

Using community science to assess the population dynamics and spatial ecology of the giant sea bass in Southern California

Andrew T. Pettit^{1,2,*}, Miranda Haggerty³, Molly R. Morse⁴, Ryan Freedman⁵, Francis Joyce⁶, Conner Jainese¹, Kaitlin Seeto¹, Jennifer E. Caselle¹, Douglas J. McCauley^{1,2}

¹Marine Science Institute, University of California, Santa Barabara, Santa Barbara CA 93106, USA
²Ecology, Evolution, and Marine Biology Department, University of California, Santa Barbara, Santa Barbara, CA 93106, USA
³California Department of Fish and Wildlife, South Coast Region, San Diego, CA 92123, USA
⁴Benioff Ocean Science Laboratory, Marine Science Institute, University of California, Santa Barbara, Santa Barbara, Santa Barbara, Santa Barbara, Santa Barbara, Santa Barbara, CA 93106, USA
⁵Channel Islands National Marine Sanctuary, National Oceanic and Atmospheric Administration, University of California, Santa Barbara, Santa Barbara, Santa Barbara, CA 93106, USA
⁶Environmental Studies Department, University of California, Santa Cruz, CA 95064, USA

ABSTRACT: The giant sea bass Stereolepis gigas is the largest marine teleost along the coast of California, USA, and is an important apex predator in kelp forests and rocky reef ecosystems. Due to their slow-growing nature and tendency to form predictable, seasonal aggregations, S. gigas were heavily overfished across much of their range from Humboldt Bay, California, to Oaxaca, Southern Mexico, and in the Gulf of California. Now listed as Critically Endangered by the IUCN and partially protected by a fishing moratorium in California, there is some evidence that the California population has begun to recover. However, there has never been a direct population assessment of their numbers in California. Using more than 1600 community-sourced photos of S. gigas from the Spotting Giant Sea Bass Project, we identified individuals through unique spot patterns on their flanks by using pattern-matching algorithms. Estimates from a POPAN mark-recapture model suggest a total population of ~1221 adult individuals in Southern California from 2015 to 2022, and a Pradel mark-recapture model indicates an increasing population trend, corroborated by a rise in incidental catch of S. gigas in set gill net landing receipts. We also found evidence of minimal population connectivity throughout Southern California with high site fidelity amongst resignted individuals. The contribution of this open and living resource of photo-identification data creates the capacity to answer critical questions regarding the life history, behavior, and population structure of *S. qiqas*, which will inform future recovery efforts for this at-risk species.

KEY WORDS: Giant sea bass \cdot Endangered species \cdot Community science \cdot Population \cdot Spatial ecology \cdot Mark-recapture \cdot Fisheries

1. INTRODUCTION

Overharvest and other anthropogenic disturbances have caused marine predator populations worldwide to decline rapidly (Pauly et al. 1998, Myers & Worm 2003), and there has been a 90% decrease in some predatory fish biomass populations from pre-industrial levels (Myers & Worm 2003). The loss of predatory fishes can cause trophic cascades (Sandin et al. 2008, Estes et al. 2011, Ripple et al. 2014, Donohue et al. 2017), reduce biodiversity (Ritchie & Johnson 2009), exacerbate marine ecosystem phase shifts (Nowicki et al. 2021), and disrupt nutrient cycling (Schmitz et al. 2010, Atwood et al. 2018). It is thus crucial to closely monitor these populations and better understand their life history and spatial ecology, especially in the context of efforts to restore these depleted populations (Myers & Worm 2003, Donohue et al. 2017). These considerations are particularly important for large-bodied or apex predators, as they exert strong ecological effects (Myers et al. 2007, Heithaus et al. 2008, Ritchie & Johnson 2009, Rizzari et al. 2014, Nowicki et al. 2021) and are often more susceptible to local extinctions due to their slow developmental growth rates (Reynolds et al. 2005, García et al. 2008), late sexual maturity (Reynolds et al. 2005) and preferentially targeting by humans for food and sport (Pauly et al. 1998). However, collecting data on the population status and biology of marine top predators can be challenging for various reasons, such as difficulty locating individuals, large home ranges, and high fieldwork costs.

An example of a top marine predator for which there is little information on its life history and current population size is the giant sea bass Stereolepis gigas. This species is the largest teleost carnivore and an important apex predator in California kelp forest and rocky reef communities, with a maximum recorded weight of 253 kg (Fitch & Lavenberg 1971, Domeier 2001, Hawk & Allen 2014). S. gigas inhabit the nearshore kelp forests and rocky reefs from Humboldt Bay, California, to Oaxaca, southern Mexico, and into the Gulf of California, with their population concentrated south of Point Conception (Crooke 1992, Domeier 2001, Hawk & Allen 2014, Ramírez-Valdez et al. 2021). Genetic evidence suggests that there is one genetically bottlenecked population throughout its range (Gaffney et al. 2007, Chabot et al. 2015). They are apex predators that consume a wide range of prey and, therefore, are anticipated to exert diverse and strong influences upon the entire food web to which they belong (Blincow et al. 2022). S. gigas reach sexual maturity between 11 and 13 yr (Fitch & Lavenberg 1971) and their maximum recorded age, determined by otolith analysis, is 76 yr (Hawk & Allen 2014). Their estimated growth rates are 6 yr to reach 14 kg, 10 yr to reach 45 kg, and 15 yr to reach 68 kg (Domeier 2001). Their slow growth rate and aggregative behavior in the summer months (Domeier 2001, Clevenstine & Lowe 2021) make them susceptible to overfishing (Colin 1992, Pauly et al. 1998, Dayton et al. 2003).

Due to their large size, *S. gigas* were commercially and recreationally fished throughout the majority of the 20th century in California. In California, the commercial fishery began in the late 1800s and peaked in 1932, with over 100 t landed (Domeier 2001). With increased fishing pressure and changes in fishing technology from handline to gill netting, the fishery crashed in the 1970s in both Mexico and California (Domeier 2001). This collapse led to the closure of the S. gigas fishery in California in 1981. However, requlations still allowed the take of 2 incidentally caught fish per vessel per trip by commercial set gill net fisheries, which primarily target California halibut Paralichthys californicus (Ayres, 1859) and white sea bass Atractoscion nobilis (Domeier 2001, Benaka et al. 2019, California Fish and Game 2023a, Haggerty & Valle 2024). Due to the continuing decline of *S. gigas*, this regulation was amended in 1988, reducing the incidental take to one fish in California waters (California Fish and Game Code §8380 [California Fish and Game 2023a] and California Code of Regulations Title 14 §28.10, https://law.resource.org/pub/us/ code/ccr/raw/2012_05/gov.ca.oal.title14.html). Since 1994, set gill nets have been banned 3 nautical miles (n miles, 5.56 km) offshore of mainland California and 1 n mile (1.85 km) from the Channel Islands (California Fish and Game Code §8610.3; California Fish and Game 2023b). Recreational fishers occasionally catch S. gigas, but the recreational take of this species is strictly prohibited in California. When fishing south of the USA-Mexico border, 2 S. gigas per angler per trip is allowed with a valid fishing license from the Mexican government (California Code of Regulations Title 14 §28.10). There are no regulations currently in place for the Mexican commercial fishery, and there is little information about the past and current status of the stock (DOF 2006, Ramírez-Valdez et al. 2021).

The International Union for the Conservation of Nature (IUCN) listed *S. gigas* as Critically Endangered in 1996, and the population was described as 'severely fragmented, leading to a continuing decline of mature individuals' (Cornish & Grouper & Wrasse Specialist Group 2004). Recent studies using longterm SCUBA monitoring surveys, experimental gill nets, and genetic sequencing suggest that the *S. gigas* population in Southern California may be increasing, likely due to the banning of inshore set gill nets in 1994, but it has not reached pre-exploitation levels (Pondella & Allen 2008, Chabot et al. 2015, House et al. 2016). However, there has never been a direct attempt to assess the population size of *S. gigas* off California.

To adequately assess the recovery of *S. gigas*, it is imperative to gain insights into their population connectivity. Gaffney et al. (2007) suggest there is only one genetically bottlenecked population of *S. gigas* that exists throughout its range, based on genetic sequencing and mitochondrial data obtained from tissue samples collected from south of Oceanside, California, to Baja California and the Gulf of California, Mexico (n = 56). Chabot et al. (2015) showed similar results from samples obtained from the Northern Channel Islands to Baja California, Mexico (n = 61). Both studies concluded that the *S. gigas* population has low genetic diversity and a small contemporary population size, likely due to intensive fishing, resulting in a bottlenecked population throughout their range (Gaffney et al. 2007, Chabot et al. 2015). To date, no genetic studies on *S. gigas* have analyzed samples north of Point Conception; however, the population is historically concentrated south of Point Conception (Crooke 1992, Hawk & Allen 2014).

A potentially powerful, yet unutilized, method for gaining new insights into the population dynamics, connectivity, and spatial ecology of *S. gigas* is direct underwater sightings. *A. gigas* have idiosyncratic markings on their flanks that remain consistent throughout their lifetime, particularly adulthood, enabling researchers to identify individuals through pattern recognition software such as the Interactive Individual Identification System (I³S; https://reijns. com/i3s/) and the modified Groth algorithm (Arzoumanian et al. 2005, Love et al. 2018). Additionally, because of their large size and charisma, *S. gigas* have become a flagship species amongst recreational divers, contributing an annual non-consumptive value of over US \$2 million to the recreational diving industry (Guerra et al. 2018) (Fig. 1). The iconic status of *S. gigas* within the recreational diving community in Southern California presents a unique opportunity to recruit community scientists to collect identifying photographs of *S. gigas* across their geographic range, enabling a cost-effective assessment of their spatial patterns and population status over time (Fig. 1). Similar methods using photo-identification repositories that utilize community science contributions have successfully been used to assess the population dynamics and movement of other marine megafauna such as whale sharks Rhincodon typus (Holmberg et al. 2009, McCoy et al. 2018, Diamant et al. 2021, Rohner et al. 2021), bottlenose dolphins Tursiops spp. (Lacetera et al. 2023), sand tiger sharks Carcharias taurus (Price et al. 2024), and manta rays Mobula alfredi (Deakos et al. 2011, Couturier et al. 2014, Setyawan et al. 2022). In this study, we use images collected by the community science platform Spotting Giant Sea Bass (SGSB), created in 2016, to identify and track resightings of S. gigas individuals, analyze these IDs to estimate the population size of the species in Southern California, and assess its



Fig. 1. Stereolepis gigas have become a flagship species for recreational divers due to their large size and charismatic nature. (A) A SCUBA diver taking a photograph of GSB187 at Cathedral Point, Anacapa Island (Douglas Klug). (B) GSB178 at Casino Point, Catalina Island (Erin Donalson). (C) GSB136 at Lion's Head, Catalina Island (Kimberly Pye). (D) A group of giant sea bass hovering under a concrete piling at Hermosa Artificial Reef (Merry Passage)

recent trends. Additionally, we contribute information on *S. gigas* spatial ecology, population connectivity, and use of marine habitats. This information can inform the management of this species in a costeffective manner.

To complement the data obtained from the community-science-based mark—recapture population assessment, we also examined *S. gigas* population dynamics from a more traditional fishery-dependent source: incidental catch of *S. gigas* documented through set gill net landing receipt records. Since most recreational diving occurs in nearshore waters where gill net fishing is prohibited, these pooled fishery-dependent and independent sources collectively provide a more complete view of *S. gigas* population dynamics throughout Southern California.

2. MATERIALS AND METHODS

2.1. Data sources

We opportunistically collected identification photos and sighting data of giant sea bass *Stereolepis gigas* from the recreational diving and fishing community through photos submitted to the photo identification repository, SGSB (https://spottinggiantseabass.msi.ucsb.edu/), a website developed with WildMe[©] utilizing Wildbook[©] software (Pettit et al. 2024). Additionally, we periodically extracted photos and videos of *S. gigas* encounters from social media platforms (Instagram, YouTube, and Facebook) and uploaded georeferenced images to our database. Each submission documenting an encounter with *S.* gigas includes the sighting location and date along with images of the fish's left and/or right flank for identification (Table S1 in the Supplement at www. int-res.com/articles/suppl/m760p151_supp.pdf). The submitter also has the option to include additional information such as the individual's estimated length, visible scars or tags, life stage, and any other relevant information (Table S1).

Wildbook[©] is an autonomous computational system that uses deep convolutional neural networks to identify individuals with distinct markings, such as spots, stripes, or wrinkles (Berger-Wolf et al. preprint doi: 10.48550/arXiv.1710.08880). For SGSB, Wildbook[©] uses a deterministic pattern-matching process that incorporates both I³S (Van Tienhoven et al. 2007) and modified Groth (Groth 1986) algorithms to suggest potential matches for newly submitted encounters based on S. gigas spot patterns in the existing image database. Upon submission, each encounter is assigned a unique alphanumeric identifier automatically generated by Wildbook[©], which remains associated with the encounter for its lifetime. Researchers then use the user interface to 'spot map' the individual (i.e. manually identify the spots on the individual's flank), run the spot pattern through the pattern-matching and computer vision software (I³S and modified Groth), and manually match the individual to suggested matches of previously identified S. gigas (Fig. 2) (Arzoumanian et al. 2005, Love et al. 2018, Berger-Wolf et al. preprint doi:10.48550/arXiv.1710.08880). $I^{3}S$ has proven to be highly effective at identifying S. gigas, with the algorithm placing a true match in the top 10 image results 95% of the time (Love et al. 2018). To increase accuracy and robustness, Wildbook[©] also



Match score: 409.75 (Match 1 of 27)

Fig. 2. A spot pattern match of GSB023, identified using the modified Groth algorithm, from encounters 5 yr apart. The match score indicates the likelihood of a true match, with higher scores reflecting greater similarity

uses the modified Groth algorithm in conjunction with I³S; this combination has been shown to be highly accurate with species with similar spot patterns, such as whale sharks *Rhincodon typus* (Arzoumanian et al. 2005, Holmberg et al. 2009, Diamant et al. 2021, Rohner et al. 2021, Berger-Wolf et al. preprint doi:10. 48550/arXiv.1710.08880) and sand tiger sharks *Carcharias taurus* (Price et al. 2024). The modified Groth algorithm has been shown to successfully match whale shark images over 90% of the time, with most match failures attributed to oblique-angle images that did not accurately capture the spot pattern (Arzoumanian et al. 2005).

Each researcher underwent extensive training on the identification process to ensure accuracy, and no images were altered. To further minimize bias, each encounter was reviewed by at least 2 trained technicians before being added to the database. If no match was found with a previously identified individual, a new unique ID was assigned to the fish. If the photos of the individual did not clearly show the spot pattern, the image was labeled as 'unidentifiable,' no individual ID was assigned, and the encounter was excluded from the population models. It is important to note that the spot patterns on the left and right flanks are distinct; therefore, each encounter is classified as having a left flank image, a right flank image, or both. Due to the difficulty of sexing S. gigas from photographs, no IDs were assigned a sex. Data from August 1997 to September 2023 were analyzed in this study; however, only data from 2015 to 2022 were included for the population models due to large gaps in sampling efforts prior to 2015. The broader data set was utilized to assess spatial patterns and connectivity. Various outreach methods were employed throughout California to raise awareness of the project and encourage the submission of images to SGSB, including presentations, posters, informational handouts, photo contests, newsletters, branded apparel, and social media campaigns.

To supplement the community science data in regions where divers and recreational fishers rarely operate, we analyzed the California Department of Fish and Wildlife's (CDFW) set gill net landing receipt data from 1994 to 2022 (Haggerty & Valle 2024). The number of *S. gigas* incidentally landed and the number of fishing trips was analyzed by year and month to assess trends.

2.2. Study area

Both community science and fisheries data sets used here focus on Southern California from Point Conception south to the border with Mexico, including the 8 Channel Islands (Fig. 3). These geographic boundaries were determined based on the highest density of reported sightings from SGSB, with only 7 reported encounters north of Point Conception and one in Mexican waters. Approximately 55 280 charter boat diver days occur annually in this region of Southern California, with diving happening year-round (Guerra et al. 2018). There is also a substantial amount of recreational shore-diving and private ves-



Fig. 3. (A) Study area in Southern California. Colored dots represent the number of left-sided encounters of *Stereolepis gigas* (n = 726) reported to the Spotting Giant Sea Bass Project from 2015 to 2022. (B) *S. gigas* set gill net incidental landings by fishing block (10 × 10 nautical miles) from 1994 to 2022. Grey blocks: *S. gigas* landings redacted from fishing blocks with fewer than 3 distinct vessels, businesses, or fishing permits to preserve confidentiality

sel diving activity in the region. Commercial set gill netting is conducted within this region outside of state waters (i.e. 3 n miles; 5.56 km) on the mainland and outside 1 n mile (1.85 km) from the Channel Islands.

2.3. Population modeling of community science data

To estimate the abundance of *S. gigas* from 2015 to 2022, we input verified sightings data from the SGSB platform into mark-recapture population models. To prevent potential double counting, only left-side encounters were analyzed for this analysis due to their larger sample size. Although SGSB was created in 2016, we received numerous retroactive encounters dating back to 1997 (with the earliest left flank encounter in 2001). From 1997 to 2014, there were large annual gaps in reported encounters, with only 71 identified left flank encounters (and 45 right flank encounters) reported. As a result, these years were excluded from the population models (see Table 1). In this analysis, a 'mark' refers to the earliest record of the individual in the study period (i.e. the first time the individual was photographed and identified). A 'recapture' is a subsequent encounter with a previously identified individual. Sightings were reported year-round, and each sampling period was defined as January–December for each year from 2015 to 2022.

Jolly-Seber (JS) models were used to estimate population size and trend using the POPAN and Pradel parameterizations, respectively (Jolly 1965, Seber 1965, Pradel 1996, Schwarz & Arnason 2016). JS formulations are open-population models that assume individuals can enter (through birth or immigration) or exit (through death or emigration) the study population. Both marked and unmarked individuals in the study population are assumed to have the same survival and capture probability (ϕ and p, respectively). All juvenile S. gigas encounters were removed from this analysis so as not to violate the assumption of equal detectability and survivability of each marked and unmarked S. gigas. S. gigas appear to exhibit high site fidelity while showing variability in their spatial ecology, with some individuals traveling long distances while others remain long-term residents of reefs (Clevenstine & Lowe 2021, Blincow et al. 2023). The tendency of recreational divers to frequent consistent locations may result in a higher likelihood of detecting *S. gigas* with high site fidelity or those residing in frequently dived areas. However, the SGSB platform also receives data from research

and commercial divers as well as fishers, spanning a wide range of locations across Southern California (Fig. 3). Through a goodness-of-fit test, there was no evidence of assumption violations.

All models were created using the R package 'RMark' (Laake 2013). Goodness-of-fit tests were implemented using the 'R2ucare' package (Gimenez et al. 2018) to examine the heterogeneity of apparent ϕ and p and to test for overdispersion, as indicated by the variance inflation factor (ĉ). An overall goodness-of-fit test was nonsignificant, indicating there is no strong evidence for a lack of good fit (p = 0.227, $\hat{c} = 1.22$; Gimenez et al. 2018). Since the overall test showed no evidence of a lack of fit or assumption violations, no model adjustments or further tests were required (Gimenez et al. 2018). However, to ensure that the equal p assumption was not violated, we ran both 'Test 2.CT' and 'Test 2.CL' (Gimenez et al. 2018). Test 2.CT assesses whether missed individuals have the same probability of being recaptured on the next occasion as currently captured individuals (Gimenez et al. 2018). Test 2.CL evaluates whether there is any difference in the timing of reencounters between individuals who were captured and not captured on occasion i_i conditional on their presence on both occasions *i* and i + 2 (Gimenez et al. 2018). Both tests were nonsignificant (p = 0.577 and p = 0.621, respectively), indicating no significant evidence that the equal catchability assumption was violated.

Akaike information criterion corrected for small sample sizes (AICc) was used as a model comparison statistic for all population models (Burnham & Anderson 1998). Akaike weights, which represent the probability that a given model is the best-fit model among the candidate set, were also calculated to facilitate the interpretation of model comparisons (Burnham & Anderson 1998, Wagenmakers & Farrell 2004, Cooch 2008). For models where $\Delta AICc \leq 2$, Akaike weights were compared, and the model with the highest Akaike weight was selected as the best-fit model (Wagenmakers & Farrell 2004, Cooch 2008). In all instances, the best-fit model had an Akaike weight that was at least twice as large as the second-largest Akaike weight (Tables S2 & S3).

The POPAN parameterization of the JS model was used to estimate the total super-population (N) (Schwarz & Arnason 1996). The POPAN formulation assumes that all individuals encountered during the survey period represent a component of a larger super-population. This super-population value represents the total number of individuals present in the study area during the study period. This parameterization also estimates p, ϕ , and the probability of entry

into the population (p_{ent}) (Schwarz & Arnason 1996); p_{ent} and ϕ were modeled as constant or variable over time. The percentage of successful identifications per year (identification rate), the number of days with reported encounters, and the number of reported encounters were calculated per year and included as covariates to estimate p (see Table 1). Capture probability was modeled as constant and dependent on the following: year, annual identification rate, the number of reported encounters per year, the number of days with reported encounters per year, and all possible combinations (including interactions) of these covariates. The size of the total super-population (N) was expressed with a 95% confidence interval based on lower and upper control limits. Due to the difficulty of sexing S. gigas, sex was not included as a grouping effect.

The Pradel parameterization of the JS model was used to estimate the rate of change (λ) in the population. This formulation estimates p, ϕ , and λ (Pradel 1996), where $\lambda < 1$ indicates a decreasing population and $\lambda > 1$ indicates an increasing population. To assess the overall trend, λ was set as constant. To explore interannual variation, λ was set as variable over time; ϕ was modeled as both constant or dependent on time since first sighting; p was modeled as constant and dependent on the following: year, annual identification rate, the number of reported encounters per year, the number of days with reported encounters per year, and all possible combinations (including interactions) of these covariates.

2.4. Population modeling of fisheries data

We also examined the S. gigas population trend from the CDFW incidental landing receipt data. Catch per unit effort (CPUE) was calculated from all commercial set gill net fishing trips between 1994 and 2022 from Point Conception to the USA-Mexico border. The initial year was chosen as 1994 to remove any confounding variables associated with the inshore set gill net ban that came into force that year (Haggerty & Valle 2024). Total set gill net trips were estimated by counting unique combinations of date, captain, vessel, and gear by year, each indicating one day of landing (i.e. one trip) by a single individual. The number of trips where an individual S. gigas was landed was estimated in the same fashion. Since it is legal to retain only one S. gigas per set gill net vessel, and landing receipts record weight and not numbers of fish, any weight landed was assumed to correspond to one S. gigas (i.e. the estimated number of trips landing *S. gigas* is also the total estimated number of *S. gigas* landed). We calculated CPUE by dividing the estimated number of *S. gigas* caught by the estimated total number of set gill net trips per year. A linear regression was fit to assess the rate of change in landings per year.

2.5. Seasonality, spatial patterns, and connectivity

To broadly characterize the seasonality of S. gigas encounters in the community science program, we examined how the cumulative number of encounters submitted to SGSB from 2015 to 2022 varied by month. We then compared these patterns to trends in the set gill net landing receipt data. To describe the long-range movement potential of individual S. gigas, we mapped all documented movements between 2 offshore islands, between the mainland and an offshore island, and notable long-distance (>50 km) movements along the mainland. We also showcased the elapsed time between resightings for each of these individuals. To do this, reported locations with resighted individuals were aggregated into 8 broader areas: San Diego (Point Loma to Solana Beach), San Clemente Island, San Onofre (San Onofre Nuclear Generating Station mitigation artificial reefs and nearby reefs), Catalina Island, Santa Barbara Island, Santa Monica Bay (Hermosa Artificial Reef [HAR] and Redondo Beach), Anacapa Island, and Santa Cruz Island. Site fidelity was evaluated by examining interannual sighting patterns at the site level to identify trends and by assessing the number of sites where each of the 3 most frequently resighted individuals were encountered. Submitted encounters from 1997 to September 2023 were also evaluated by reef type (natural vs. artificial) to assess trends in S. gigas hotspot activity and site status. Artificial reefs are defined as structures intentionally created from natural or man-made materials (e.g. concrete pilings, tires, cars) that are deployed on the seafloor to influence physical, biological, or socioeconomic processes associated with marine resources (Seaman & Jensen 2000).

3. RESULTS

3.1. Community science database

The SGSB repository received 1621 encounters of giant sea bass *Stereolepis gigas* from 1997 to 2022 from 330 individual submitters. Of these reported encounters, 1031 were identifiable; 600 contained an

identifiable left spot pattern, and 593 contained an identifiable right spot pattern. Only 7 out of the 1621 encounters were north of Point Conception. To date, SGSB has received only one reported sighting event in Mexico off the Coronado Islands, on 4 August 2023, involving the observation of 8 individuals (the SGSB platform is not yet formally designed to incorporate encounters in Mexico, but this is an anticipated area of expansion).

3.2. Population modeling of community science data

From 2015 to 2022, 327 left-sided *S. gigas* individuals were identified out of 726 encounters with varied survey effort per year (Table 1). The discovery curve of new *S. gigas* identifications in the SGSB community science repository versus the cumulative number of identifications continually increased and did not approach an asymptote (Fig. S1). The best-supported POPAN model, with a model weight of 0.42, had ϕ as constant, *p* dependent on the number of submitted encounters per year, and *p*_{ent} as constant (Table S2, Fig. S2). The estimated super-population was 1221 individuals (95% CI = 988–1537; Table 2).

The best-supported Pradel model had a constant λ_r constant ϕ , and *p* as dependent on the number of reported encounters per year with a model weight of 0.37 (Table S3, Fig. S2). In this model, λ was 1.08 (1.02–1.14), indicating an increasing population (Table 2). When λ was modeled as time-variable, the best-supported model had a weight of 0.22 with a constant ϕ and *p* dependent on the number of reported encounters per year. Starting from 2015-2016, there was an increasing rate of change in the population $(\lambda = 1.05, 0.59 - 1.86)$, with the following 4 periods continuing this increasing rate of change with the highest rate of change in 2018–2019 ($\lambda = 1.59, 0.95$ – 2.63; Table 2). In 2020-2021, we saw a decreasing rate of change ($\lambda = 0.79$, 0.50–1.25; Table 2). From 2021– 2022, the population essentially did not change ($\lambda =$ 0.99, 0.72-1.35; Table 2).

3.3. Population modeling of fisheries data

Overall, CPUE increased from 1994 to 2022 with the lowest number of *S. gigas* caught per set gill net trip in 1994 at 0.01 and the highest number of *S. gigas* caught per trip in 2022 at 0.11 (Fig. 4). Annual CPUE increased slightly over time with an increase of 0.002 *S. gigas*

 Table 1. Giant sea bass Stereolepis gigas survey effort, encounters, and identification rates per year from 2015 to 2022. Identified encounters are encounters with an attributed ID

Year	2015	2016	2017	2018	2019	2020	2021	2022	Total
Days with reported encounters	24	15	26	48	47	68	50	50	328
No. of submitted encounters	40	32	59	80	164	179	88	84	726
No. of identified encounters	30	30	44	53	129	122	65	55	528
Identification rate (%)	0.75	0.94	0.75	0.66	0.79	0.68	0.74	0.65	
No. of S. gigas identified	20	20	31	42	85	88	56	48	
New to database	20	19	29	35	74	75	38	37	
Re-sighted individuals	0	1	2	7	11	13	18	11	
Cumulative population	20	39	68	103	177	252	290	327	

Table 2. Model estimates of giant sea bass population size (N) and yearly abundance Ny from the best-fit POPAN model and the rate of change (λ) from the Pradel models. The annual estimates of λ are from the best-fit Pradel model with λ set as time-dependent. The λ value from the entire study period is from the best-fit Pradel model with λ set as constant. Note that λ is the rate of population change between years and not within a year. N and Ny are expressed with a 95% confidence interval based on lower and upper control limits

Model	Parameter	2015	2016	2017	2018	2019	2020	2021	2022	Overall
POPAN	Ny and N	283.7 (109.0— 458.5)	357.6 (201.1– 514.2)	415.9 (259.5– 572.2)	461.8 (294.3– 629.4)	498.1 (313.6— 682.5)	526.6 (323.2– 730.1)	549.1 (326.7– 771.6)	566.9 (326.5— 807.3)	1221.1 (987.5— 1537.4)
Pradel	λ		1.05 (0.59— 1.86)	1.42 (0.84— 2.4)	1.17 (0.75— 1.80)	1.59 (0.95— 2.63)	1.04 (0.80— 1.36)	0.79 (0.50— 1.25)	0.99 (0.72— 1.35)	1.08 (1.02— 1.14)



Fig. 4. Catch-per-unit-effort (CPUE) of giant sea bass *Stereolepis gigas* from set gill net landings from 1994 to 2022. CPUE was calculated as the number of individual giant sea bass caught per commercial set gill net fishing trips per year



caught per trip per year (r = 0.70, F = 27.36, p < 0.001; Fig. 4). From 1994 to 2022, an average of 126 *S. gigas* were landed per year with a mean weight of 42.3 pounds (19.2 kg) (Table S4).

3.4. Seasonality

S. gigas encounters in our community science data were documented throughout the year, with a noticeable increase in reported encounters in late summer and early fall, particularly from August to October. September has the highest number of reported encounters (Fig. 5A). We did not see a discernible seasonal trend in the set gill net landings data, with the number of S. gigas caught per set gill net trip ranging from 0.023 in October to 0.063 in November, while the catch per trip remained relatively consistent from June to September over the years of this study (Fig. 5B). However, the average weight landed per set gill net trip per month exhibited a trend similar to the SGSB data, with the highest average landings by weight occurring from June to September (Fig. S3A) (Haggerty & Valle 2024).

3.5. Spatial patterns and connectivity

There were 8 notable movements (i.e. long-distance movements of >50 km and movements between islands and the mainland) observed in the SGSB database from 7 unique individuals from 2015 to 2022 (Fig. 6). Individu-

Fig. 5. (A) Cumulative number of submitted Stereolepis gigas encounters by month from 2015 to 2022 from the Spotting Giant Sea Bass repository. (B) Average number caught per commercial set gill net trip by month from 1994 to 2022, calculated as the cumulative estimated number of *S. gigas* caught per month divided by the cumulative reported number of commercial fishing trips per month. Error bars: ±SE of the number of *S. gigas* caught per set gill net trip



Fig. 6. (A) All detected movements of *Stereolepis gigas* between islands, between islands and the mainland, and over long distances (>50 km). Arrow indicates where the individual was encountered following its previous encounter. It is important to clarify that these directional cues do not represent real-time tracking data. Rather, they offer a long-term, generalized view of movement patterns. Bubble size is correlated with the number of resighting events within the given area (i.e. the number of reported encounters of individuals seen more than once). (B) Abacus plot depicting the encounter timeline for the 7 individuals that moved between islands, between islands and the mainland, or over long distances (>50 km). Each colored line indicates a single reported encounter; line color correlates with the general geographic area where the individual was encountered

al 'HERART003' was observed performing 2 long-distance movements. The fish was first encountered at HAR on 9 September 2015, re-sighted at San Onofre

HAR on 9 September 2015, re-sighted at San Onofre on 25 September 2020, and later reseen at HAR on 11 February 2021, covering a straight-line distance of approximately 92 km each way (Fig. 6).

The longest overall straight-line distance traveled between resightings was by HERART004, seen at HAR on 6 September 2015 and resighted in La Jolla on 22 September 2021, at approximately 155 km (Fig. 6). Notably, with regard to speed of movement, GSB187 was encountered off Anacapa Island on 4 July 2021 and quickly resighted off Santa Cruz Island on 19 July 2021, a distance of approximately 10 km (Fig. 6). Of the 189 individuals encountered at least twice, only 4.7% of individuals have a documented long-distance movement (Fig. S4).

Based on the most encountered individuals, S. gigas seem quite individualistic regarding long-term site fidelity. Our most-often encountered individual, GSB136, has been encountered 19 times at 7 dive sites throughout the north side of Catalina Island, with the most distant sites being approximately 22 km apart. This fish was encountered consistently from 2017 to 2023 but only during March through November (Fig. S4). In contrast, the second-most encountered individual, GSB178, was encountered 18 times from 2017 to 2023, solely at Casino Point Dive Park off Catalina Island, with encounters from March through October (Fig. S4). Our third-most encountered individual, GSB023, has been observed 11 times from 2014 to 2023, all on the northeast shore of Catalina Island: Little Farnsworth to Casino Point Dive Park (approximately 2.2 km apart), with encounters from June through October. To date, the longest time elapsed between sightings is for GSB191, which was first encountered in 2001 at the Italian Gardens dive site off Catalina Island and resighted in 2005 and 2021 at the same location (Fig. S4). When considering all the submitted encounters from August 1997 to September 2023, a large number of encounters occurred in unprotected artificial reefs, almost entirely at HAR (Fig. S5).

4. DISCUSSION

The development of the SGSB open repository, comprising community-science-derived images of giant sea bass *Stereolepis gigas*, has created a unique opportunity to generate important new insights into the population dynamics and spatial ecology of this understudied and historically overfished species. Our analysis of this repository indicates that the *S. gigas*

population is increasing in Southern California. Additionally, we examined incidental catch of *S. gigas* through set gill net landing receipts, which further validate the increasing trend of the *S. gigas* population in this region.

4.1. Population modeling of community science data

Our population models estimate the S. gigas population in Southern California to be approximately 1221 adult individuals, representing the first direct population estimate of S. gigas. Chabot et al. (2015) estimated the contemporary effective population size of the entire population to be approximately 500 individuals; however, given the life history of S. gigas (i.e. long lifespan, long generation times, and low fecundity), accurately estimating the census population size from the effective population size remains challenging. To contextualize this approximation of the current adult S. gigas population size in California for management applications, it is essential to understand the pre-exploitation baseline population of this species. Such contextualizations are challenging. California landings of S. gigas peaked at approximately 100 t in 1932 (Domeier 2001). Recreational landings in California peaked in 1963, with over 500 individual S. gigas landed (Domeier 2001). The average weight of S. gigas caught from 1994-2022 reported as not gutted or dressed was approximately 19.2 kg (42.3 lbs). If the same sizes were caught in 1932, approximately 4730 S. gigas would have been caught that year alone, which is nearly 4 times the estimated total size in Southern California from our best-fit POPAN model-and this would have been after approximately 62 yr of intense commercial fishing. This estimated average weight of 42.3 lbs (19.2 kg) only reflects S. gigas that were reported on landing receipts as whole and not gutted or dressed. That being said, when there was an active S. gigas fishery, S. gigas were commonly dressed prior to landing to conserve space on fishing vessels (Crooke 1992). However, after decades of extensive overfishing, the S. gigas caught today are likely younger and smaller than the S. gigas caught in the 1930s, with 58% of S. gigas landed in set gill nets from 1994-2022 being only approximately 2-7 yr old (Haggerty & Valle 2024). Therefore, the available data suggest that our estimated population size is substantially below preexploitation baselines.

Our best-fit Pradel model with constant λ (rate of change over the entire time period) suggests a pop-

ulation increase from 2015 to 2022 (Table 2). The bestfit time-dependent Pradel model showed an annual population increase until 2020–2021, where λ (rate of population change between years) was less than 1 (Table 2). From 2021 to 2022, λ was equal to 1, indicating a possible decline in the population in 2021, followed by a stable population in 2022. However, we caution against overinterpretation of such shorterterm time periods. This putative decrease may have been associated with a change or reduction in SCUBA observer effort during the COVID-19 pandemic (despite the fact that the model accounts for the number of submitted encounters). Additionally, in September 2019, the sinking of the 'Conception' dive boat near the Northern Channel Islands tragically claimed the lives of 33 divers and a crew member, greatly disrupting the local recreational diving industry. This event, coupled with the onset of the COVID-19 pandemic, led to a dramatic decrease in dive trips, particularly overnight trips. By 2022, recreational diving activity had only partially recovered, with dive trips reaching just 54% of 2016 levels (Morse et al. 2024). During this period of reduced diving activity, many of the 2020 encounters were submitted by a smaller group of dedicated community scientists due to the broader inaccessibility of recreational diving during the pandemic. This redistribution of sampling effort may have influenced population trend assessments. Furthermore, the estimated population decline may also have been affected by the onset of La Niña oceanographic conditions in 2020, which continued through 2022, marking the longest consecutive stretch of negative Oceanic Niño Index since 1998-2001 (Thompson et al. 2024) and may have affected *S. gigas* behavior in nearshore waters. There was a La Niña event from 2016-2017 but it was considered weak (Wells et al. 2017). With more years of sighting data, the confidence intervals of the annual rate of change of the population will shrink. That said, the lowest lower bound of the confidence interval for the best-fit model for the overall rate of change is 1.02. This suggests that at the lower confidence limit, the population of *S. gigas* still grew from 2015 to 2022.

4.2. Population modeling of fisheries data

When evaluating a species' current conservation management, utilizing multiple data sources will ensure a comprehensive understanding of its population status. CPUE data from the CDFW landing receipts confirms our mark—recapture model results showing a slight population increase from 1994 to 2022 (Fig. 4). Interestingly, CPUE shows an increase in landings during the 2020–2021 period, in which the Pradel model suggested a population decrease. Our findings reinforce the conclusion that there is a trend towards an increasing *S. gigas* population in California. Pondella & Allen (2008) observed similar results through the use of experimental set gill nets throughout Southern California from 1995 to 2004 and SCUBA surveys off Palos Verdes Point from 1974 to 2004. They noted a significant increase in CPUE over the course of the study, with giant sea bass being observed in Palos Verdes for the first time in 2002 and every year thereafter.

On average, 126 S. gigas were landed as incidental catch annually from the set gill net fishery in California from 1994 to 2022 (Table S4). This may represent an underestimate as it does not include any S. gigas discarded as bycatch that may occur in these fisheries. A discard usually occurs when the fisher has caught more than one S. gigas and is legally required to return the rest to the ocean. While set gill net fisheries in California do not currently carry fisheries observers, past Federal observer data from 1990-2017 indicated that 31 out of 43 captured S. gigas were returned dead (Haggerty & Valle 2024). If indeed the discard rate for this species is 72%, it is possible that the total number of *S. gigas* removed from the population (i.e. landings plus bycatch) is substantially higher than 126. The clear alignment between trends in the CPUE data, prior research, and our community science population assessments suggest that the S. gigas population is increasing, and thus recruitment is likely outpacing potential impacts from incidental or discarded bycatch. However, it remains unclear if and how the annual removal of 126 individuals, and likely more, from a population estimated at 1221 adult individuals is influencing this rate of population increase and recovery toward the historic baseline.

4.3. Patterns of seasonality

In addition to the ability to estimate population size and dynamics of *S. gigas*, the community science data also provides valuable insight into how this difficultto-study species uses marine space. Reported encounters in the SGSB data set spanning more than a decade demonstrate a clear seasonal pattern, with the majority of sightings occurring between July and October and peak encounters occurring in September (Fig. 5A). Such patterns are consistent with previous findings that *S. gigas* aggregate in nearshore waters from June to October (House et al. 2016, Clark & Allen 2018, Clevenstine & Lowe 2021, Spector et al. 2022). Furthermore, Clevenstine & Lowe (2021) discovered that out of the 34 S. gigas they tagged at Catalina Island, some remained resident year-round whereas others left the island in the late fall or winter and returned in the spring and early summer, indicating annual fidelity and partial seasonal migration. Interpretation of such seasonal behaviors for S. gigas from community science records needs to be considered in light of the fact that due to favorable conditions, more recreational diving typically occurs during the summer and fall. However, the alignment between our observations and seasonality and noncommunity science records suggests such biases may be minor.

The S. gigas bycatch landings record showed some, but not all, of these same trends in seasonality. Landing rates for S. gigas by weight per set gill net trip showed a similar clear seasonal trend, with the largest landing by weight caught between June and September (Fig. S3) (Haggerty & Valle 2024). Ramírez-Valdez et al. (2021) also found that the largest proportion of landings in Mexico and the USA were reported in the summer months from 2000-2016. The estimated number of individual S. gigas caught per set gill net trip (Fig. 5B) was, however, relatively consistent from March to September, with the highest landings rates in November, followed by July. This observed November peak in the estimated number of individuals caught could arise from the fact that set gill net fisheries are required to fish in deeper offshore waters, capturing more *S. gigas* during their hypothesized movements from shallower coastal waters to deeper waters in the late fall and winter months (Burns et al. 2020, Clevenstine & Lowe 2021, Peria 2023). Additionally, throughout the year, there are differences in set gill netting locations that could be impacting the seasonal differences observed in S. gigas landings. The trend in S. gigas catch is similar to the seasonality of all set gill net landings, with January–March having a higher proportion of S. gigas landed in blocks south of Huntington Beach and April–July displaying a greater proportion of landings in blocks off Ventura, with less discernible changes in location the remainder of the year.

4.4. Inter-island and long-distance movements

We documented 8 instances of movements between islands, between islands and the mainland, and longdistance (>50 km) movements along the mainland amongst 7 individuals (Fig. 6). Such observations of long-range movement hold both ecological and applied significance. They showcase patterns of potential genetic connectivity within the population, inform our understanding of which sites and regions may be recolonized and recovered, and guide the use of place-based conservation measures to support the management of this species.

However, only 4.7% of resighted individuals exhibited long-distance movements, suggesting potentially limited genetic exchange among sites and making S. gigas susceptible to localized genetic bottlenecks. This concern is reinforced by previous research that found high site fidelity among *S. gigas* at Santa Barbara Island (Spector et al. 2022), Catalina Island (Clevenstine & Lowe, 2021), and La Jolla (Blincow et al. 2023) during the spawning months. Returning to the same site every year during the spawning months can potentially limit genetic exchange within the larger population, as individuals are likely spawning with many of the same individuals every year. Gaffney et al. (2007) reported extremely low mitochondrial sequence diversity amongst the S. gigas population and identified limited differentiation at nuclear loci between the Pacific Coast and Sea of Cortez (Gulf of California) populations, suggesting low genetic exchange, possibly due to natural selection or genetic drift in populations with low effective numbers of males. Chabot et al. (2015) observed a lack of population structure and no evidence of isolation by distance among sample localities from the Northern Channel Islands to northern Baja California, concluding that S. gigas constitutes a single population with shallow genetic divergence and little phylogeographic structure. Additionally, estimates of contemporary migration indicated gene flow generally occurring in a north-to-south direction, with estimates of self-recruitment (i.e. individuals returning to their locality of origin) being highest within all sampling localities (Chabot et al. 2015). This pattern aligns with S. gigas tagging studies and reinforces concerns about local genetic bottlenecks. Therefore, these long-distance movements of select individuals may help facilitate gene flow and maintain one genetically similar population throughout their geographic range, even if genetic exchange is limited.

The observation that multiple *S. gigas* were able to travel long distances (i.e. at least 155 km), move between islands over relatively short time periods (i.e. 15 d), and traverse deepwater regions (e.g. >900 m) suggests a relatively high capacity for regional mobility by this species. There are only 6 species in the Family Polyprionidae, which contains only 2 genera

worldwide (Oh et al. 2021). Little is known about the spatial ecology amongst these species; however, a multi-year mark-recapture study assessing the spatial ecology of hāpuku, Polyprionidae oxygeneios, showed variable movement patterns, with some individuals recaptured at the same location, whereas others traveled substantial distances up to 1389 km (Beentjes & Francis 1999). The Atlantic goliath grouper Epinephelus itajara has a similar biology to S. gigas as the largest teleost apex predator in coral reefs in the Caribbean, reaching over 400 kg (Koenig et al. 2007). Tagging studies have shown that E. itajara are capable of traveling distances greater than 170 km (Pina-Amargós & González-Sansón 2009, Koenig et al. 2011). Both juvenile and adult E. itajara exhibit high site fidelity, with the vast majority (82%) of recaptured adults being re-sighted less than 1 km from the previous sighting location (Koenig et al. 2011).

These community science-based observations align with others that have utilized acoustic tracking data to reveal similar kinds of long-distance (>50 km) movements across deepwater channels by S. gigas (Burns et al. 2020, Clevenstine & Lowe 2021). They also provide behavioral evidence that corroborates the finding of contemporary gene flow across the population reported by Chabot et al. (2015). These patterns provide some preliminary insight into which regions may be especially important hubs for movement that help to facilitate connectivity across their range. Anacapa Island, for example, seems to be one such area of high population connectivity. Four fish undertaking long-range movements were detected leaving and arriving at Anacapa Island. HAR also seems to be an important site for population connectivity, with 3 fish observed moving to and from this site. However, it is important to note that there is considerable diving effort along Anacapa Island and HAR, potentially introducing a bias in sightings and, subsequently, our understanding of *S. gigas* hotspots.

Analyses of patterns of relative mobility versus site fidelity for *S. gigas* suggest heterogeneity between individuals. Among the 90 individual *S. gigas* encountered over multiple years, only 7.8% exhibited longdistance (>50 km) or cross-channel movements (Fig. 6). This observation is consistent with Clevenstine & Lowe (2021), who discovered individualistic movement patterns off Catalina Island via acoustic tagging. Blincow et al. (2023) found that some *S. gigas* are long-term residents of the La Jolla kelp forest, exhibiting high site fidelity within small, well-defined areas. However, one out of the 7 tagged *S. gigas* emigrated from La Jolla to Del Mar (8 km), and 2 fish left the array within 2 d and were not detected by any other deployed receivers across Southern California. The findings align with studies of other marine megafauna (e.g. some sharks) in which only a subset of individuals undertake long-distance movements (White et al. 2017). Factors potentially contributing to these individualistic spatial patterns could include sex or life stage. While it is not possible to accurately determine the sex of *S. gigas* through imagery alone, future studies conducted in conjunction with SGSB could be employed to assess if and how sex contributes to their spatial ecology.

Continued image collection by SGSB, along with expanding outreach and image collection north of Point Conception and south to Mexico, will enable researchers to better understand movements to and from these northerly and southerly populations. The latter may be especially helpful as researchers attempt to understand if and how S. gigas may undertake any range shifts to adjust to climate-changeinduced alterations in oceanographic conditions. Furthermore, Mexican waters comprise 73% of this species' range, making it essential to collect data in this region for a comprehensive picture of population status, to assess how local populations might respond to varying national fishery management, and to address the uneven distribution of research effort on S. gigas throughout its range (Ramírez-Valdez et al. 2021).

S. gigas have historically rarely been observed north of Point Conception (Crooke 1992, Domeier 2001, Hawk & Allen 2014). However, 7 encounters were reported during the course of this study. All such encounters exclusively involve sexually immature sub-adults weighing between approximately 7 and 23 kg. These individuals were captured and released by recreational hook and line fishers fishing for California halibut *Paralichthys californicus* (Ayres, 1859) offshore Half Moon Bay, Santa Cruz, Monterey, and Cayucos. The first reported sighting occurred in July 2019, with subsequent encounters documented annually. As with other examples of California marine predatory fishes, such as white sharks Carcharodon carcharias (Tanaka et al. 2021) and the California sheephead Semicossyphus pulcher (Cowen 1985, Tegner & Dayton 1987), these encounters might suggest a northward expansion of the S. gigas range due to climate change or El Niño events, which cause warmer temperatures and a change in current conditions, increasing recruitment north of Point Conception. Sustained reporting and monitoring of the survival and persistence of S. gigas in Northern California will be required to substantiate the possibility of a northward reintroduction and potential expansion.

4.5. Artificial reefs

The 0.2 ha HAR (Lewis 1989), approximately 1 n mile (1.85 km) off Hermosa Beach, has the second most reported encounters in the SGSB repository (21% of all encounters), with reported sightings every month of the year (Fig. 3A). Notably, the largest aggregation reported to SGSB occurred at HAR, with 22 individuals observed in a single dive. It is important to note that the disproportionate number of encounters may be driven by frequent diving efforts from a few avid local divers seeking the seemingly ever-present S. gigas. However, there are considerable gaps in reporting and much lower survey effort compared to more popular dive sites in Southern California, with only 57 d of reported encounters from 2015 to 2022. In comparison, Casino Point has the most reported encounters (24% of all encounters) yet has 205 d with reported encounters from 2015 to 2022. Additionally, 2 avid divers conducted 66 dives at HAR from 2015 to 2022, encountering at least one S. gigas 86% of the time. It is unclear why S. gigas are so consistently present at this location. However, our resighting data highlights its potential importance for population connectivity (e.g. as a stopover hub), as fish encountered at HAR have been sighted off Catalina Island, San Onofre, and San Diego (Fig. 6). Similar patterns have been observed for S. gigas at Wheeler North Artificial Reef off San Onofre (Burns et al. 2020). Further research is needed to understand what specifically attracts *S. gigas* to certain artificial reefs, how they utilize these reefs, and identify any additional deleterious outcomes that may arise from their associations with such reefs (Burns et al. 2020).

4.6. Future research

With continued community involvement, the SGSB platform and data set present a growing opportunity for others to continue to answer key questions regarding the life history, population dynamics, behavior, and management of *S. gigas*.

One priority area for future research would be investigating how *S. gigas* interact with marine protected areas (MPAs). As reported herein, *S. gigas* are a highly mobile species capable of long-distance movements. This may make it challenging for fixed spatial protections, such as MPAs, to confer meaningful population-level protection to S. gigas—especially in Southern California, where the average MPA size (all MPA types pooled) is approximately 17.8 km² (CDFW 2024). However, even small MPAs may provide enhanced spatial protection for S. gigas spawning aggregations, as they do for other ecologically similar species that range widely but periodically aggregate, such as the Nassau grouper Epinephelus striatus (Chiappone & Sealey 2000, Kadison et al. 2010, Heppell et al. 2012), red hind grouper Epinephelus guttatus (Beets & Friedlander 1999, Nemeth 2005), and mutton snapper Lutjanus analis (Burton et al. 2005). One notable observation in these data is that nearly 3 times as many S. gigas encounters in natural habitats (i.e. kelp forests and rocky reefs) occurred within MPAs (inclusive of all types of California state MPAs pooled) versus outside of MPAs (Fig. S5). Potential non-mutually exclusive explanations for elevated sightings of S. gigas in MPAs could include (1) that this arises from the known tendency of the recreational SCUBA diving community to preferentially use MPAs in California (Morse et al. 2024); (2) improved foraging opportunities by S. gigas on prey (e.g. lobster and finfish) that have been shown to increase in size and density within California MPAs (Caselle et al. 2015, Hamilton & Caselle 2015, Lenihan et al. 2021); or (3) that added protection from fishing of all types as is afforded in many MPAs. Additional research can resolve the rigor of this putative affiliation between S. gigas and MPAs, discern whether different levels of protection (e.g. marine reserves, parks, conservation areas, or recreational management areas) influence the number of sightings, investigate the causative mechanisms that may be driving these trends, and assess if and how MPAs could better support management goals for this species.

Additional future research on *S. gigas* that could leverage this living data set includes but is not limited to social network analyses, tracking their potential range shift with climate change, exploring interactions between *S. gigas* and recreational fisheries (e.g. analyzing the prevalence of fishing gear and wounds on sighted fish), and quantifying the prevalence of parasites on *S. gigas*, such as the host-specific parasite *Lepeophtheirus longipes* (Passarelli et al. 2021).

4.7. Limitations

We encourage caution when comparing these fishery-dependent and independent data sources. Given the intensity and geographic distribution of submissions, we assume that the majority of the *S. gigas* population in Southern California is being sampled, but we cannot know this for certain. If the majority of adult S. gigas spend a portion of the year in nearshore waters where they are observable by SCUBA divers and recreational fishers, these estimates may be comprehensive for this portion of their range. Conversely, if some individuals reside exclusively in deeper reefs, these models may be potentially underestimating the population. Not much is known about the sister species of S. gigas, the striped jewfish S. doederleini, except that they inhabit depths of 400-600 m (Oh et al. 2021), suggesting that S. gigas may also utilize deepwater reefs, with a portion of their population potentially inhabiting these regions exclusively. Amongst other species in the wreckfish (Polyprionidae) family, the hāpuku Polyprionidae oxygeneios inhabits depths ranging from 50-600 m (Beentjes & Francis 1999), and the bass groper Polyprion americanus inhabits a depth range from 50 to 1000 m (Wakefield et al. 2013). Electronic tagging studies of S. gigas indicate that most tagged individuals, which were tagged in the nearshore, either remain nearshore year-round or migrate to offshore waters in late fall or early winter and return in spring or early summer (Clevenstine & Lowe 2021, Spector et al. 2022). However, further information on these nearshore and offshore and deep reef movements is needed.

Importantly, a shortcoming in this particular study is the omission of sightings data from the S. gigas population found in Mexico (Ramírez-Valdez et al. 2021). Even though the majority of the species' range is in Mexico, very little is known about their transboundary movement (Gaffney et al. 2007, Chabot et al. 2015, Ramírez-Valdez et al. 2021). Additionally, recent research suggests that the S. gigas population might not have experienced as drastic a collapse in Mexican waters and may be at healthier levels, with over 4200 individuals estimated to be landed per year in Mexico (Ramírez-Valdez et al. 2021). Therefore, linking sightings data from Mexico is imperative to generate comprehensive population estimates for the species across its entire range, to understand how local populations respond to differences in national fishery management, and to better comprehend how climate change may influence transboundary S. gigas movement, among other important research questions.

5. CONCLUSIONS

This study presents important findings on the population dynamics and spatial ecology of this Critically Endangered and economically significant species. Alongside these population-centric inquiries, we share new findings regarding temporal and spatial patterns of adult *S. gigas* in Southern California, many of which have a direct significance for their management. This collection of results reaffirms the significant role that non-invasive, cost-effective community science efforts can play in monitoring rare, at-risk, and data-poor species while also highlighting the importance of integrating these findings with more traditional sources to ensure robust assessments.

The transition of this repository into a dynamic, publicly accessible data set creates an opportunity for it to become an increasingly valuable resource for researchers and conservation practitioners (Pettit et al. 2024). By making this data publicly available, we anticipate that a myriad of future insights will emerge from this open and community-driven data resource. Advancing our knowledge of *S. gigas* will be crucial given the dynamically evolving landscape of management for this species, as well as the changing environment in which *S. gigas* occur.

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