Mar Biol
DOI 10.1007/s00227-008-0924-0

RESEARCH ARTICLE

The decline and recovery of four predatory fishes from the Southern California Bight

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Received: 29 May 2007 / Accepted: 29 January 2008

Abstract What to do about fisheries collapse and the decline of large fishes in marine ecosystems is a critical debate on a global scale. To address one aspect of this debate, a major fisheries management action, the removal of gill nets in 1994 from the nearshore arena in the Southern California Bight (34°26′30″N, 120°27′09″W to 33°32′03″N, 117°07′28″W) was analyzed. First, the impetus for the gill net ban was the crash of the commercial fishery for white seabass (*Atractoscion nobilis*; Sciaenidae) in the early 1980s. From 1982 to 1997 catch remained at a historically low level (47.8 ± 3.0 mt) when compared to landings from 1936–1981, but increased significantly from 1995–2004 ($r = 0.89$, $P < 0.01$) to within the 95% confidence limit of the historic California landings. After the white seabass fishery crashed in the early 1980s, landings of soupfin (*Galeorhinus galeus*; Triakidae) and leopard shark (*Triakis semifasciata*; Triakidae) also significantly declined ($r = 0.95$, $P < 0.01$ and $r = 0.91$, $P < 0.01$, respectively) until the gill net closure. After the closure both soupfin and leopard shark significantly increased in CPUE ($r = 0.72$, $P = 0.02$ and $r = 0.87$, $P < 0.01$, respectively). Finally, giant sea bass (*Stereolepis gigas*; Polyprionidae) the apex predatory fish in this ecosystem, which was protected from commercial and recreational fishing in 1981, were not observed in a quarterly scientific SCUBA monitoring program from 1974 to 2001 but reappeared in 2002–2004. In addition, CPUE of giant seabass increased significantly from 1995 to 2004 ($r = 0.82$, $P < 0.01$) in the gill net monitoring program. The trends in abundance of these fishes return were not correlated with sea surface temperature (SST), the Pacific Decadal Oscillation (PDO) or the El Niño/Southern Oscillation (ENSO). All four species increased significantly in either commercial catch, CPUE, or in the SCUBA monitoring program after the 1994 gill net closure, whereas they had declined significantly, crashed, or were absent prior to this action. This suggests that removing gill nets from coastal ecosystems has a positive impact on large marine fishes.

Communicated by J.P. Grassle.

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Introduction

It is increasingly apparent that world-wide fisheries are over-exploited and this exploitation typically begins with large apex predators (Jackson et al. 2001; Dayton et al. 2002). Large fishes, hallmarks of these ecosystems, are declining in abundance at a rapid rate and these declines may be especially problematic for long-lived species with low fecundity such as sharks (Baum et al. 2003; Worm et al. 2006; Myers et al. 2007). While some debate persists about these conclusions (Burgess et al. 2005), there is little evidence that fisheries, especially those of long-lived and slow-growing species, have the ability to rebound from over exploitation (Hutchings 2000). Nonetheless, various management approaches have been designed to address this problem varying in scope from fishery closures to marine reserves.

The Southern California Bight has experienced a variety of cyclic changes and episodic events resulting in reduced productivity. Manifestations of these phenomena were above normal sea surface temperatures, increased frequencies and strength of El Niños, and reduced regional productivity due to the weakening of the California current (McGowan et al. 1998). In addition this transition zone was suffering the possible negative effects of anthropogenic global warming (Hughes 2000; Field et al. 2006). As a result, documented declines in nearshore regional fish populations, both fished and non-fished stocks (Holbrook et al. 1997; Love et al. 1998; Brooks et al. 2002; Pondella et al. 2002), nearshore plankton volumes and larval fish abundances (Stephens and Pondella 2002) leave a potentially grim prognosis for the long-term fate of this ecosystem and certainly a difficult template for fish and fisheries recovery.

In the Southern California Bight the state outlawed gill nets and trammel nets within its waters (within 3 nautical miles of the mainland and 1 nautical mile of the islands) in 1990 (California State Proposition 132). This ban went into effect in 1994 at a time when the nearshore gill net fishery (16.5-cm square mesh, surface to bottom) was targeting white seabass, *Atractoscion nobilis*, in spawning aggregations on rocky headlands and kelp beds. Giant seabass, *Stereolepis gigas*, also aggregated to spawn in these habitats. The nearshore gillnet fishery also landed soupfin, *Galeorhinus galeus*, and leopard shark, *Triakis semifasciatus*. Because the continental shelf in the Southern California Bight is relatively abrupt (Emery 1960; Hickey 1993), this ban moved the commercial fishery out of the nearshore ecosystem throughout most of the region (Fig. 1). On the mainland, this fishery continued primarily on the continental shelf outside of 3 nautical miles offshore of the San Pedro Bay and Ventura Flats. While at the southern California islands, the fishery transitioned to a drift net fishery outside of the closure area.

The effects of this closure are the subjects of this paper. For white seabass and giant seabass this fishery adjustment protected their susceptible spawning aggregations that were traditionally targeted. The apex predator on these nearshore reefs was giant sea bass. This grouper-esque wreckfish virtually disappeared from the commercial fishery in California waters by the mid 1970s. Protected initially in 1981, each commercial boat could land two incidentally captured fishes per day. The California Department of Fish and Game amended this restriction to one fish per day in 1988. While these laws reduced the landings of this species, it did not appear to reduce the mortality associated with the bycatch of the nearshore fishery, as the fishing effort and locations were not restricted (Domeier 2001). Giant sea bass remained rare in southern California throughout the 1970s–1990s (Domeier 2001). While spawning aggregations of soupfin and leopard sharks have not been documented in this region, these sharks were known to utilize the shallow environment of the Southern California Bight for pupping (Ebert 2003). We hypothesize that the gill net fishery was negatively affecting these stocks and that this management action released these nearshore fishes from the major portion of their fishing mortality in 1994. In this paper, we examine this hypothesis by analyzing all of the available fishery-dependent and fishery-independent data from the Southern California Bight.

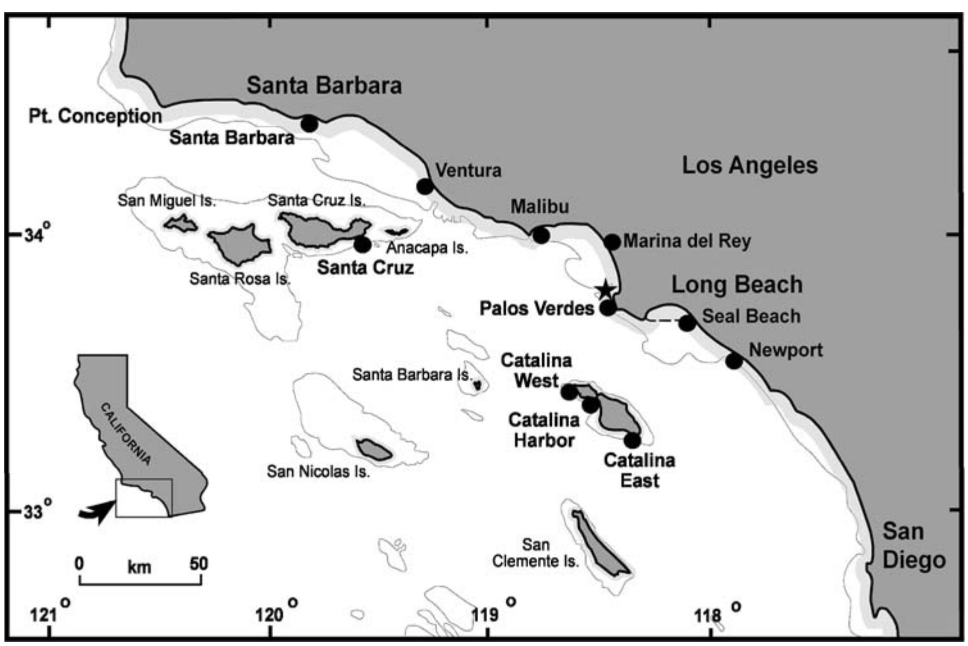

Fig. 1 Locations of ten gill net monitoring stations in the Southern California Bight (*filled circles*). Palos Verdes Point (*filled star*) was the location of a 1974–2004 SCUBA rocky reef monitoring site. *Continuous line* shows 100-m isobath and closure areas are *shaded*

Methods

We downloaded the fishery-dependent data directly from Pacific States Marine Fisheries Commission RecFIN and PacFIN databases at http://www.psmfc.org and collected fishery-independent data from two sources. The first was a gill net monitoring program for white seabass conducted for the Ocean Resource Enhancement Hatchery Program (OREHP), California Department of Fish and Game throughout the Southern California Bight (34°26′30″N, 120°27′09″W to 33°32′03″N, 117°07′28″W). In the months of April, June, August and October from 1995 to 2004, we sampled using horizontal experimental gill nets at 10 stations (Fig. 1). Six nets (replicates) were employed [45.7 in length and 2.4 m in depth, consisting of six, 7.62 m panels of three different mesh sizes (two each of 25.4, 38.2, and 50.8-mm square mesh)] at each station. In 1995 and 1996, the pilot years of this program, 27 and 37 net sets, respectively were completed. Each station was sampled four times ($n = 40$) in the 7 subsequent years. Nets were set perpendicular to the coastline according to a stratified random design. Sets were made an hour before sunset and retrieved an hour after sunrise. Stations were chosen at random from 1 km blocks of coastline using a random number table and nets were set equidistant from each other within each block in sand/rock or reef/kelp habitat in 5–14 m (MLLW) in depth. Marina del Rey and Seal Beach have sand or sand–mud bottoms. At all other stations nets were set on or immediately adjacent to rocky reefs (Pondella and Allen 2000). Catch per unit effort (CPUE) was calculated as the number of individuals captured per set (station).

The second fishery-independent data set used was the SCUBA surveys of Palos Verdes Point (Fig. 1). Fishes were surveyed quarterly on SCUBA from 1974 to 2004 following previously described protocols (Stephens et al. 1984, 1994; Pondella et al. 2002). Palos Verdes Point is a prominent rocky headland that typically supports a vibrant-kelp bed community. We completed three replicated belt transects at the 3, 6, 9 and 13 m isobaths each quarter. For giant sea bass the number per survey were calculated by dividing the abundance by the four annual survey periods.

All the statistical analyses were completed in Statistica 7.0 (Stat Soft, Inc.). Long-term trends were described using regressions. Prior to regression, data were tested for normality using the Shapiro-Wilks w statistic (Legendre and Legendre 1998). All the data sets were found to be normally distributed. Temperature data (Newport Pier), ENSO and PDO indices were downloaded from http://www.sccoos.org, http://www.cpc.ncep.noaa.gov and http://jisao.washington.edu/pdo, respectively. For environmental indices the annual mean value was calculated and correlations to catch data were conducted with a Bonferroni correction ($\alpha' = \alpha/k$) (Legendre and Legendre 1998).

Results

From 1936 to 1981 a mean of 488 mt of white seabass, *Atractoscion nobilis* were landed annually in California. Of this catch 282 mt were from the California fishery with the remaining fishes caught in Mexico and landed in southern California (Fig. 2). The landings of white seabass caught in California began to decline in the mid 1970s. Demonstrating the magnitude of this decline, the 22 lowest catches of the entire time series from California occurred between 1975 and 1997. As the California fishery waned, the pressure on the fishes taken in Mexican water increased dramatically. In 1974, California fishers landed 39.0 mt from Mexico; in 1975 this had increased to 366.0 mt the second highest reported catch from Mexico. From 1975 until the

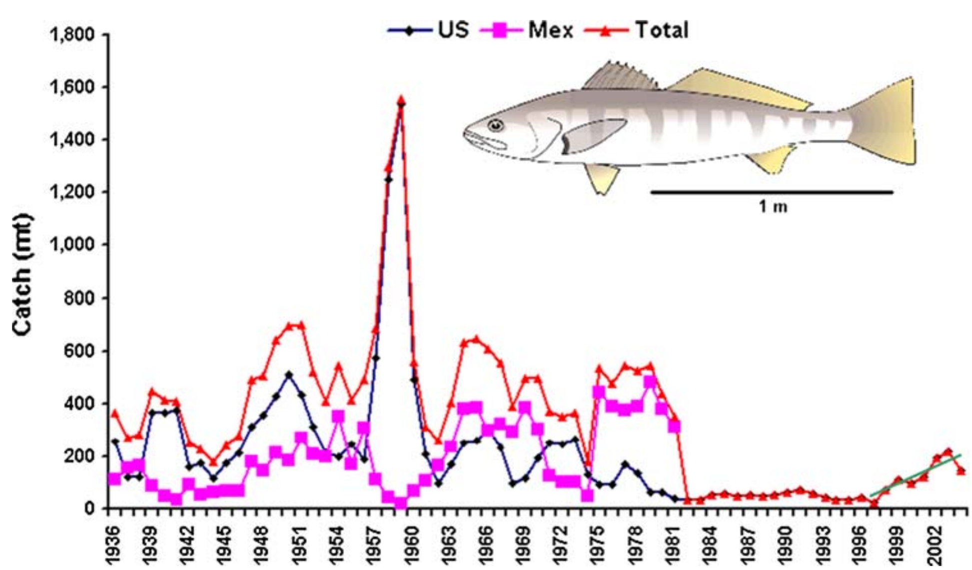

Fig. 2 Commercial catch (mt) of white seabass landed in California from 1936 to 2004. Catch of white seabass caught in the US is in *black diamonds*; catch of white seabass from Mexico landed in the US is in *pink squares*; and, the total landings in the US is in *red triangles*. The significantly positive regression from 1997 through 2004 is in *green*

Table 1 Commercial catch and CPUE from the OREHP program was correlated with the mean annual sea surface temperature (SST) from Newport Beach, the Pacific Decadal Oscillation index (PDO), and the El Niño Southern Oscillation index (ENSO)

Time series	SST		PDO		ENSO	
	r	*P*	*r*	*P*	*r*	*P*
1995–2004						
Soupfin (*Galeorhinus galeus*)	−0.232	0.519	−0.065	0.859	−0.078	0.830
Leopard shark (*Triakis semifasciata*)	−0.508	0.134	−0.374	0.288	0.147	0.686
Giant sea bass (*Stereolepis gigas*)	−0.338	0.340	−0.284	0.426	−0.036	0.921
White seabass (*Atractosciaon nobilis*)	−0.377	0.283	−0.336	0.342	−0.166	0.646
1977–2004						
Soupfin (*Galeorhinus galeus*)	0.107	0.588	−0.087	0.660	0.066	0.738
Leopard shark (*Triakis semifasciata*)	0.388	0.041	0.259	0.184	−0.208	0.289

The correlation coefficients and associate *P* values were presented. We assessed significance at $P = 0.003$ using the Bonferroni correction ($\alpha' = \alpha/k$)

exclusion of California fishers in Mexican waters in 1982 an average of 327.0 mt (±16.9) of white seabass were landed. The four highest catches of Mexican white seabass were recorded during this period. In 1982, the lowest catch (26.1 mt) of the century at that time was landed by the commercial fishery. This was eclipsed in 1997 when only 21.7 mt of white seabass were landed. From 1982 to 1997 catch remained at this low level (47.8 ± 3.0 mt) and then increased significantly from 1995 to 2004 ($r = 0.89$, $P < 0.01$). This trend was not a correlate of SST, PDO or ENSO (Table 1). In 2003, the commercial fishery landed 219.4 mt, which was within the 95% confidence level of the historic pre-collapse California landings.

Prior to 1977 the landings of sharks were reported as a single category. Catch of soupfins, *Galeorhinus galeus*, ranged between 74 and 126 mt from 1977 to 1985 at which point catch declined significantly ($r = 0.95$, $P < 0.01$) through 1995 (Fig. 3). The annual landings for leopard shark ranged between 6.3 and 46.0 mt with a mean of 20.5 mt between 1977 and 1999 (Fig. 3). Catch declined significantly ($r = 0.91$, $P < 0.01$) from 1983 to 1996. With the exception of the landings for leopard shark, *Triakis semifasciata*, which was weakly associated with variation in SST ($r = 0.388$, $P = 0.04$), the commercial catches for soupfin and leopard shark (1977–2004) were not correlated with SST, PDO or ENSO values (Table 1).

Soupfins did not appear in the OREHP gill net monitoring program until 1997, when 0.20 individuals/station were caught. CPUE increased to a high of 0.48 individuals/station in 2002. There was a significant increase in CPUE from 1995 to 2004 (Fig. 4; $r = 0.72$, $P = 0.02$) and this change in CPUE was not correlated with changes in SST, PDO or ENSO (Table 1). The lowest CPUE of large leopard sharks (>1 m TL) was in 1995 and 1997 (0.19 and 0.13 individuals/station, respectively). From 1995 to 2004 CPUE increased significantly (Fig. 4; $r = 0.87$, $P < 0.01$) with the highest CPUE (1.58 individuals/station) in 2003. The CPUE of giant sea bass, *Stereolepis gigas*, did not correlate with the studied oceanographic variables, but did follow the same pattern of a significant positive increase in CPUE over the course of the decade (Fig. 4; $r = 0.82$, $P < 0.01$).

Beginning in July 1974, 1,319 replicated belt transects were completed at Palos Verdes Point, Rancho Palos Verdes (Fig. 1) through November 2004. Conditions permitting an equal number of transects were completed in all four seasons during this time period by trained scientific divers. 131,691 fishes representing 82 species and 35 families were surveyed. The first giant sea bass observed during this monitoring program was on May 29, 2002 at a depth of 7 m and then again on September 24, 2002. This was 21 years after receiving protected status and 12 years after the gill net closure. Giant sea bass were observed at depths between 3 and 9 m in May and September in 2003 and 2004 (Fig. 5).

Discussion and Conclusions

The commercial fishery for white seabass, *Atractoscion nobilis*, based in southern California, historically consisted of landings from the Southern California Bight and Mexico (Fig. 2). The Mexican catch was fishes caught in Mexican waters by California fishers and landed in southern California (Vojkovich and Crooke 2001). By 1981 that fishery had collapsed to 10% of its historic catch and remained at this level for 15 years. This situation led to the 1994 nearshore gill net ban. In 1982 the Mexican government excluded the US fleet concentrating the fishery in California waters (Vojkovich and Crooke 2001; Allen et al. 2007). The increased catch beginning in 1997 could have been partially due to fishes migrating from the south, where fishing mortality had likely declined after the exclusion of the US fleet, but this was 15 years later. This lag suggests that migration from Mexico alone cannot explain the increased catch after 1997.

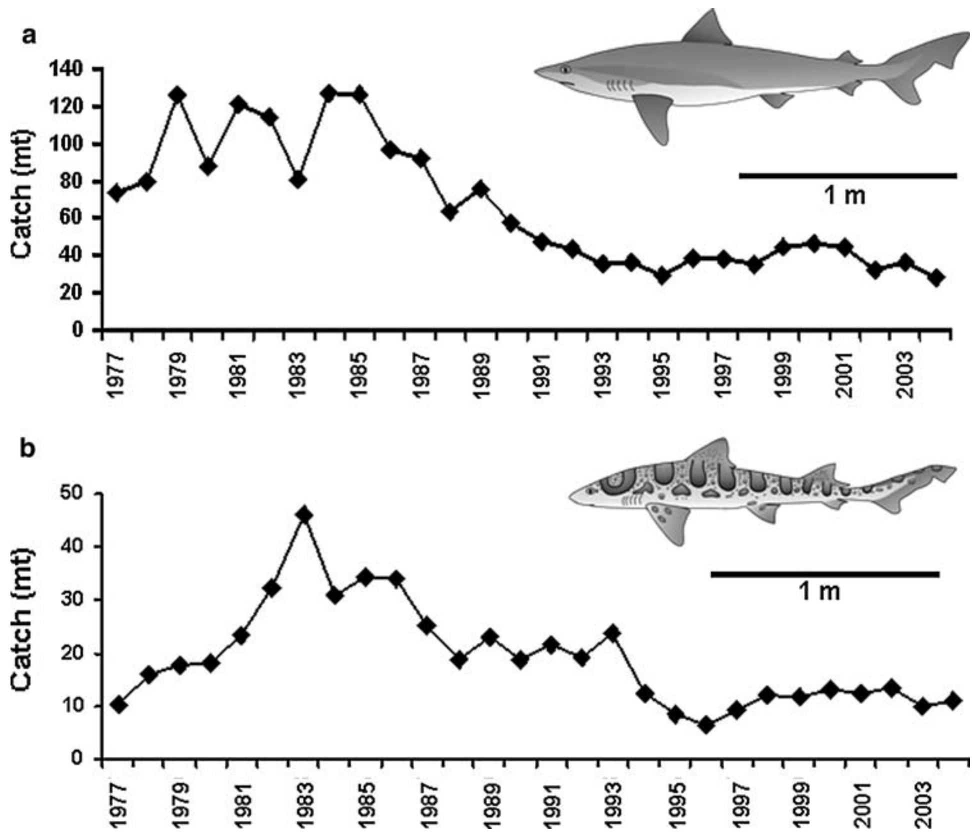

Fig. 3 Commercial catch (mt) of soupfin **a** and leopard shark **b** landed in California from 1997 to 2004

The increase, coming 20 years after a regime shift in the Southern California Bight (Stephens et al. 1984; Holbrook et al. 1997) was also not explained by the environmental variables tested in the present study, suggesting that the gill net closure was largely responsible for their recovery.

The fishery for soupfin, *Galeorhinus galeus*, also based in southern California collapsed due to over-fishing 1938–1944 (Ripley 1946). Annual landings prior to this exploitation (1930–1937) were 285 mt (±27 mt). This fishery, which peaked in 1939 with nearly 4,186 mt recorded, landed 19,096 mt from 1938 to 1944 and disappeared by the end of the war (Ripley 1946). Commercial landings of soupfin were not reported again until 1977 when they reemerged at less than 50% (range 74–126 mt) of their pre WWII landings until 1985 at which point catches started declining precipitously through 1995 (Fig. 3). At no point did the fishery return to the pre-exploitation level, and these nearshore sharks were not observed in local coastal monitoring programs during this period (Stephens et al. 1984, 1994). Soupfins absent in 1995 and 1996 in the gill net monitoring program significantly increased in CPUE beginning in 1997. They were reported in kelp beds for the first time during scientific SCUBA monitoring programs in 2002 (Pondella et al. 2005).

The commercial catch of leopard shark, *Triakis semifasciata*, followed a similar pattern. Catch declined significantly from 1983 to 1996. It appeared that fishers continued to target soupfin, leopard shark and giant sea bass, *Stereolepis gigas*, as the white seabass fishery waned and this pressure through the 1980s and early 1990s led to the precipitous decline or continued suppression of these stocks. Adult leopard sharks as well as giant sea bass followed the soupfin and white seabass pattern of a significant positive linear increase in CPUE post-closure (Fig. 3). What was even more striking than these increases in CPUE was the dramatic appearance of giant sea bass (Fig. 5) on nearshore reefs where they had been absent for decades (Stephens et al. 1994; Holbrook et al. 1997). By 2004 the numbers of four major predatory fish had increased in this nearshore, rocky reef ecosystem.

To determine if these trends were due to population dynamics of these taxa, optimally demographic parameters such as net reproductive rates (R_0), generation times (G) and instantaneous population growth coefficients (r), at a minimum, would be pertinent in this evaluation. Unfortunately, none of these parameters are known for these southern Californian stocks. For leopard sharks in central California, demographic parameters were $R_0 = 4.467$, $G = 22$–35 years and $r = 0.067$ (Cailliet 1992). This gave a population doubling time estimate of 10 years with a 7% increase per year in the absence of fishing mortality. The increase we observed was 14.8% per year, approximately double the central coast estimate. While these study methods differed, the higher rate of return described for southern

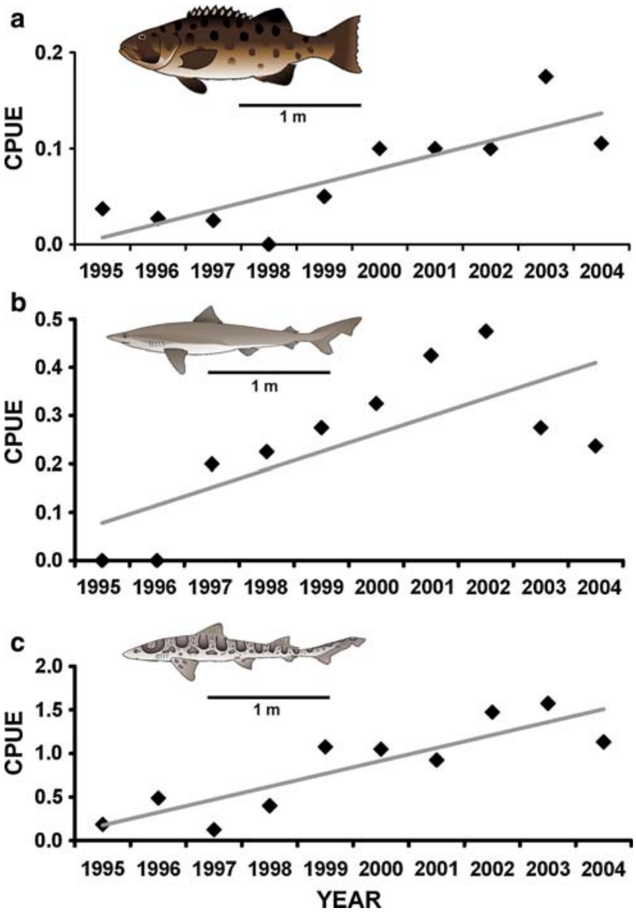

Fig. 4 Catch per unit effort (individuals/station) from the OREHP monitoring program of giant sea bass (**a**), soupfin (**b**) and leopard shark (**c**) from 1995–2004 for the Southern California Bight. Regression statistics for soupfin ($r = 0.72$, $P = 0.02$), leopard shark ($r = 0.87$, $P < 0.01$), and giant sea bass *(*$r = 0.82$, $P = 0.0034$*)* were all significant

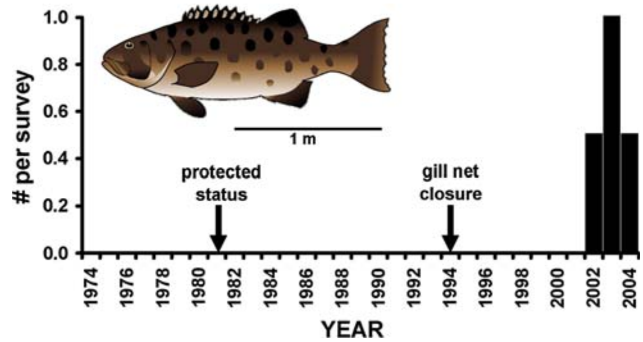

Fig. 5 Number of giant sea bass observed per quarterly survey at Palos Verdes Point, Rancho Palos Verdes, Los Angeles County from 1974 to 2004

California stock could be explained by variation in demographic parameters for southern California that may include the possibility of faster growth, different migration rates and fecundity. While these parameters and the demographic parameters of the soupfin, giant seabass and white seabass were not known for southern California, the increases reported herein do not seem unreasonable and highlight the need for demographic surveys in the Southern California Bight. Nonetheless, four very different species responded similarly and the most parsimonious interpretation of these changes is the gill net closure.

An alternative explanation for the return of these species to the nearshore arena of the Southern California Bight may be immigration into the region, perhaps a result of an environmental change such as warming in the Southern California Bight. These patterns could also be a result of changes in the onshore/offshore distribution of the fishes. The Southern California Bight, classically defined as a transitional zone between the cold temperate Oregonian fauna to the north and the warm temperate San Diegan fauna, is susceptible to faunal shifts associated with macro scale environmental changes (Horn et al. 2006). The regime shifted due to the Pacific Decadal Oscillation (PDO) in 1977 (McGowan et al. 1998; Chavez et al. 2003) preempting a nearshore ichthyofauna transition to a warm temperate fauna (Stephens et al. 1994; Holbrook et al. 1997). Above average sea surface temperatures (SST) augmented with an increased frequency and strength of El Nino Southern Oscillation (ENSO) events characterized the Southern California Bight. As with most large marine fishes, these taxa have high vagility and their abundance may be affected by such perturbations. Clearly these fishes have the potential to immigrate and emigrate from this area. Thus, it is not surprising that catch and CPUE of these fishes from 1977 to 2004 was not correlated with environmental variables (SST, PDO index, ENSO index; Table 1). The major regime shift had already occurred. SST was weakly associated with the landings of leopard shark (1997–2004), an indicator of possible onshore/offshore movement and/or coastal migratory patterns, a finding consistent with a previous study in San Francisco Bay (Smith and Abramson 1990). While these analyses were not optimal tests of these alternative hypotheses, longer time series and more demographic information would be appropriate; however, these kinds of data are not available. Nonetheless, it is reasonable to conclude that the release in fishing pressure was the single greatest contributor to their recovery and return to the Southern California Bight.

In 1994, when the gill net ban was implemented, all of these nearshore fisheries had either collapsed or were declining significantly. Exclusion of gill nets in state waters, while a contentious issue at the time, did not negatively affect the commercial white seabass fishery and it now appears to be directly responsible for its recovery. In addition, increases in abundance of four large predatory fishes in these nearshore rocky reefs occurred at a time of declining fisheries productivity throughout California waters, especially in the Southern California Bight. Thus, in this coastal area, the banning of the gill net fishery was effective in allowing the recovery of several large apex predators.

Acknowledgments The Ocean Resource Enhancement Hatchery Program, California Department of Fish and Game, funded the nearshore gill net monitoring program. The SCUBA surveys over the last three decades were funded by several sources including Southern California Edison, the US Army Corps of Engineers, California Department of Fish and Game, and Chevron Texaco Products Company, where we would like to acknowledge the support of Wayne Ishimoto. All research was conducted under permits to both authors from the California Department of Fish and Game.

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