ICES Journal of Marine Science, 57: 1218–1226. 2000
doi:10.1006/jmsc.2000.0808, available online at http://www.idealibrary.com on IDEAL®

An assessment of the effectiveness of Marine Protected Areas in the San Juan Islands, Washington, USA

Fernando C. Tuya*, Mark L. Soboil, and Janine Kido

Tuya, F. C., Soboil, M. L., and Kido, J. 2000. An assessment of the effectiveness of Marine Protected Areas in the San Juan Islands, Washington, USA. – ICES Journal of Marine Science, 57: 1218–1226.

Marine protected areas (MPAs) can serve as effective sites for sheltering targeted populations. Furthermore, populations can experience increased abundance and body size in such areas. In this study we compared the abundance and body sizes of commonly collected fish and invertebrate species inside three marine preserves established eight years prior to our survey (University of Washington marine research preserves), two newly established MPAs, and three unprotected sites in the San Juan Archipelago. To determine whether population abundance and individual size were greater in protected areas within each site, these two parameters were measured for the red urchin (*Strongylocentrotus franciscanus*; Agasizz, 1865), sea cucumber (*Parastichopus californicus*; Stimpson, 1857), scallops (*Chlamys rubida*; Hinds, 1845; *Chlamys behringiana*; Midendorff, 1849; *Hinnites giganteus*; Gray, 1825), copper rockfish (*Sebastes caurinus*, Richardson, 1845), quillback rockfish (*Sebastes maliger*, Jordan & Jilbert, 1880), China rockfish (*Sebastes nebulosus*; Ayres, 1854) and lingcod (*Ophiodon elongatus*; Girard, 1854). The newly established MPAs and unprotected sites showed similar levels of abundance and size frequency distributions for the target species. Differences in abundance and size of small red urchins, scallops, rockfishes, and lingcod were not found among the three categories of sites, which could be attributable to a lack of effective protection within the protected sites. The UW marine preserves exhibited greater abundance of medium and large sized red urchins, which is explained because these areas lie within an urchin fishery closure zone established in the late 1970s. Ordination techniques showed that fishing pressure was the main cause of the decreased target populations assemblages. This study suggests that marine preserves have been effective in enhancing the medium and large size classes of the red urchin populations.

Key words: Marine Protected Areas (MPAs), marine reserves, temperate reef fisheries, visual censuses, rockfish, red urchin, San Juan Islands, Washington.

Received 1 June 1999; accepted 9 May 2000.

Fernando C. Tuya: Marine Sciences Faculty, Biology Department, Campus de Tafira s/n, 35017, Las Palmas de G.C., Canary Islands, Spain. E-mail: ftuya@ciemar.ccbb.ulpgc.es. Mark L. Soboil: Friday Harbor Labs, University Road 620, University of Washington, WA 98250, USA; E-mail: mls@axe.humboldt.edu. Janine Kido: Department of Biological Sciences, California State University, Fullerton, P.O. Box 6850, Fullerton, California, USA; E-mail: janinekido@hotmail.com

Introduction

The Food and Agricultural Organization of the United Nations (FAO) recently reported that 69% of the worlds marine stocks are either fully to heavily exploited, overexploited, or depleted (United Nations Food and Agriculture Organization, 1995). The decrease in marine stocks may be due to anthropogenic causes, such as over-fishing, habitat loss, and pollution. This worldwide exploitation of marine organisms has prompted actions to minimize the decline of economically valuable populations.

One such action being used to protect exploited species from over-fishing is the establishment of Marine Protected Areas (MPAs). Protected areas which have been implemented according to the life histories of target species can provide effective protection of critical nursery grounds, spawning grounds, and sites of high

*Author for correspondence.

1054–3139/00/041218 + 09 $30.00/0

species diversity (Allison *et al.*, 1998). MPAs may also limit control human activities that could negatively affect the density and average size of organisms (Polunin and Roberts, 1993). Dugan and Davis (1993) reviewed studies of existing marine parks and reserves that impose harvest restrictions and concluded that the higher abundance of target species within protected areas was the most detectable effect. The abundance and mean individual size of target species of fish, crustaceans, and molluscs were as much as 25 times and 200% greater, respectively.

Species of the Pacific Northwest coast are commonly fished by both commercial and recreational fishers. Although MPAs have been established to protect target species, few studies have investigated their success. This study looked at the impact of both commercial and recreationally fished species in areas that have received varying degrees of protection for different amounts of time.

Materials and methods

The abundance and mean size of the red urchin (*Strongylocentrotus franciscanus*; Agasizz, 1865), sea cucumber (*Parastichopus californicus*; Stimpson, 1857), scallops (*Chlamys rubida*; Hinds, 1845; *Chlamys behringiana*; Midendorff, 1849; *Hinnites giganteus*; Gray, 1825), copper rockfish (*Sebastes caurinus*, Richardson, 1845), quillback rockfish (*Sebastes maliger*, Jordan and Jilbert, 1880), China rockfish (*Sebastes nebulosus*; Ayres, 1854) and lingcod (*Ophiodon elongatus*; Girard, 1854) populations were determined at three categories of site. These included three University of Washington marine research preserves (UWPs), two MPAs and three unprotected sites (UPs). The first category of study sites include: Shady Cove (48°33′30″N, 123°00′20″W), Point George (48°33′45″N, 122°59′00″W) and Yellow Island (48°35′35″N, 123°01′40″W), all of which lie within the UWPs, established eight years ago. Collection of all marine organisms is prohibited in these areas, except for scientific research as permitted by the Director of the Friday Harbor Laboratories. The second category of sites include: Bell Island (48°35′50″N, 122°57′40″W) and Lime Kiln (48°31′15″N, 123°09′15″W) were sites have been designated MPAs since 1997. These are voluntary no-take zones where the collection of finfish with the exception of salmon is prohibited. Cliff Island (48°35′30″N, 123°00′20″W), Bell Island (outside the MPA) and Lime Kiln (outside the MPA) are sites where no protection exists.

Random sub-tidal censuses were carried out from 6 July to 13 July 1998 on reef slopes in the San Juan Archipelago (Fig. 1). Our census method included positioning a 50 m × 1/2 inch fibreglass tape randomly at no deeper than 20 m (Brock, 1954). Each transect was

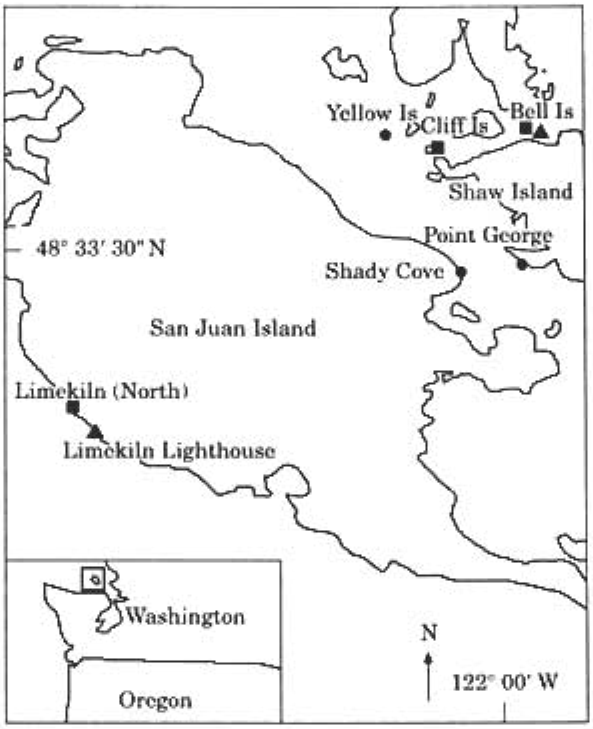

Figure 1. Location of sub-tidal censuses on the reef slopes of the San Juan Archipelago.

50 m × 6 m, thus 300 m^2 was investigated at each visual census, with one 50 m × 3 m strip transect (*sensu* Kingsford, 1998) being completed per diver. Four visual censuses were carried out at each site. Divers started at the same end of each transect, with one diver counting target fish species and the other counting and measuring target invertebrate species. All observations were made during daytime diving sessions using scuba equipment. Consequently diurnally exposed species were selected to avoid biases (Brock, 1982), as well as those species that would most likely show changes in the population structure attributable to fishing restrictions over the short duration of the study (Edgar and Barret, 1997).

Transects were placed parallel to the shoreline in order to maximize homogeneous benthic conditions (Willan *et al.*, 1979). A depth limit of 20 m was imposed by the safety constraints on a diver performing repetitive dives. Scallops and fishes encountered along each transect were placed in one of three size classes: small (juveniles), medium-size individuals and large (adults) according to Moulton (1977). Red urchins were categorized according to commercial fishing regulation: small (<102 mm = unfished lower size limit), medium-size (102–140 mm = fished sizes) and large (>140 mm = fished sizes). Cucumbers were not distinguished in size classes. The four scallops species were pooled as one category (scallops).

Table 1. Surveyed sites in the UWPs, MPAs and UPs. Reef type (r=rocky, s=sand/mud, w=sand/rock), slope angle (deg), visibility (m), temperature (C°) and depth (m).

	Reef type	Slope angle	Visibility	Temperature	Depth
UWPs					
Shady Cove	w and r	0	4	12	18–20
Point George	r	0–45	6	11	18
Yellow Island	s and w	0	4	12	10
MPAs					
East side of Bell Island	w	45–90	5	11	17–22
Lime Kiln	w	0–45	5	12	17
UPs					
Cliff Island	w	0	5	12	12
Bell Island unprotected zone	r	45–90	5	11	18–23
Lime Kiln unprotected zone	w and r	0–90	5	12	15–18

Substrate, temperature, slope, depth, visibility and fishing pressure were recorded for each site (Table 1). Salinity was not measured because its influence has not been shown to influence the abundance of rocky reef fish communities (Sanders, 1985). Fishing pressure was quantified according to the protective status of each site (Bell, 1992). To simplify and standardize recordings, three classes of substrates and slope gradients (Bortone *et al.*, 1994; Falcon *et al.*, 1996) were also defined along the transects.

Sites and species ordination analyses were performed by a two *row-centering* strategy principal component analysis (PCA) with a standardized function (F) equaling 1 (with a Pearson product-moment correlation matrix), and two hierarchical classification analyses using Euclidean distance (ED) were chosen as exploratory analyses to evaluate site and species relationships. Multiple regression models were run for the first two principal components found in the PCA to interpret which abiotic or anthropogenic variables were represented. ANOVA was used to test the significance of each variable that significantly influenced the clustering of sites and species within the PCAs.

A Pearson product-moment correlation matrix was used to find significant relationships between the different abiotic variables.

Multiple regression models were used to evaluate relationships between target species abundance and abiotic or anthropogenic factors.

Single factor ANOVA was used to test significant differences for the three size classes of urchins among UWPs, MPAs and UPs. Tukey's test was used for comparisons of significant effects after ANOVA. Before using ANOVA the assumptions of normality and homoscedasticity were tested by Kolgomorov–Smirnov and Cochran's test, respectively. All statistics analyses were carried out using SYSTAT 7.0. p-values were selected at a significance level of $\alpha=0.05$.

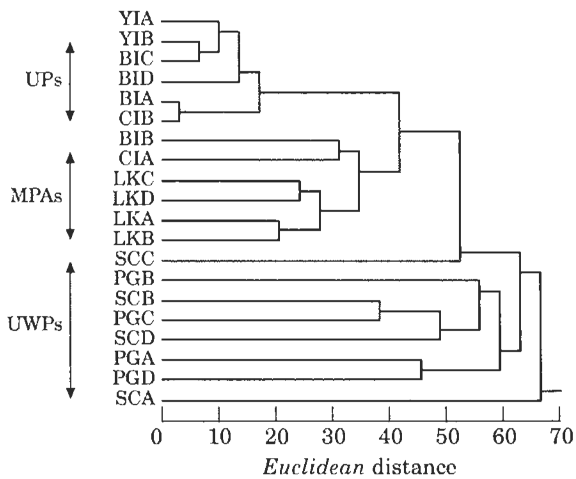

Figure 2.

Results

Sites hierarchical classification analysis

Two main clusters at ED$\simeq$60 were identified. The first cluster contained all UWPs, and the second included all MPAs and UPs. From the second cluster two subclusters were also identified at ED$\simeq$40. The first cluster contained all MPAs, and the second contained all UPs (Fig. 2). These clusters distinguish sites based on the duration of protected status.

Species hierarchical classification analysis

Two main clusters at ED$\simeq$20 were identified. The first cluster contained cucumbers, medium and large urchins, all of which have been or are currently commercially fished. The second cluster contained small sea urchins

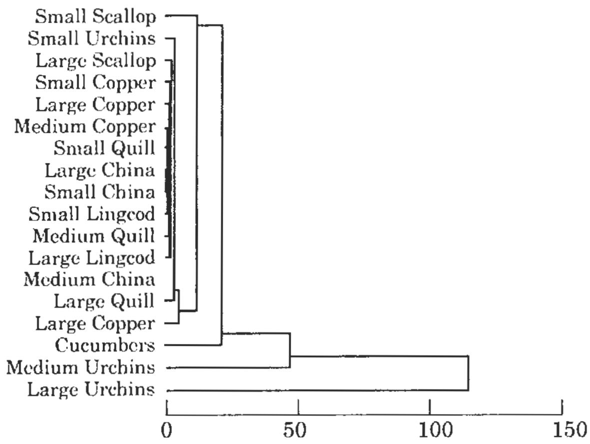

Figure 3.

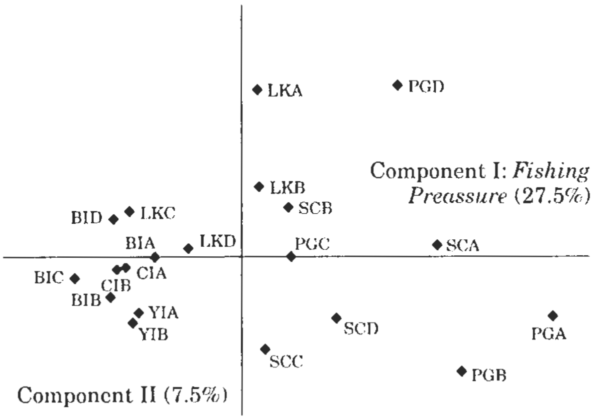

Figure 4.

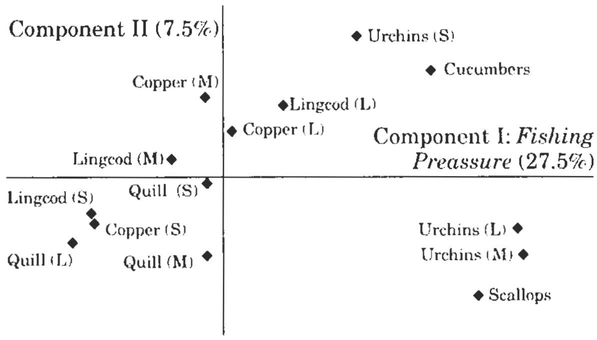

Figure 5.

(not harvested), scallops, lingcod and rockfish, which are only recreationally fished (Fig. 3).

Ordination analysis (PCA)

The first two principal components (PC1 and PC2) accounted for 35% of the variance within the data. The multiple regression model (adjusted R<0.20, p>0.05) for the PC2 and the abiotic variables suggested that no abiotic or anthropogenic variables affected site and species relationships along the PC2. Therefore, no environmental role was interpreted for the PC2. The multiple regression model (adjusted R>0.67, p<0.05) for the PC1 and abiotic variables suggested that fishing pressure had the greatest significance on site and species relationships (p<0.002) (Figs 4 and 5). The fishing pressure multiple regression coefficient (−0.328) showed a significant negative relationship within PC1.

Pearson product-moment correlation matrix

A correlation analysis among all abiotic and anthropogenic parameters showed a significant negative relationship between temperature and depth (p<0.01). Therefore, temperature and depth were grouped together as one abiotic variable in all multiple regression models.

Multiple regression models

The first multiple regression models were used to compare all abiotic or anthropogenic variables with the total abundance of the target species (all size classes were pooled for each species). These models suggested that abiotic or anthropogenic factors do not affect abundance along sites for lingcod, scallops and china, copper and quillback rockfishes (p>0.05, Table 2). For red urchins and cucumbers the fishing pressure and habitat parameters produced significant changes in abundance (p<0.05, table 2). From this first model, visibility and slope were shown to have no significant relationship with any of the target species (p>0.05, Table 2), so a second multiple regression models were run without these parameters for red urchins and cucumbers. These second models showed a significant negative relationship between urchins and cucumbers with increasing fishing pressure (p<0.05, Table 2) and significant relationship between urchin and cucumber abundance with habitat type (p<0.05, Table 2). The relationship between urchins and cucumbers with habitat type shows an increase in abundance with rocky sites and a decrease in rock/sand sites. Finally, when the simple linear regression was run for urchins and cucumbers with the fishing pressure parameter, only urchins exhibited any negative significant relationship (p<0.05, Table 2). Furthermore, fishing pressure plays a more

Table 2. Multiple regression models. y: target species abundance, x_1: fishing pressure, x_2: depth, x_3: slope, x_4: habitat, x_5: visibility.

Target species	Regression models																	
	$y=yo+ax_1+bx_2+cx_3+dx_4+ex_5$								$y=yo+ax_1+bx_2+dx_4$						$y=yo+ax_1$			
	R^2	F	yo	a	b	c	d	e	R^2	F	yo	a	b	d	R^2	F	yo	a
Urchin	0.49	5.33*	−307.57	−154.14*	−4.47	0.86	143.85*	34.52	0.38	3.46*	720.61	−177.99**	−4.30	43.84*	0.51	19.23**	248.53	−110.74
Cucumber	0.53	5.97*	30.56	−14.96*	0.8	−5.39	18.65*	−8.36	0.52	9.30*	−25.40	−15.6*	0.68	28.97*	0.27	6.87 (n.s)	26.82	−9.62
China	−0.06	0.78 (n.s.)	0.94	−0.77	−0.08	0.005	−0.92	1.3										
Quillback	0.09	1.34 (n.s.)	−27.11	−2.98	0.3	−0.19	1.48	1.64										
Copper	0.13	1.53 (n.s.)	−29.58	0.69	0.05	−0.67	1.36	5.81										
Lingcod	0.02	1.19 (n.s)	−1.38	0.21	0.06	0.11	0.32	0.45										
Scallop	−0.041	0.78 (n.s)	−22.9	−0.068	4.89	6.52	5.31	0.31										

$*=0.01<p<0.05$; $**=p<0.01$; n.s.=not significant.

significant role in determining urchin abundance among sites than it does for the abundance of the other target species.

Single factor ANOVA

The previous analyses suggested that the main difference in target species abundance among UWPs, MPAs and UPs was the abundance of red urchins. The UWPs showed significantly greater abundance for medium and large size red urchin when compared to the MPAs and UPs (Table 3; Fig. 6b,c). The small urchin abundance was not different among the three sites categories (Table 3, ANOVA: $F=2.37$, $p>0.05$; Fig. 6a), whereas the medium and large size classes showed significant (Table 3, $p<0.05$; Fig. 6b,c) differences. The posteriori Tuckey's test showed no significant difference of urchin abundance for the medium and large size categories between MPAs and UPs (Table 3, $p>0.05$, Fig. 6b,c).

Discussion

Both the ordination and the classification multivariate analyses showed that the study sites were more similar with regard to protective status than any other variables. The heterogeneity was a consequence of the difference in the red urchin abundance. This result contrasts with Edgar and Barret (1997). Their study sites showed greater similarity according to the abiotic parameters than by protective status.

With the exception of Yellow Island, the abundance of medium and large urchins was significantly different among UWPs, MPAs and UPs. The low abundance at Yellow Island may be explained by the presence of harbor seals and/or little to no rocky habitat for fish and invertebrate assemblages.

One-way ANOVA, the regression models and multivariate species analyses showed that the abundance of medium and large urchins is significantly affected by fishing pressure in the MPAs and unprotected areas. Due to the depletion of stocks by over-fishing of other economically important invertebrates in the past (e.g. abalone), invertebrate fisheries have now focused on other currently abundant species, such as urchins. Such a trend has been occurring in California. Species that once provided major value and biomass to commercial and recreational landings, such as abalone (*Haliotis* spp.) and giant sea bass (*Stereolepis gigas*) have been replaced by rock crabs (*Cancer* spp.), red urchins and rockfish (*Sebastes* spp.) (Dugan and Davis, 1993). The economic replacement has maintained the fishery's economic stability and reduced pressure to restore exhausted stocks resulting from serial depletion.

In addition, urchin abundance was significantly higher within the UWPs than in MPAs and UPs. The short period of time the MPAs have been protected may explain why the abundance and population sizes of the urchins are not as great as in the UWPs. This fact is also explained because all the UWPs lie within an urchin fishery closure zone, which has been closed to the commercial fishing since the late 1970s (Klinger, personal communication). This closure is regulated and enforced by the Washington State Department of Fish and Wildlife. The trends in urchin populations might be attributable to this closure, therefore, rather than protection under the UWPs programme. Palsson and Pacunski (1995) found similar results when they compared the abundance of rockfish and lingcod assemblages in newly established refuges to long-established refuges. Because rockfish, lingcod and urchins grow slowly, the new MPAs have not allowed abundance and size composition to recover to unfished conditions.

Because none of the protected areas (UWPs and MPAs) had a significantly larger abundance in small urchins, scallops, sea cucumbers, lingcod and rockfish, it could imply that the UWPs and MPAs, have not enhanced population abundance within the refuges. This result was similar to that obtained by Cole *et al.* (1990), except that his study showed an increase in the number of lobsters in the protected areas.

This heterogeneity could be attributed to poorly situated protected areas in which larval supply is inadequate due to unfavorable currents and poor recruitment. It might also stem from inadequate sampling: the patchy distribution of fish species may contribute to the lack of any statistical significance (Cole *et al.*, 1990) and the change in mean size may be easier to detect than the change in total abundance (Edgar and Barret, 1997). It could also be attributable to a lack of effective protection within the protected areas. In the new MPAs, a programme has been established to estimate the level of voluntary compliance, but no data are available yet (Klinger, personal communication). Therefore, we could suppose that enforcement has not been effectively achieved within these areas.

Within the UWPs there were approximately 60 times more large urchins than small urchins. The extremely low abundance of small urchins within UWPs, MPAs and UPs could be due to the fact that broadcast spawners exhibit reduced fertilization efficiencies as densities decline (Levitan *et al.*, 1992). For red urchins, fertilization is unlikely over distances of more than a metre or two due to the mixing of the water column, which quickly dilutes gametes (Denny and Shibata, 1989). Recruitment can be decreased further with increased fishing pressure because safe havens for juveniles under spines or adults could be disrupted (Rogers-Bennet *et al.*, 1996; Tegner and Dayton, 1997). Therefore, if an area exhibits low-level adult abundance it may not experience enough recruitment to offset adult mortality from fishing (Tegner and Dayton, 1997). The

Table 3. Summary of one-way ANOVA and posteriori Tukey's test showing the effect of status protection on the three size classes of urchins.

	Small					Medium					Large				
Source	DF	SS	MS	F	P	DF	SS	MS	F	P	DF	SS	MS	F	P
Sites	2	24.50	12.25	2.37	0.14	2	7545	3772	8.16	0.010	2	116 735	58 367	6.71	0.016
Error	9	46.50	5.17			9	4162	462			9	78 297	8700		
Total	11	71.00				11	11 707				11	195 031			
						UWPs>MPAs (t-stat=3.06, 2-sided p=0.011*)					UWPs>MPAs (t-stat=2.97, 2-sided p=0.013*)				
						UWPs≫UPs (t-stat=3.81, 2-sided p=0.003**)					UWPs>UPs (t-stat=3.34, 2-sided p=0.069*)				
						MPAs=UPs (t-stat=0.76, 2-sided p=0.28 n.s.)					MPAs=UPs (t-stat=0.37, 2-sided p=0.35 n.s.)				

*=0.01<p<0.05; **=p<0.01; n.s.=not significant.

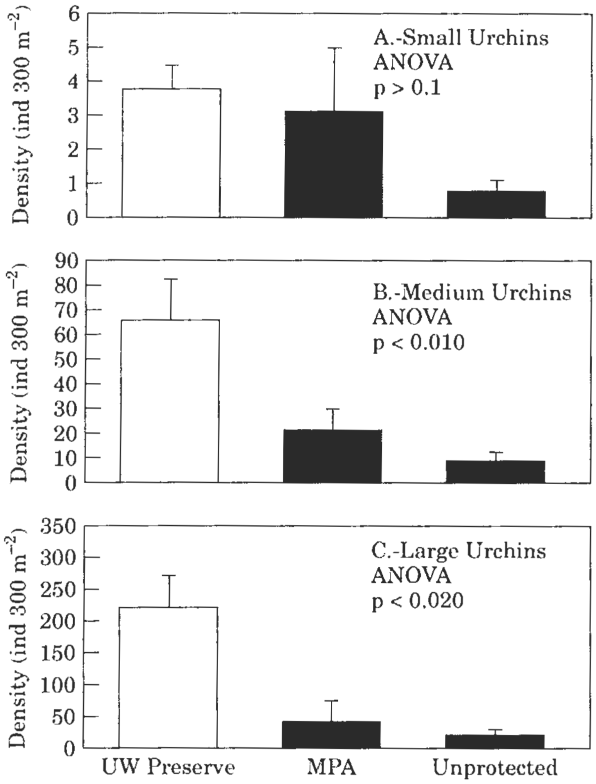

Figure 6.

low abundance variation between sites could also be due to the fact that the protected sites (UWPs and MPAs) are successfully operating as "feeder areas" that are producing adult fish and juveniles which are naturally migrating ("spillover effect") into the unprotected areas (Russ and Alcala, 1996). Secondly, since the eggs and larvae of many of the target species have an extended pelagic stage, many may be dispersing outside the protected areas. Since many of the target species grow slowly and mature late in life (Leaman, 1991), so that quantitative results may take many years to collect, long-term strategies for managing and monitoring the success of protected areas may be required. Long-term studies on recruitment would also enhance our understanding of the factors that determine species abundance (Cole *et al.*, 1990).

Finally, as shown in this study as well by Bortone *et al.* (1994) and Falcon *et al.* (1996), since the regression models were capable of explaining only a small amount of the variation in the dependent variables of the target species populations, other variables that were not examined may also influence the structure of the target populations. Therefore, future research should include measurements such as the dynamics of human activities, biogeography characteristics, number and size of refuges, atmospheric-oceanographic parameters, current patterns (Lindquist and Pietrafesa, 1989), biocomplexity (Luckurst and Luckurst, 1978), distance to shore and water quality.

Acknowledgements

We thank Terrie Klinger, Laura Rogers-Bennet, Jennifer Reusink, and Leah Gerber for their guidance and support throughout this project. We also thank the Friday Harbor Marine Labs for the use of their diving facilities. Most importantly, we would like to acknowledge the companionship and contributions of all the students in the summer Marine Conservation course of 1998.

References

Allison, G. W., Lubchenco, J., and Carr, M. H. 1998. Marine reserves are necessary but not sufficient for marine conservation. Ecological Applications, 8: 79–92.

Bell, J. D. 1992. Effects of depth and marine reserve fishing restrictions on the structure of a rocky reef fish assemblage in the Northern-Western Mediterranean Sea. Journal Applied Ecology, 20: 357–369.

Bortone, S. A., Van Tasell, J., Brito, A., Falcón, J. M., and Bundrick, C. M. 1991. A visual assessment of the inshore fishes and fishery resources off El Hierro, Canary Islands: A baseline survey. Scientia Marina, 55: 529–541.

Brock, V. E. 1954. A preliminary report on a method of estimating reef fish populations. Journal Wildlife Management, 18: 297–308.

Brock, R. E. 1982. A critique of the visual census method for assessing coral reef fish populations. Bulletin of Marine Science, 32: 269–276.

Cole, R. G., Ayling, T. M., and Cuesse, R. G. 1990. Effects of marine reserve protection at Goat Coast, northern New Zealand. New Zealand Journal of Marine and Freshwater Resources, 24: 197–210.

Denny, M. W., and Shibata, M. F. 1989. Consequences of surf-zone turbulence for settlement and external fertilization. American Naturalist, 117: 838–840.

Dugan, J. E., and Davis, G. E. 1993. Applications of marine refugia to coastal fisheries management. Canadian Journal of Fisheries and Aquatic Sciences, 50: 2029–2042.

Edgar, G. J., and Barret, N. S. 1997. Short term monitoring of biotic change in Tasmanian marine reserves. Journal of Experimental Marine Ecology, 213: 261–279.

Falcón, J. M., Bortone, S. A., Brito, A., and Bundrick, C. M. 1996. Structure of and relationships within and between the littoral, rock-substrate fish communities off four islands in the Canarian Archipelago. Marine Biology, 125: 215–231.

Kingsford, M., and Battershill, E. 1998. Studying Temperate Marine Environments. Canterbury University Press, New Zealand. 334 pp.

Leaman, B. M. 1991. Reproductive styles and life history variables relative to exploitation and management of *Sebastes* stocks. Environmental Biology of Fishes, 30: 253–271.

Levitan, D. R., Sewell, M. A., and Chia-Fu-Shiang 1992. How distribution and abundance influence fertilization success in the sea urchin *Parastichopus californicus*. Ecology, 73: 248–254.

Lindquist, D., and Pietrafesa, L. 1989. Current vortices and fish aggregations: the current field and associated fishes around a tugboat wreck in Oslow Bay, North Carolina. Bulletin of Marine Science, 44: 533–544.

Luckurst, B. E., and Luckurst, K. 1978. Analysis of the influence of substrate variables on coral reef fish communities. Marine Biology, 49: 317–323.

Moulton, L. L. 1977. An ecological analysis of fishes inhabiting the rocky nearshore regions of northern Puget Sound, Washington. Ph.D. Dissertation, University of Washington, Seattle, WA.

Palsson, W. A., and Pacunski, R. E. 1995. The response of rocky reef fishes to harvest refugia in Puget Sound. Puget Sound Research '95 Proceedings, 1: 224–234.

Polunin, N. V. C., and Roberts, C. M. 1993. Greater biomass and value of target coral-reef fishes in two small Caribbean marine reserves. Marine Ecology Progress Series, 100: 167–176.

Rogers-Bennet, L., Bennet, W. A., Fastenau, H. C., and Dewees, C. M. 1995. Spatial variation in red sea urchin reproduction and morphology. Implications for harvest refugia. Ecological Applications, 5: 1171–1186.

Russ, G. R., and Alcala, A. C. 1996. Do marine reserves export adult fish biomass? Evidence from Apo Island, central Philippines. Marine Ecology Progress Series, 132: 1–9.

Sanders, R. 1985. Hydrological, dial and lunar factors affecting fishes on artificial reefs off Panama city, Florida. Bulletin of Marine Science, 37: 318–328.

Tegner, M. J., and Dayton, P. K. 1977. Sea urchin recruitment patterns and implications of commercial fishing. Science, 196: 324–326.

United Nations Food and Agriculture Organization 1995. The state of world fisheries and aquaculture. Rome, Italy.

Willan, R. C., Dollimore, J. M., and Nicholson, J. 1979. A survey of fish populations at Karikari Peninsula, Northland, by scuba diving. New Zealand Journal of Marine and Freshwater Resources, 13: 447–458.