 14 since 1991, they have been sporadic or small. Since 2006, between 88% and ~100% of the cumulative landings in Quebec have come from areas 8 and 9, mainly in subareas 8D, 8E, and 9-1 (Tables 1 and 2, Figures 4 to 7).

In subarea 8D (Table 2; Figure 5), the fishing data from logbooks for the period prior to 2018 are uncertain or have gaps due to logbook irregularities (DFO 2022). Average annual landings between 2022 and 2024 were 119 t, above the 103 t average for the previous period (2018–2021).

Landings in subarea 8E (Table 2; Figure 5) were relatively low and sporadic prior to 2008, but, between 2010 and 2012, increased to the highest values in the time series. Landings declined in 2013 and 2014, but have been fairly stable since 2015, due to the division of area 8 into subareas and the introduction of various management measures. Average landings in subarea 8E during the 2022–2024 period were 103 t, below the historical average of 111 t for the previous period (2003–2021).

In area 9, the majority of annual landings come from subarea 9-1 (Tables 1 and 2; Figures 4 and 6). Although annual landings between 1996 and 2003 were low, they subsequently increased significantly to a peak of 685 t in 2007. Afterward, they declined sharply, but began to increase again in 2009. Since that time, they have remained relatively stable at around 300–400 t. Average annual landings in subarea 9-1 during the 2022–2024 period were 306 t, below the average of 325 t for the previous period (2004–2021).

In area 10, landings have been very sporadic and low, with values below 5 t in the years from 1996 to 1999, in 2007 and in 2023–2024. Landings were also sporadic and low in area 11 between 1991 and 2015 (Tables 1 and 2; Figures 4 and 7). Landings peaked at almost 58 t in 2016 and then declined, remaining around 40 t until 2019. However, they have fallen sharply since then. Average annual landings from 2022 to 2023 (no fishing in 2024) were 14 t, up from the reference average of 12 t (2002–2021).

Total effort in diver-hours in subareas 8D, 8E and 9-1 and in area 11 is shown in Table 3.

#### Catch per unit effort

In subareas 8D and 8E, CPUE values have been increasing since 2019–2020. During the 2022–2024 period, the average annual CPUE value was 343 kg/dh in subarea 8D and 210 kg/dh in subarea 8E, which are comparable to the annual average of 316 kg/dh for the respective reference period in 8D (2018–2021) and 208 kg/dh in 8E (2003–2021) (Table 4; Figures 8 and 9).

In subarea 9-1, CPUE values were high between 2010 and 2017, and subsequently declined. CPUE values have been rising since 2020. The average CPUE for 2022–2024 (356 kg/dh) was comparable to the reference average (367 kg/dh; 2004–2021) (Table 4; Figure 10).

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Lastly, in area 11, annual CPUE values have been increasing since 2016. The average annual CPUE value during the 2022–2023 period (275 kg/dh) is higher than the reference average (206 kg/dh; 2002–2021) (Table 4; Figure 11).

### **Size structure**

The size structure of green sea urchins in subareas 8D, 8E and 9-1 showed little interannual variations, with no clear trends observed. No multimodal size distributions have been observed, which prevents cohort tracking. The proportion of sublegal-size sea urchins (< 50 mm) in landings averages less than 5% (Figure 12). Average sizes in 2022 were 63 mm, 61 mm and 65 mm in subareas 8D, 8E and 9-1, respectively (Figure 13); in 2023, 64 mm, 60 mm and 62 mm in 8D, 8E and 9-1, respectively (Figure 14); and, in 2024, 65 mm, 60 mm and 65 mm in 8D, 8E and 9-1, respectively (Figure 15). The number of samples and sea urchins measured are shown in Table 5.

### **FISHERY-INDEPENDENT DATA**

Average sizes have been relatively stable throughout the study area over the time series (Figure 16). Mean densities of legal-size sea urchins (> 50 mm) have been high and relatively stable since 2016 (Figures 17 and 18) based on the dataset for subarea 9-1. The high values for the standard deviation and 95% confidence interval (95% CI) for these two indices reflect the aggregation behaviour of sea urchins, as well as the lack of consistency in following the sampling plan over time.

### **ADVICE ON STOCK STATUS**

The absence of indicators and reference points for sea urchin stocks prevents the assessment of their status as per the Fishery Decision-Making Framework Incorporating the Precautionary Approach.

The commercial CPUE, calculated from logbook data, varies according to the management measures implemented, weather conditions, harvesters' experience, and market factors. It therefore represents an index of fishery performance, and does not appear to be a good indicator of stock status. It could, however, shed light on the risk of local depletion when combined with information on the spatial distribution of fishing activities and the size structure of commercial catches.

### **SOURCES OF UNCERTAINTY**

The quality of the data acquired from the logbooks is uncertain. For all fishing areas, information on landings and the duration of dives is not always recorded for each individual dive for each diver but aggregated by diver or fishing trip. In addition, a duplicate values for landings and effort occur regularly in logbooks. For example, the same landings and effort values may be recorded for several trips on the same vessel, or the data for a single dive may be duplicated for all or part of the fishing trip (e.g., instead of having 16 separate values for a fishing trip, with four divers making four dives each, the logbooks show the same values 16 times). The presence of these duplicate values in the data used to calculate annual indices and trends makes it difficult to interpret these results. Notably, there is a risk of introducing significant bias and of masking the true variance in population dynamics (Sarracino and Mikucka 2017), which can make it difficult to detect trends in stock productivity and to assess the real impact of the fishery. The significant presence of duplicates in the source data could explain, in part, the low 95% CI values for the average annual CPUE (Figures 8 to 11).

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Catch rates (CPUE) used to assess stock status can be influenced by a range of factors, including harvesters' behaviour, technological advances, management measures, market factors, and environmental variations. In addition, the phenomenon of hyperstability can keep CPUE values artificially high despite a real decline in abundance. For these reasons, CPUE should be considered an index of fishery performance rather than a reliable indicator of stock status (Miller and Nolan 2000; Chen and Hunter 2003). Aside from the well-known problems with the interpretation and usefulness of commercial CPUE values (Hilborn and Walters 1992; Hoyle et al. 2024), data from the sea urchin fishery do not respect one of the underlying assumptions made when CPUE is used as a stock status indicator—namely the random sampling of the population. Harvesters target a very limited portion of the sea urchin population, concentrated in areas of dense aggregations, and make a selection of individuals. This behaviour increases the risk of hyperstability in this index.

Cases of hyperstability in CPUE values have been documented in regions adjacent to the EGSL, where sea urchin populations have collapsed without this decline being reflected in the CPUE. For example, in Maine (United States), landings fell after 1992 due to overfishing, but CPUE values remained stable despite the decline in the stock (Chen and Hunter 2003; Johnson et al. 2012).

In Nova Scotia, mass mortality events caused by the pathogenic amoeba *Paramoeba invadens* have been observed. In the early 1980s, this pathogen is thought to have caused the death of around 270 kilotonnes of sea urchins, or 10 to 100 times the biomass harvested in the fishery between 1995 and 2000. Despite this high mortality, CPUE values remained stable (Miller and Nolan 2000; Ling et al. 2015; Scheibling et al. 2020).

CPUE should be considered an index of fishery performance rather than a reliable indicator of stock status (Miller and Nolan 2000; Chen and Hunter 2003). However, when combined with information on the spatial distribution of fishing activities and the size structure of commercial catches, CPUE can help in assessing the risk of local depletion.

No data are available on sea urchin bycatch in other directed fisheries, or on the impacts of discards at sea. The mortality associated with these activities, including discards from all sources, remains unknown. The extent and impact of removals under provincial licences are also uncertain. The exploitation of wild sea urchins in the provincial fishery (i.e., in aquaculture) is another source of uncertainty surrounding the total catches in the EGSL and the reliability of CPUE values.

Rates of natural mortality ( $M$ ) and fishing mortality ( $F$ ); recruitment; demographic structures in the population; and indices of relative abundance are not known for green sea urchins in the EGSL due to a lack of data.

Lastly, the extent and magnitude of the overlap in distribution of, and the hybridization between, the green sea urchin (*S. droebachiensis*) and the pale sea urchin (*S. pallidus*) in the Estuary and Gulf of St. Lawrence are unknown (Swan 1962; Addison and Hart 2005; Addison and Kim 2018, 2022).

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## TABLES

*Table 1. Annual landings (t) of sea urchins by sea urchin fishing area in the Quebec Region. N/A column: information not available.*

<b>Year</b>	<b>3</b>	<b>7</b>	<b>8</b>	<b>9</b>	<b>10</b>	<b>11</b>	<b>12</b>	<b>14</b>	<b>N/A</b>	<b>Total</b>
1991	-	-	-	-	-	6.69	-	-	-	6.69
1994	-	-	-	-	-	13.61	-	-	-	13.61
1995	-	-	116.59	-	-	4.65	-	-	0.45	5.10
1996	-	-	59.25	34.92	0.29	3.04	-	-	1.40	98.90
1997	0.28	3.83	61.22	20.26	4.93	23.87	-	28.37	16.46	159.22
1998	-	-	5.92	0.74	0.24	1.21	1.01	-	0.68	9.80
1999	-	-	15.63	0.50	0.28	1.59	-	-	3.63	21.63
2000	-	-	9.64	5.63	-	-	-	-	3.99	19.26
2001	-	-	12.71	11.36	-	-	-	-	0.09	24.16
2002	-	-	40.83	0.91	-	0.48	-	-	5.08	47.30
2003	-	-	92.63	49.55	-	1.32	-	-	2.22	145.72
2004	-	-	73.46	111.60	-	0.72	-	-	2.38	188.16
2005	-	-	43.63	132.93	-	3.58	-	-	2.34	182.48
2006	-	-	69.52	599.52	-	14.61	-	-	6.21	689.86
2007	-	-	63.34	685.14	2.00	6.21	-	-	4.52	761.21
2008	-	-	179.79	249.65	-	17.60	-	-	3.17	450.21
2009	-	-	186.50	153.31	-	0.93	-	-	5.00	345.74
2010	-	-	340.04	266.64	-	0.53	-	-	8.08	615.29
2011	-	-	259.60	402.14	-	0.81	-	-	7.06	669.61
2012	-	-	287.47	398.60	-	0.57	-	-	20.09	706.73
2013	-	-	150.88	423.28	-	0.25	-	-	35.49	609.90
2014	-	-	67.10	427.92	-	0.23	-	-	67.14	562.39
2015	-	-	173.03	416.06	-	0.56	-	-	17.12	606.77
2016	-	-	255.41	472.26	-	58.33	-	-	18.52	804.52
2017	-	-	217.04	421.23	-	35.36	-	-	13.72	687.35

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<b>Year</b>	<b>3</b>	<b>7</b>	<b>8</b>	<b>9</b>	<b>10</b>	<b>11</b>	<b>12</b>	<b>14</b>	<b>N/A</b>	<b>Total</b>
2018	-	-	184.17	291.33	-	37.44	-	-	11.82	524.76
2019	-	-	228.39	299.87	-	52.64	-	-	14.84	595.74
2020	-	-	162.12	245.01	-	6.62	-	-	8.95	422.70
2021	-	-	260.27	315.27	-	4.34	-	-	6.16	586.04
2022	-	-	214.53	367.56	-	11.13	-	-	6.28	599.50
2023	-	-	216.19	392.26	0.08	17.25	-	-	7.03	632.81
2024	-	-	234.96	385.82	0.48		-	-	-	621.26

*Table 2. Annual landings (t) of sea urchins in the most heavily fished sea urchin fishing areas and subareas in the Quebec Region, by area or subarea.*

Year	8	8D*	8E	9	9-1	11	N/A
1991	-	-	-	-	-	6.69	-
1994	-	-	-	-	-	13.61	-
1995	65.06	51.53	-	-	-	4.65	0.45
1996	59.25	-	-	34.92	-	3.04	1.40
1997	61.22	-	-	19.86	0.39	23.87	16.46
1998	1.08	4.84	-	0.74	-	1.21	0.68
1999	15.63	-	-	0.50	-	1.59	3.63
2000	9.64	-	-	5.63	-	-	3.99
2001	12.71	-	-	11.36	-	-	0.09
2002	-	40.83	-	0.91	-	0.48	5.08
2003	-	15.56	77.07	49.55	-	1.32	2.22
2004	2.16	52.41	18.89	48.00	63.60	0.72	2.38
2005	-	22.29	21.34	24.93	107.99	3.58	2.34
2006	-	69.52	-	0.47	599.05	14.61	6.21
2007	1.86	61.48	-	0.49	684.66	6.21	4.52
2008	-	93.08	86.70	31.72	217.93	17.60	3.17
2009	1.59	41.46	143.45	10.68	142.63	0.93	5.00
2010	-	105.04	235.00	-	266.64	0.53	8.08
2011	-	52.39	207.21	3.86	398.29	0.81	7.06
2012	-	53.82	233.64	0.29	398.31	0.57	20.09
2013	-	40.63	110.25	-	423.28	0.25	35.49
2014	-	50.62	16.48	8.54	419.38	0.23	67.14
2015	-	85.75	87.28	0.38	415.68	0.56	17.12
2016	-	132.47	122.94	17.78	454.48	58.33	18.52
2017	-	110.59	106.45	85.56	335.67	35.36	13.72
2018	-	78.60	105.58	53.16	238.17	37.44	11.82
2019	-	117.99	110.40	-	299.87	52.64	14.84
2020	-	67.69	94.42	4.04	240.97	6.62	8.95
2021	-	149.58	110.68	-	315.27	4.34	6.16
2022	-	110.46	104.06	41.70	325.85	11.13	6.28
2023	-	119.95	96.23	106.18	286.08	17.25	7.03
2024	-	126.00	108.96	78.75	307.07	-	-

\* Values for subarea 8D prior to 2018 are considered uncertain.

*Table 3. Total annual fishing effort (diver-hours) in the sea urchin fishery in subareas 8D, 8E and 9-1 and area 11.*

<b>Year</b>	<b>8D*</b>	<b>8E</b>	<b>9-1</b>	<b>11</b>
2002	215.00	-	-	3.00
2003	60.00	312.00	-	11.73
2004	234.50	69.00	208.00	2.67
2005	114.00	86.00	325.00	7.00
2006	286.88	-	1555.25	69.82
2007	143.83	-	2640.52	34.10
2008	233.17	211.50	718.05	58.27
2009	174.50	439.40	518.13	5.88
2010	216.00	992.03	649.67	2.58
2011	439.42	952.00	855.92	4.43
2012	828.00	1271.00	792.30	2.88
2013	87.33	576.50	962.78	1.52
2014	17.50	104.92	1039.12	0.97
2015	-	628.28	939.17	2.92
2016	-	777.30	1063.30	299.33
2017	-	679.63	857.17	160.82
2018	160.00	814.62	760.25	188.92
2019	416.67	837.47	985.30	234.33
2020	296.75	527.17	864.63	29.67
2021	593.33	575.00	1080.53	19.42
2022	349.50	550.62	898.33	40.92
2023	316.00	416.58	899.88	68.33
2024	362.00	521.02	882.62	-

\* Values for subarea 8D prior to 2018 are considered highly uncertain.

*Table 4. Annual average catch per unit effort (CPUE) (kg/diver-hours), an indicator of the performance of the sea urchin fishery, in subareas 8D, 8E and 9-1 and area 11.*

<b>Year</b>	<b>8D*</b>	<b>8E</b>	<b>9-1</b>	<b>11</b>
2002	242.05	-	-	105.99
2003	267.42	259.98	-	167.97
2004	221.48	283.35	311.21	272.38
2005	193.35	222.98	355.47	136.64
2006	301.12	-	329.99	172.15
2007	427.78	-	297.20	182.65
2008	396.22	405.32	315.46	314.18
2009	239.52	327.46	285.01	193.89
2010	462.77	285.35	406.71	226.76
2011	187.95	224.99	466.72	179.83
2012	65.00	178.97	499.02	227.66
2013	390.53	179.03	445.86	172.11
2014	817.32	161.81	398.76	225.19
2015	-	135.49	443.84	236.98
2016	-	133.61	433.34	196.08
2017	-	141.85	394.59	219.77
2018	502.17	124.78	318.89	207.06
2019	287.47	116.23	312.65	232.26
2020	229.14	164.04	291.40	229.21
2021	245.69	190.26	299.46	215.24
2022	314.41	188.27	368.92	292.54
2023	370.90	231.31	342.52	257.34
2024	343.44	210.09	355.83	-

\* Values for subarea 8D prior to 2018 are considered highly uncertain.

Table 5. Statistics on the number of samples (N) and number of specimens (n) obtained in dockside sampling and used to determine sea urchin size structures, by fishing area or subarea. Numbers could not be determined for subareas 8D, 8E and 9-1 before they were established, as samples could not be associated with the correct subarea.

Year	8		8D		8E		9		91		11		Total N	Total n
	N	n	N	n	N	n	N	n	N	n	N	n		
1996	13	3625	-	-	-	-	4	1208	-	-	1	217	18	5050
1997	4	1053	-	-	-	-	-	-	-	-	9	2007	13	3060
1998	9	2328	-	-	-	-	-	-	-	-	-	-	9	2328
1999	11	2897	-	-	-	-	-	-	-	-	1	250	12	3147
2000	7	1841	-	-	-	-	2	557	-	-	-	-	9	2398
2001	12	3265	-	-	-	-	2	366	-	-	-	-	14	3631
2002	11	2992	-	-	-	-	-	-	-	-	-	-	11	2992
2003	12	3419	-	-	-	-	1	253	-	-	-	-	13	3672
2004	14	3705	-	-	-	-	3	692	-	-	-	-	17	4397
2005	10	2430	-	-	-	-	2	322	-	-	-	-	12	2752
2006	9	1837	-	-	-	-	12	1854	-	-	-	-	21	3691
2007	9	2321	-	-	-	-	19	3012	-	-	-	-	28	5333
2008	9	1489	-	-	-	-	14	2200	-	-	-	-	23	3689
2009	10	1696	-	-	-	-	6	912	-	-	-	-	16	2608
2010	20	3166	-	-	-	-	-	-	12	1860	-	-	32	5026
2011	11	1706	-	-	-	-	-	-	13	1969	-	-	24	3675
2012	-	-	8	1266	6	902	-	-	19	2841	-	-	33	5009
2013	-	-	6	902	6	900	-	-	11	1657	-	-	23	3459
2014	-	-	4	900	2	300	1	163	8	1431	-	-	15	2794
2015	-	-	6	1521	5	978	-	-	8	1238	-	-	19	3737
2016	-	-	8	1701	6	1201	-	-	8	1274	-	-	22	4176
2017	-	-	6	1727	7	1550	-	-	6	949	-	-	19	4226
2018	-	-	6	1800	6	1300	-	-	10	1565	-	-	22	4665
2019	-	-	9	2251	5	751	-	-	6	951	1	150	21	4103

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Year	8		8D		8E		9		91		11		Total N	Total n
	N	n	N	n	N	n	N	n	N	n	N	n		
2020	-	-	7	1050	4	600	-	-	5	784	-	-	16	2434
2021	-	-	8	1200	3	450	-	-	2	305	-	-	13	1955
2022	-	-	8	1616	6	1045	1	150	3	454	-	-	18	3265
2023	-	-	5	776	6	974	-	-	8	1219	-	-	19	2969
2024	-	-	10	1540	7	1080	-	-	7	974	-	-	24	3594

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## FIGURES



*Figure 1. Green sea urchin (Strongylocentrotus droebachiensis). Photo credit: Claude Nozères.*

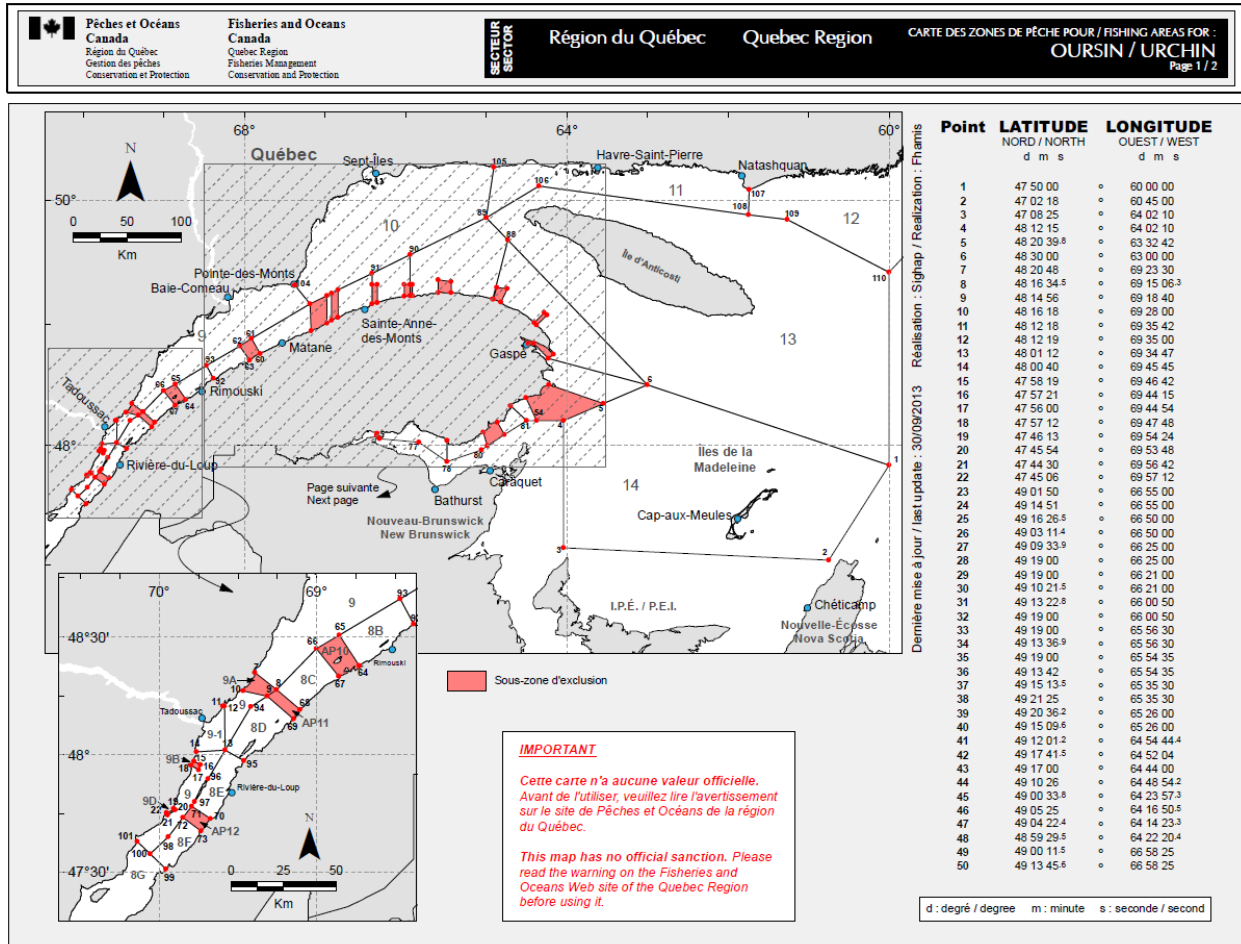


Figure 2. Map of the 14 sea urchin fishing areas in the Quebec Region, including subareas, exclusion subareas (in red) and the geographical coordinates used to delineate their boundaries.

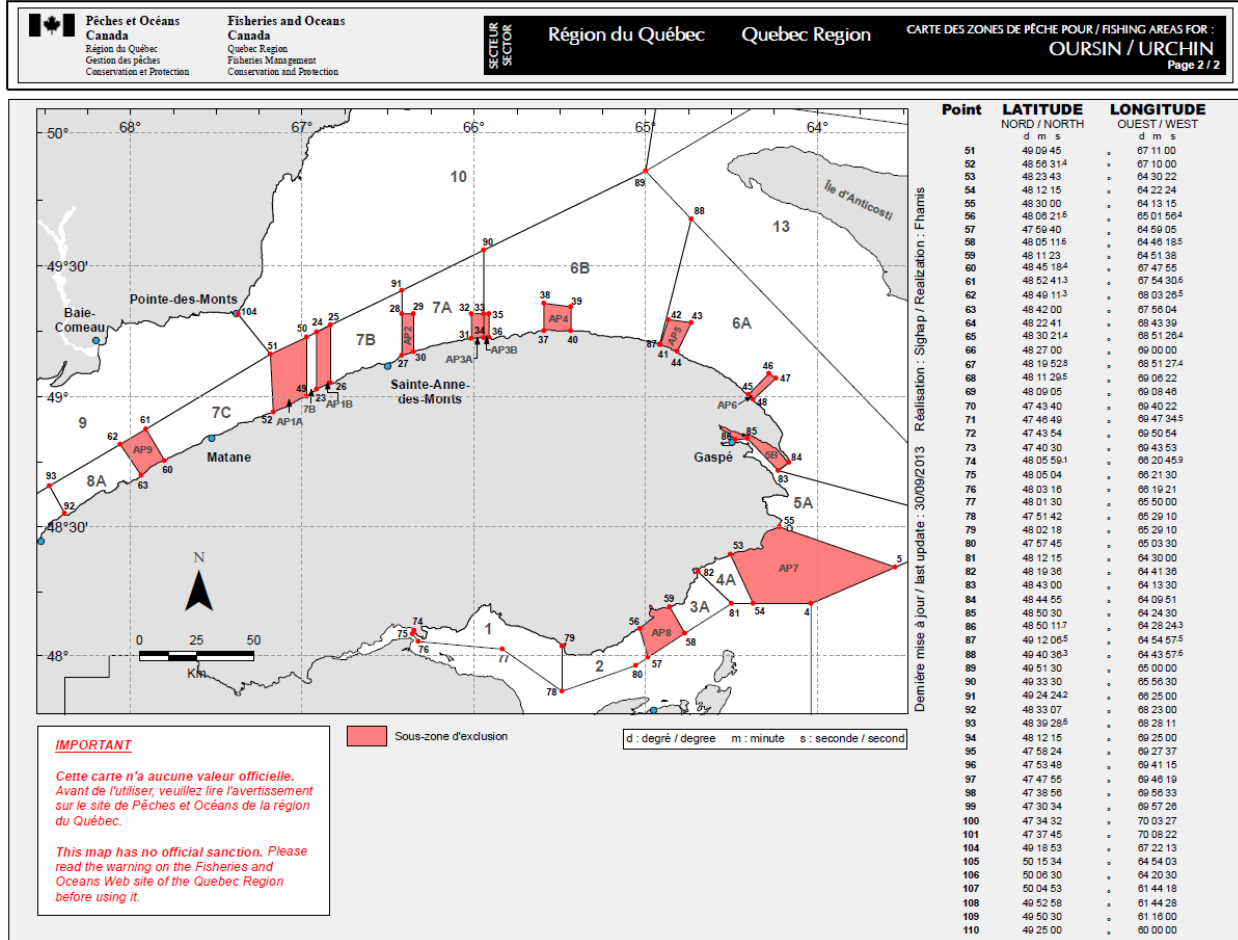


Figure 3. Enlarged map of sea urchin fishing areas around the Gaspé Peninsula in the Quebec Region, including subareas, exclusion subareas (in red) and the geographical coordinates used to delineate their boundaries.

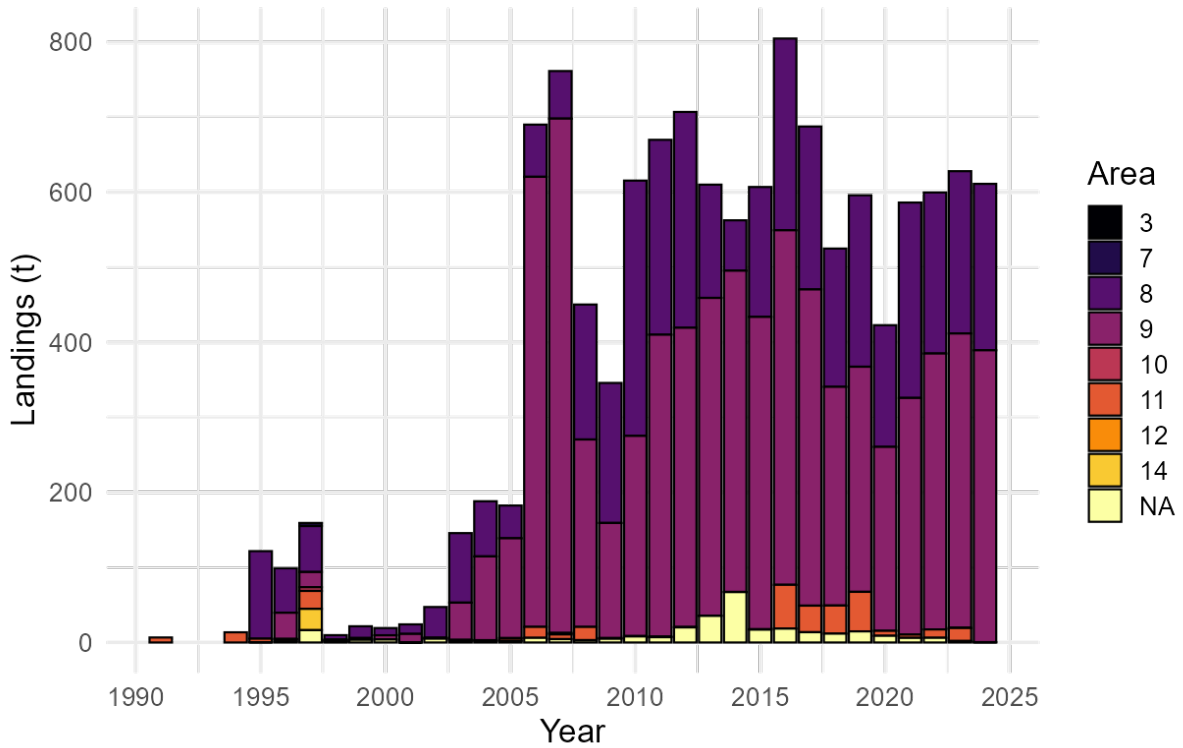


Figure 4. Annual landings (t) of green sea urchins in sea urchin fishing areas in the Quebec Region. NA = not attributable.

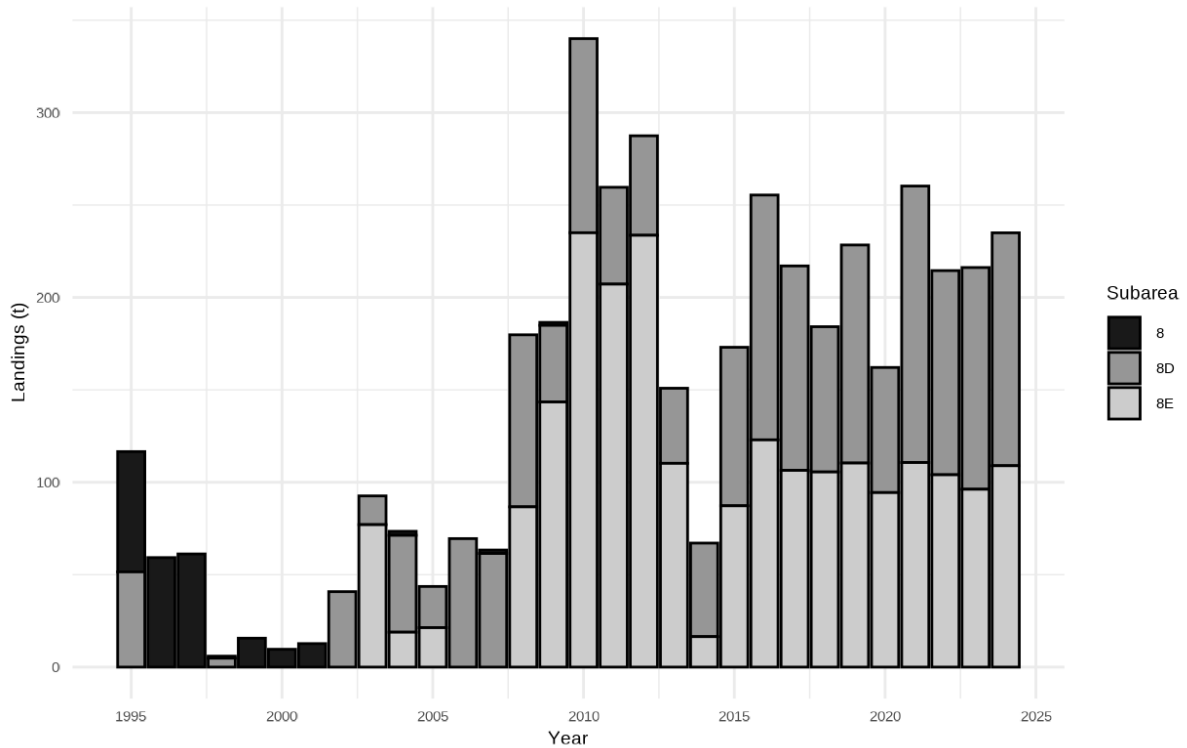


Figure 5. Annual landings (t) of green sea urchins in area 8 (black), subarea 8D (dark grey) and subarea 8E (light grey).

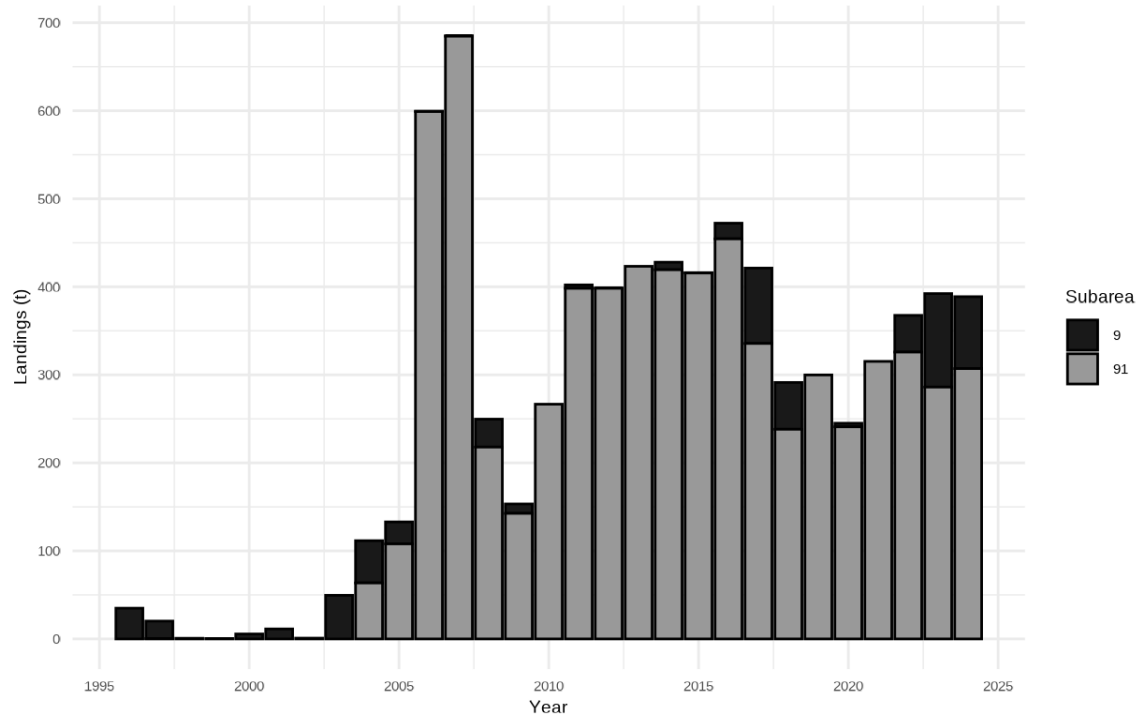


Figure 6. Annual landings (t) of green sea urchins in area 9, broken down into subarea 9-1 (grey) and the rest of area 9 (black).

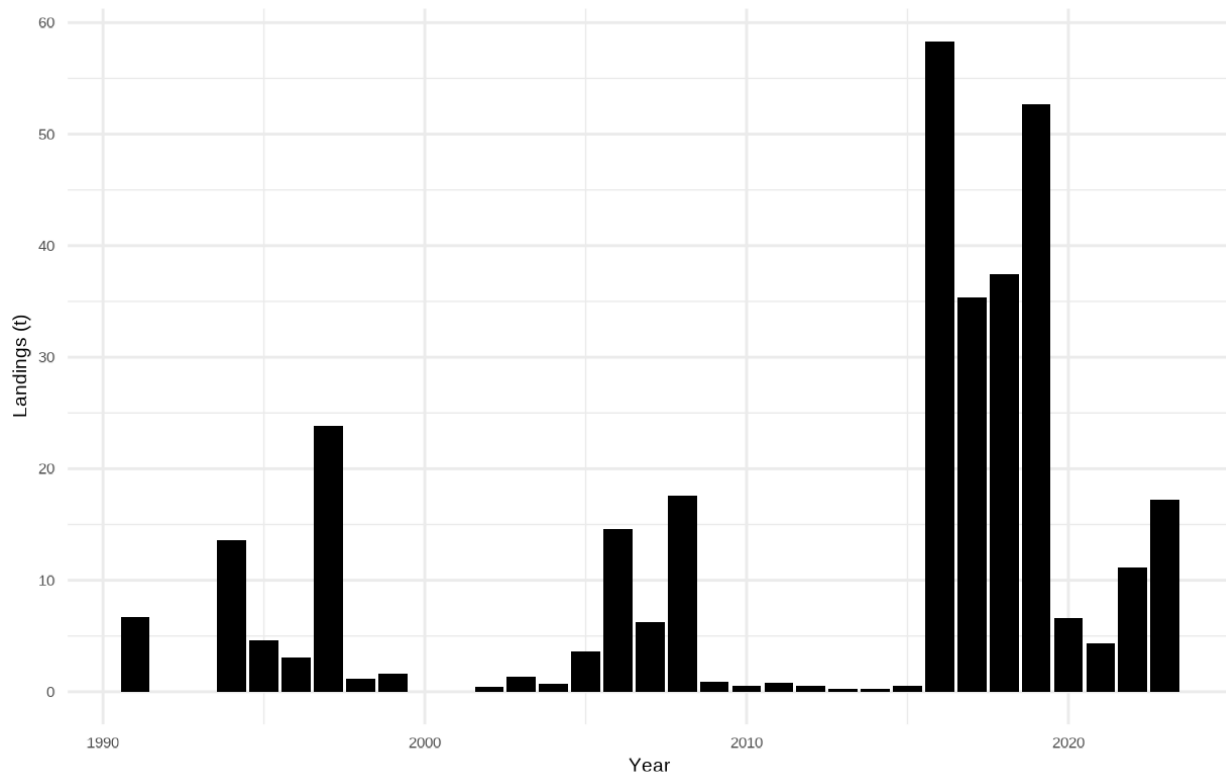


Figure 7. Annual landings (t) of green sea urchins in area 11.

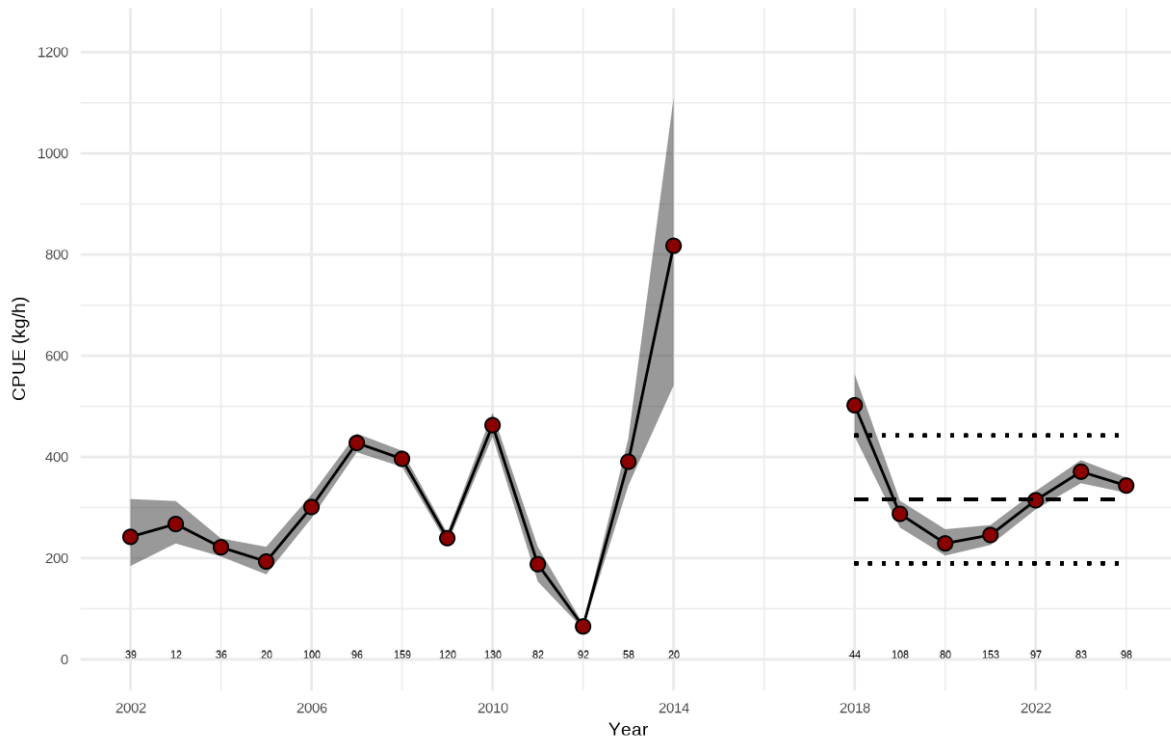


Figure 8. Catch per unit effort (CPUE: kg/dh; circles) and the associated 95% CI (grey band) in subarea 8D. The dashed horizontal lines represent the historical average for the CPUE, and its associated standard deviation, for the 2018–2021 reference period.

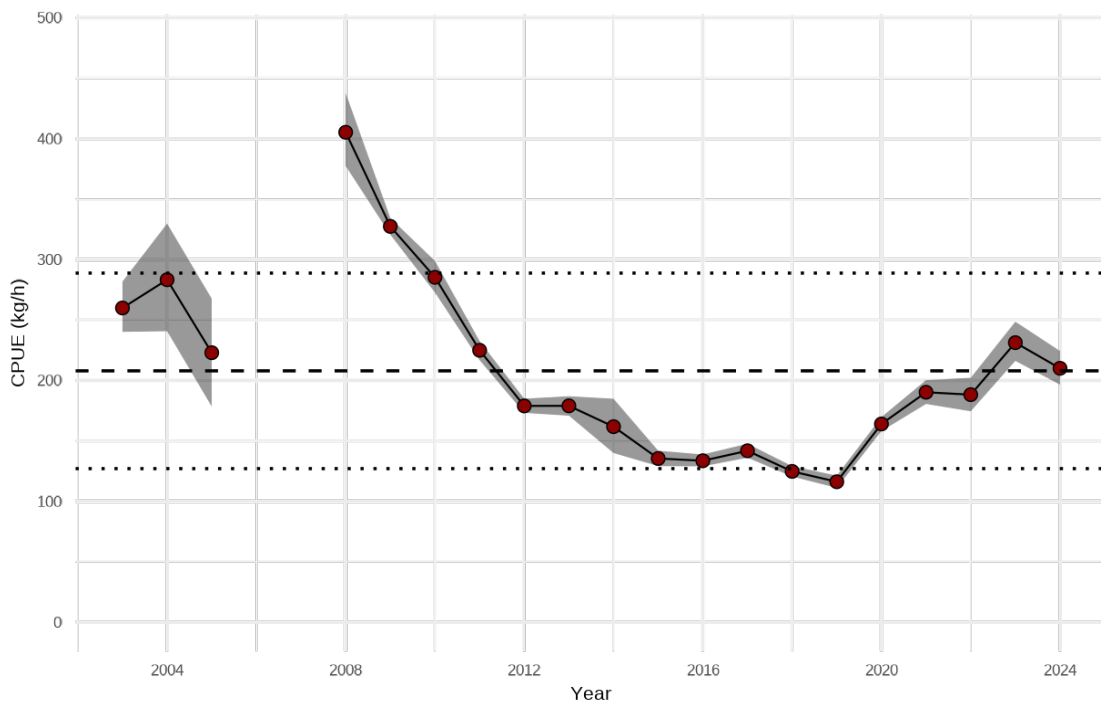


Figure 9. Catch per unit effort (CPUE: kg/dh; circles) and the associated 95% CI (grey band) in subarea 8E. The dashed horizontal lines represent the historical average for the CPUE, and its associated standard deviation, for the 2003–2021 reference period.

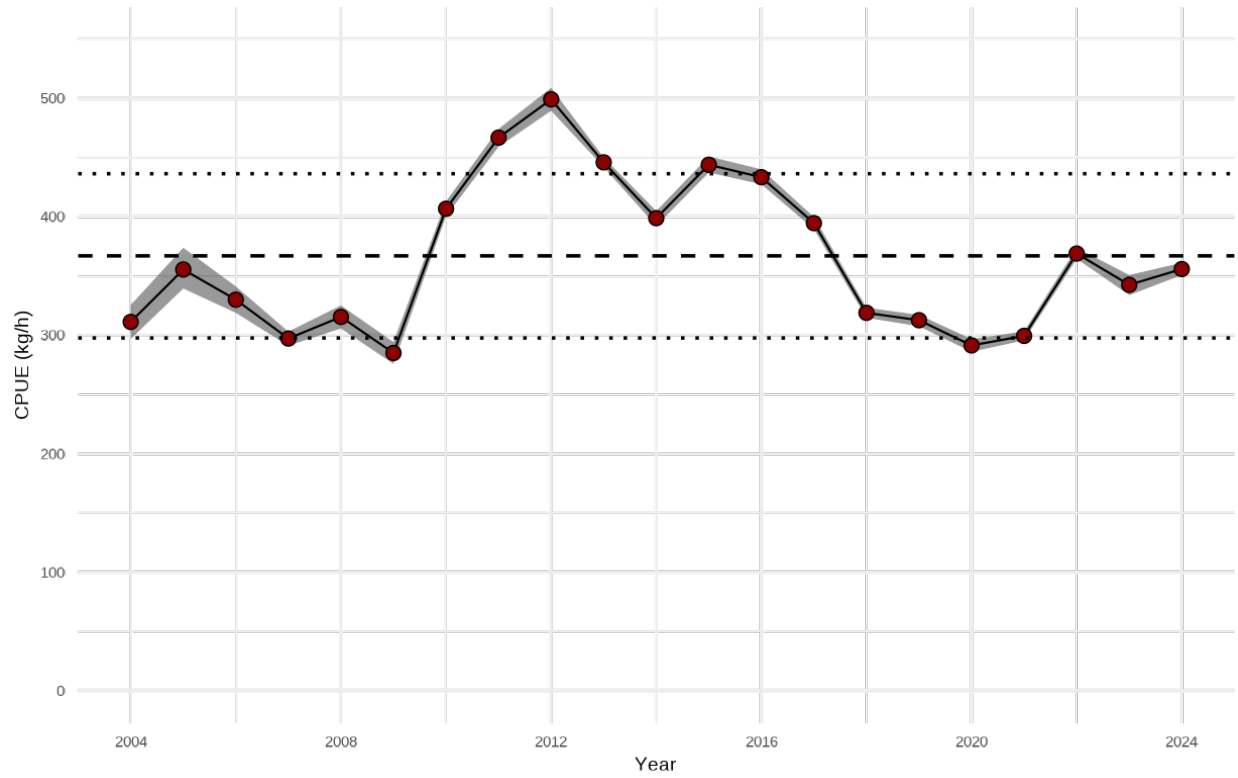


Figure 10. Catch per unit effort (CPUE: kg/dh; circles) and the associated 95% CI (grey band) in subarea 9-1. The dashed horizontal lines represent the historical average for the CPUE, and its associated standard deviation, for the 2004–2021 reference period.

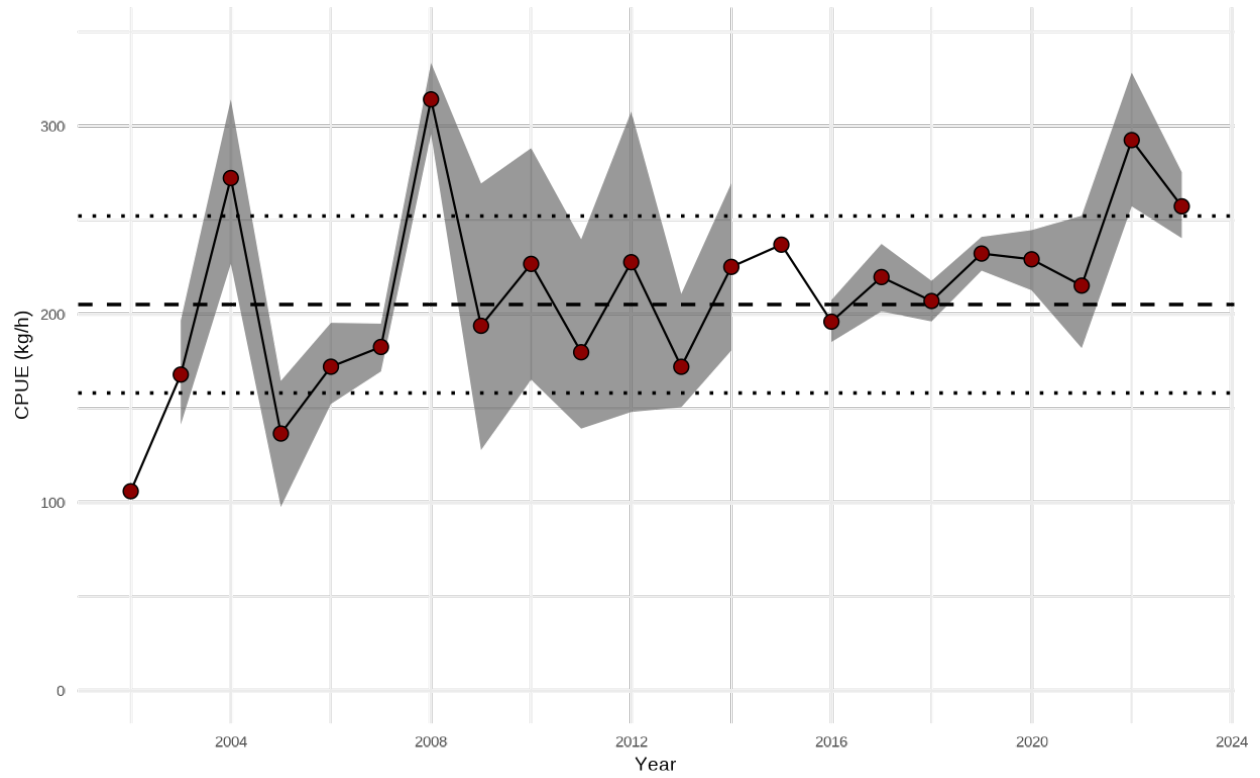


Figure 11. Catch per unit effort (CPUE: kg/dh; circles) and the associated 95% CI (grey band) in area 11. The dashed horizontal lines represent the historical average for the CPUE, and its associated standard deviation, for the 2002–2021 reference period.

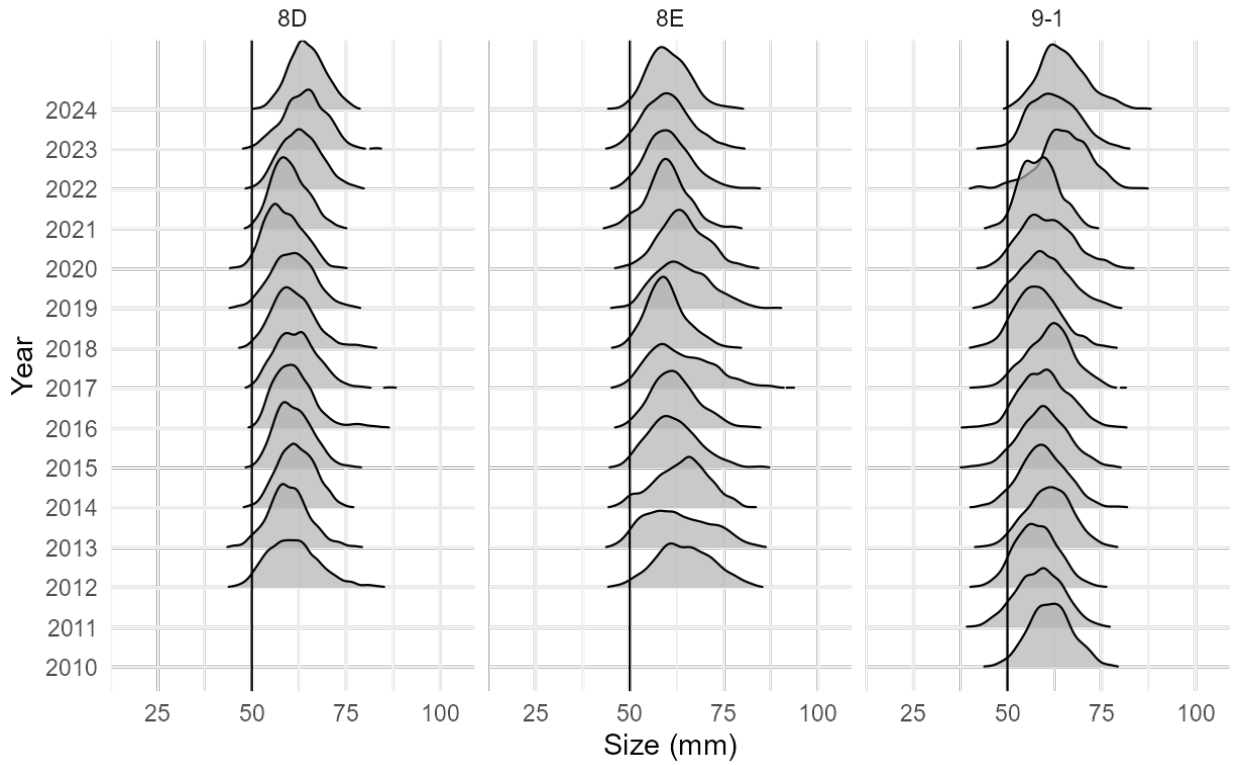


Figure 12. Size distribution (test diameter in mm) of sea urchins measured in DFO's dockside sampling program, by fishing area and subarea. The vertical black line indicates the commercial size (50 mm).

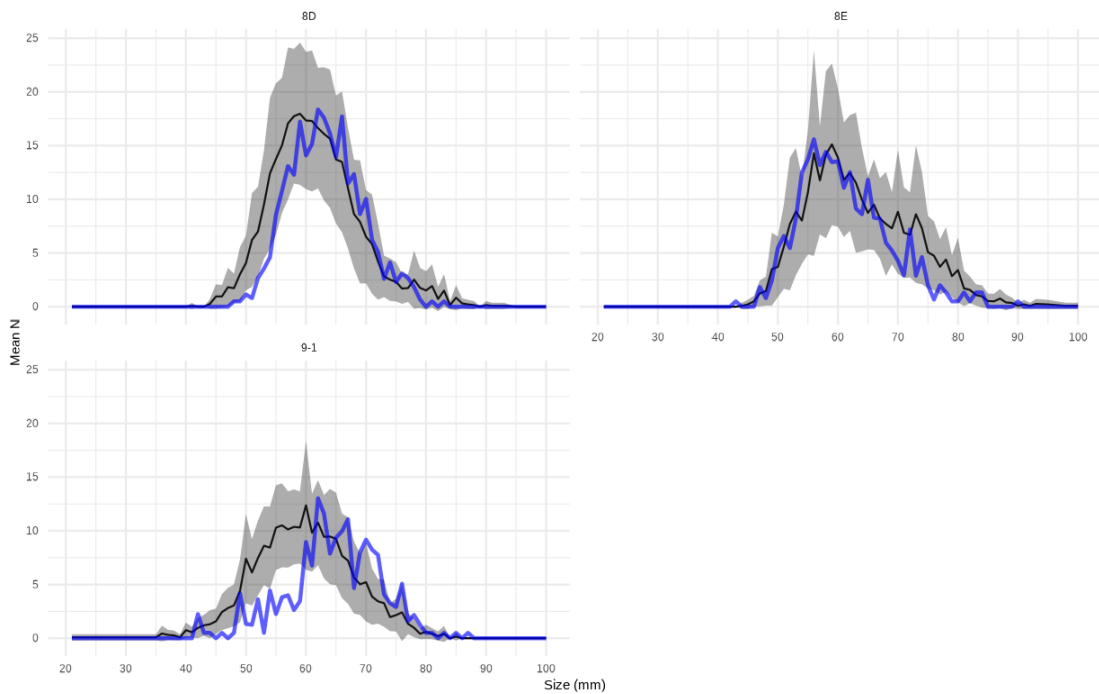


Figure 13. Historical average (2010–2021) size (black lines), and the associated standard deviation (grey band), of test diameters (1 mm classes) of green sea urchins measured in DFO's dockside sampling program in subareas 8D, 8E and 9-1 (panels). The average numbers in 2022 are shown in blue.

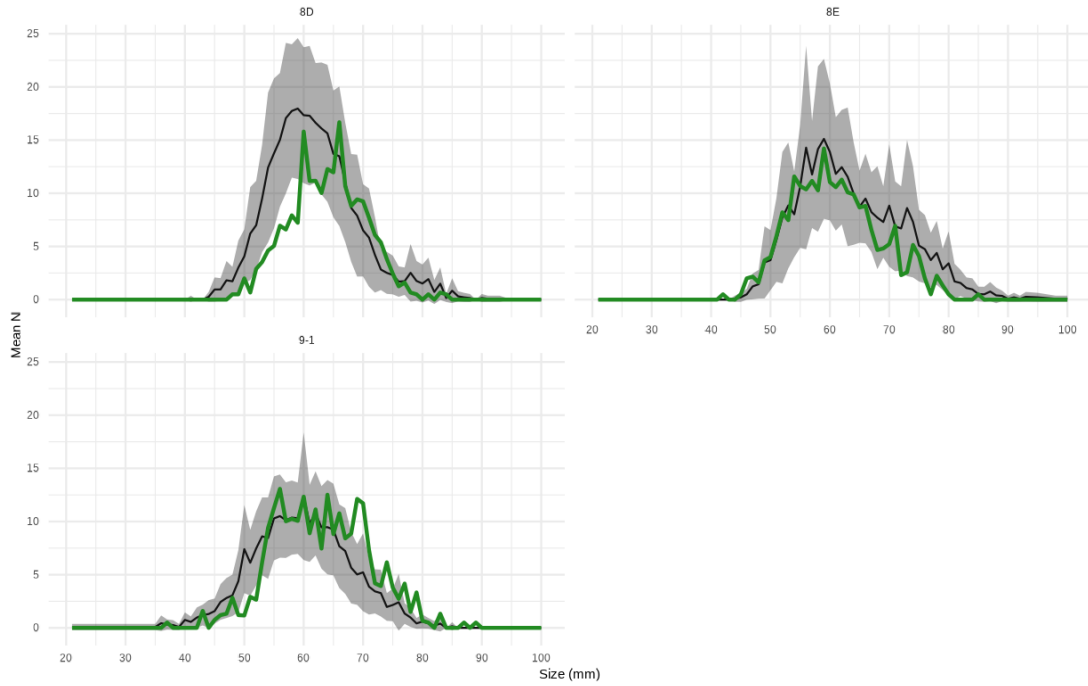


Figure 14. Historical average (2010–2021) size (black lines), and the associated standard deviation (grey band), of test diameters (1 mm classes) of green sea urchins measured in DFO’s dockside sampling program in subareas 8D, 8E and 9-1 (panels). The average numbers in 2023 are shown in green.

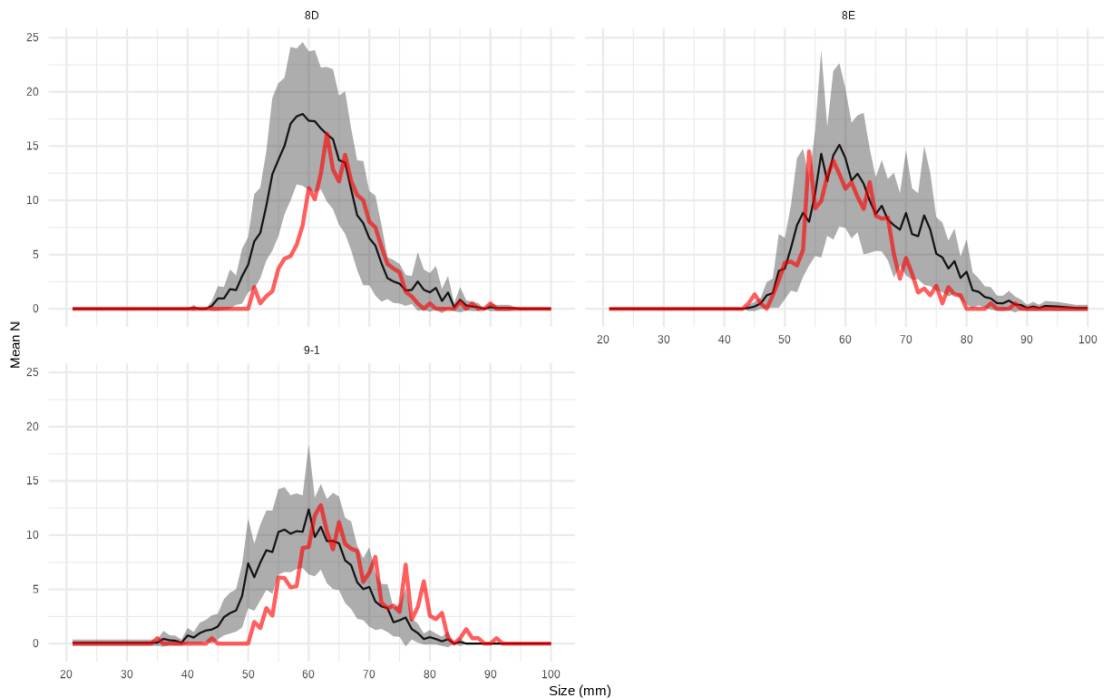


Figure 15. Historical average (2010–2021) size (black lines), and the associated standard deviation (grey band), of test diameters (1 mm classes) of green sea urchins measured in DFO’s dockside sampling program in subareas 8D, 8E and 9-1 (panels). The average numbers in 2024 are shown in red.

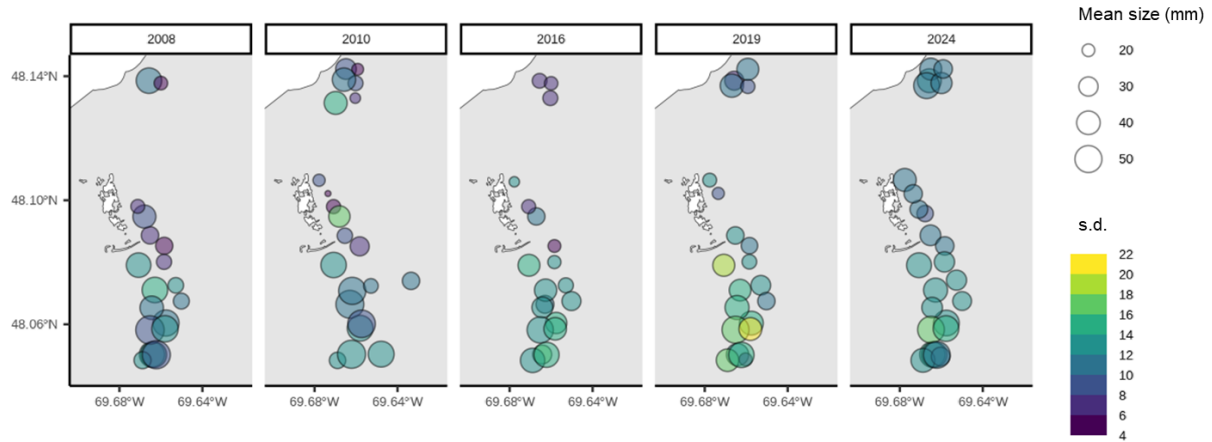


Figure 16. Positions of standard transects sampled during the research survey at the mouth of the Saguenay River in subarea 9-1. The size of the circles represents the average size of the sea urchins sampled and the colour of the circle, the associated standard deviation.

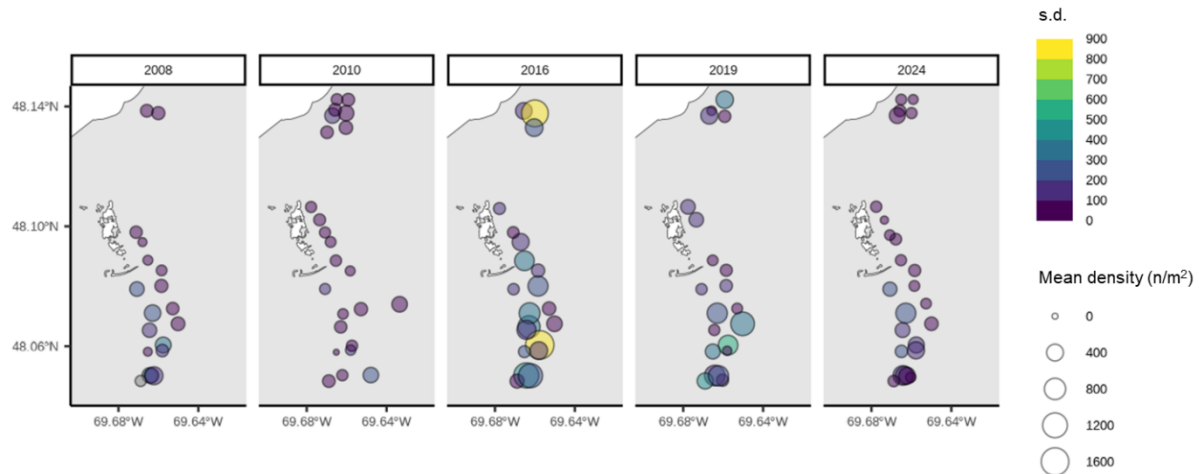


Figure 17. Positions of standard transects sampled during the research survey at the mouth of the Saguenay River in subarea 9-1. The size of the circle represents the average density (n/m<sup>2</sup>) of sea urchins and the colour of the circle, the associated standard deviation.

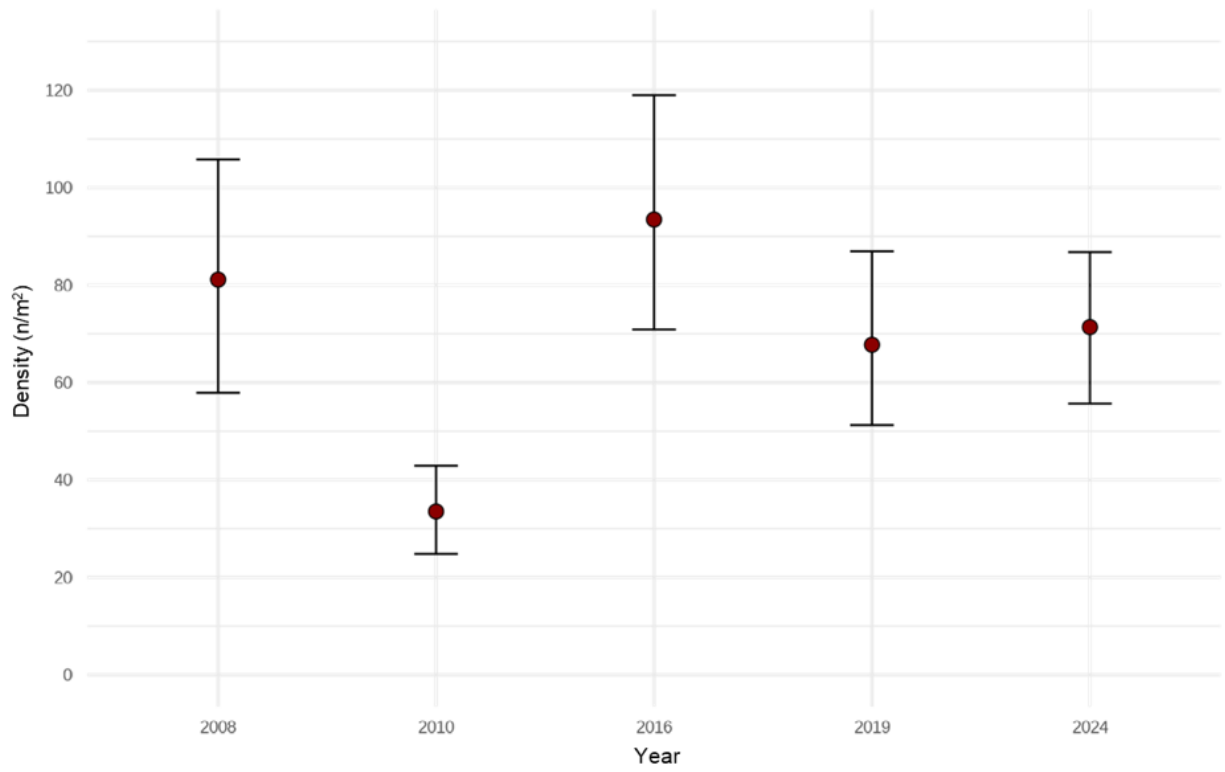


Figure 18. Density (number/m<sup>2</sup>) of sea urchins larger than the legal size of 50 mm and the associated 95% CI.