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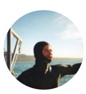
Article *in* Canadian Journal of Fisheries and Aquatic Sciences · April 2011

DOI: 10.1139/cjfas-58-9-1773



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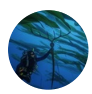

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The use of marine reserves in evaluating the dive fishery for the warty sea cucumber (*Parastichopus parvimensis*) in California, U.S.A.

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Abstract: Management of sustainable fisheries depends upon reliable estimates of stock assessment. Assessment of many stocks is based entirely on fishery-dependent data (e.g., catch per unit effort), which can be problematic. Here we use fishery-independent data on stock size, collected within and outside of no-take reserves before and after the onset of fishing, to evaluate the status of the dive fishery for warty sea cucumbers, *Parastichopus parvimensis*, in southern California. Long-term monitoring data showed that abundance decreased throughout the Channel Islands within 3–6 years after the onset of fishing. No significant changes in the abundance of *P. parvimensis* were observed at the two non-fished reserve sites, although densities tended to increase following onset of the fishery. Before–after, control–impact (BACI) analyses of seven fished and two non-fished sites implicated fishing mortality as the cause of 33–83% stock declines. In sharp contrast, stock assessment based on CPUE data showed no declines and a significant increase at one island. To date, most discussion on marine reserves has focused on the protection and enhancement of exploited populations. Our study demonstrates the critically important, but often overlooked, role that marine reserves can play in providing reliable information on stock assessment.

Résumé : Une gestion durable des pêches nécessite des estimations fiables de l'évaluation des stocks. L'évaluation de nombreux stocks est basée entièrement sur des données provenant de la pêche commerciale (e.g. la capture par unité d'effort), ce qui peut poser un problème. Nous utilisons ici des données de taille du stock indépendantes de la pêche, récoltées à l'intérieur et à l'extérieur des zones protégées sans récolte, avant et après le début de la pêche, pour évaluer l'état de la pêche commerciale en plongée du concombre de mer verruqueux *Parastichopus parvimensis* dans le sud de la Californie. Des données provenant d'un suivi à long terme indiquent que la densité a diminué dans toute la région des îles de Santa Barbara en moins de trois à six ans après le début de la pêche. Aucun changement significatif de densité de *P. parvimensis* n'a été observé à deux réserves sans pêche, bien que les densités aient eu tendance à augmenter après le début de la pêche. Des analyses de type BACI (avant-après; témoin-impact) de sept sites de pêche et de deux sites non-exploités identifient la mortalité due à la pêche comme la cause de 33 à 83% du déclin des stocks. En revanche, les estimations de stock basées sur des données de CPUE ne révèlent aucun déclin et elles indiquent même une augmentation significative de densité à une des îles. Jusqu'à présent, la plupart des discussions sur les réserves marines ont traité de la protection et du redressement des populations exploitées. Notre étude démontre le rôle critique, mais souvent méconnu, que les réserves marines jouent en fournissant des données fiables sur l'estimation des stocks.

[Traduit par la Rédaction]

Introduction

Managing a sustainable fishery requires some means of assessing the current status of a population and how it changes over time. Only then can the impacts of fishing be evaluated and effective fishery management implemented. Typically, stock assessment is based on fishery-dependent data such as catch per unit effort (CPUE), which are much less expensive to obtain than direct data on stock size. The use of CPUE data to evaluate the status of exploited stocks can be problematic because of difficulties in standardizing effort (particularly in light of changing fishing technologies and catch efficiency) and the reluctance of some fishermen to accurately report their catch for fear of divulging trade secrets (Hillborn and Walters 1992).

Comparisons involving no-take marine reserves that make use of fishery-independent data (e.g., direct estimates of population abundance) offer an alternative approach to stock

Received February 21, 2001. Accepted July 4, 2001. Published on the NRC Research Press Web site at http://cjfas.nrc.ca on August 17, 2001.
J16232

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 DOI: 10.1139/cjfas-58-9-1773

assessment. Numerous studies have reported higher abundances and larger sizes of fish inside no-take reserves relative to outside, while others have reported increases in the abundance and size of fish in an area following its designation as a no-take zone (reviewed in Rowley 1992; Halpern 2002). Such comparisons between impacted (i.e., fished) and control (i.e., non-fished) areas that are separated either in space or time are generally considered inadequate for assessing most environmental impacts, which are best evaluated by comparing population changes in impacted and non-impacted areas before and after the impact occurs (Underwood 1993; Stewart-Oaten and Bence 2001). Unfortunately, the use of a before–after, control–impact design (i.e., BACI) to evaluate impacts from fishing is not feasible for most established fisheries owing to a lack of data on stocks before fishing and (or) the lack of data from a suitable non-fished control site. Such may not be the case for emerging fisheries, however, and no-take reserves that use adequate monitoring programs offer a powerful means of assessing the status of harvested stocks and their likelihood for sustainability.

Here we take a BACI approach to evaluate the status of the recently developed dive fishery for the warty sea cucumber, *Parastichopus parvimensis*, in southern California. Fishery-independent data used in our analyses were obtained from long-term kelp forest monitoring programs conducted by the National Park Service (NPS) and the United States Geological Survey (USGS). These data consisted of estimates of *P. parvimensis* abundance in fished and non-fished areas before and after the onset of the fishery. As mentioned above, stock assessment based on fishery-dependent data is often suspect. To evaluate if this was true for the warty sea cucumber fishery, we compared results on the status of the fishery based on long-term population monitoring with those estimated by data on CPUE.

Methods

History of the fishery

Parastichopus parvimensis is a large sea cucumber that occurs from Baja California to Monterey Bay, California. It is most common south of Point Conception (Brumbaugh 1980) and lives in a variety of habitats from the intertidal zone to at least 30-m depth (Muscat 1983; Kalvass 1992). It is most abundant on shallow reefs where it feeds on organic detritus contained in thin layers of soft sediments that accumulate on the bottom. In 1982, an experimental dive fishery for *P. parvimensis* began in southern California. Like other large sea cucumbers, *P. parvimensis* is harvested for its body wall and musculature, which are dried and exported to Asian markets. Initially, fishing was sporadic, often consisting of a single day's catch by a single boat. Harvesting permits were required beginning in 1992 as the Asian demand for imported sea cucumbers steadily increased as a result of the overharvesting of traditional fishing grounds in the Indo-Pacific. The majority of the permit holders in southern California are also sea urchin fishermen and most of the landings for *P. parvimensis* take place from June through August when the commercial harvest of sea urchins is prohibited (D. Ono, unpublished data). The commercial fishery for *P. parvimensis* began in earnest in the mid-1990s and is largely unregulated (i.e., there are no restrictions on the size, season, or amount of harvest). Most of the catch is taken from the Channel Islands located off the coast of Santa Barbara, California.

Data on *P. parvimensis* abundance

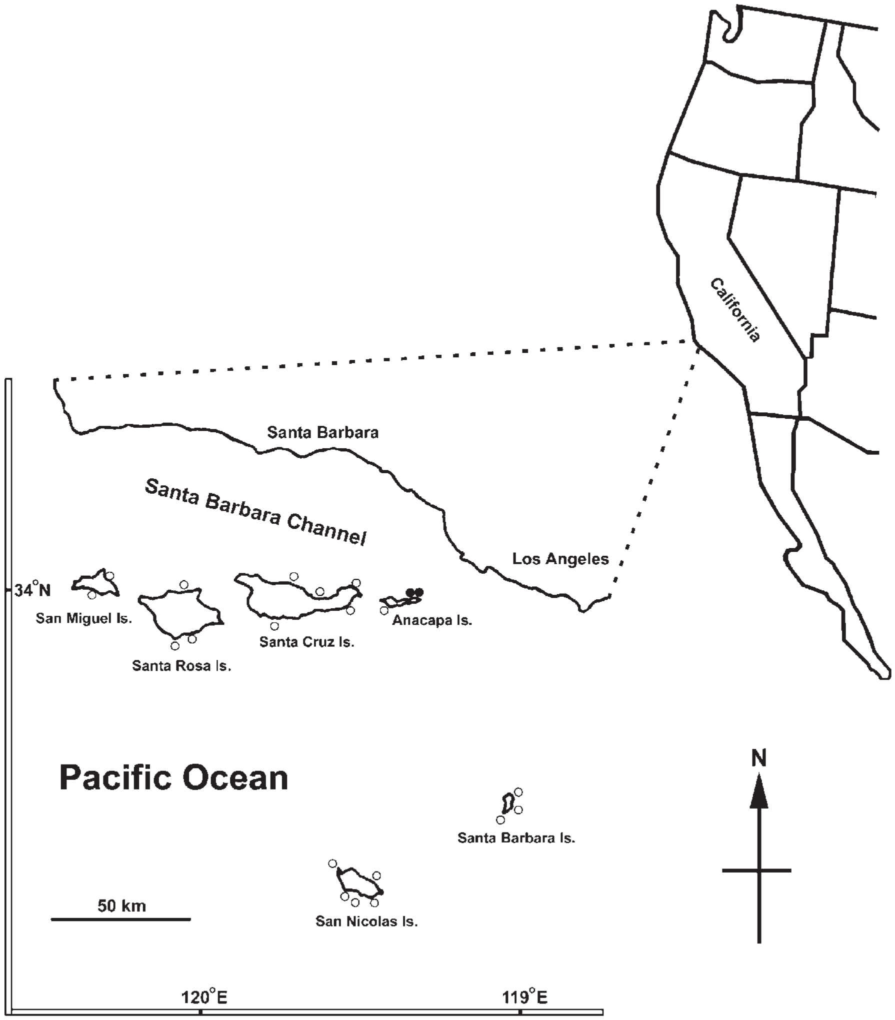
Data on *P. parvimensis* abundance used in our evaluation of the fishery were collected at the five northern Channel Islands (San Miguel Island (SMI), Santa Rosa Island (SRI), Santa Cruz Island (SCI), Anacapa Island (ANI), and Santa Barbara Island (SBI)) and at nearby San Nicolas Island (SNI) (Fig. 1). Approximately two-thirds of the *P. parvimensis* harvest in California has come from these six islands (California Department of Fish and Game (CDFG) CMASTR data). Data on *P. parvimensis* abundance from the northern Channel Islands were obtained from the NPS, which has collected data annually on the abundance of a wide variety of species that inhabit kelp forests at 16 sites around the five islands since 1982 (Davis et al. 1997). Two of the sites on the northeast side of ANI (Landing Cove and Cathedral Cove) are in a marine reserve where fishing has not been allowed since 1978. Fishing is permitted at the other 14 sites. *Parastichopus parvimensis* abundance at each of the 16 sites was estimated from non-destructive sampling of non-cryptic individuals. Data were collected once per year in the summer (June to August) by divers in 1 m^2 or 2 m^2 quadrats (n = 12 to 40 quadrats per site), which were randomly placed within a permanent 100 m × 20 m sampling area that defined each site. Sites for long-term monitoring were not specifically chosen by the NPS for the purpose of monitoring *P. parvimensis* abundance, but rather they were selected to represent the broad range of environmental conditions and biological assemblages in the park (Davis et al. 1997). Monitoring sites were generally located in areas of continuous reef with relatively low coverage of sand and cobble. The location and physical characteristics of each site are summarized in Reed et al. (2000).

Comparable data on *P. parvimensis* abundance for five sites off SNI were obtained from the USGS. The five sites were distributed around the island and were chosen by the USGS to represent the broad range of oceanographic conditions at SNI. The USGS has been conducting semi-annual (April and October) monitoring of the kelp forest communities at these sites since 1980. On each survey, *P. parvimensis* are counted by divers in five permanent 10 m × 2 m quadrats at each site using a non-destructive sampling procedure similar to that used by the NPS. We used only data from October surveys at SNI in our analyses to maintain compatibility with annual data from the northern Channel Islands. Fishing is permitted at all sites at SNI.

Before–after, control–impact (BACI) analyses

BACI analyses were done to estimate the effects of fishing on populations of *P. parvimensis* at the permanent monitoring sites on the six islands. The BACI analysis and its assumptions are described in detail elsewhere (Stewart-Oaten et al. 1986; Stewart-Oaten and Bence 2001) and are summarized briefly below. In a BACI design, control and impact sites are sampled concurrently on multiple dates before and after an impact (i.e., onset of fishing) occurs. Each survey date during the before period provides an estimate of the spatial difference between the impact and control sites. Continued sampling during the after period yields a time series of differences between the impact and control sites from the before and after periods. In our analyses, the impact was the commercial dive fishery for *P. parvimensis*. The start of the commercial fishery marked the beginning of the after period and was defined for each island as the year in which the cumulative catch equaled or exceeded 5% of the total cumulative catch through 1999. The no-take reserves at Landing Cove and Cathedral Cove on ANI served as our control (non-fished) sites, while all other sites were considered impact (i.e., fished) sites that varied in their intensity of fishing pressure. We used the mean annual abundance of *P. parvimensis* at Landing and Cathedral Coves as the control in all BACI analyses. The mean difference between the impact and control sites in the before period was compared statistically with that in the after period using a t test to determine if there has been an impact and the

Fig. 1. Map of the study area showing the locations of long-term monitoring sites where data on the abundance of *P. parvimensis* have been collected. Fishing has been prohibited since 1978 in the two sites at Anacapa Island designated by solid circles. Sites designated by open circles have had no restrictions on the *P. parvimensis* harvest.



relative size of its effect. Separate BACI analyses were done for each fished site. In all BACI analyses, *t* tests were one-tailed to test the null hypothesis that there was no decline in the abundance of *P. parvimensis* following the onset of commercial fishing.

Some of the underlying assumptions of the BACI model are (*i*) the data are additive (i.e., the effects of location (fished vs. non-fished) and period (before vs. after the onset of fishing) are constant and can be added to the underlying mean abundance to obtain the expected abundances in the different cells of the design); (*ii*) the random errors in the differences between fished and non-fished sites are independent; and (*iii*) the random errors are normally distributed with equal variances and means of zero. BACI analyses were performed only on those sites for which data met these assumptions. The statistical procedures that we used to test these assumptions are described in Stewart-Oaten (1987), Carpenter (1989), and Stewart-Oaten et al. (1992).

Absolute changes in the abundance of *P. parvimensis* following the onset of fishing were evaluated with *t* tests that compared the average abundance of *P. parvimensis* in the before and after periods. Such before–after *t* tests were done for all 19 fished sites and the average of the two non-fished control sites. In all before–after comparisons, *t* tests were two-tailed to test the null hypothesis of no change in the abundance of *P. parvimensis* following the onset of commercial fishing.

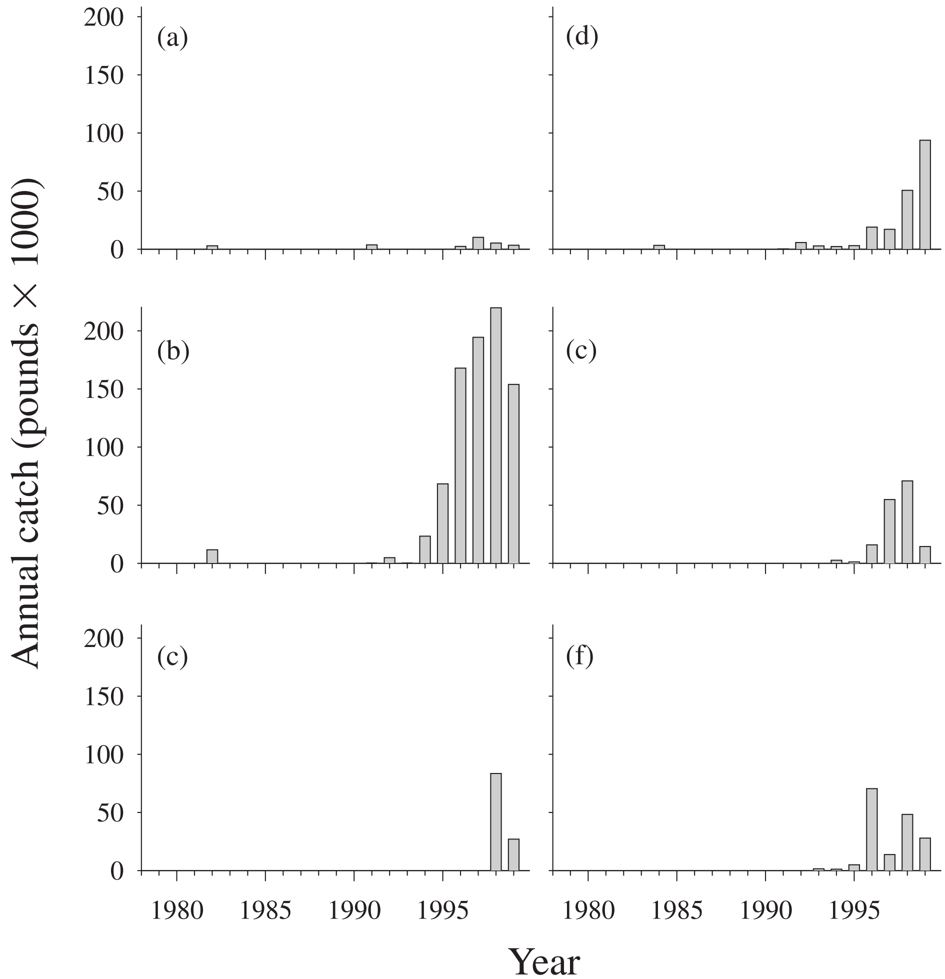
Data on CPUE

Data on CPUE were obtained from the CDFG CMASTR database and represent catch (in pounds) per boat per day. The data are based on landings weighed at the dock by the processor, who records the data on a “fish ticket” which is given to the CDFG. Fishermen are required to assign their landings to 10 × 10 nautical mile fish blocks and report the fishing effort (i.e., number of boat days) allocated to their catch. Because of the relatively low spatial resolution of these data, CPUE could only be estimated at the scale of islands and not sites within islands. The CMASTR data used in our analyses of CPUE covered the entire after period for each of the six islands.

Results

As mentioned above (see Methods), we chose the first year in which the cumulative catch on each island equaled or exceeded 5% of the total cumulative catch as the beginning of the after period in our BACI analyses. This occurred in 1993 for SCI and SRI, 1995 for SMI, ANI, and SNI, and 1997 for SBI (Fig. 2). As of 1999, approximately 56% of the total cumulative catch taken from all six islands was taken

Fig. 2. Annual catch of warty sea cucumbers (*P. parvimensis*) by island at five northern Santa Barbara Channel Islands and San Nicolas Island: (*a*) San Miguel Island; (*b*) Santa Cruz Island; (*c*) Santa Barbara Island; (*d*) Santa Rosa Island; (*e*) Anacapa Island; and (*f*) San Nicolas Island.



from SCI, followed by 13% from SRI, 11% from each of SNI and ANI, 7% from SBI, and <2% from SMI.

Monitoring data suggest that the abundance of *P. parvimensis* decreased throughout the islands at fished sites following the onset of fishing (Table 1). Within 3 to 6 years of the start of the fishery, significant declines in sea cucumber abundance (ranging from 25% to 100%) occurred at 8 of 19 sites spread among five islands. An additional seven sites showed non-significant (i.e., $P > 0.05$) reductions in sea cucumber abundance, while four sites showed non-significant increases. In contrast, there was a trend of increasing abundance of *P. parvimensis* at the two non-fished reserve sites after 1993, 1995, and 1997, which marked the onset of fishing at SCI and SRI, ANI, SNI, and SMI, and SBI, respectively.

Only 7 of the 19 fished sites met the underlying assumptions of BACI (Table 2). The use of BACI to estimate impacts from fishing was excluded at nine sites because the data in the before period were non-additive. Non-additivity can be expected in cases where abundance at the impact site is consistently much lower than at the control site. Under such conditions, the difference between impact and control sites will be much greater during years when sea cucumber abundance is generally high compared with years when sea cucumber abundance is generally low. Such was the case for eight of the nine sites that failed to meet the assumption of additivity. These eight sites were probably not subjected to intensive fishing pressure because of their inherently low sea cucumber stocks, and five of them showed no significant difference in sea cucumber abundance between the before and after periods (in all cases, $P > 0.10$, Table 1). Pelican Bay on Santa Cruz Island was the lone exception to this rule as data from this site were non-additive despite relatively high abundances in the before period. The high densities of *P. parvimensis* at Pelican Bay and its close proximity to mainland ports make it an attractive site for fishing. The significant 65% decline in sea cucumber abundance observed at this site from the before to after periods (Table 1) coupled with the high landings reported for Santa Cruz Island (Fig. 2) suggest that it was probably heavily fished. Three additional sites with relatively high densities of *P. parvimensis* (Nav Fac, Admiral's Reef, and Gull Island) were excluded from analyses using BACI because they showed positive trends in abundance during the before period (Table 2).

All seven sites that met the underlying assumptions of BACI showed significant declines (relative to non-fished sites) following the onset of fishing (Table 3, Fig. 3). The

Table 1. Percentage change in the mean annual abundance of warty sea cucumbers (*P. parvimensis*) at the 19 fished and 2 non-fished island sites following the onset of fishing.

Island	Site	Onset of fishing	% change	*t* value	*P*
	Fished				
San Nicolas	West Dutch Harbor	1995	–47	–3.1	**0.007**
	East Dutch Harbor	1995	–3	–0.2	0.857
	West End	1995	–24	–1.3	0.209
	Nav Fac	1995	+31	0.7	0.519
	Daytona Beach	1995	+12	0.8	0.452
Santa Barbara	Arch Point	1997	–48	–1.6	0.128
	Cat Canyon	1997	+52	2.0	0.071
	SE Sea Lion Rookery	1997	–31	–2.7	**0.017**
Anacapa	Admiral's Reef	1995	+18	1.4	0.171
Santa Cruz	Scorpion Anchorage	1993	–62	–3.0	**0.009**
	Pelican Bay	1993	–73	–3.4	**0.002**
	Fry's Harbor	1993	–77	–7.8	**<0.002**
	Yellowbanks	1993	–37	–1.9	0.082
	Gull Island	1993	–34	–1.3	0.225
Santa Rosa	Rodes Reef	1993	–100	–3.0	**0.010**
	Johnson's Lee North	1993	–49	–3.5	**0.003**
	Johnson's Lee South	1993	–9	–0.2	0.867
San Miguel	Hare Rock	1995	–70	–2.5	**0.026**
	Wyckoff Ledge	1995	–10	–0.5	0.649
	Non-fished				
Anacapa	Landing and Cathedral Coves	1993	87	0.5	0.648
		1995		2.1	0.052
		1997		1.1	0.272

Note: Onset of fishing is defined as the first year that the total cumulative catch at an island equaled or exceeded 5%. Percentage change at the non-fished sites was based on an average of Landing and Cathedral Coves. Because the onset of fishing varied among islands, the percentage change in *P. parvimensis* abundance at Landing and Cathedral Coves was calculated for three different time periods to allow comparisons with sites having different onsets of fishing. *P* values < 0.05 are bolded.

Table 2. Results for tests of additivity and no trends in the data on the annual abundance of warty sea cucumbers (*P. parvimensis*) at 19 island sites subjected to fishing, collected before the onset of fishing.

Island	Fished site	Additivity	No trends	Abundance prior to fishing
San Nicolas	**West Dutch Harbor**	**Yes**	**Yes**	**5.36**
	East Dutch Harbor	No	Yes	2.27
	West End	No	Yes	1.68
	Nav Fac	Yes	No (+)	5.69
	Daytona Beach	No	Yes	1.45
Santa Barbara	Arch Point	No	Yes	2.39
	Cat Canyon	No	Yes	3.02
	SE Sea Lion Rookery	**Yes**	**Yes**	**9.28**
Anacapa	Admiral's Reef	Yes	No (+)	12.47
Santa Cruz	**Scorpion Anchorage**	**Yes**	**Yes**	**6.83**
	Pelican Bay	No	Yes	10.67
	Fry's Harbor	**Yes**	**Yes**	**21.11**
	Yellowbanks	**Yes**	**Yes**	**7.50**
	Gull Island	Yes	No (+)	10.28
Santa Rosa	Rodes Reef	No	Yes	0.23
	Johnson's Lee North	**Yes**	**Yes**	**5.23**
	Johnson's Lee South	**Yes**	**Yes**	**1.74**
San Miguel	Hare Rock	No	Yes	1.72
	Wyckoff Ledge	No	Yes	2.05

Note: "Yes" indicates that the data for a given site passed the underlying assumption of BACI and "No" indicates that the data failed the underlying assumption of BACI. (+) indicates trend was positive. BACI tests to evaluate the effects of fishing on the abundance of *P. parvimensis* were done for sites that passed tests for both additivity and no trends (shown in bold). Abundance prior to fishing is the mean annual density of *P. parvimensis* (number·10 m^{-2}) in the before period.

Table 3. Relative changes in the density of warty sea cucumbers (*P. parvimensis*) following the onset of commercial fishing at seven island sites.

Island	Site	Relative % change	df	*t* value	*P*
San Nicolas	West Dutch Harbor	–67	15	–5.3	<0.001
Santa Barbara	SE Sea Lion Rookery	–52	16	–1.7	0.027
Santa Cruz	Scorpion Anchorage	–76	16	–3.8	<0.001
	Fry's Harbor	–83	7	–4.9	<0.001
	Yellowbanks	–53	12	–2.1	0.015
Santa Rosa	Johnson's Lee South	–33	16	–1.4	0.044
	Johnson's Lee North	–51	16	–2.9	0.003

Note: Relative changes represent the difference in sea cucumber density between a given fished site and the mean of non-fished control sites in the before period relative to the difference between the fished and non-fished sites in the after period. Separate BACI *t* tests that compared the average deltas (fished – non-fished) from the before and after periods were used to determine the statistical significance of relative change at each site. *t* tests are one-tailed (testing for declines at fished sites).

decline in the relative abundance of *P. parvimensis* at these sites (as estimated by BACI) ranged from 33% to 83%. The smallest relative decline occurred at Johnson's Lee South, which supported sparse densities of *P. parvimensis* compared with the other six sites evaluated using BACI (Table 2 and Fig. 3). This was the only one of the seven sites that did not show a significant decline in the absolute density of *P. parvimensis* from the before to after period (Table 1). The absolute density of *P. parvimensis* at the other six sites declined by 25% to 67% following the onset of fishing.

An evaluation of *P. parvimensis* stocks based on analyses using fishery-dependent data contrasted sharply with that based on BACI and before–after analyses of fishery-independent data. CPUE did not decline at any of the six islands during the 3- to 6-year period following the onset of fishing despite a general decline in *P. parvimensis* abundance during this time (Fig. 4). Surprisingly, Santa Rosa Island actually showed a significant increasing trend in CPUE.

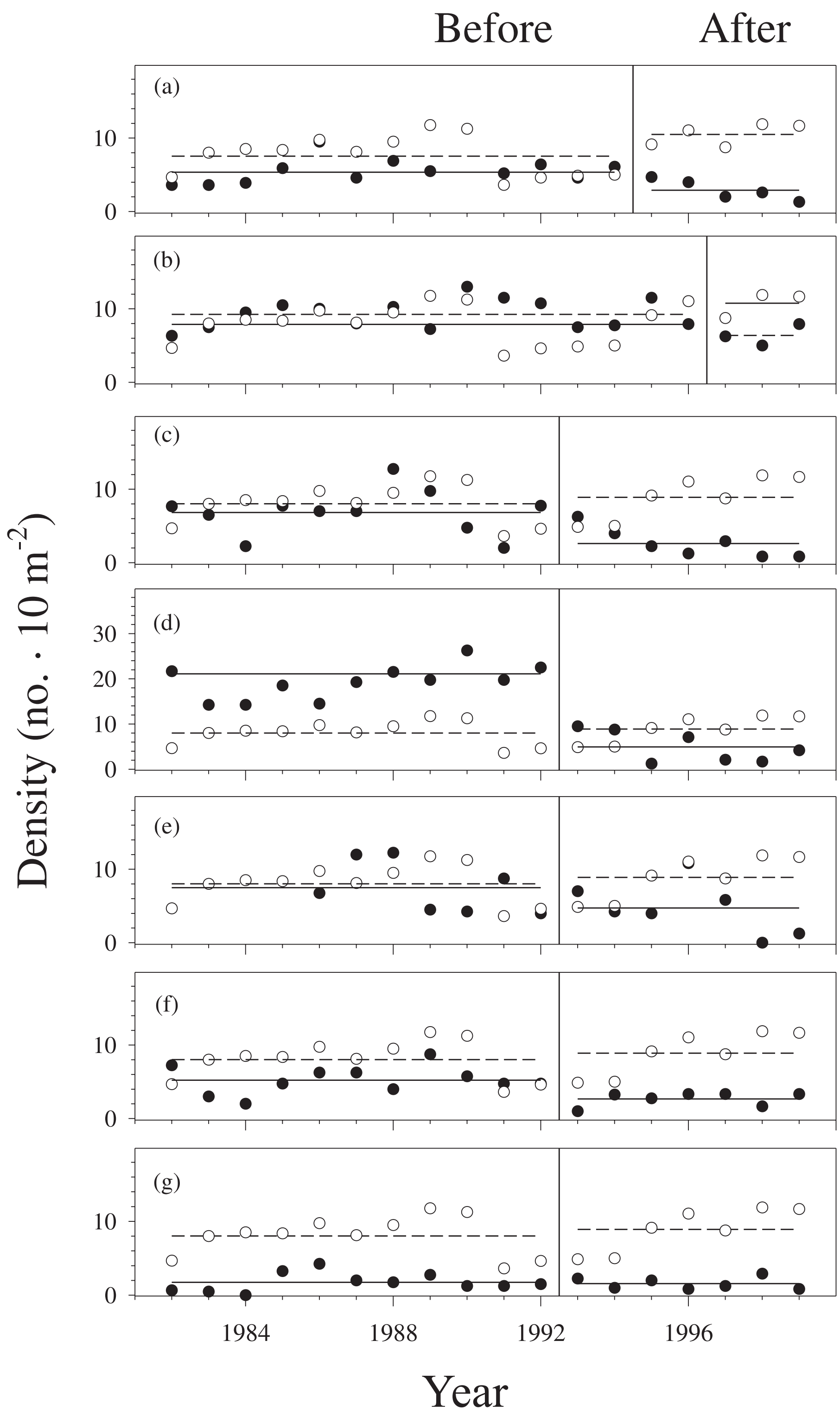
Discussion

Data on the abundance of *P. parvimensis* collected from the NPS and the USGS show that substantial and statistically significant population declines occurred in the Channel Islands within 3 to 6 years of the rapid expansion of the commercial dive fishery. The BACI analyses done on seven sites implicate fishing as a significant cause of these declines. An explicit assumption in our BACI analyses was that no fishing occurred in the two no-take zones on Anacapa Island, which were used as control sites. It is worth noting that any poaching done at these sites would have biased the BACI analyses and led to an underestimate of the impact of fishing. Thus, in this sense, the relative declines in sea cucumber abundance identified by BACI should be viewed as being conservative.

Regulations for most fisheries (including the dive fishery for *P. parvimensis*) require fishermen to report the amount, location (i.e., fish block), and date of their catch. These data are frequently used to calculate CPUE, which is used as an indicator of stock size. The expectation is that CPUE should decline with decreases in stock size. Our analyses show this not to be the case for *P. parvimensis* populations at the Channel Islands. CPUE for *P. parvimensis* either showed little change or significantly increased (e.g., Santa Rosa Island) despite general declines in sea cucumber abundance as indicated by the fishery-independent monitoring data. The specific reasons for the lack of correspondence between CPUE and stock size in this case are not known. The pattern of CPUE declining more slowly than abundance has been reported elsewhere by Hillborn and Walters (1992) who attributed it to two factors: (*i*) spatially structured stocks with substocks of differing densities and differing catchability and (*ii*) changes in catchability over time. Both factors appear to apply to the *P. parvimensis* fishery at the Channel Islands. Data collected prior to the fishery show densities of *P. parvimensis* differed greatly among sites and islands. Moreover, it is reasonable to assume that fishermen have become more efficient at locating and harvesting *P. parvimensis* over the recent history of the fishery, resulting in changes in catchability over time. Other factors could have lead to a constant CPUE in the face of declining stock size as well. For instance, search and handling times by fishermen might remain constant over a range of high stock densities. Furthermore, small vessels with limited holding capacity would tend to stabilize CPUE over a range of stock densities, especially where travel time between the port and fishing grounds is significant. Good correspondence between changes in CPUE and stock size may in fact only occur at low stock densities, where search time becomes a relatively large component of fishing effort.

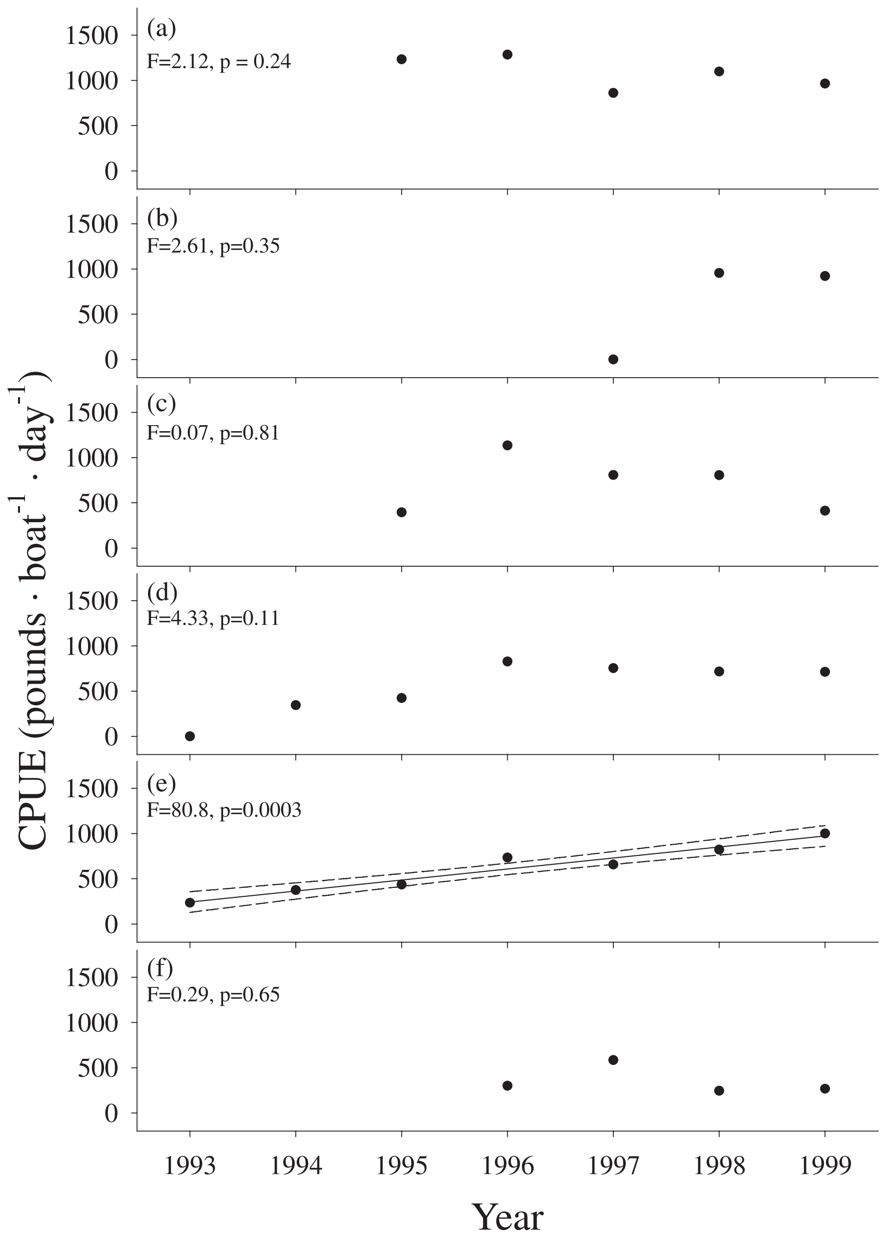
There are some limitations to using long-term monitoring data in fishery management. Foremost among these is that few long-term monitoring programs have been designed specifically for the purpose of fishery evaluation. Often, data collected by entities not associated with the fishing industry (e.g., universities, non-fishery agencies, private industry, etc.) are the only fishery-independent data available for a given stock. The locations, dates, and sampling frequency at which such data are collected are often suboptimal for evaluating the effects of fishing on stock size, particularly if BACI analyses are desired (e.g., McClanahan 1989). Such was the case for *P. parvimensis* in California. The only data on the long-term abundance of *P. parvimensis* are those of the NPS and the USGS. The monitoring programs of these two agencies were established primarily to track long-term changes in kelp forest communities at a wide array of sites that were selected to reflect the broad range of environmental conditions and biological assemblages at the Channel Islands. Consequently, only 7 of 19 sites met the statistical assumptions of our BACI analy-

Fig. 3. Densities of warty cucumbers (*P. parvimensis*) at fished sites on San Nicolas ((*a*) West Dutch Harbor), Santa Barbara ((*b*) Southeast Sea Lion Rookery), Santa Cruz ((*c*) Scorpion Anchorage, (*d*) Fry's Harbor, and (*e*) Yellowbanks), and Santa Rosa Islands ((*f*) Johnson's Lee North and (*g*) Johnson's Lee South). The annual mean for each fished site (solid circles) is plotted with the annual mean of the two non-fished reserve sites on Anacapa Island (open circles). Horizontal lines are means for fished (solid) and non-Fished (dotted) sites for the before and after periods (separated by vertical line).



ses for *P. parvimensis*. Nonetheless, an important advantage of these monitoring programs with respect to their use in evaluating the *P. parvimensis* fishery is that they began in advance of the fishery. Typically, monitoring programs are not begun until there is a perceived problem with an exploited population.

Fig. 4. Annual CPUE data of warty cucumbers (*P. parvimensis*) during the after period at six islands: (*a*) San Nicolas Island; (*b*) Santa Barbara Island; (*c*) Anacapa Island; (*d*) Santa Cruz Island; (*e*) Santa Rosa Island; and (*f*) San Miguel Island. Shown are *F* and *P* values for tests of significance of linear trends in CPUE vs. Time. For Santa Rosa Island, where the *P* value is <0.05, the linear fit (solid line) and upper and lower 95% confidence intervals (dotted lines) are shown.



Another limitation in using BACI has to do with assigning causality. Although the BACI analysis avoids the pitfalls of pseudoreplication (sensu Hurlbert 1984) associated with before–after or control–impact studies, it nonetheless is not a manipulative experiment and cannot completely rule out coincidental time by location interactions that coincide with the onset of the impact (Stewart-Oaten and Bence 2001). Schroeter et al. (1993) recognized this fact and addressed it by considering information in addition to BACI analyses that could either corroborate or lead to rejection of conclusions based on the BACI alone. The present study provides such additional information. The criteria used by the NPS and the USGS to select locations for long-term monitoring resulted in a dataset consisting of a large number of widely distributed sites that varied greatly in their abundance of *P. parvimensis* as well as in their distance from fishing ports. Such conditions undoubtedly resulted in a wide range of fishing pressures, which could be useful in isolating the effects of fishing (Pfister and Bradbury 1996; Kalvass and Hendrix 1997). In our study, sites closest to mainland ports (and coincidentally closest to the non-fished reserved sites) with high abundances of *P. parvimensis* in the before period (e.g., sites on SCI) were heavily fished and consequently suffered some of the greatest declines. By contrast, sites far from port (and coincidentally, far from non-fished reserve sites) with relatively low densities of sea cucumbers (e.g., sites on SMI and several on SNI) were fished less intensively and showed little change in the abundance of

P. *parvimensis* from the before to the after periods. Such patterns corroborate the results of our BACI analyses and provide additional evidence that implicates fishing as a major cause of declining stocks of *P*. *parvimensis* at the Channel Islands. Moreover, the spatial pattern of larger declines at fished sites closer to the non-fished control sites argues against the alternative hypothesis that the relative declines at fished sites were caused by site-specific oceanographic conditions in the after period that were favorable only to *P. parvimensis* in the vicinity of the control sites.

The use of no-take marine reserves to enhance depleted stocks has been the subject of much discussion and debate (Davis 1989; Dugan and Davis 1993; NRC 2001). The use of reserves for purposes of stock assessment, however, has received far less attention (NRC 2001). Our study demonstrates the value of an established marine reserve in assessing the status of the recent and largely unregulated dive fishery for *P. parvimensis*, a relatively sedentary sea cucumber. In the absence of a no-take reserve and long-term monitoring, catch data would have provided the only means for assessing *P. parvimensis* stocks in California. Our analyses show that management of the *P. parvimensis* fishery based solely on CPUE would lead to an inaccurate assessment of the stock, which could lead to the demise of the fishery. No-take reserves may also prove valuable in assessing the stock of established fisheries. Comparisons of stock abundance between protected and non-protected areas before and after a reserve is established can provide valuable information on the extent to which a fishery has caused stocks to decline, and on the resiliency of exploited stocks to recover in the absence of fishing. Whether used to evaluate fisheries with a long or short history, the value of no-take reserves to fishery managers is critically dependent on rigorous enforcement of no-take regulations, and on comprehensive long-term monitoring. Future plans involving the establishment of no-take zones for purposes of fishery enhancement should recognize the value of reserves in stock assessment and require the enforcement and monitoring that is needed to successfully achieve this important but often overlooked objective.

Acknowledgements

We thank R. Warner for insightful comments and discussion, D. Huang for assistance with graphics, and M. Kenner for making available the data from San Nicolas Island. This study was funded by the National Marine Fisheries Service (NOAA) under the Saltonstall–Kennedy Program (Award No. NA86FD0070) and the National Science Foundation (Award No. OCE-9982105)..

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