OCS Study
MMS 2005-002

Effect of Temporal and Spatial Separation of Samples on Estimation of Impacts

Final Technical Summary

Final Study Report

U.S. Department of the Interior
Minerals Management Service
Pacific OCS Region

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Prepared under MMS Cooperative
Agreement No. 14-35-0001-30758
by
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U.S. Department of the Interior
Minerals Management Service
Pacific OCS Region

Camarillo
April 2005

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Suggested Citation

The suggested citation for this report is:

Minchinton, Todd E., and Peter T. Raimondi. Effect of Temporal and Spatial Separation of Samples on Estimation of Impacts. MMS OCS Study 2005-002. Coastal Research Center, Marine Science Institute, University of California, Santa Barbara, California. MMS Cooperative Agreement Number 14-35-0001-30758. 85 pages.

Table of Contents

List of Tables

List of Figures

FINAL TECHNICAL SUMMARY

STUDY TITLE: Effect of Temporal and Spatial Separation of Samples on Estimation of Impacts

REPORT TITLE: Effect of Temporal and Spatial Separation of Samples on Estimation of Impacts

CONTRACT NUMBER: 14-35-0001-30758

SPONSORING OCS REGION: Pacific

APPLICABLE PLANNING AREA(S): Southern and Central California

FISCAL YEAR(S) OF PROJECT FUNDING: FY 99/00, FY 00/01

COMPLETION DATE OF REPORT: April 2005

COSTS: FY 99/00 - $34,293; FY 00/01 - $34,293

CUMULATIVE PROJECT COST: $68,568

PROJECT MANAGER: Russell J. Schmitt

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KEY WORDS: Abalone, Anemone, BACI design, Barnacle, Baseline data, Before-After design, California, El Niño, Environmental impacts, Intertidal, Limpet, Marine algae, Marine invertebrates, Monitoring, Mussel, Power analysis, Rockweed, Seasonal trend, Sea star, Surfgrass, Temporal Trend, Turfweed, Withering Syndrome

BACKGROUND: Since 1991 the Minerals Management Service has funded a monitoring program for Santa Barbara County, California, U.S.A, called the "Shoreline Inventory". The inventory was to provide baseline data on rocky intertidal communities that could be used in the event of an oil spill to calculate losses due to the impact. The main logistical task of the inventory was to set-up and sample at regular intervals 9 permanent sampling sites in the rocky intertidal zone of Santa Barbara County. At each site species assemblages dominated by 13 species were targeted and sampled in permanent plots in spring and autumn of each

year. From 1991 to 1995 the Shoreline Inventory was expanded to 18 mainland sites across 4 counties (San Luis Obispo County, Santa Barbara County, Ventura County, Los Angeles County) and 8 offshore sites on two islands (Catalina Island, Santa Cruz Island). This report summarizes the spatial and temporal trends in abundance for the 13 target species at the 18 mainland sites and assesses using power analysis the ability of the Shoreline Inventory monitoring program to detect changes in the abundance of the target species.

OBJECTIVES: The primary objective of this study was to use the database of the Shoreline Inventory to test quantitatively the critical assumption that estimation of consequences after an impact is possible for sites that were monitored before the impact. Specifically, we answer the following two questions:

1. What are the spatial and temporal patterns of abundance for the 13 target species at the 18 mainland sites?
2. Based upon power analyses following simulated impacts to target species at each site, should there be any modifications to the sampling design of the monitoring program?

DESCRIPTION: The Shoreline Inventory includes 18 mainland sites extending from central to southern California. Five sites were established in autumn 1995 in San Luis Obispo County (Point Sierra Nevada, Piedras Blancas, Cayucos Point, Hazards, Shell Beach), 9 sites in spring 1992 in Santa Barbara County (Occulto, Purisima Point, Stairs, Boathouse, Government Point, Alegria, Arroyo Hondo, Coal Oil Point, Carpinteria), 2 sites in autumn 1994 in Ventura County (Mussel Shoals, Old Stairs), and 2 sites in autumn 1994 in Los Angeles County (Paradise Cove, White's Point). The Shoreline Inventory includes 13 key target species comprising both sessile and mobile macroinvertebrates and macroalgae that are generally the dominant occupiers of primary space at particular heights and exposures in the intertidal zone. Four target species (or species groups) are sessile or sedentary macroinvertebrates: (1) barnacles (the acorn barnacles *Chthamalus fissus/dalli* and *Balanus glandula*), (2) the gooseneck barnacle *Pollicipes polymerus*, (3) mussels (primarily *Mytilus californianus*), and (4) the sea anemones *Anthopleura elegantissima* and *A. sola*. Six target species are macroalgae: (5) the brown fucoid alga or rockweed *Silvetia compressa* (formerly *Pelvetia compressa*, *Pelvetia fastigiata*), (6) the brown fucoid alga or rockweed *Hesperophycus californicus* (formerly *H. harveyanus*), (7) the red alga *Mastocarpus papillatus*, (8) the red alga or turfweed *Endocladia muricata*, (9) the "rainbow" alga *Mazzaella* spp. (formerly *Iridaea* spp.), and (10) the marine angiosperm or surfgrass *Phyllospadix torreyi/scouleri*. Three target species are mobile invertebrates: (11) the ochre sea star *Pisaster ochraceus*, (12) the giant owl limpet *Lottia gigantea*, and (13) the black abalone *Haliotis cracherodii*. At each site permanent areas were established where target species were the dominant organisms, and then their abundances were sampled in spring and autumn of each year.

To examine variation in the abundance of target species in space and time, summary statistics (mean and standard error of the mean) for all sites and sample periods were calculated and plotted graphically. Analyses of covariance (ANCOVA) procedures were used to assess seasonal and temporal patterns of abundance for each of the target species at each site. The

ability of the monitoring program to estimate the effects of impacts was done using power analyses. Impacts to the abundance of target species were simulated and then the ability to detect these impacts was assessed under a before-after design. Based on these results, we then determined whether there should there be modifications to the sampling design of the Shoreline Inventory monitoring program.

STUDY RESULTS AND SIGNIFICANT CONCLUSIONS: The Shoreline Inventory monitoring program has produced what is, in terms of its spatial and temporal extent, one of the most comprehensive sets of data ever collected for species living in rocky intertidal communities. Visual inspection of graphs and statistical analyses of the data reveal that the current sampling design and monitoring protocols have a great ability to show the natural limits of variation and to detect trends in the abundance of target species. Three types of temporal trends were clear: (1) short-term seasonal trends, (2) intermediate-term fluctuations on time scales ranging from one to several years, and (3) long-term changes for five or more years due to changes in abundance over the entire sampling period. First, short-term seasonal trends were apparent at a minority of sites for the acorn barnacles, the gooseneck barnacle *Pollicipes* and the sea star *Pisaster*, with abundances greater in autumn than in spring. The macroalgae exhibited more pronounced seasonal trends than the macroinvertebrates. At most sites, the cover of *Silvetia*, *Mastocarpus* and *Phyllospadix* was greater in autumn than in spring, whereas *Endocladia* was more abundant in spring than in autumn. A second striking and common trend in the data were intermediate-term fluctuations in abundance where there were relatively rapid declines sometimes followed by more gradual increases. These relatively rapid and common declines occurred from 1997 to 1998, which corresponds to the 1997-98 El Niño event, which was the strongest of the century. Many target species experienced declines at some sites during the 1997/98 El Niño event, including barnacles, mussels, *Endocladia*, *Silvetia*, *Phyllospadix*, *Anthopleura* and *Haliotis*. The third obvious and consistent trend was the long-term declines in abundance over the entire monitoring period for some target species at several sites, including *Haliotis*, which experienced population crashes due to the fatal disease, withering syndrome. Detection of these temporal trends in abundance (seasonal, El Niño, disease) highlights the efficacy and utility of the current Shoreline Inventory monitoring program.

For all target species that met assumptions of statistical tests (10 of 13 species; see section 3.2.2), there was 80% power to detect 50% changes in abundance at $\alpha = 0.05$. Indeed, for the majority of species and sites, changes in abundance of only 20% could be detected with 80% power at $\alpha = 0.05$. Power to detect change varied among target species (but not among sites) and was greatest for mussels and least for *Pisaster*. Some species (*Silvetia*, *Hesperophycus*, *Haliotis*) and sites (e.g., Stairs) could not be assessed through power analysis because they failed to meet the assumptions of statistical tests. Alternative means should be considered to determine whether impacts could be detected for these species and sites should they occur. Indeed, additional more sophisticated analyses and designs should be explored to make greater use of the entire range of data. No changes should be made to the current monitoring program, but additional sites and target species, including means to monitor recruitment, should be added as resources permit.

STUDY PRODUCTS:

Publications:

Wilson, C. T., P. T. Raimondi, J. A. Altstatt, and T. E. Minchinton. Changes in intertidal community structure following mass mortality of the black abalone, *Haliotis cracherodii*, and implications for abalone recovery. Conservation Biology. In Review.

Raimondi, P. T., C. M. Wilson, R. F. Ambrose, J. M. Engle and T. E. Minchinton. 2002. Continued declines of black abalone along the coast of California: are mass mortalities related to El Niño events? Marine Ecology Progress Series **242**:143–152.

Minchinton, T. E. and P. T. Raimondi. 2001. Long-term Monitoring of Rocky Intertidal Communities at the Channel Islands National Park: Summary of Spatial and Temporal Trends and Statistical Power Analysis. Report and presentation for the Scientific and Management Review of Monitoring Protocols for the Channel Islands National Park. 29 pp.

Research Presentations:

Minchinton, T. E. and P. T. Raimondi. 2001. Long-term Monitoring of Rocky Intertidal Communities at the Channel Islands National Park: Summary of Spatial and Temporal Trends and Statistical Power Analysis. Report and presentation for the Scientific and Management Review of Monitoring Protocols for the Channel Islands National Park. 29 pp.

Minchinton, T. E., P. T. Raimondi, D. V. Richards, G. E. Davis and K. D. Lafferty. 2001. Detecting Environmental Impacts: Monitoring Programs and Power Analysis. Poster presented at UC Toxic Substances Research & Training Program 14th Annual Research Symposium. Lake Tahoe, California.

Minchinton, T. E., P. T. Raimondi, M. Wilson, R. F. Ambrose and J. M. Engle. 2000. Continued Declines of Black Abalone due to Withering Syndrome along the Coast of California. Poster presented at UC Toxic Substances Research & Training Program 13th Annual Research Symposium. San Diego, California. April 2000.

Becker, B. J. and MARINE members. 2001. MARINE (Multi-Agency Rocky Intertidal Network): Coordinated ecological monitoring on a regional scale. Displayed at various symposia.

FINAL STUDY REPORT

PART I: INTRODUCTION, OBJECTIVES, AND APPROACH

1.1 Introduction

Monitoring programs designed to detect natural and anthropogenic disturbances to populations and species assemblages in natural communities are becoming increasingly popular. The major assumption of all pre-impact monitoring studies is that the data collected will allow estimation of the consequences of an impact to ecological communities. One key assumption is that estimation of consequences after an impact will be possible for the sites that were monitored prior to the impact. The value of a coastline monitoring program depends on the assumption that estimations of impacts can be done accurately. Here, we report the results of data and statistical analyses that provide a sound basis for judging the utility of pre-impact monitoring programs to estimate accurately the consequences of impacts to ecological communities. Without testing the fundamental assumptions of such programs, any conclusion of ecological impact following perturbation will be challenged. Clearly, no local, State or Federal monitoring program will have served its purpose if the conclusions derived from the results cannot be vigorously defended.

Since 1991 the Minerals Management Service has funded a monitoring program for Santa Barbara County, California, U.S.A, called the "Shoreline Inventory". The inventory was to provide baseline data on rocky intertidal communities that could be used in the event of an oil spill to calculate losses due to the impact. The main logistical task of the inventory was to set-up and sample at regular intervals 9 permanent sampling sites in the rocky intertidal zone of southern and northern Santa Barbara County. At each site species assemblages dominated by 13 species were targeted and sampled in permanent plots in spring and autumn of each year. From 1991 to 1995 the Shoreline Inventory was expanded to 18 mainland sites across 4 counties (San Luis Obispo County, Santa Barbara County, Ventura County, Los Angeles County) and 8 offshore sites on two islands (Catalina Island, Santa Cruz Island). An important feature of all 26 sites is that they were selected, set-up and sampled using the same protocol. Therefore, the results from all sites are directly comparable. In its current form the Shoreline Inventory is one of the largest intertidal monitoring programs ever conducted.

1.2 Objectives

The primary objective of this study was to use the database of the Shoreline Inventory to test quantitatively the critical assumption that estimation of consequences after an impact is possible for sites that were monitored before the impact.

Specifically, we answer the following two questions:

1. What are the spatial and temporal patterns of abundance for the 13 target species at the 18 mainland sites?

2. Based upon power analyses following simulated impacts to target species at each site, should there be any modifications to the sampling design of the monitoring program?

1.3 Approach

1.3.1 Background

The methods used in the Shoreline Inventory monitoring program, including a discussion of the pros and cons of different sampling approaches and protocols, can be found in Ambrose et al. (1995). Details of the monitoring program, which was developed based on those used by the National Park Service intertidal monitoring program (see Richards and Davis 1988), and rational for its design can be found in Engle et al. (1994). A detailed discussion of the selection of target species, sampling procedures and survey sites for the Shoreline Inventory is given in Engle et al. (1994) and Raimondi et al. (1999). The approach used for gathering data for the Shoreline Inventory is briefly explained below, but see the above references for more detail.

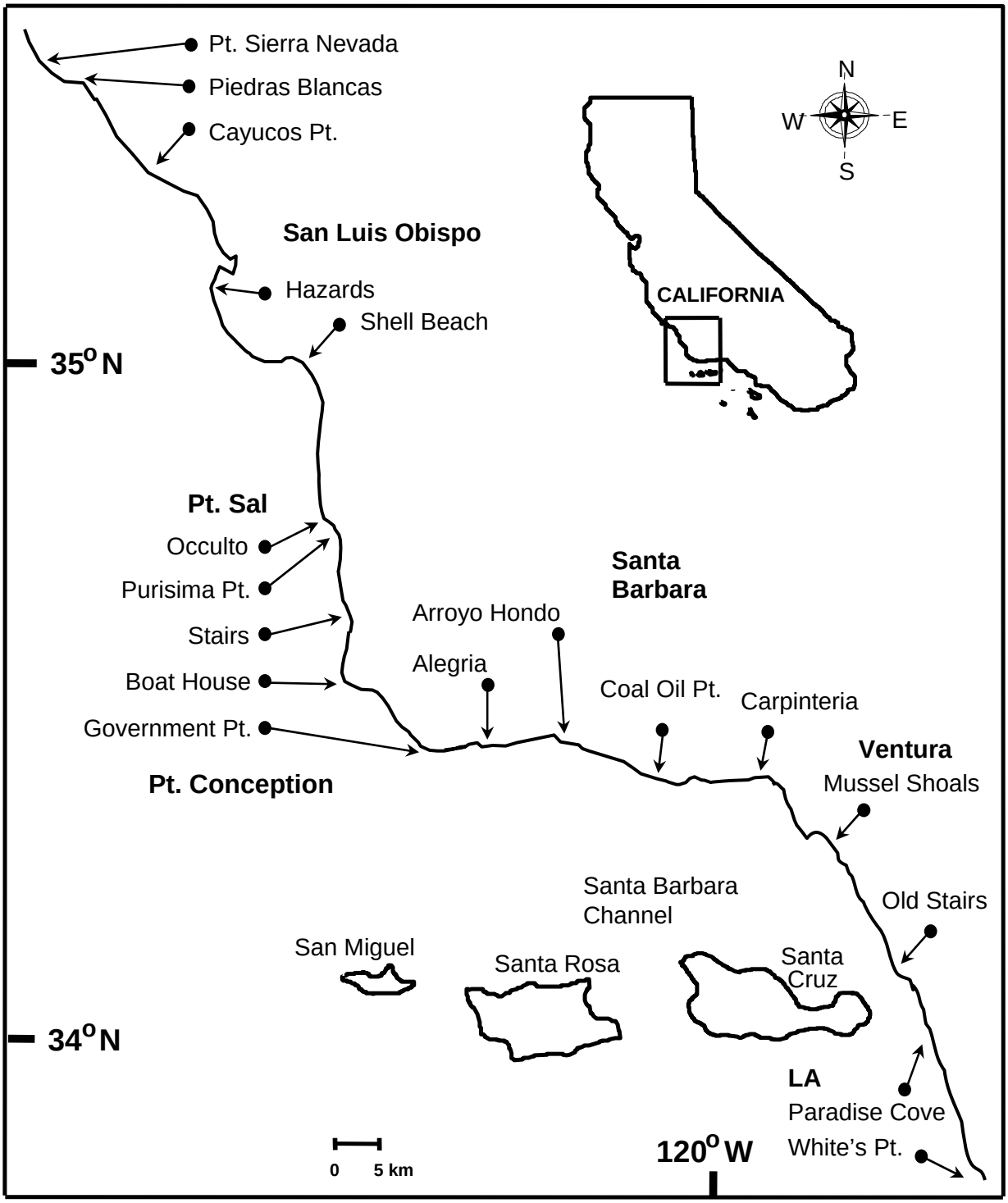
1.3.2 Study Sites

Study sites were selected based upon key criteria. Sites had to be at locations with (1) species assemblages and ecological conditions representative of the southern and central California coastline, (2) relatively pristine habitats that were susceptible to human impacts, including oil spills, and (3) optimal conditions for long-term monitoring (accessible, abundant target species, large enough to establish permanent sampling areas).

The Shoreline Inventory includes 8 island sites and 18 mainland sites (although new sites have been recently added). The 8 sites on Santa Cruz Island (Orizaba Cove, Prisoner's Harbor, Scorpion Rock, Willows Anchorage, Trailer, and Fraser Cove) and 2 sites on Catalina Island (Bird Rock, Little Harbour) were established in autumn 1994. The 18 mainland sites extend from central to southern California and include 5 sites established in autumn 1995 in San Luis Obispo County (Point Sierra Nevada, Piedras Blancas, Cayucos Point, Hazards, Shell Beach), 9 sites established in spring 1992 in Santa Barbara County (Occulto, Purisima Point, Stairs, Boathouse, Government Point, Alegria, Arroyo Hondo, Coal Oil Point, Carpinteria), 2 sites established in autumn 1994 in Ventura County (Mussel Shoals, Old Stairs), and 2 sites established in autumn 1994 in Los Angeles County (Paradise Cove, White's Point).

Only the 18 mainland sites are included in this study. The island sites were excluded from the study for two reasons. First, we wanted to focus on the mainland sites, which have received less attention than the island sites. Indeed, a similar study to this one includes all sites on Santa Cruz Island (see Minchinton and Raimondi 2001). In particular, we wanted to compare the response of target species between the 9 mainland sites north of Point Conception, an important biogeographic break, and the 9 sites south of Point Conception. Second, the mainland sites have been sampled most consistently and therefore present the most complete and comparable set of data, which is critical for data and statistical analysis of spatial and temporal patterns and power analysis. Location of the mainland study sites along the southern and central coastline of California is presented in Figure I-1.

Figure I-1. Rocky intertidal monitoring sites along the southern and central coast of California.



1.3.3 Target Species

Target species were selected based upon key criteria. Criteria used for species selection included: (1) ecological importance in structuring intertidal communities (abundant species, competitively dominant species, ecosystem engineers, major predators, etc.), (2) interesting local and regional distributions (e.g., species characteristic of particular intertidal heights or microhabitats or those at the limits of the biogeographic range), (3) well-studied with extensive literature available, and (4) human interest (vulnerable species, threatened species, harvested species, etc.).

The Shoreline Inventory includes 13 key target species comprising both sessile and mobile macroinvertebrates and macroalgae that are generally the dominant occupiers of primary space at particular heights and exposures in the intertidal zone (see Raimondi et al. (1999) for a detailed discussion of the natural history of these target species). Four target species (or species groups) are sessile or sedentary macroinvertebrates: (1) barnacles (the acorn barnacles *Chthamalus fissus/dalli* and *Balanus glandula*), (2) the gooseneck barnacle *Pollicipes polymerus*, (3) mussels (primarily *Mytilus californianus*), and (4) the sea anemones *Anthopleura elegantissima* and *A. sola*. Six target species are macroalgae: (5) the brown fucoid alga or rockweed *Silvetia compressa* (formerly *Pelvetia compressa*, *Pelvetia fastigiata*), (6) the brown fucoid alga or rockweed *Hesperophycus californicus* (formerly *H. harveyanus*), (7) the red alga *Mastocarpus papillatus*, (8) the red alga or turfweed *Endocladia muricata*, (9) the "rainbow" alga *Mazzaella* spp. (formerly *Iridaea* spp.), and (10) the marine angiosperm or surfgrass *Phyllospadix torreyi/scouleri*. Three target species are mobile invertebrates: (11) the ochre sea star *Pisaster ochraceus*, (12) the giant owl limpet *Lottia gigantea*, and (13) the black abalone *Haliotis cracherodii*.

All 13 target species are included in this study, but not all target species are present at each site (generally about 4 to 10 target species at per site). The specific target species sampled at each of the 18 mainland study sites are listed in Table I-1.

Table I-1. Target species sampled at each of 18 mainland sites.

County & Site	*Anthopleura*	Barnacles	*Pollicipes*	Mussels	*Silvetia*	*Hesperophycus*	*Mastocarpus*	*Endocladia*	*Mazzaella*	*Phyllospadix*	*Pisaster*	*Lottia*	*Haliotis*
San Luis Obispo													
Point Sierra Nevada		x		x	x	x	x		x	x	x		x
Piedras Blancas													x
Cayucos Point		x		x	x	x		x		x	x	x	x
Hazards		x		x	x			x	x		x	x	
Shell Beach		x		x	x		x	x			x		
Santa Barbara													
Occulto		x		x				x					
Purisima Point													x
Stairs		x		x	x			x		x	x	x	x
Boathouse	x	x		x	x			x			x	x	x
Government Point		x	x	x	x			x		x	x	x	x
Alegria	x	x	x	x								x	
Arroyo Hondo		x		x						x	x		
Coal Oil Point	x									x			
Carpinteria	x	x	x							x			
Ventura													
Mussel Shoals	x	x		x						x	x		
Old Stairs	x	x		x				x			x		
Los Angeles													
Paradise Cove		x		x				x		x			
White's Point		x		x				x					

1.3.4 Sampling Methods

At each site permanent areas were established where target species were the dominant organisms, and then their abundances were sampled in spring and autumn of each year. For this study, this comprised 22 sample periods from spring 1992 to autumn 2002 (although the number of sample periods varied depending on when the site was established). The abundances of sessile and sedentary macroinvertebrates (barnacles, *Pollicipes*, *Anthopleura*, mussels) and macroalgae (*Silvetia*, *Hesperophycus*, *Mastocarpus*, *Endocladia*, *Mazzaella*) were quantified by assessing their percent covers from photographs of 5 replicate quadrats for each species at each site. The abundance of *Phyllospadix* was quantified in situ by determining its percent cover in 3 replicate point-intercept line transects per site. The abundances of mobile species (*Pisaster*, *Lottia*, *Haliotis*) were quantified by counting them in situ in 3 replicate plots (5 for *Lottia*) and band transects (for *Pisaster* at some sites) for each species at each site.

For logistical reasons, some sites and target species could not be sampled for some sample periods. When this happened frequently, the target species at that site was omitted because the data were not deemed sufficient for presentation and analysis. Of the 97 combinations of sites and species sampled, this occurred in only four instances: mussels and *Pisaster* at Carpinteria and *Lottia* at Old Stairs and Paradise Cove. This left 93 target species and site combinations for analysis (see Table I-1). On 16 occasions, an individual quadrat, transect or plot was not sampled for a particular target species at a particular site on a particular date. On 6 occasions, a replicate quadrat, transect or plot was damaged, probably by storms. For three of these replicates, a new area with the same abundance of the target species was located and subsequently sampled. For the others, the reduced number of replicates was subsequently sampled. No attempt was made to account for any of these missing data, but given that they occurred sporadically across species, sites and sample periods, and the entire set of data comprised thousands of data points, their absence had no important influence on data and statistical analyses.

Part II: SPATIAL AND TEMPORAL PATTERNS

2.1 Introduction

The first objective was to determine the spatial and temporal patterns of abundance for the 13 target species at the 18 mainland sites. Although this objective has previously been addressed for some of the sites and sample periods (see Ambrose et al. 1995, Raimondi et al. 1999), the current report analyzes data from a greater range of sites over a longer period. Indeed, this report updates these previous reports, providing the most current and comprehensive summary of the spatial and temporal patterns of abundance of target species for sites of the Shoreline Inventory.

The purpose of summarizing these spatial and temporal trends is to examine changes in the structure and dynamics of the target species. Given the relatively long temporal series of data over a relatively large number of sites, it is possible to identify both short- (e.g., seasonal) and longer-term (e.g., El Niño events) changes in abundance over both local (e.g., sites separated by kilometers) and regional (e.g., sites separated by the biogeographic break at Point Conception) spatial scales. Because of the relatively large number of target species, it is possible to identify whether changes are species-specific or common across taxa. Investigating such spatial and temporal patterns of species abundance can help identify dramatic changes outside normal limits of variation and lead to considerable insights about the factors which might be responsible for such fluctuations (e.g., changes due to human impacts such as oil spills).

2.2 Methods

To examine variation in the abundance of target species in space and time, summary statistics (mean and standard error of the mean) for all sites and sample periods were calculated and plotted in two ways. First, summary statistics were calculated for each combination of site and sample period. Here replication comes from individual quadrats, plots or transects at a given site and sample period. Second, to examine broader patterns of abundance over time (i.e., for all sites combined), summary statistics were calculated for each sample period. Here replication comes from individual sites during a sample period. Because sites were established at different times, however, the level of replication varies with sample period. All sites were established by autumn 1995 and, therefore, except where data for a site were occasionally missing, the number of replicate sites remains constant after this date.

Analyses of covariance (ANCOVA) procedures were used to assess seasonal and temporal patterns of abundance for each of the target species at each site using statistical software program SYSTAT. All available sample periods were used in analyses and data were not transformed. In this model, season was included as a categorical variable and time (i.e., sample period) was included as the covariate. This analysis allows determination of linear changes in abundance over time and of seasonal effects after removal of (linear) temporal trends. These linear analyses give an indication of whether abundances increased or decreased over time from the initiation of sampling until the most current sample. They do not, however, detect non-linear patterns in the data. For example, if the cover a target species decreased and then increased, which might

happen when there is an impact followed by a recovery, this might be detected as no trend. Given the large numbers of sites and extensive time series of data, such non-linear trends were apparent for species at some sites. It was beyond the scope of this report to use more sophisticated time series analyses to determine the underlying non-linear patterns in these data. Instead, these trends are described by visual inspection of the data, pointing out common patterns across species, space and time.

Visual inspection of the graphs was done to determine whether temporal changes in abundance showed any spatial consistency among sites (i.e., geographic patterns such as latitudinal trends or differences around the biogeographic break at Point Conception). To facilitate these geographic comparisons, graphs and presentation of the data for sites are arranged from north to south.

2.3 Results

2.3.1 *Anthopleura*

The anemone *Anthopleura* was sampled at Boathouse, Alegria, Coal Oil Point, and Carpinteria in Santa Barbara County and at Mussel Shoals and Old Stairs in Ventura County (Figure II-1a, b). On average for all sample dates, cover varied widely among sites from about 10 to 70%. There were no significant seasonal trends in the cover of *Anthopleura* (Figure II-1a, b; Table II-1a). Nevertheless, abundances at some sites (e.g., Boathouse, Carpinteria) were consistently greater in spring than in autumn, but this pattern was not obvious when all sites were combined (Figure II-14a). At all 6 sites the cover of *Anthopleura* decreased over time, and this trend was significant at Boathouse, Alegria and Coal Oil Point (Figure II-1a, b; Table II-1a), but was not apparent when all sites were combined (Figure II-14a). The most consistent and striking declines in cover occurred at some sites from 1997 to 1998 (e.g., Alegria, Carpinteria, Mussel Shoals, Old Stairs), but declines in cover were also apparent in the early and mid 1990s. At some sites cover increased after declines in 1997/98 (Figure II-1a, b).

Figure II-1a: Mean (±se) percent cover of *Anthopleura* in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time. Note differences in scale on y-axis.

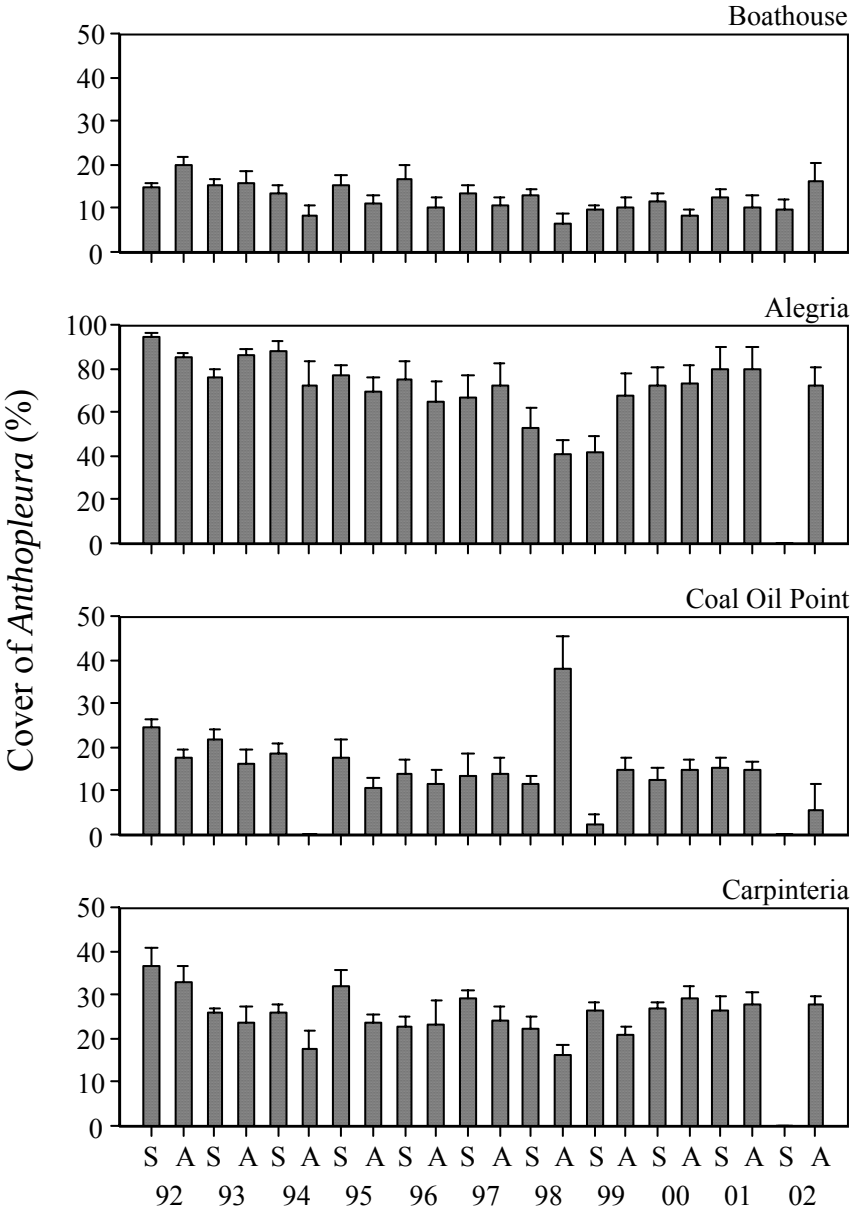


Figure II-1b: Mean (±se) percent cover of *Anthopleura* in spring (S) and autumn (A) from 1992 to 2002 at each of two sites. Blank spaces indicate that sampling was not done at that time.

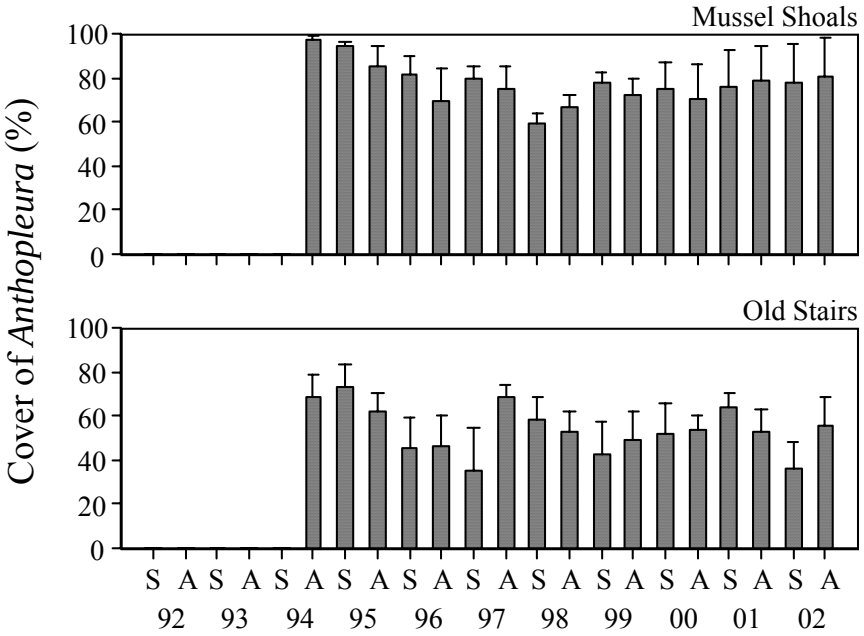


Table II-1a. Results of ANCOVA analyses examining seasonal and temporal patterns of abundance of target species. (S) indicates greater abundances in spring. (A) indicates greater abundances in autumn. (+) indicates an increasing linear trend over time. (–) indicates a decreasing linear trend over time. Significant results are in bold.

Species	Site	By Season	Over Time
Anthopleura	Boathouse	0.171	**0.003 (–)**
Anthopleura	Alegria	0.864	**0.005 (–)**
Anthopleura	Coal Oil Point	0.393	**0.009 (–)**
Anthopleura	Carpinteria	0.069	0.383
Anthopleura	Mussel Shoals	0.935	0.154
Anthopleura	Old Stairs	0.311	0.208
Barnacles	Point Sierra Nevada	0.174	**< 0.001 (–)**
Barnacles	Cayucos Point	**0.036 (A)**	**< 0.001 (–)**
Barnacles	Hazards	0.422	**0.016 (–)**
Barnacles	Shell Beach	0.604	0.681
Barnacles	Occulto	0.428	**< 0.001 (–)**
Barnacles	Stairs	0.410	**< 0.001 (–)**
Barnacles	Boathouse	0.866	0.940
Barnacles	Government Point	0.404	**< 0.001 (–)**
Barnacles	Alegria	**0.014 (A)**	0.097
Barnacles	Arroyo Hondo	0.124	**0.005 (–)**
Barnacles	Carpinteria	0.223	**< 0.001 (–)**
Barnacles	Mussel Shoals	**< 0.001 (A)**	0.549
Barnacles	Old Stairs	**0.030 (A)**	**< 0.001 (–)**
Barnacles	Paradise Cove	0.599	**0.007 (–)**
Barnacles	White's Point	0.347	**< 0.001 (–)**
Pollicipes	Government Point	0.718	**0.024 (–)**
Pollicipes	Alegria	0.181	0.386
Pollicipes	Carpinteria	**0.001 (A)**	**< 0.001 (–)**
Mussels	Point Sierra Nevada	0.537	0.155
Mussels	Cayucos Point	0.692	**< 0.001 (–)**
Mussels	Hazards	0.449	**< 0.001 (–)**
Mussels	Shell Beach	0.916	**< 0.001 (–)**
Mussels	Occulto	0.355	0.301
Mussels	Stairs	0.977	**< 0.001 (–)**
Mussels	Boathouse	0.719	**0.001 (–)**
Mussels	Government Point	0.769	**< 0.001 (–)**
Mussels	Alegria	0.128	**< 0.001 (–)**
Mussels	Arroyo Hondo	0.194	**< 0.001 (–)**
Mussels	Mussel Shoals	0.601	**0.008 (–)**
Mussels	Old Stairs	0.356	0.810
Mussels	Paradise Cove	0.375	**0.038 (–)**
Mussels	White's Point	0.961	0.809

2.3.2 Barnacles

Barnacles (*Chthamalus fissus/dalli* and *Balanus glandula*) were sampled at all sites except Piedras Blancas, Purisima Point and Coal Oil Point (Figure II-2a-d). On average for all sample dates, cover varied considerably among sites from about 25 to 70%. At 4 of 15 sites (Cayucos Point, Alegria, Mussel Shoals, Old Stairs) there were significant seasonal differences in abundance, with cover consistently greater in autumn than in spring (Figure II-2a-d; Table II-1a). This was a minor seasonal trend with changes in cover by on average 5 to 10% and this trend was also apparent when all sites were combined (Figure II-14a). The cover of barnacles decreased significantly at 11 of 15 sites (Figure II-2a-d; Table II-1a) and this decrease was clear for all sites combined (Figure II-14a). At some sites (e.g., Stairs, Boathouse) cover declined dramatically from 80 to 10%, whereas at others the decline was less striking or not apparent at all. There were two times of synchronous decline among some sites, the periods from 1992 to 1993 and 1997 to 1998, and this is reflected for all sites combined (Figure II-2a-d; Figure II-14a). There were also dynamic trends in the data, with frequent and substantial decreases and increases in the cover of barnacles over time. At some sites declines were followed by increases in cover in subsequent years (e.g., Boathouse, Alegria, Arroyo Hondo, White's Point), whereas at others cover did not increase after decline (e.g., Occulto, Stairs).

Figure II-2a: Mean (±se) percent cover of barnacles in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time. Note differences in scale on y-axis.

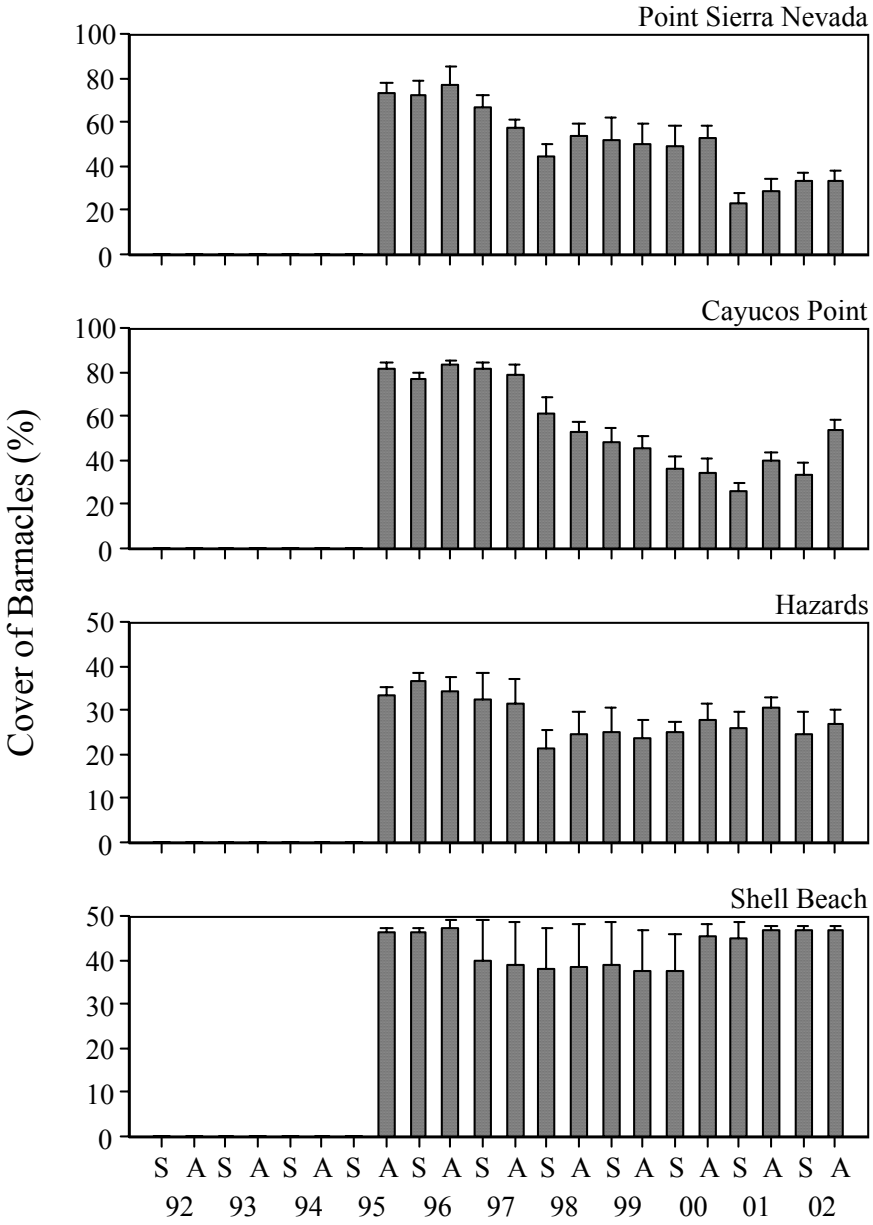


Figure II-2b: Mean (±se) percent cover of barnacles in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time.

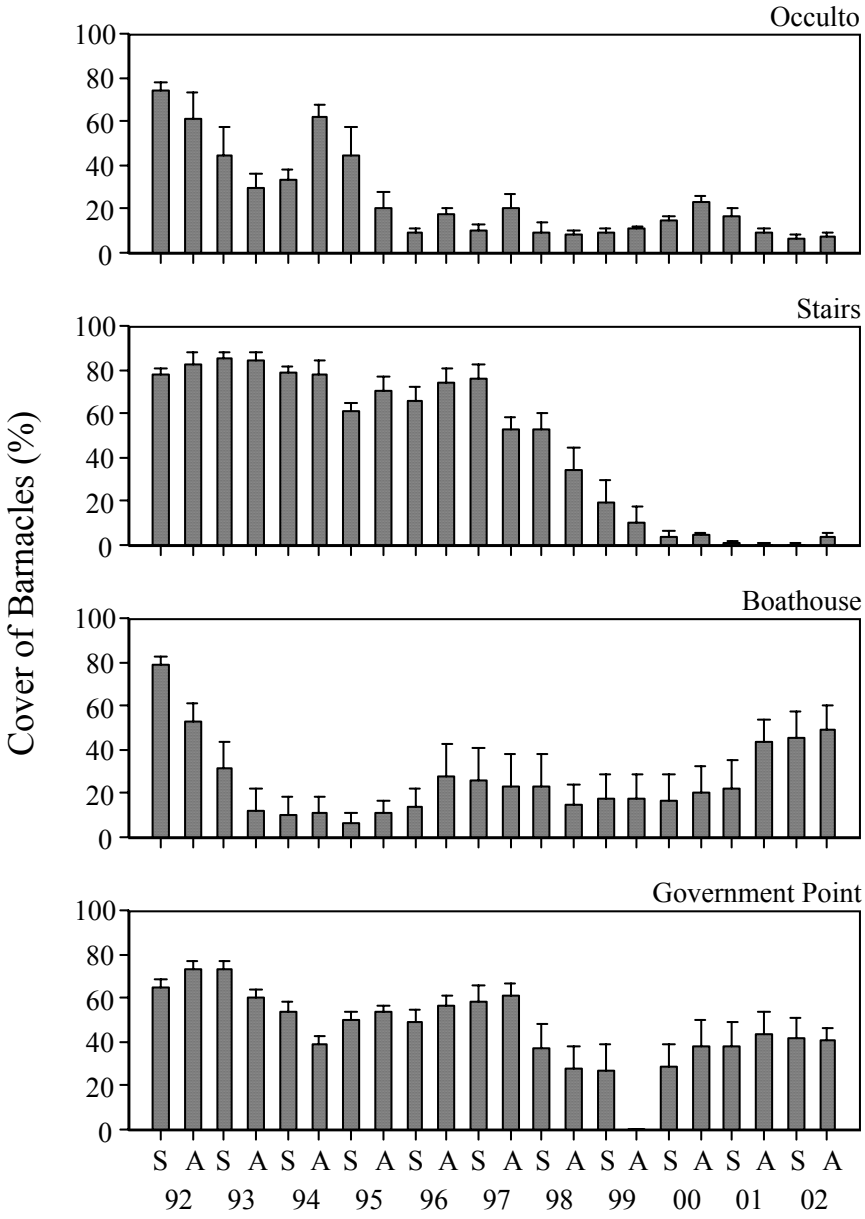


Figure II-2c: Mean (±se) percent cover of barnacles in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time.

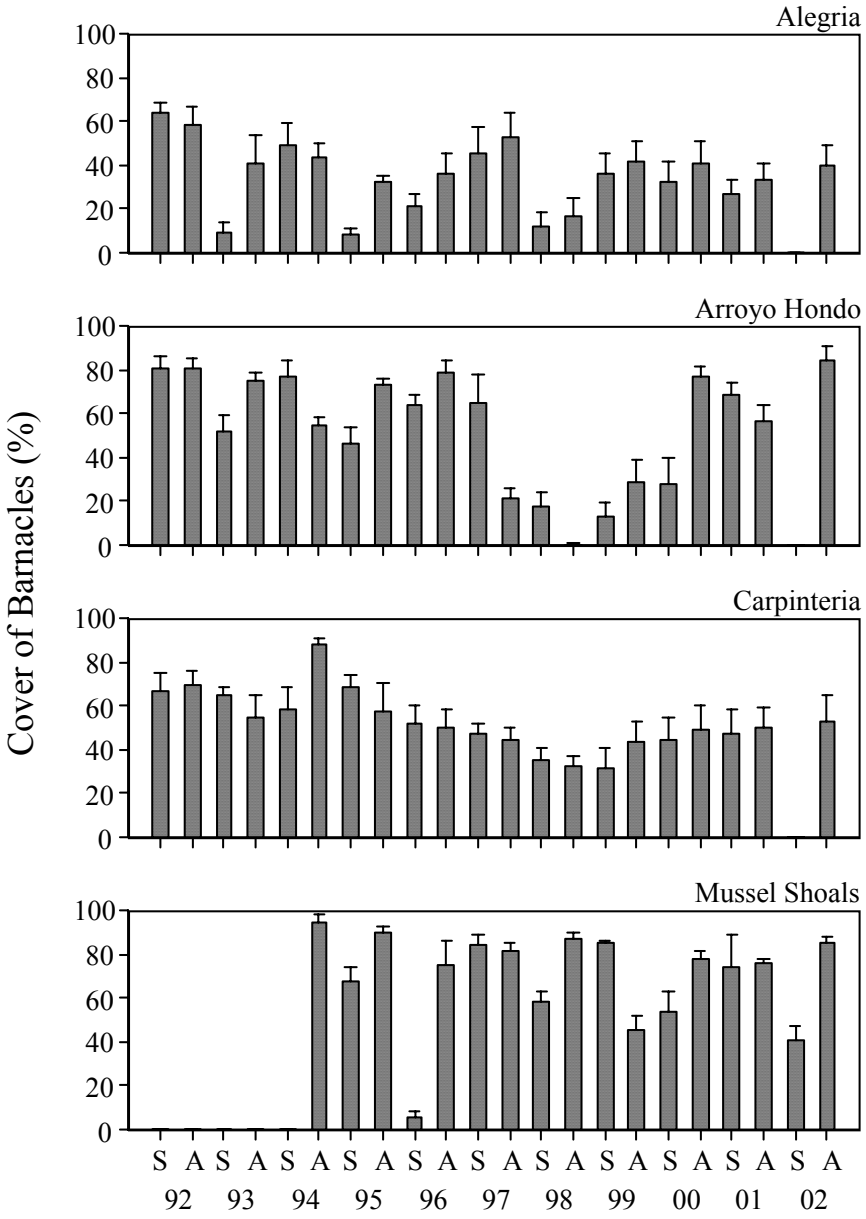
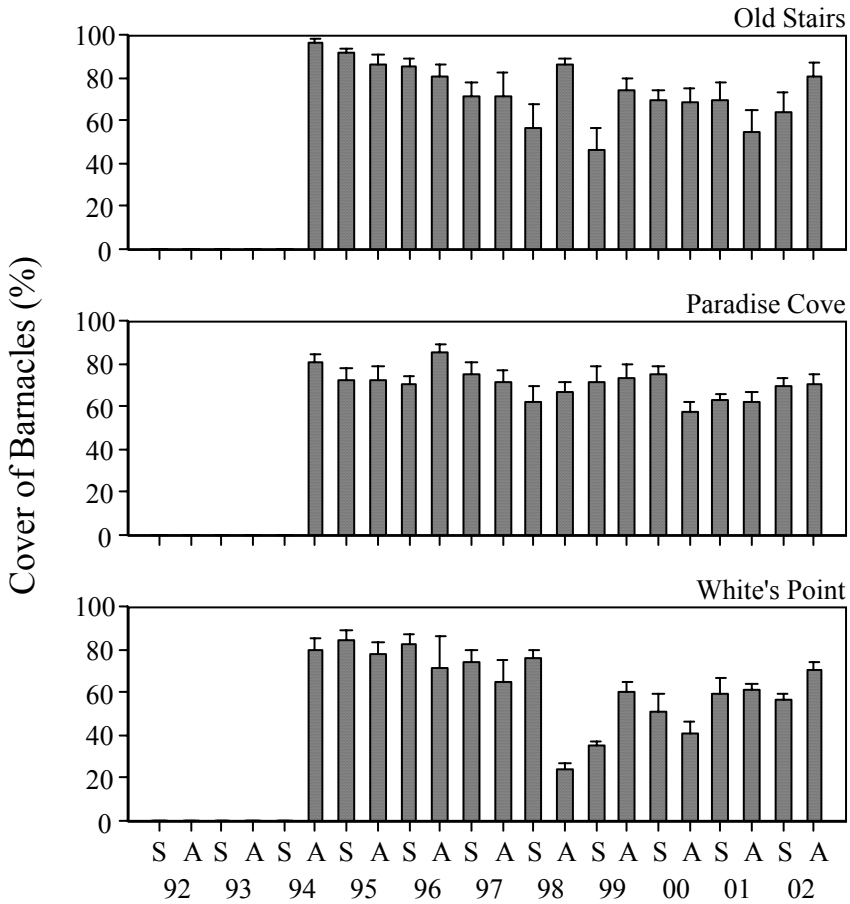


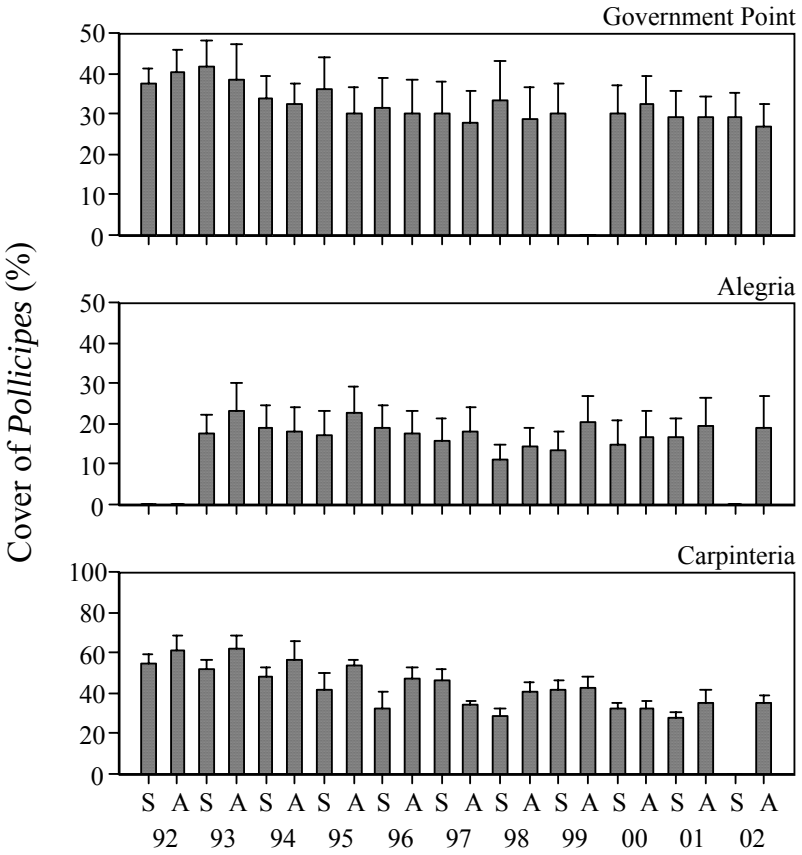
Figure II-2d: Mean (±se) percent cover of barnacles in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time.



2.3.3 *Pollicipes polymerus*

The gooseneck barnacle *Pollicipes* was only sampled at Government Point, Alegria, and Carpinteria in Santa Barbara County (Figure II-3). On average for all sample dates, cover varied among sites from about 15 to 40%. The cover of *Pollicipes* was significantly greater in autumn than in spring at Carpinteria, and this seasonal trend was apparent at Alegria but not at Government Point (Figure II-3; Table II-1a; Figure II-14a). The cover of *Pollicipes* declined gradually and significantly over time at Government Point and Carpinteria, but remained relatively consistent at Alegria (Figure II-1; Table II-1a).

Figure II-3: Mean (±se) percent cover of *Pollicipes* in spring (S) and autumn (A) from 1992 to 2002 at each of three sites. Blank spaces indicate that sampling was not done at that time. Note differences in scale on y-axis.



2.3.4 Mussels

Mussels (primarily *Mytilus californianus*) were sampled at all sites except Piedras Blancas, Purisima Point, Coal Oil Point and Carpinteria (sampled too infrequently for presentation) (Figure II-4a-d). On average for all sample dates, cover was consistently high and varied among sites from about 40 to 90%. There were no significant seasonal trends in mussel cover (Figure II-4a-d; Table II-1a). At 10 of 14 sites, mussel cover declined significantly over time (Figure II4a-d; Table II-1a; Figure II-14a). At most sites, the cover of mussels remained remarkably consistent from one sample period to the next, but at several sites cover was reduced by more than 50% (e.g., Shell Beach, Stairs, Arroyo Hondo, Old Stairs and also at Carpinteria, which is not presented). Declines were apparent at several sites from 1997 to 1998, and this was reflected when all sites were combined (Figure II-4a-d, Figure II-14a), but cover also decreased at other times or gradually over time. Similar to barnacles, decreases in cover were sometimes followed by increases, highlighting dynamic temporal patterns.

Figure II-4a: Mean (±se) percent cover of mussels in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time.

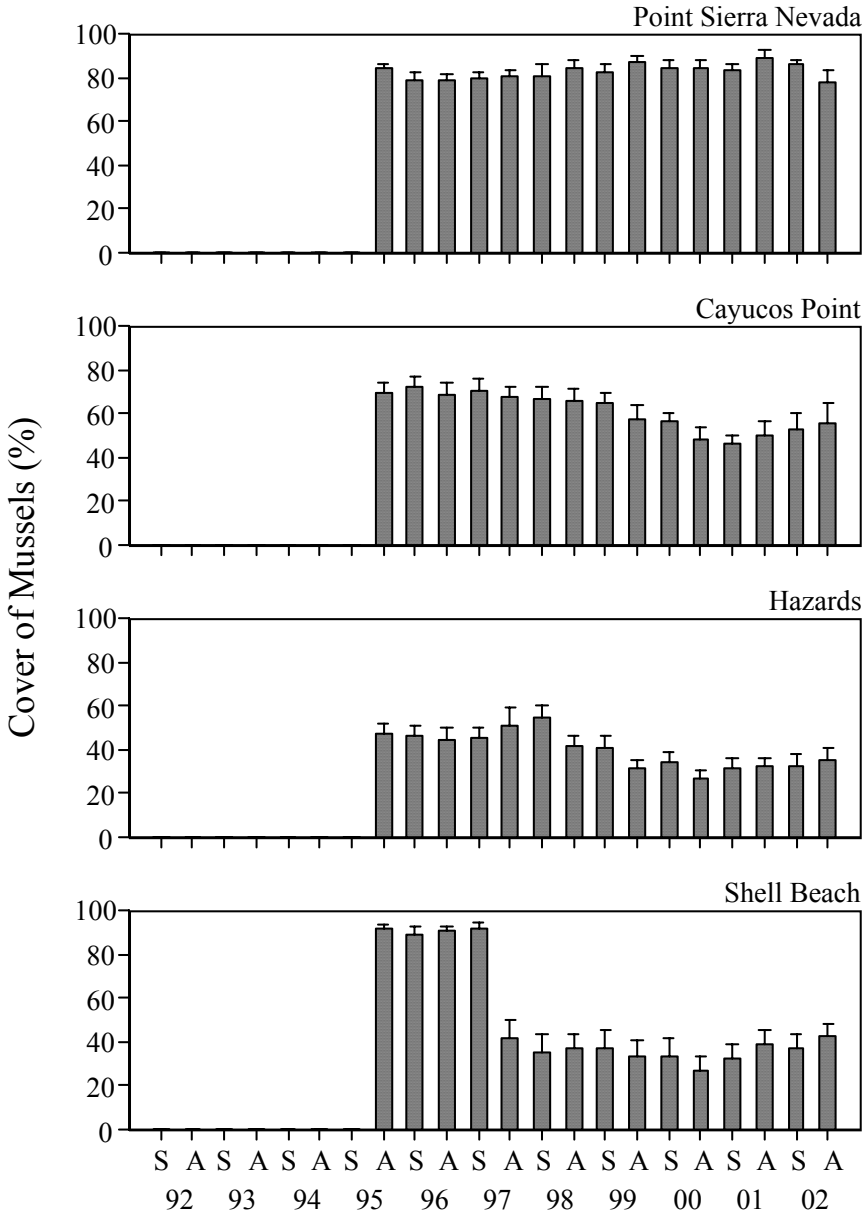


Figure II-4b: Mean (±se) percent cover of mussels in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time.

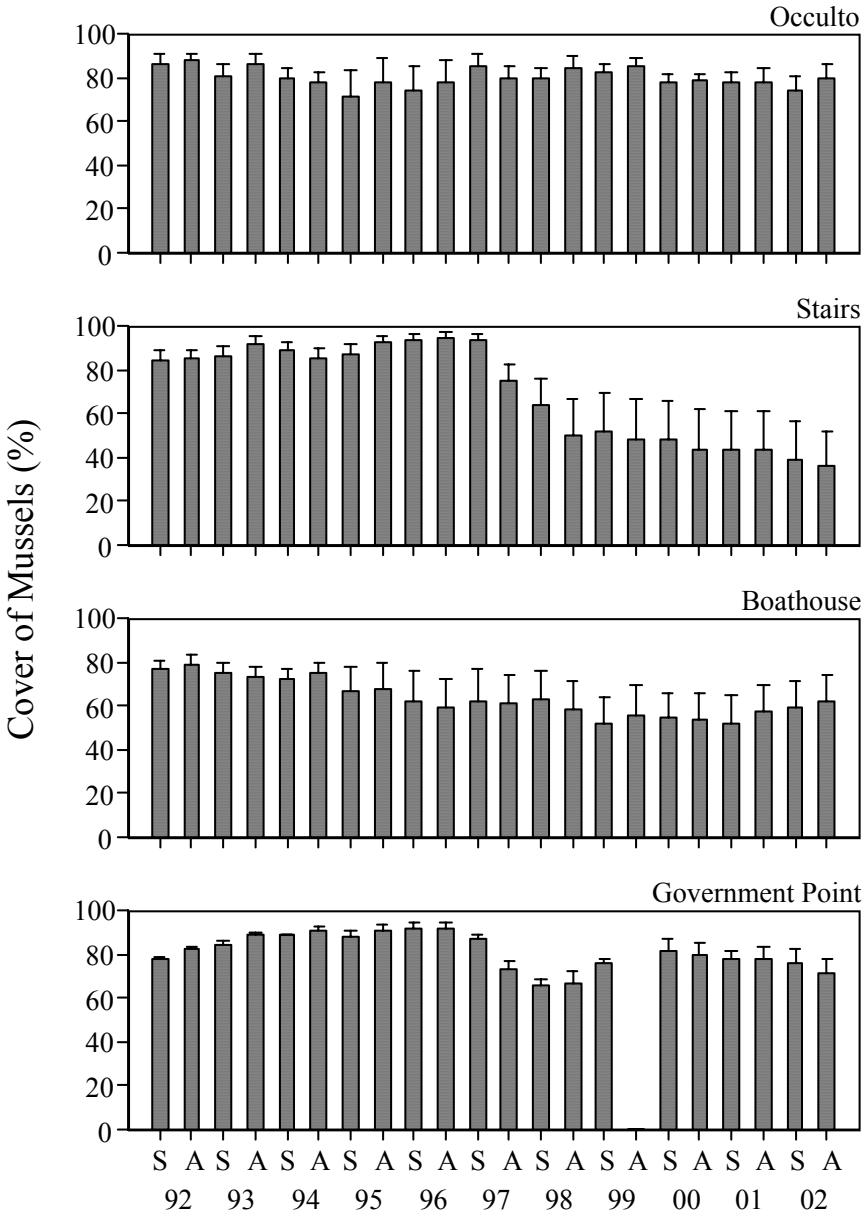


Figure II-4c: Mean (±se) percent cover of mussels in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time.

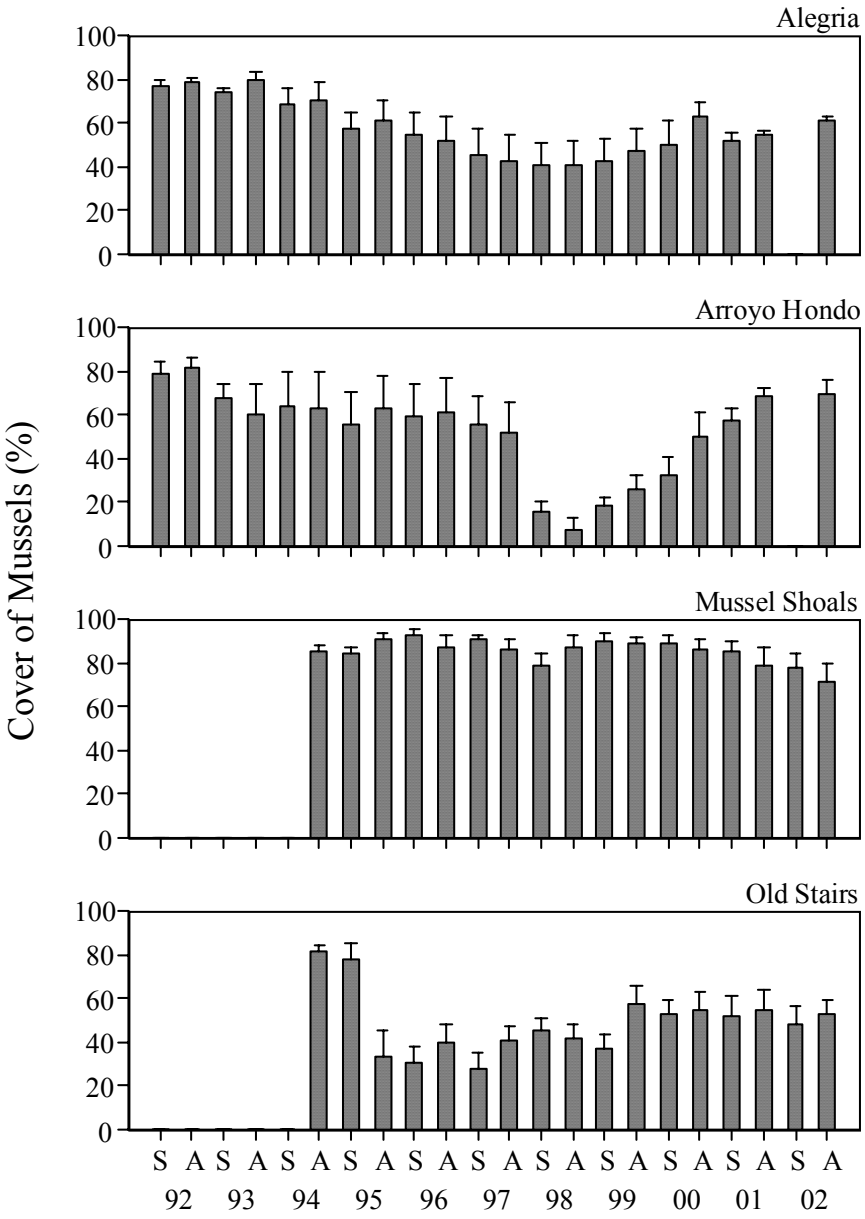
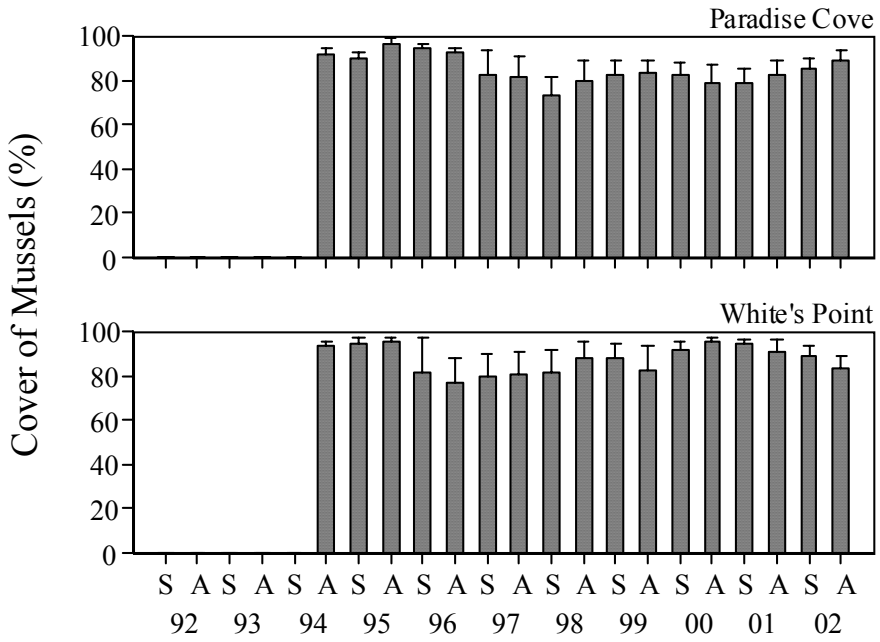


Figure II-4d: Mean (±se) percent cover of mussels in spring (S) and autumn (A) from 1992 to 2002 at each of two sites. Blank spaces indicate that sampling was not done at that time.



2.3.5 *Silvetia compressa*

The brown fucoid alga or rockweed *Silvetia compressa* was sampled at Point Sierra Nevada, Cayucos Point, Hazards and Shell Beach in San Luis Obispo County and at Stairs, Boathouse and Government Point in Santa Barbara County (Figure II-5a, b). On average for all sample dates, cover was consistently high and varied among sites from 50 to 80%. There were strong seasonal trends in the cover of *Silvetia*, with 4 of 7 sites having greater cover in autumn than in spring by on average 20% (Figure II-5a, b; Table II-1b). Except at Boathouse, the cover of *Silvetia* declined significantly over time at the other 6 sites (Figure II-5a, b; Table II-1b). All sites were established with relatively great amounts of cover and, at most sites, cover appeared to decrease gradually over time, with reductions by as much as 50%. There was, however, a relatively minor but synchronous decline in cover at most sites from 1997 to 1998. These seasonal and temporal patterns were reflected for all sites combined (Figure II-14b).

Figure II-5a: Mean (±se) percent cover of *Silvetia* in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time.

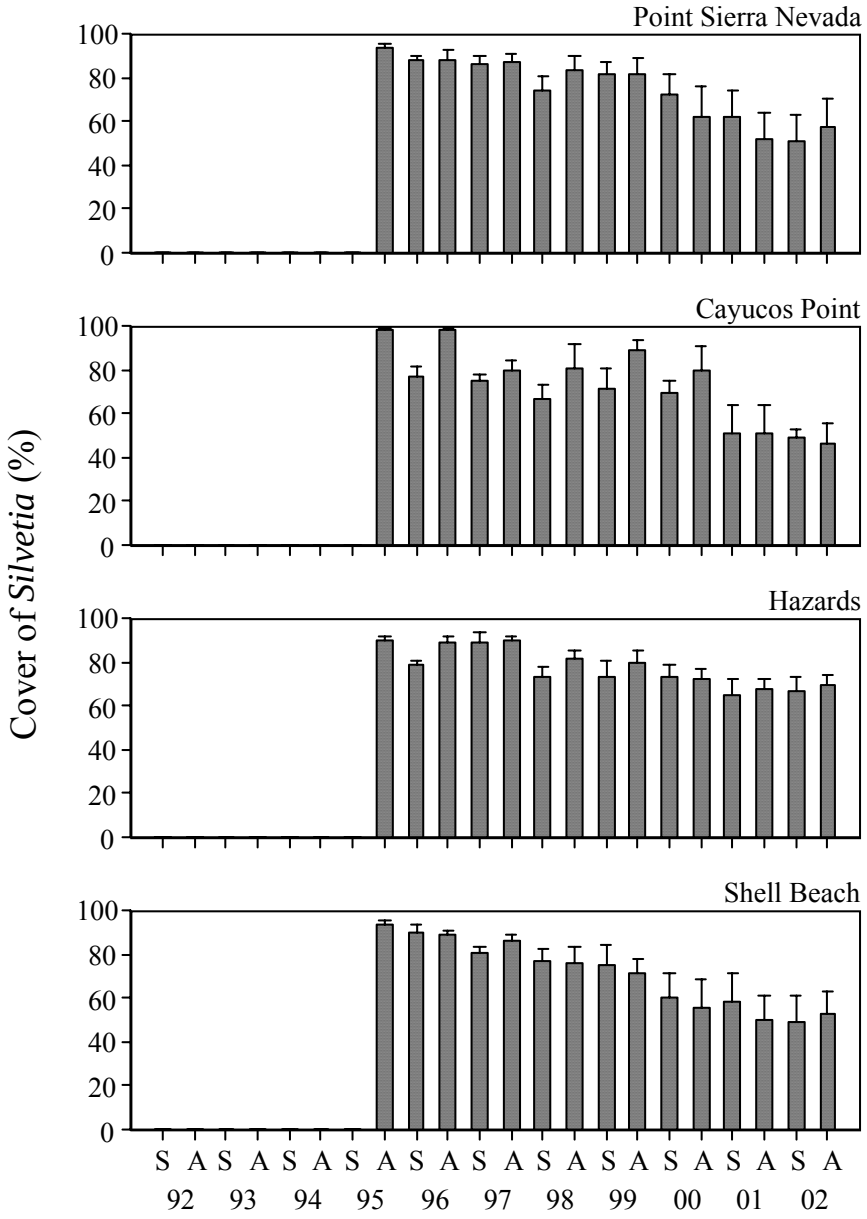


Figure II-5b: Mean (±se) percent cover of *Silvetia* in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time.

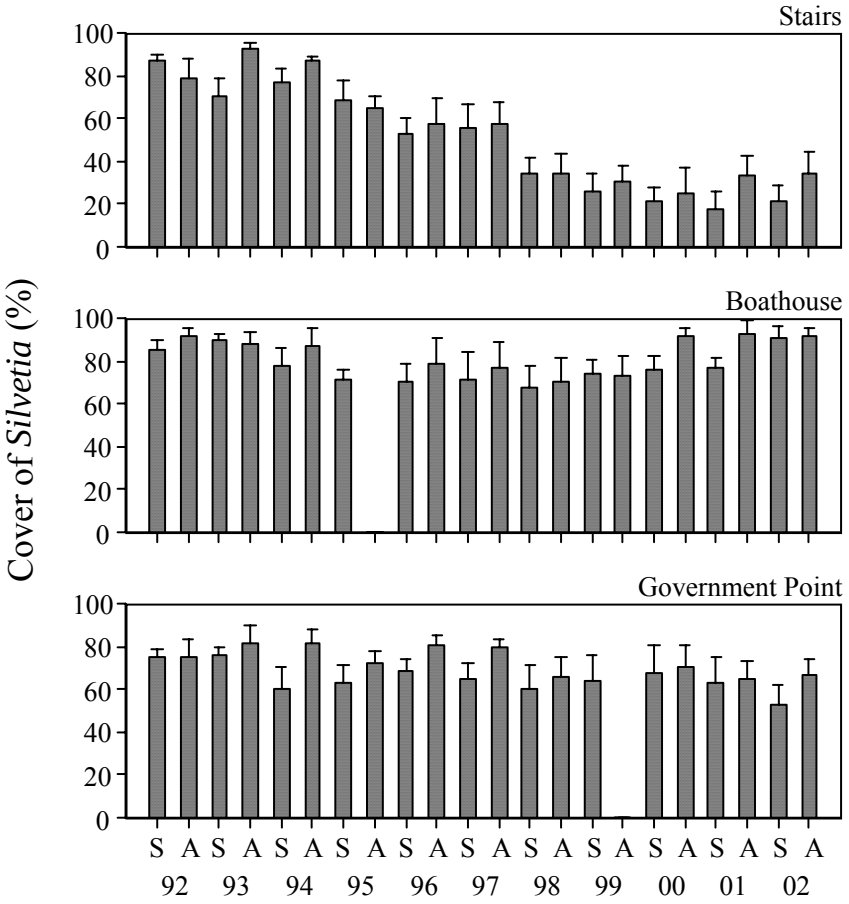


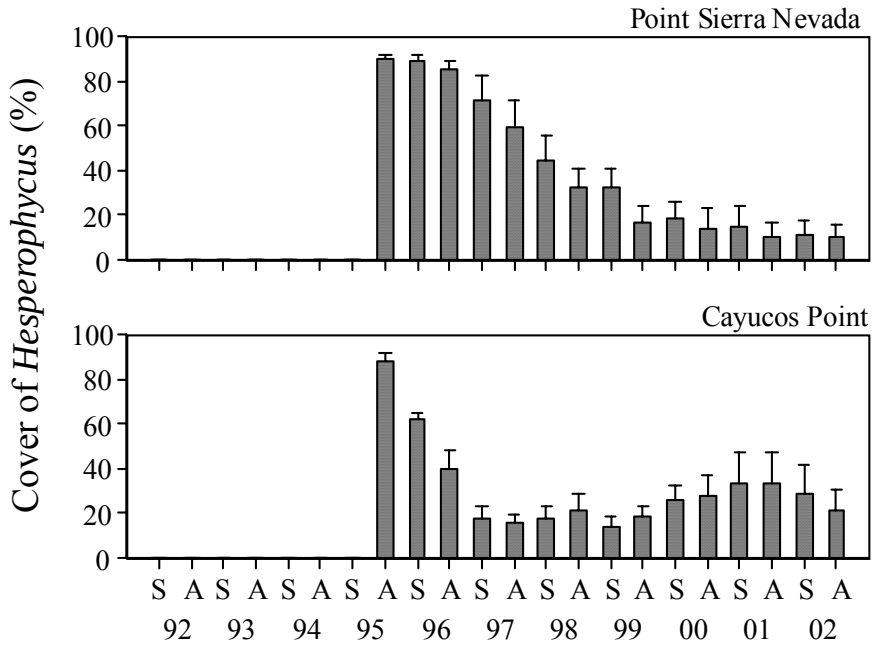
Table II-1b. Results of ANCOVA analyses examining seasonal and temporal patterns of abundance of target species. (S) indicates greater abundances in spring. (A) indicates greater abundances in autumn. (+) indicates an increasing linear trend over time. (–) indicates a decreasing linear trend over time. Significant results are in bold.

Species	Site	By Season	Over Time
Silvetia	Point Sierra Nevada	0.638	**< 0.001 (–)**
Silvetia	Cayucos Point	**0.004 (A)**	**< 0.001 (–)**
Silvetia	Hazards	**0.022 (A)**	**< 0.001 (–)**
Silvetia	Shell Beach	0.675	**< 0.001 (–)**
Silvetia	Stairs	**0.010 (A)**	**< 0.001 (–)**
Silvetia	Boathouse	0.051	0.827
Silvetia	Government Point	**0.009 (A)**	**0.011 (–)**
Hesperophycus	Point Sierra Nevada	0.898	**< 0.001 (–)**
Hesperophycus	Cayucos Point	0.408	**0.002 (–)**
Mastocarpus	Point Sierra Nevada	**< 0.001 (A)**	**< 0.001 (–)**
Mastocarpus	Shell Beach	0.719	**< 0.001 (–)**
Endocladia	Cayucos Point	**0.041 (S)**	**< 0.001 (–)**
Endocladia	Hazards	**< 0.001 (S)**	**< 0.001 (–)**
Endocladia	Shell Beach	**< 0.001 (S)**	**< 0.001 (–)**
Endocladia	Occulto	**< 0.001 (S)**	**< 0.001 (–)**
Endocladia	Stairs	**< 0.001 (S)**	**0.020 (–)**
Endocladia	Boathouse	0.100	**< 0.001 (–)**
Endocladia	Government Point	**< 0.001 (S)**	**0.003 (–)**
Endocladia	Old Stairs	0.645	**< 0.001 (–)**
Endocladia	Paradise Cove	0.141	**< 0.001 (+)**
Endocladia	White's Point	**< 0.001 (S)**	**< 0.001 (–)**
Mazzaella	Point Sierra Nevada	0.221	0.622
Mazzaella	Hazards	0.157	0.148
Phyllospadix	Point Sierra Nevada	**0.021 (A)**	0.440
Phyllospadix	Cayucos Point	0.090	0.148
Phyllospadix	Stairs	**0.020 (A)**	**< 0.001 (–)**
Phyllospadix	Government Point	**0.002 (A)**	0.118
Phyllospadix	Arroyo Hondo	**< 0.001 (A)**	0.180
Phyllospadix	Coal Oil Point	0.135	**< 0.001 (–)**
Phyllospadix	Carpentaria	**< 0.001 (A)**	**0.003 (–)**
Phyllospadix	Mussel Shoals	0.057	**< 0.001 (–)**
Phyllospadix	Paradise Cove	**0.004 (A)**	0.162

2.3.6 *Hesperophycus californicus*

The fucoid alga or rockweed *Hesperophycus californicus* was only sampled Point Sierra Nevada and Cayucos Point in San Luis Obispo County (Figure II-6). On average for all sample dates, cover was about 25% at both sites. In contrast to *Silvetia*, there was no seasonal pattern in the abundance of *Hesperophycus* (Figure II-6, Table II-1b). Similar to *Silvetia*, the cover of *Hesperophycus* decreased significantly over time, with cover at both sites declining dramatically from about 90 to 20% (Figure II-6, Table II-1b).

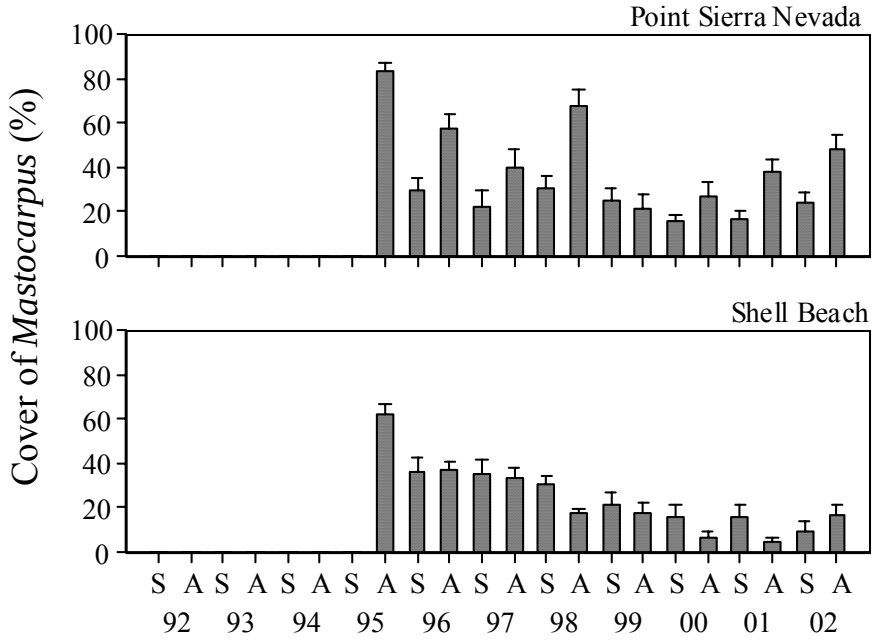
Figure II-6: Mean (±se) percent cover of *Hesperophycus* in spring (S) and autumn (A) from 1992 to 2002 at each of two sites. Blank spaces indicate that sampling was not done at that time.



2.3.7 *Mastocarpus papillatus*

The red alga *Mastocarpus papillatus* was only sampled at Point Sierra Nevada and Shell Beach in San Luis Obispo County (Figure II-7). On average for all sample periods, cover at both sites was about 20 to 30%. At Point Sierra Nevada there was a strong seasonal trend that was not apparent at Shell Beach, with cover of *Mastocarpus* in autumn greater than in spring by on average about 20% (Figure II-7; Table II-1b). At both sites, *Mastocarpus* decreased gradually and significantly over time, with no apparent pattern to the decline (Figure II-7; Table II-1b).

Figure II-7: Mean (±se) percent cover of *Mastocarpus* in spring (S) and autumn (A) from 1992 to 2002 at each of two sites. Blank spaces indicate that sampling was not done at that time.



2.3.8 *Endocladia muricata*

The red alga or turfweed *Endocladia muricata* was sampled at Cayucos Point, Hazards and Shell Beach in San Luis Obispo County, Occulto, Stairs, Boathouse and Government Point in Santa Barbara County, Old Stairs in Ventura County and Paradise Cove and White's Point in Los Angeles County (Figure II-8a-c). On average for all sampling dates, cover varied among sites from 20 to 50%. There were significant seasonal trends in the abundance of *Endocladia* at 7 of 10 sites, with cover greater by about 10 to 25% in spring than in autumn (Figure II-8a-c; Table II-b). At 9 of 10 sites, *Endocladia* cover gradually and significantly decreased over time, whereas at Paradise Cove the cover of *Endocladia* increased significantly (Figure II-8a-c; Table II-b). At 8 of 10 sites there was a clear decline in abundance from 1997 to 1998, followed by a slight increase in cover at some sites (Figure II-8a-c). A synchronous pattern of decline among sites was also evident in 1992 and 1993 at sites in Santa Barbara County (Figure II-8a-c). The increased cover in autumn and temporal pattern of decline are evident when all sites are combined (Figure II-14b).

Figure II-8a: Mean (±se) percent cover of *Endocladia* in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time.

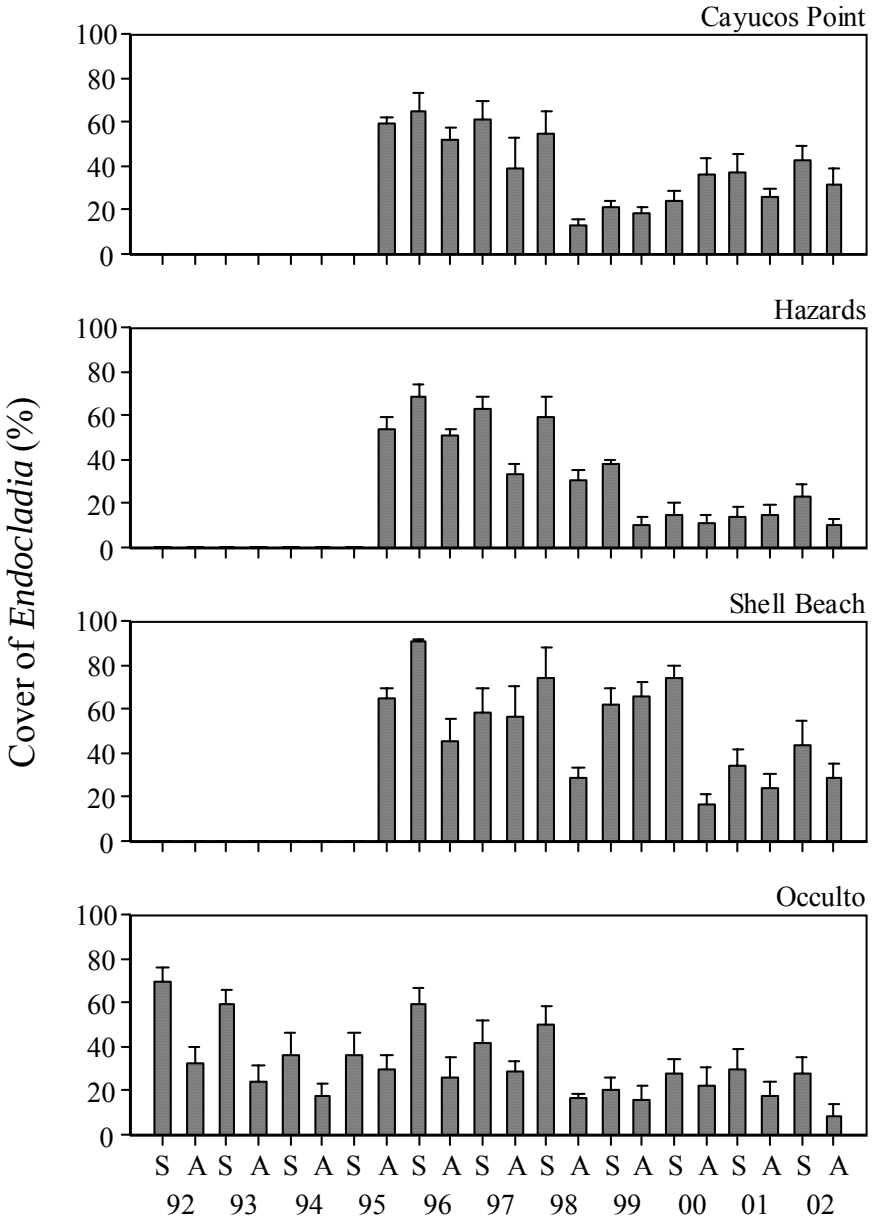


Figure II-8b: Mean (±se) percent cover of *Endocladia* in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time.

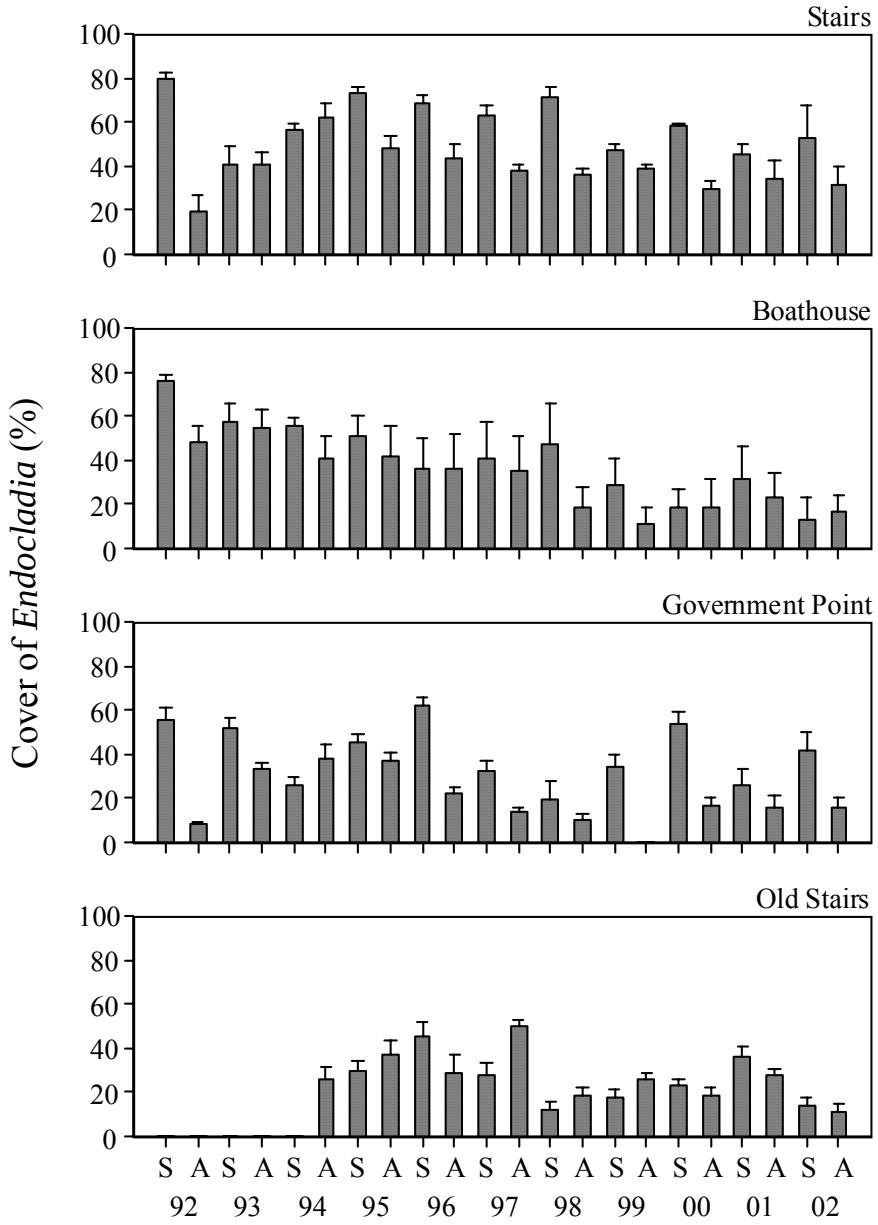
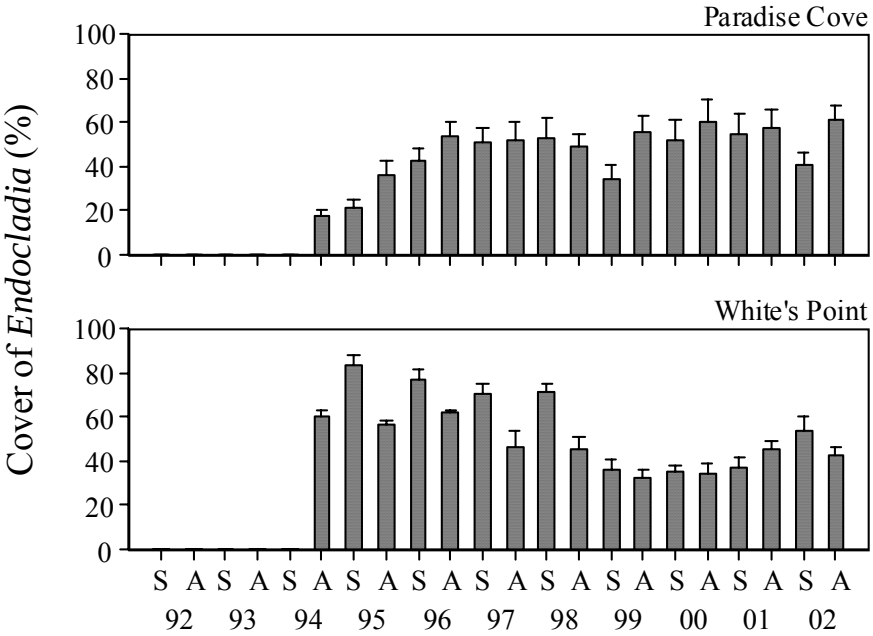


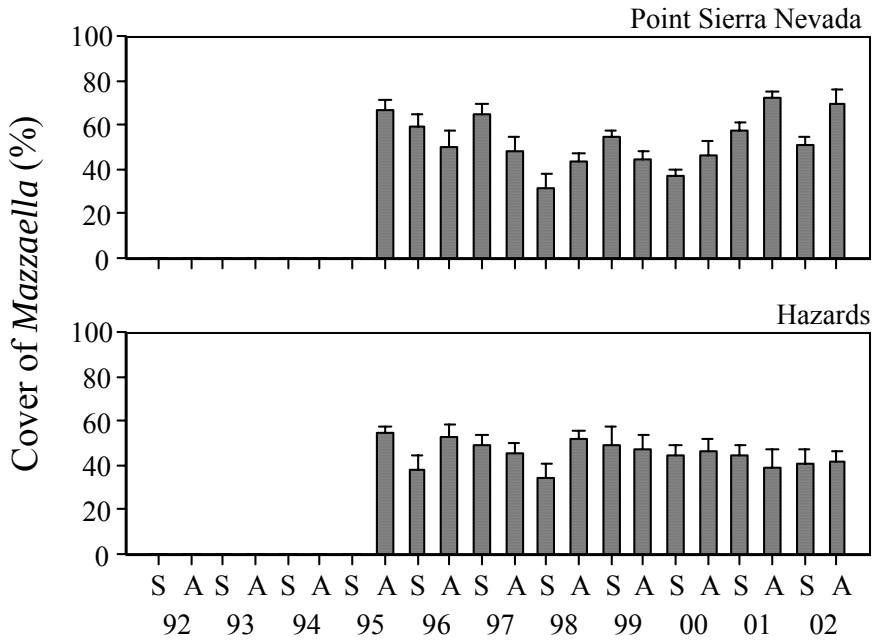
Figure II-8c: Mean (±se) percent cover of *Endocladia* in spring (S) and autumn (A) from 1992 to 2002 at each of two sites. Blank spaces indicate that sampling was not done at that time.



2.3.9 *Mazzaella* spp.

The alga *Mazzaella* spp. was only sampled at Point Sierra Nevada and Hazards in San Luis Obispo County (Figure II-9). On average for all sample periods, cover varied among sites from 40 to 50%. There were no significant seasonal (although cover was on average greater in the autumn) or temporal trends for this suite of species (Figure II-9; Table II-1b; Figure II-14c). Nevertheless, the abundance of *Mazzaella* declined marginally at both sites from 1997 to 1998 and then increased slightly thereafter (Figure II-9).

Figure II-9: Mean (±se) percent cover of *Mazzella* in spring (S) and autumn (A) from 1992 to 2002 at each of two sites. Blank spaces indicate that sampling was not done at that time.



2.3.10 *Phyllospadix scouleri/torreyi*

The surfgrass *Phyllospadix scouleri/torreyi* was sampled at Point Sierra Nevada and Cayucos Point in San Luis Obispo County, Stairs, Government Point, Coal Oil Point and Carpinteria in Santa Barbara County, Mussel Shoals in Ventura County and Paradise Cove in Los Angeles County (Figure II-10a-c). On average for all sample periods, cover was generally great and varied among sites from 50 to 90%. There were dramatic seasonal trends in the abundance of *Phyllospadix*, with 6 of 9 sites having on average about 5 to 20% and significantly greater cover in autumn than in the spring and the other sites, with the exception of Cayucos Point, showing a similar pattern (Figure II-10a-c; Table II-1b). At 4 of 9 sites there was a dramatic and significant decline in the cover of *Phyllospadix* (Figure II-10a-c,; Table II-1b). Reductions in cover occurred primarily from 1997 to 1998, and similar patterns were observed at two additional sites (Figure II-10a-c). At several sites (e.g., Carpinteria, Stairs), there were substantial increases in cover following these declines. These prominent seasonal and temporal trends remained clear when all sites were combined (Figure II-14c).

Figure II-10a: Mean (±se) percent cover of *Phyllospadix* in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time.

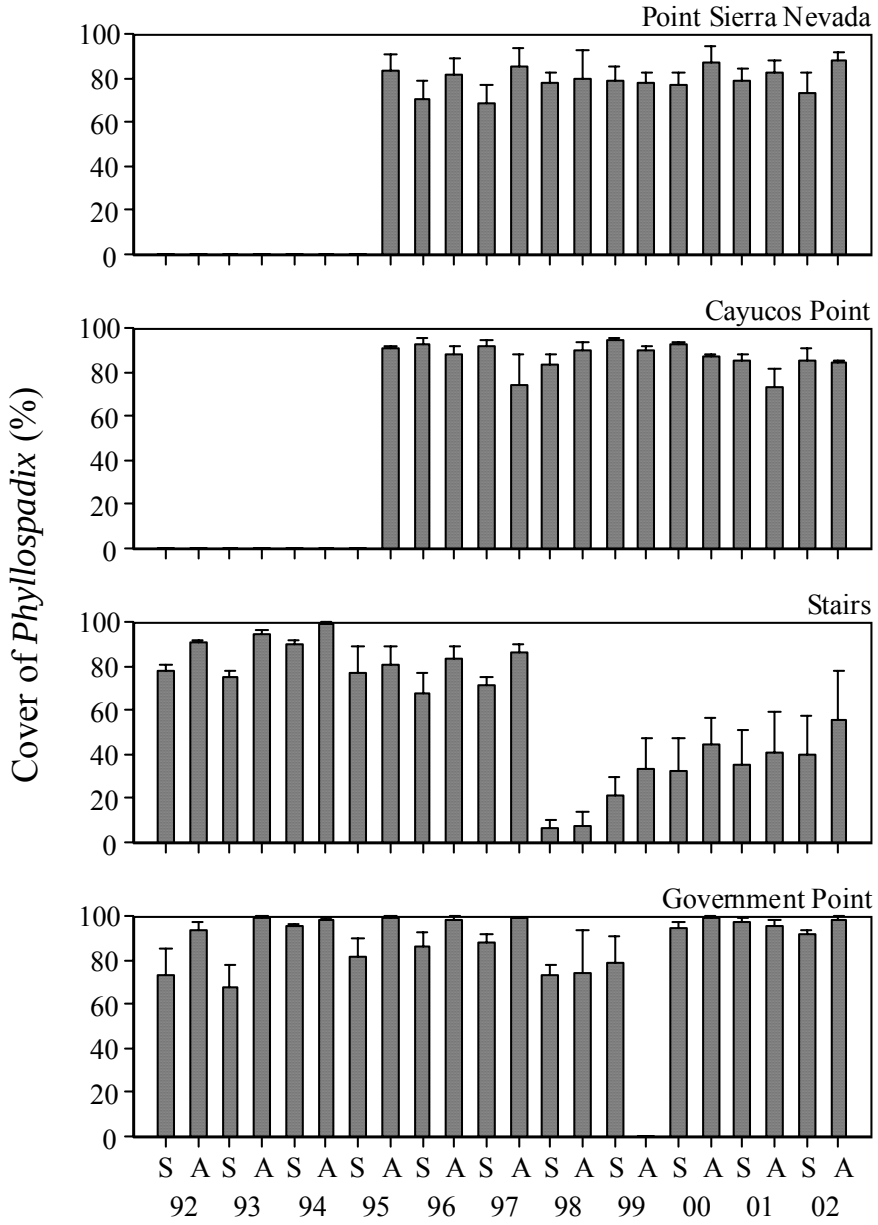


Figure II-10b: Mean (±se) percent cover of *Phyllospadix* in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time.

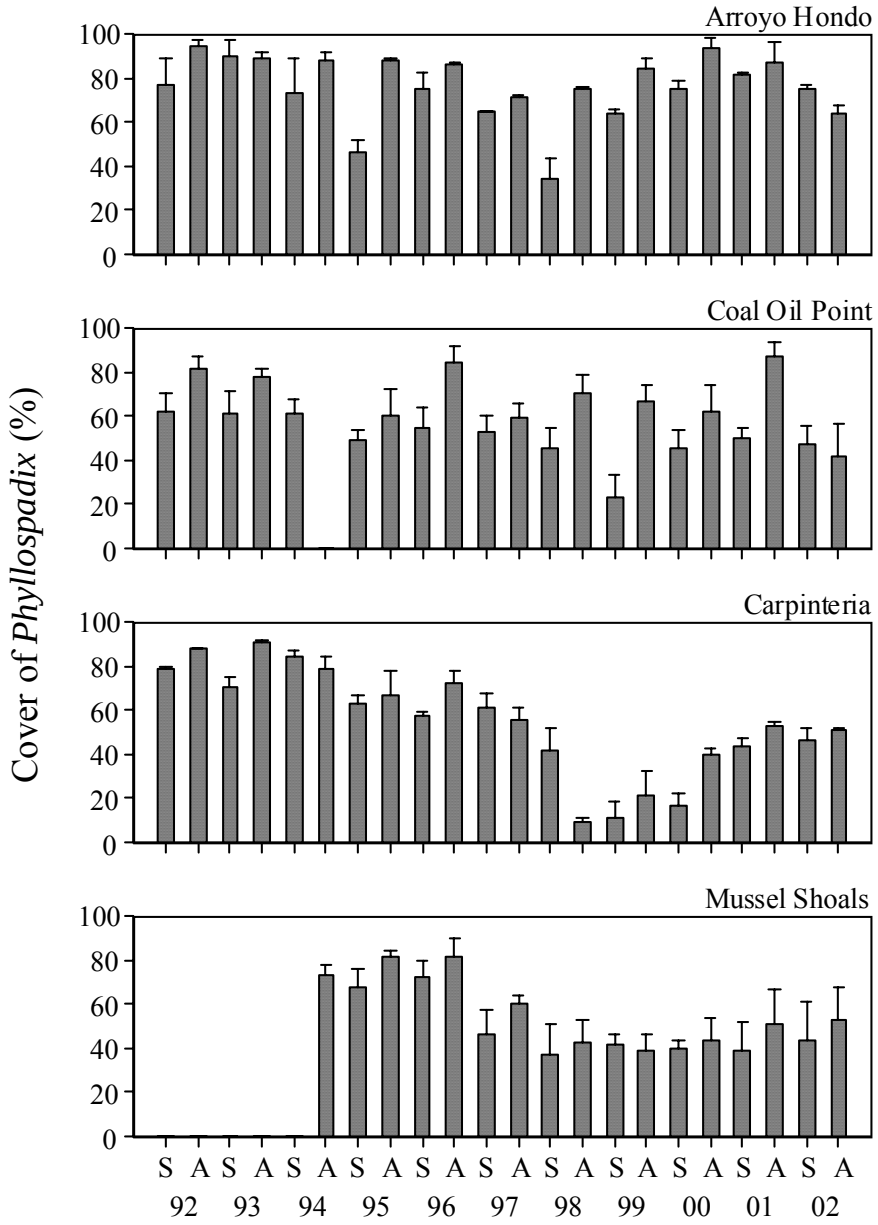
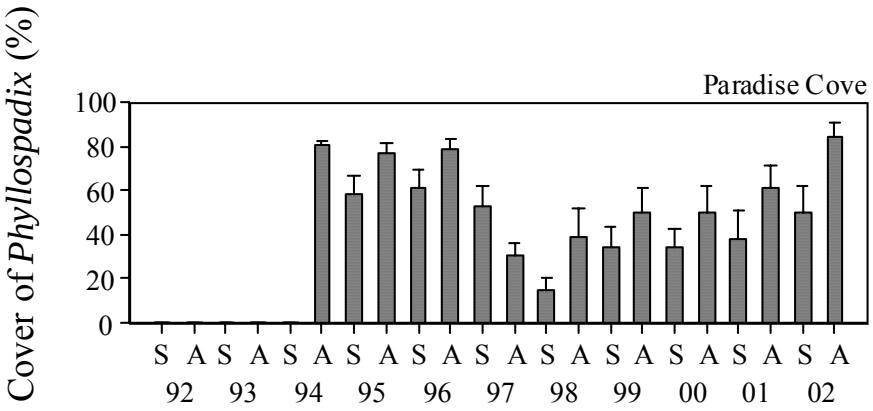


Figure II-10c: Mean (±se) percent cover of *Phyllospadix* in spring (S) and autumn (A) from 1992 to 2002 at each of two sites. Blank spaces indicate that sampling was not done at that time.



2.3.11 *Pisaster ochraceus*

The ochre sea star *Pisaster ochraceus* was sampled at Point Sierra Nevada, Cayucos Point, Hazards and Shell Beach in San Luis Obispo County, at Stairs, Boathouse, Government Point and Arroyo Hondo in Santa Barbara County and at Mussel Shoals and Old Stair in Ventura County (Figure II-11b). On average for all sample periods, numbers varied among sites from less than 10 to about 30. In general, numbers were quite variable among plots and sites over time. At 2 northern sites in San Luis Obispo County, densities were significantly greater in autumn than in spring (Figure II-11a-c; Table II-1c). There were no other clear seasonal patterns at the other sites. *Pisaster* densities significantly increased at Point Sierra Nevada and Cayucos Point, significantly decreased at Mussel Shoals, and displayed a variety of temporal patterns at the other sites (Figure II-11a-c; Table II-1c). For all sites combined, there appears to be a slight decrease in the abundance of *Pisaster* from 1992 to 1993 and again after 1997, which is then followed by an increase (Figure II-14d)

Figure II-11a: Mean (±se) density of *Pisaster* in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time. Note differences in scale on y-axis.

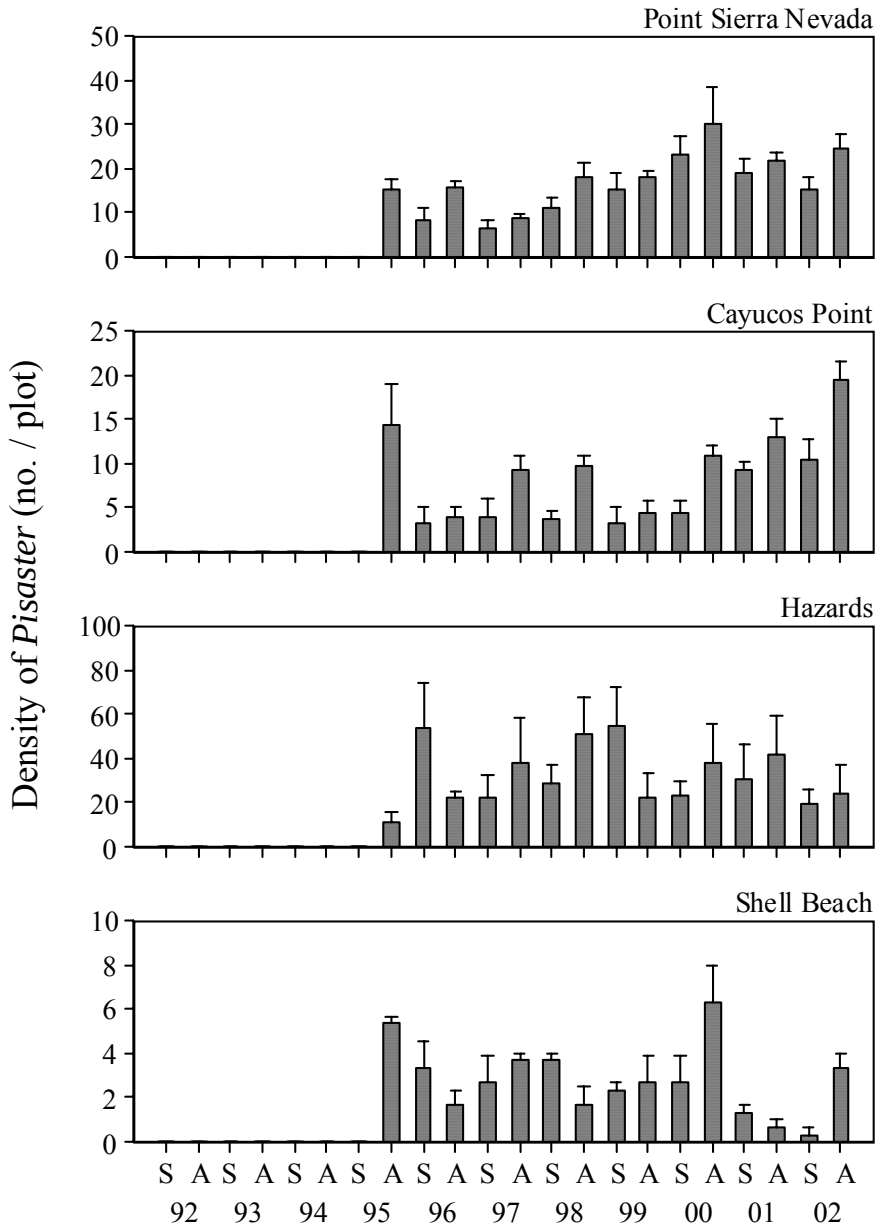


Figure II-11b: Mean (±se) density of *Pisaster* in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time. Note differences in scale on y-axis.

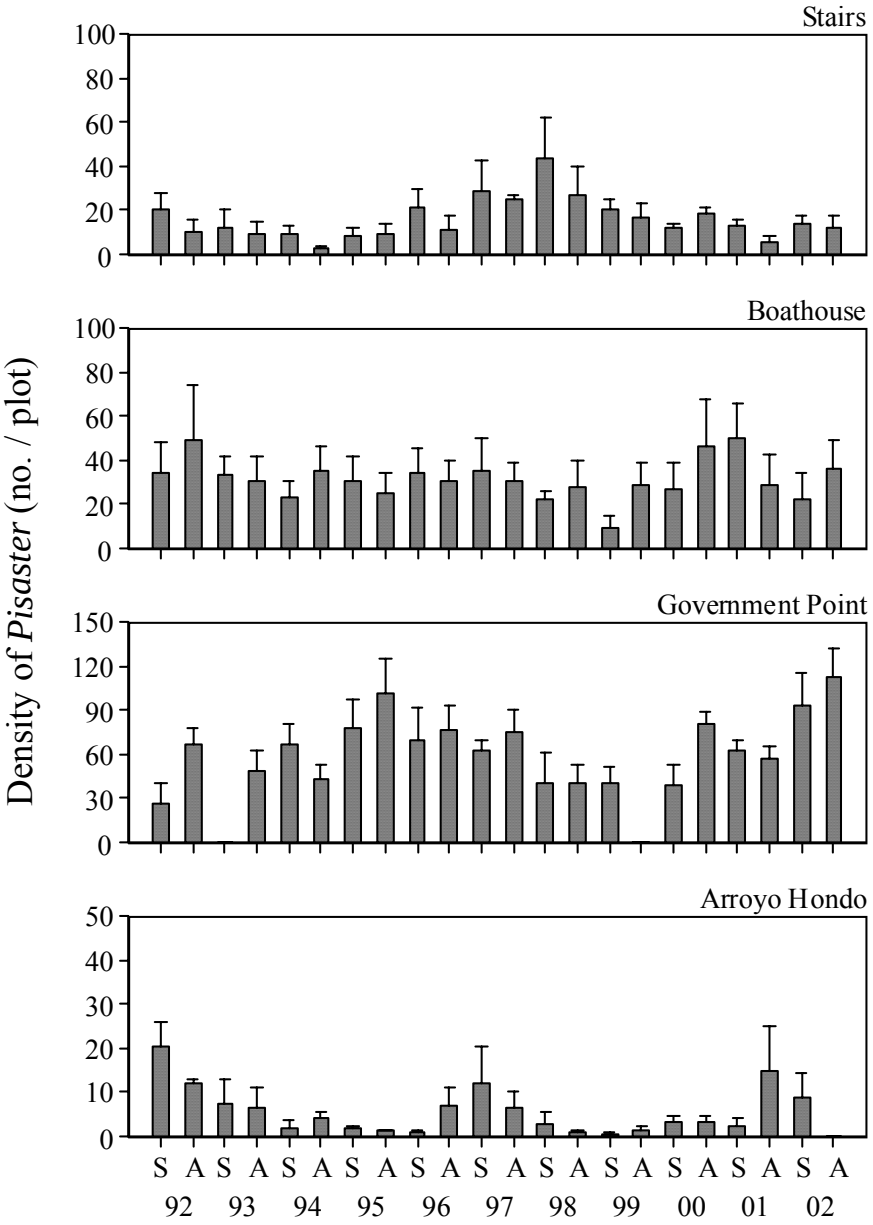


Figure II-11c: Mean (±se) density of *Pisaster* in spring (S) and autumn (A) from 1992 to 2002 at each of two sites. Blank spaces indicate that sampling was not done at that time. Note differences in scale on y-axis.

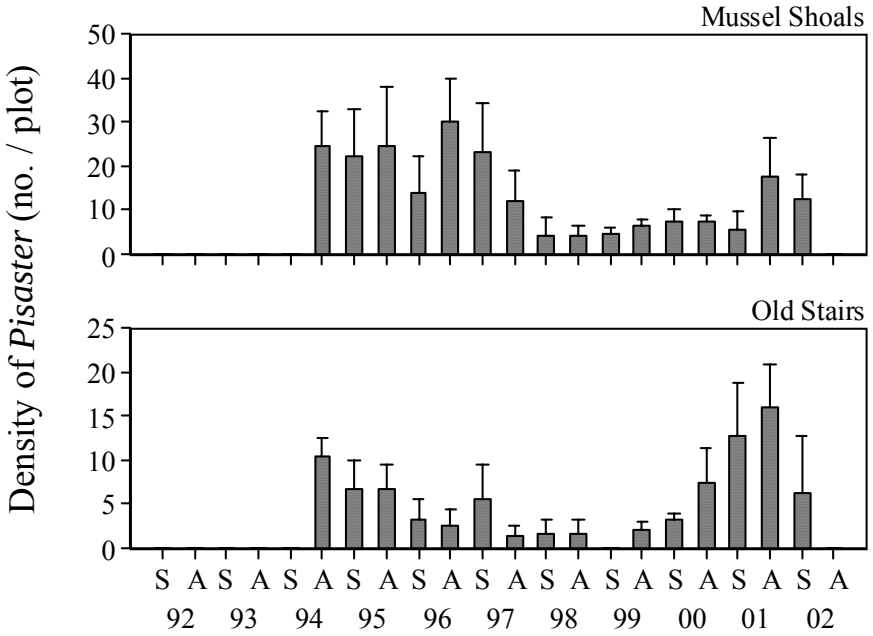


Table II-1c. Results of ANCOVA analyses examining seasonal and temporal patterns of abundance of target species. (S) indicates greater abundances in spring. (A) indicates greater abundances in autumn. (+) indicates an increasing linear trend over time. (–) indicates a decreasing linear trend over time. Significant results are in bold.

Species	Site	By Season	Over Time
Pisaster	Point Sierra Nevada	**0.014 (A)**	**< 0.001 (+)**
Pisaster	Cayucos Point	**< 0.001 (A)**	**< 0.001 (+)**
Pisaster	Hazards	0.752	0.904
Pisaster	Shell Beach	0.154	0.063
Pisaster	Stairs	0.124	0.399
Pisaster	Boathouse	0.380	0.703
Pisaster	Government Point	0.131	0.083
Pisaster	Arroyo Hondo	0.996	0.194
Pisaster	Mussel Shoals	0.427	**0.009 (–)**
Pisaster	Old Stairs	0.501	0.266
Lottia	Cayucos Point	0.885	**0.010 (–)**
Lottia	Hazards	0.480	0.335
Lottia	Stairs	0.404	**< 0.001 (+)**
Lottia	Boathouse	0.798	0.366
Lottia	Government Point	0.465	0.085
Lottia	Alegria	0.734	**< 0.001 (+)**
Haliotis	Point Sierra Nevada	0.533	**0.006 (+)**
Haliotis	Piedras Blancas	0.977	0.614
Haliotis	Cayucos Point	0.788	**< 0.001 (–)**
Haliotis	Purisima Point	0.822	**< 0.001 (–)**
Haliotis	Stairs	0.825	**< 0.001 (–)**
Haliotis	Boathouse	0.827	**< 0.001 (–)**
Haliotis	Government Point	0.483	**< 0.001 (–)**

2.3.12 *Lottia gigantea*

The giant owl limpet *Lottia gigantea* was sampled at Cayucos Point and Hazards in San Luis Obispo County and Stairs, Boathouse, Government Point and Alegria in Santa Barbara County (Figure II-12b). On average for all sample periods, numbers varied among sites from 20 to 50. Densities were relatively consistent from one sample period to the next and there were no seasonal patterns of abundance (Figure II-12b; Table II-1c). The density of *Lottia* declined significantly at Cayucos Point, remained relatively constant at Hazards, Boathouse and Government Point and increased significantly at Stairs and Alegria (Figure II-12b; Table II-1c). Despite these patterns, there was a general decline at several sites from 1995 to 1997 and an overall increase in *Lottia* density from 1997 to 1998, which was particularly evident when all sites were combined (Figure II-14d).

Figure II-12a: Mean (±se) density of *Lottia* in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time. Note differences in scale on y-axis.

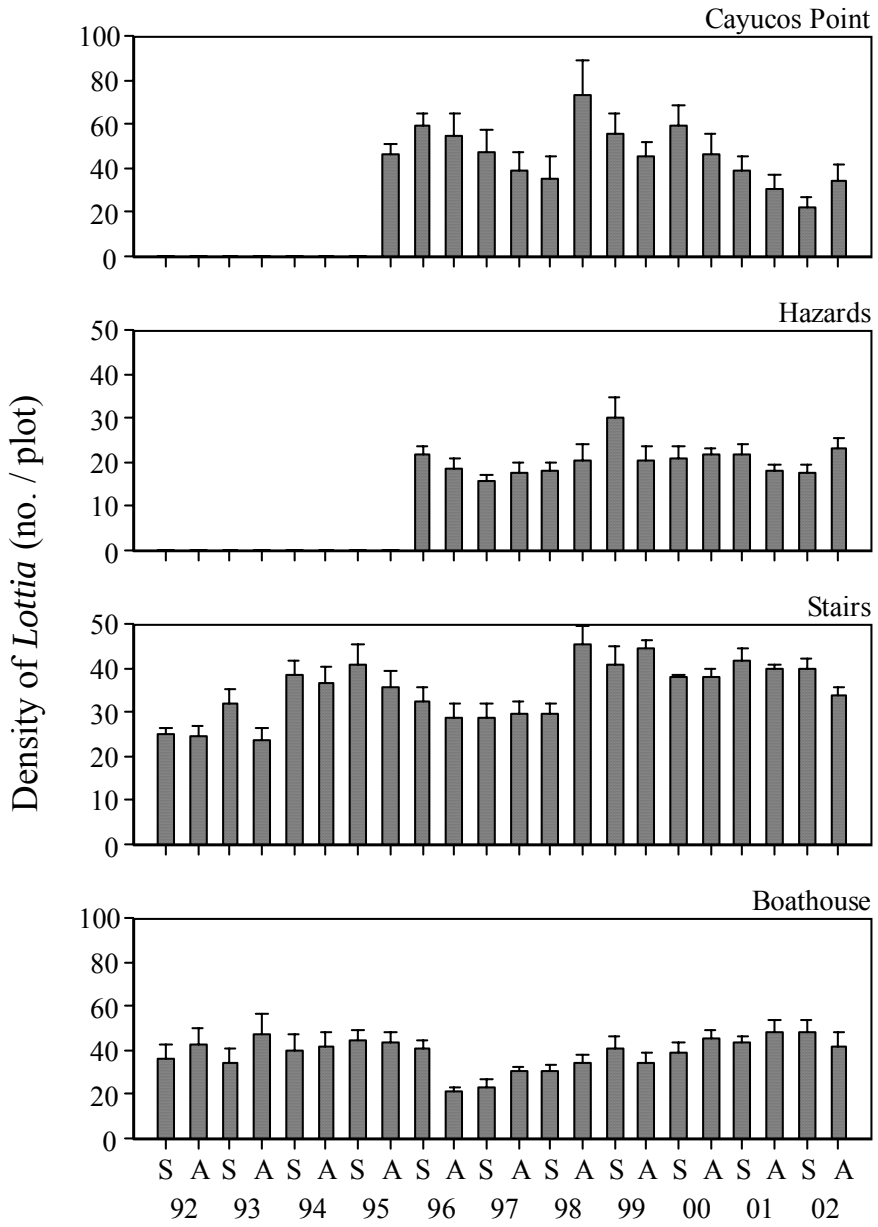
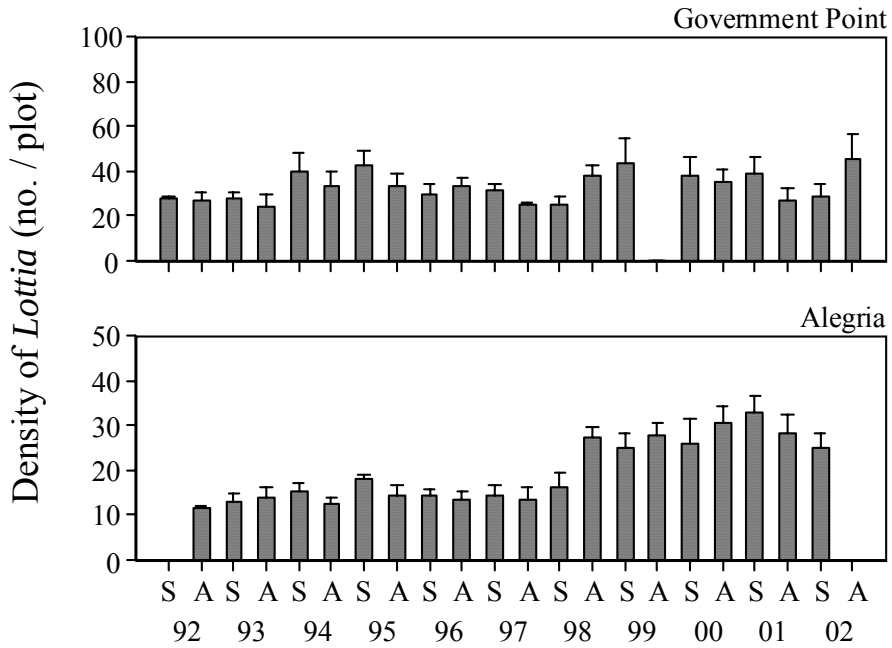


Figure II-12b: Mean (±se) density of *Lottia* in spring (S) and autumn (A) from 1992 to 2002 at each of two sites. Blank spaces indicate that sampling was not done at that time. Note differences in scale on y-axis.



2.3.13 *Haliotis cracherodii*

The black abalone *Haliotis cracherodii* was sampled at Point Sierra Nevada, Piedras Blancas, Cayucos Point and Purisima Point in San Luis Obispo County and at Stairs, Boathouse and Government Point in Santa Barbara County (Figure II-13a, b). On average for all sample periods, *Haliotis* numbers varied among sites from about 70 to less than 10 individuals per plot. There were no seasonal trends in abundance (Figure II-13a, b). There were dramatic and significant declines in the abundance of *Haliotis* at the 5 most southern sites, but not at the 2 most northern sites, Piedras Blancas and Point Sierra Nevada, the latter site exhibiting a significant increase in *Haliotis* abundance (Figure II-13a, b; Table II-1c; Figure II-14d). The timing of declines varied with latitude, with declines happening sooner at the southern than the northern sites.

Figure II-13a: Mean (±se) density of *Haliotis* in spring (S) and autumn (A) from 1992 to 2002 at each of four sites. Blank spaces indicate that sampling was not done at that time. Note differences in scale on y-axis.

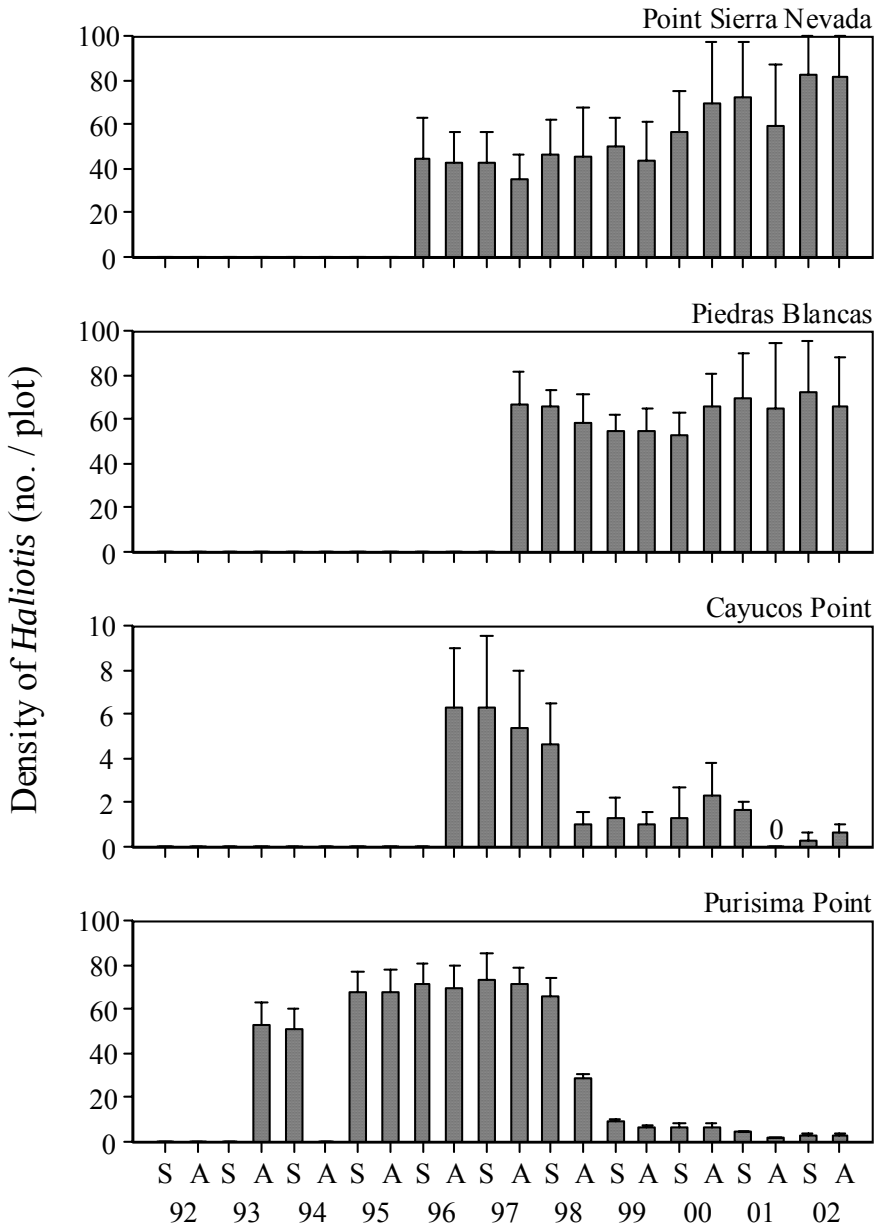


Figure II-13b: Mean (±se) density of *Haliotis* in spring (S) and autumn (A) from 1992 to 2002 at each of three sites. Blank spaces indicate that sampling was not done at that time. Note differences in scale on y-axis.

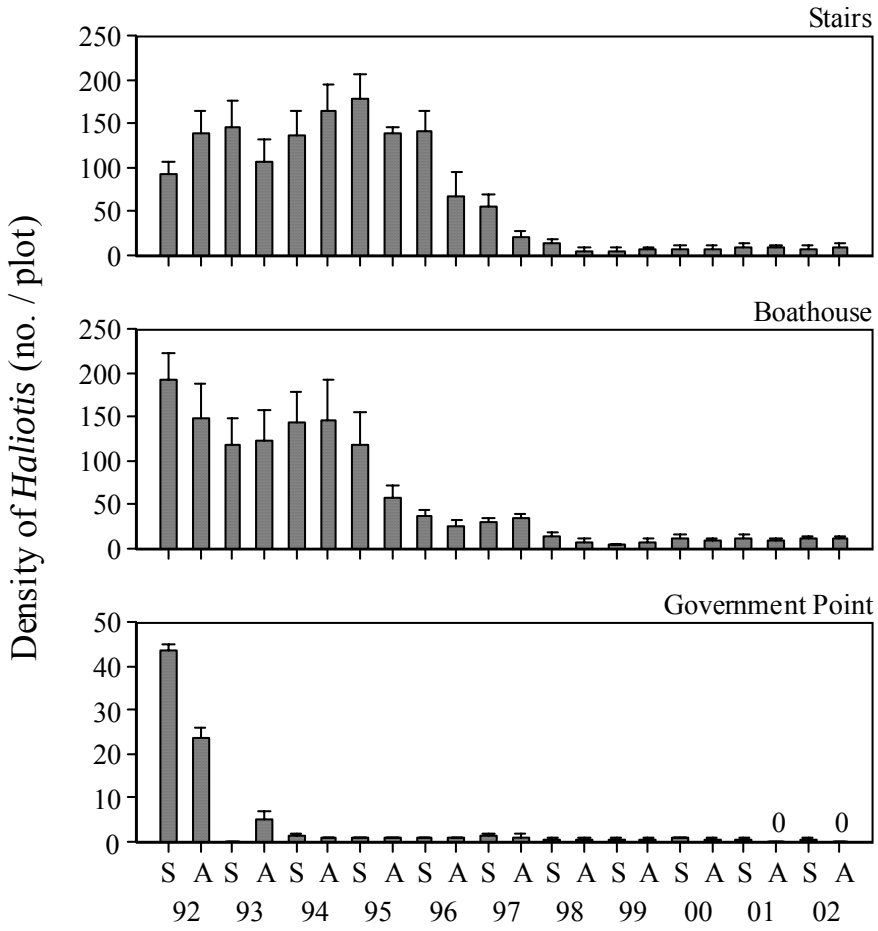


Figure II-14a: Mean (±se) percent cover of various species in spring (S) and autumn (A) from 1992 to 2002 for all sites (sites are replicates).

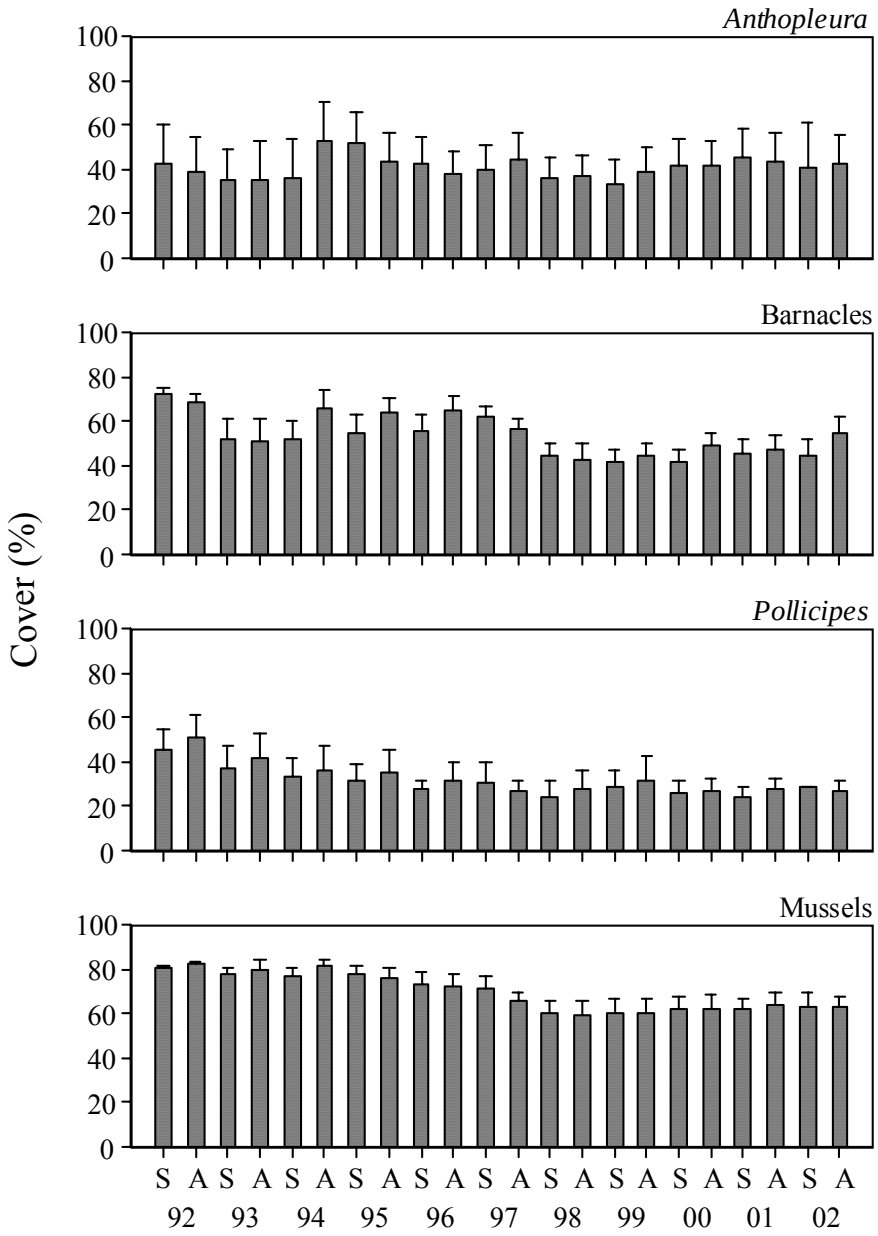


Figure II-14b: Mean (±se) percent cover of various species in spring (S) and autumn (A) from 1992 to 2002 for all sites (sites are replicates). Blank spaces indicate that sampling was not done at that time.

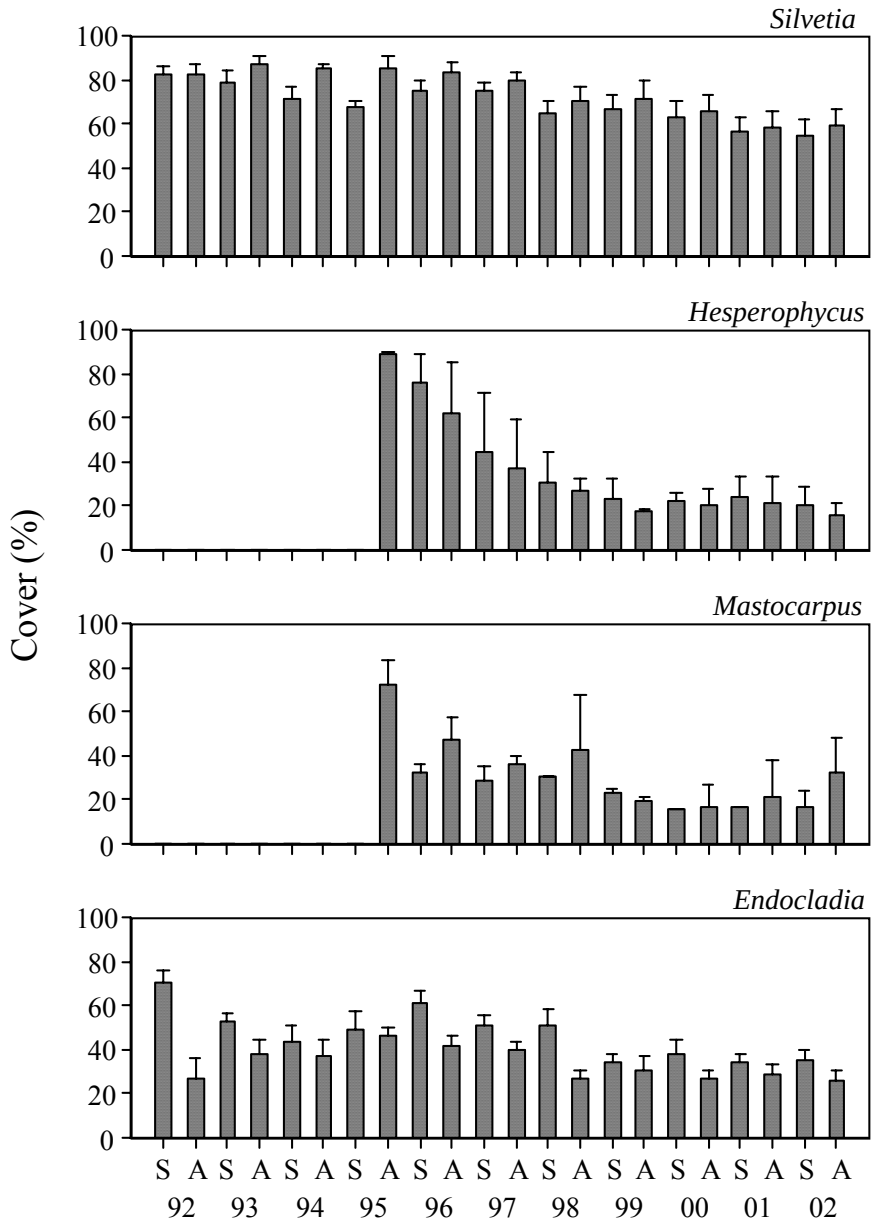


Figure II-14c: Mean (±se) percent cover of various species in spring (S) and autumn (A) from 1992 to 2002 for all sites (sites are replicates). Blank spaces indicate that sampling was not done at that time.

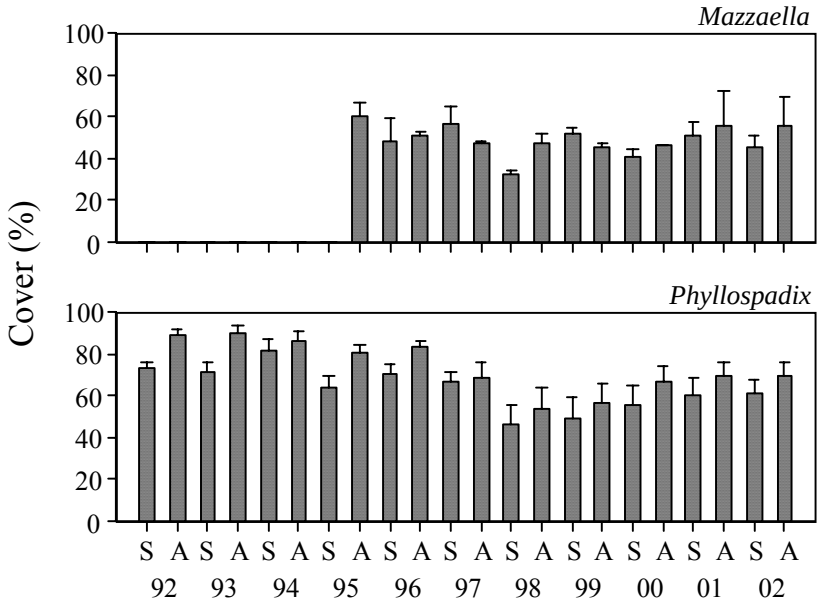
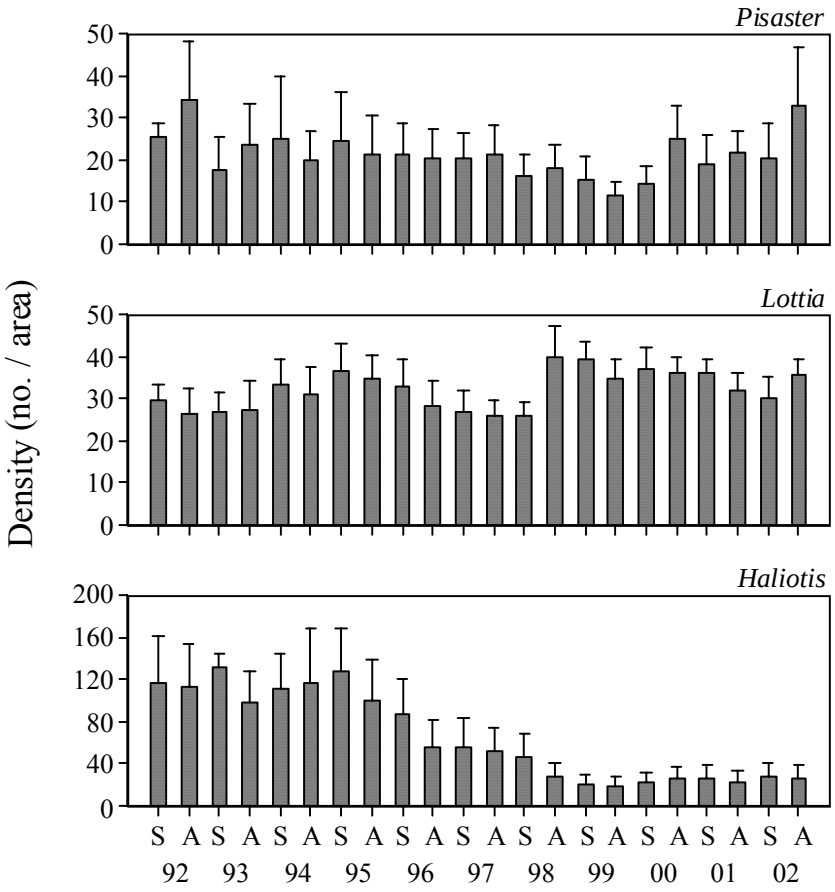


Figure II-14d: Mean (±se) density of various species in spring (S) and autumn (A) from 1992 to 2002 for all sites (sites are replicates).



2.4 Discussion

The Shoreline Inventory monitoring program has produced what is, in terms of its spatial and temporal extent, one of the most comprehensive sets of data ever collected for species living in rocky intertidal communities. A primary objective of this long-term ecological monitoring program, and of this report, is to evaluate whether the amassed data has the ability to detect changes and show trends in the abundance of target species. To detect changes and show trends in the context of this monitoring program means that the current sampling design can provide an accurate picture of the natural variation in abundance of the target species. For the target species in this region, there are three known sources of natural of variation that can produce significant trends: (1) seasonal variation due to such factors as the growth of organisms, recruitment events, and winter storms, (2) variation due to episodic events such as years of great recruitment or mass mortalities, and (3) variation due to El Niño events (and longer-term phenomena at the scale of decades might also prove important). Only by establishing "normal" limits of variation can changes due to "abnormal conditions" be detected. Abnormal conditions might include oils spills (see Raimondi et al. 1999), disease such as *Haliotis* withering syndrome (see Raimondi et al. 2002), illegal harvesting, or longer-term changes due to unknown sources of variation such as rises in sea surface temperatures due perhaps to global warming. Finally, data for the monitoring program must allow the success of any management actions to be assessed.

Visual inspection of graphs and statistical analyses of the data reveal that the current sampling design has a great ability to show the natural limits of variation and to detect trends in the abundance of target organisms. Three types of temporal trends were clear: (1) short-term seasonal trends, (2) intermediate-term fluctuations on time scales ranging from one to several years, and (3) long-term changes (particularly gradual decreases, but also increases) for five or more years due to changes in abundance over the entire sampling period for a particular species at a particular site. Interestingly, many of the spatial and temporal patterns of abundance and data and statistical analyses based on these trends were similar to those reported by Raimondi et al. (1999), even though the current data set included 4 more years of information (8 sample periods). Raimondi et al. (1999) have thoroughly discussed potential factors responsible for changes to the abundance of target species and community structure resulting from these changes, so these will not be repeated here (see also Ambrose et al. 1995). Only new and significant changes for sites and times not examined by Raimondi et al. (1999) are discussed.

Many species exhibited short-term seasonal trends, but these were not consistent among sites or biogeographic regions north and south of Point Conception. Short-term seasonal trends were apparent at a minority of sites for the acorn barnacles, the gooseneck barnacle *Pollicipes* and *Pisaster*, with abundances greater in autumn than in spring. The macroalgae exhibited more pronounced seasonal trends than the macroinvertebrates. At most sites, the cover of *Silvetia*, *Mastocarpus* and *Phyllospadix* was greater in autumn than in spring, whereas the *Endocladia* was more abundant in spring than in autumn. Identical strong seasonal trends have been documented for *Endocladia* and *Silvetia* at several rocky intertidal sites on the Channel Islands (Minchinton and Raimondi 2001). The magnitude of these changes was relatively small with on average about 5 to 20% changes in abundance among seasons. Seasonal changes for species such as *Pollicipes*, *Silvetia* and *Mastocarpus*, where abundances are greater in autumn than winter, have been attributed to summer growth for *Silvetia* and winter storms that removed

barnacles and algae from the rock surface (Raimondi et al. 1999). Similarly, the cover of surfgrass *Phyllospadix* is reduced during winter by storms, which can move sand causing periodic burial and scouring of blades (Raimondi et al. 1999).

A second striking and common trend in the data were intermediate-term fluctuations in abundance where there were relatively rapid declines sometimes followed by more gradual increases. These relatively rapid and common declines occurred from 1997 to 1998, which corresponds to the 1997-98 El Niño, which was the strongest of the century. Along the California coast, these events are associated with increased sea surface temperature, decreased nutrients, and increased frequency and intensity of storms, which can generate large amplitude swells, increased rainfall which leads to increased sediment load and reduce salinities in nearshore waters, and more floating debris (Raimondi et al. 1999). Raimondi et al. (1999) provided a detailed quantitative assessment of the effects of the 1997/98 El Niño event on rocky intertidal communities in southern and central California. They concluded that the abundances of barnacles, mussels, *Endocladia*, *Silvetia*, *Phyllospadix*, but not *Lottia* declined during the 1997/98 El Niño event, and these conclusions are confirmed by the subsequent sampling presented in this report. In addition, similar but less spatially consistent declines occur at some sites for *Anthopleura* and *Haliotis* during this event. Indeed, a detailed analysis has shown that although declines of *Haliotis* due to withering syndrome over the past decade are not exclusively linked to times of El Niño, they may occur faster during times of El Niño events (see Raimondi et al. 2002). Interestingly, at several sites there appeared to be an increase in the density of *Lottia* (e.g., Cayucos Point, Stairs, Alegria) during this event. This increase might represent recruitment, but detailed analysis of the size-structure of the population is required to be certain and this should be done.

Storm-related effects of the strong 1997-98 El Niño and other El Niño events during the 1980s and 1990s have also been shown to significantly reduce the abundance of these same target species inhabiting rocky shore communities along the coasts of the Channel Islands (Minchinton and Raimondi 2001). Minchinton and Raimondi (2001) found that the abundances of barnacles, mussels, rockweeds including primarily *Silvetia*, and *Endocladia* experienced declines at some sites during some of these periods of El Niño. Studies of rocky intertidal communities in California following the 1982-83 El Niño (Gunnill 1985, Murray and Horn 1989) also detected changes in species abundances, but these were not as pronounced as those reported by Raimondi et al. (1999) or Minchinton and Raimondi (2001).

Interestingly, there was considerable variation in declines due to El Niño among species and sites (see also Raimondi et al. 1999, Minchinton and Raimondi 2001). At some sites, a particular target species declined during El Niño (e.g., barnacles), whereas at other sites its abundance remained constant. At some sites there were declines for some target species but not others. Finally, at some sites there were striking declines for all target species (e.g., Arroyo Hondo). Minchinton and Raimondi (2001), studying rocky shore communities in the Channel Islands, suggested that sites closer together and with the same aspect appeared to behave more similarly in their patterns of decline during El Niño, but they also pointed out a general lack of spatial consistency in the declines among sites. No common spatial trends, such as differences between sites north and south of Point Conception, in the declines during El Niño were observed here or by Raimondi et al. (1999), but this may require closer scrutiny linking environmental conditions

such as wave amplitude and sea surface temperatures to declines at individual sites. These results again emphasize the importance of studying several target species at multiple sites. The ability of the monitoring program to detect changes due to El Niño is a significant result. Clearly, the sampling program is capable of detecting relatively small temporal changes in the abundance of target organisms.

The third obvious and consistent trend was the long-term declines in abundance over the entire monitoring period for many target species at many sites. Of the 93 separate analyses for each species and site combination, abundances declined 56 times, increased on only 6 occasions, and for 31 analyses there was no detectable linear trend. The overwhelming number of decreases compared to increases may not be surprising because plots were initially chosen to contain substantial abundances of the target species and these plots were repeatedly sampled over time. For relatively long-lived species (relative to the current duration of the monitoring program), such as mussels and *Silvetia*, these declines may at least partially be an artifact of initial conditions as abundances are bounded at 100% cover and therefore there is a greater probability of decline than increase. For other shorter-lived organisms, such as barnacles and *Endocladia* that would have likely turned over during the monitoring program, the long-term declines in abundance may be a "real" phenomenon. Indeed, with the exception of *Haliotis*, 5 of the 6 increases were for the mobile species *Lottia* and *Pisaster*. Frequent declines might also be expected, however, because the largest El Niño event of the last century occurred in the middle of the sampling period and many target species appeared to be negatively affected by conditions during this period of anomalous environmental conditions and have not recovered.

Nevertheless, particular species exhibited strong patterns of decline throughout the sampling period and these should be given particular consideration. Populations of the black abalone *Haliotis* were crashing during this time, a phenomenon also well documented at sites on the Channel Islands (see Raimondi et al. 2002 and references therein). The demise of the population appears to be due a fatal disease called withering syndrome caused by the infection of *Haliotis* by a bacterium. The observed spatial pattern of decline, with the abundance of *Haliotis* decreasing from south to north is clearly detected by the sampling program and might suggest potential patterns of transmission and infection for the causative agent. As noted by Raimondi et al. (1999), there has been a consistent and continued decrease in mussel cover at the majority of sites, and the same could be said for several species of macroalgae. Raimondi et al. (1999) suggested that this might be due to removal of mussels by storms followed by a lack of survivorship of new recruits. Although cover remains high at some sites, these declines should be monitored for mussels as well as other species that appear to be decreasing for no apparent reason. For many long-lived species, the monitoring program is probably not long enough to make sound judgements about the factors responsible for declines, but certainly a lack of recruitment or mortality of recruits should be noted. Indeed, it is advised that the Shoreline Inventory be expanded to sample recruitment of target species where this is not being done already through size measurements. Impacts may indeed affect the larval and juvenile stages and not the adults of target species.

Detection of these temporal trends in abundance (seasonal, El Niño, disease) highlights the efficacy and utility of the current Shoreline Inventory monitoring program. Expansion of the program to a broader range of sites so that larger-scale regional patterns of abundance of target

species (and species diversity) is advisable and would wisely recognizes the need for long-term ecological monitoring that is currently unsupported by mainstream funding agencies.

Part III: STATISTICAL POWER ANALYSIS

3.1 Introduction, Objectives, and Approach

One of the primary objectives of the long-term monitoring program is to compile a set of data that can be used to estimate the consequences of impacts (e.g., oil spill) to the target organisms living in the rocky intertidal community. It is often assumed that the pre-impact program will collect data that have the ability to detect changes in abundance of the target organisms should an impact occur. Without assessment of this assumption, however, there is no way of knowing whether the monitoring program has been adequately designed to meet its expressed purpose. In this part of the report we test this critical assumption by using power analysis to determine the ability of the data collected from the long-term monitoring program to detect impacts to target organisms. The purpose of this analysis is to determine whether modifications to the sampling design of the monitoring program are necessary so that impacts may be adequately detected.

The ability of the monitoring program to estimate the effects of impacts will depend on the statistical power to determine changes in the abundance of the target organisms. Statistical power is the ability of the sampling design to detect an impact when one has truly occurred. For example, if a monitoring program has statistical power of 80% at α = 0.05 to detect a 50% change in abundance of one of the target organisms, this means that if an impact actually did occur that produced a 50% change in abundance, then this monitoring program has an 80% probability of correctly identifying that change. Within the scope of the monitoring program, there are several common sampling designs that can be used to detect impacts to target organisms (see Schmitt and Osenberg 1996). Two of the most common and widely used are the Before-After design and the Before-After / Control-Impact (or BACI) design. In a Before-After design, impacts are detected by statistically comparing differences in the abundance of organisms at a single site before and after the impact. In a BACI design, impacts are detected by statistically comparing differences in the abundance of organisms between control and impact sites before the impact to differences between these sites after the impact (Stewart-Oaten et al.1986). Here we provide, using only the before-after design, an assessment of the statistical power of tests to detect differences in the abundance of target species from data collected under the Shoreline Inventory.

The questions we address are: (1) what is the statistical power to detect changes to the abundances of target species following an impact at one of the monitored sites, and (2) based on the results of power analyses, should there be modifications to the sampling design of the monitoring program. To do this, we first have to simulate future impacts to target species. Next, we determine the statistical power of tests to detect differences in abundance due to the impact. In particular, we ask two questions: (1) with what level of power can we detect a given level of change in abundance (10% to 50% change) at alpha levels of 0.05, 0.10, and 0.20, and (2) what level of change in abundance can we detect with 80% power at alpha levels of 0.05, 0.10 and 0.20. We also consider how the results vary among the target species and spatially by considering whether impacts are easier to

detect for some species and sites. The answers to these questions will help guide modifications to the current sampling design of the monitoring program for each target species. Two particular modifications to the design (and thus sampling effort) to be considered are: (1) does the temporal sampling frequency (i.e., number of times sampled) provide adequate statistical power to detect impacts, and (2) does the spatial extent of the sampling program (i.e., number of sites sampled) provide adequate statistical power to detect impacts.

3.2 Methods

3.2.1 Before-After Design: Simulating Impacts and Calculating Power

In a before-after design, impacts are detected statistically by comparing mean abundances at a single site before and after an impact, typically with a *t*-test. Note that replication within a site and its associated variance is not used, as they are only relevant for ensuring that the mean abundance at a site is accurately represented. Because we are simulating impacts at these sites, all data fall in the pre-impact or before period. Therefore, to determine the ability of the monitoring program to detect differences in the abundance of target species in the event of a future impact, we simulated impacts, which were done as follows. First, the mean and standard deviation of the mean abundances at each sampling date in the before period were calculated (using untransformed data). This mean and standard deviation represented the mean and standard deviation for the before period. To enable comparisons of results of the power analyses among species and sites, only data from autumn 1995 to autumn 2002 were used. This was done because sites were established at different times from 1992 to 1995 and, therefore, some had been sampled more often than others. During this period of 7.5 years there was a continuous record of sampling over 14 or 15 sample periods for all species and site combinations.

Impacts were simulated by decreasing the mean abundance calculated for the before period by increments of 10%, yielding values from 0 to 90% of the before period mean. This gave ten means for the after period representing impacts ranging from 10 to 100% of the mean in the before period. The standard deviation and sample size (n = 14 or 15) associated with these means was assumed to be the same as those for the mean in the before period. Using the differences between the means in the before and after period, the standard deviation, and the sample size, we calculated the power to detect significant impacts by using a two-sided, one sample *t*-test. The null hypothesis tested was that there is no impact. Power was calculated for each species at each site and for all species and sites combined with alpha levels of 0.05, 0.10, and 0.20 using the power analysis software program PASS.

These analyses generated a power curve for each of the three levels of alpha, with the percent change in abundance on the *x*-axis (ranging from 0 to 100% of the mean abundance at the impacted site) and power on the *y*-axis. We extracted particular information from these curves, which are not shown, and present this summarized information. First, we determined how many of the before-after comparisons could detect 10, 20, 30, 40 and 50% changes (which was an appropriate range given the results)

with 80% power. Second, for each before-after comparison we determined the mean level of power to detect 10, 20, 30, 40 and 50% changes in abundance. Third, for each before-after comparison we determined the mean level of change in abundance that could be detected with 80% power.

3.2.2 Before-After Design: Tests of Assumptions

Before we can proceed with power analyses using *t*-tests, we have to determine whether several important assumptions are satisfied by the data in the before period. Two important assumptions are lack of trends and independence of random errors. Mean abundances at the site can trend positively or negatively over time. If this were to continue after the impact, then detection of an effect of the impact would be confounded. We tested for trends in mean abundance by linear regression of the mean abundances against sample period. Serial correlation (i.e., non-independence) of errors occurs when the differences in mean abundances vary as a function of time between the samples. Serial correlation of errors can lead to invalid estimates of the error variance and thus affect the outcome of statistical tests. We tested for serial correlation using Durbin-Watson tests for mean abundances against sample period ($\alpha = 0.05$). Inconclusive results were considered as failures. Significant trend or serial correlation ($\alpha = 0.05$) excluded this before-after comparison from power analysis.

Two aspects of the data compromise these tests and must be noted. First, for some species and site combinations, there is a missing sample period and this leads to irregular spacing among samples. How this affects the outcomes of these tests is not known. Omitting data so that samples were regularly spaced would result in a severe loss of data, so this was not done. Second, there are clearly large, long-lasting trends at some sites, likely due to El Niño events or other factors. The presence of these trends would likely result in the detection of more cases of non-independence than would otherwise be the case without these trends. Introducing a covariate to attempt to account for these trends was beyond the scope of this report (Stewart-Oaten et al. 1986).

A final assumption that errors are normally distributed was not tested because the *t*-test is robust to all but the most extreme violations of this assumption (Stewart-Oaten et al. 1992). For more detailed explanation of these assumptions, see Stewart-Oaten et al. (1986, 1992).

3.3 Results

For all species only 29% (27 of 93) of the before-after comparisons passed the tests for both assumptions (Table III-1). For 62% (52 of 93) of comparisons there were significant trends in the data, and the assumption of independence was violated for 77% (47 of 93) of comparisons (Table III-1). Consequently, for all species, power analyses could be done only for 27 before-after comparisons.

Table III-1a. Results of statistical tests of assumptions before power analysis (see text for details). Data used were mean abundances at each site over time from autumn 1995 to autumn 2002. NS = not significant. * = significant. INCON = inconclusive results. Results in bold passed both tests of assumptions and the target species at that site was used in power analyses.

Species	Site	Trend	Independence
Anthopleura	Boathouse	NS	NS
Anthopleura	Alegria	NS	*
Anthopleura	Coal Oil Point	NS	INCON
Anthopleura	Carpinteria	NS	NS
Anthopleura	Mussel Shoals	NS	NS
Anthopleura	Old Stairs	NS	NS
Barnacles	Point Sierra Nevada	*	NS
Barnacles	Cayucos Point	*	*
Barnacles	Hazards	*	INCON
Barnacles	Shell Beach	NS	*
Barnacles	Occulto	NS	NS
Barnacles	Stairs	*	*
Barnacles	Boathouse	*	*
Barnacles	Government Point	NS	*
Barnacles	Alegria	NS	NS
Barnacles	Arroyo Hondo	NS	*
Barnacles	Carpinteria	NS	*
Barnacles	Mussel Shoals	NS	NS
Barnacles	Old Stairs	NS	NS
Barnacles	Paradise Cove	NS	NS
Barnacles	White's Point	NS	NS
Pollicipes	Government Point	NS	NS
Pollicipes	Alegria	NS	NS
Pollicipes	Carpinteria	*	NS
Mussels	Point Sierra Nevada	NS	NS
Mussels	Cayucos Point	*	*
Mussels	Hazards	*	INCON
Mussels	Shell Beach	*	*
Mussels	Occulto	NS	NS
Mussels	Stairs	*	*
Mussels	Boathouse	*	*
Mussels	Government Point	NS	*
Mussels	Alegria	NS	*
Mussels	Arroyo Hondo	NS	*
Mussels	Mussel Shoals	*	*
Mussels	Old Stairs	*	NS
Mussels	Paradise Cove	NS	*
Mussels	White's Point	NS	*

Table III-1b. Results of statistical tests of assumptions before power analysis (see text for details). Data used were mean abundances at each site over time from autumn 1995 to autumn 2002. NS = not significant. * = significant. INCON = inconclusive results. Results in bold passed both tests of assumptions and the target species at that site was used in power analyses.

Species	Site	Trend	Independence
Silvetia	Point Sierra Nevada	*	NS
Silvetia	Cayucos Point	*	NS
Silvetia	Hazards	*	NS
Silvetia	Shell Beach	*	NS
Silvetia	Stairs	*	NS
Silvetia	Boathouse	*	NS
Silvetia	Government Point	*	*
Hesperophycus	Point Sierra Nevada	*	*
Hesperophycus	Cayucos Point	NS	*
Mastocarpus	Point Sierra Nevada	NS	NS
Mastocarpus	Shell Beach	*	NS
Endocladia	Cayucos Point	*	NS
Endocladia	Hazards	*	NS
Endocladia	Shell Beach	*	NS
Endocladia	Occulto	*	*
Endocladia	Stairs	NS	*
Endocladia	Boathouse	*	NS
Endocladia	Government Point	NS	NS
Endocladia	Old Stairs	*	NS
Endocladia	Paradise Cove	NS	NS
Endocladia	White's Point	*	NS
Mazzaella	Point Sierra Nevada	NS	INCON
Mazzaella	Hazards	NS	NS
Phyllospadix	Point Sierra Nevada	NS	*
Phyllospadix	Cayucos Point	NS	NS
Phyllospadix	Stairs	NS	*
Phyllospadix	Government Point	NS	NS
Phyllospadix	Arroyo Hondo	NS	NS
Phyllospadix	Coal Oil Point	NS	*
Phyllospadix	Carpentaria	NS	*
Phyllospadix	Mussel Shoals	*	INCON
Phyllospadix	Paradise Cove	NS	*

Table III-1c. Results of statistical tests of assumptions before power analysis (see text for details). Data used were mean abundances at each site over time from autumn 1995 to autumn 2002. NS = not significant. * = significant. INCON = inconclusive results. Results in bold passed both tests of assumptions and the target species at that site was used in power analyses.

Species	Site	Trend	Independence
Pisaster	Point Sierra Nevada	*	NS
Pisaster	Cayucos Point	NS	NS
Pisaster	Hazards	NS	NS
Pisaster	Shell Beach	NS	NS
Pisaster	Stairs	NS	INCON
Pisaster	Boathouse	NS	NS
Pisaster	Government Point	NS	*
Pisaster	Arroyo Hondo	NS	INCON
Pisaster	Mussel Shoals	*	INCON
Pisaster	Old Stairs	NS	*
Lottia	Cayucos Point	*	NS
Lottia	Hazards	NS	NS
Lottia	Stairs	NS	INCON
Lottia	Boathouse	*	INCON
Lottia	Government Point	NS	NS
Lottia	Alegria	*	INCON
Haliotis	Point Sierra Nevada	*	NS
Haliotis	Piedras Blancas	NS	*
Haliotis	Cayucos Point	*	*
Haliotis	Purisima Point	*	*
Haliotis	Stairs	*	*
Haliotis	Boathouse	*	*
Haliotis	Government Point	*	NS

Some species passed the tests of assumptions more frequently than others. The sea anemone *Anthopleura* and the barnacle *Pollicipes* passed both assumptions for more than 50% of comparisons. Barnacles, the red alga *Mastorcarpus*, the alga *Mazzaella*, surfgrass *Phyllospadix* and the sea star *Pisaster* successfully passed both assumptions for between 25 and 50% of comparisons. Mussels and the turfweed *Endocladia* passed both tests for less than 25% of comparisons. Data for comparisons with the brown fucoid algae or rockweeds *Silvetia* and *Hesperophycus* and the black abalone *Haliotis* never passed the tests for both assumptions. For the 5 species that had been monitored at 9 or more sites, both tests of assumptions were passed most frequently for barnacles (6 of 15 comparisons) and *Pisaster* (4 of 10 comparisons) and least frequently for mussels (2 of 14 comparisons).

The 27 before-after comparisons that passed both tests were spread out roughly evenly across sites (Table III-1). Four sites were not involved in any of these comparisons: Piedras Blancas and Purisima Point, where only the black abalone *Haliotis* had been monitored, and Stairs and Coal Oil Point (Table III-1). For the 10 sites where at least 5 target species had been monitored, both tests of assumptions were passed most frequently

at Government Point (4 of 9 comparisons) and least frequently at Stairs (0 of 8 comparisons.)

3.3.1 *Anthopleura*

Under the before-after design, 67% (4 of 6) of comparisons passed the tests of both assumptions (Table III-1a). For all 4 sites a 20% change in the cover of *Anthopleura* was detectable with 80% power at all levels of alpha (Figure III-1a). Indeed, on average for all sites, the power to detect a 20% change was almost 100%, and 10% changes in abundance could be detected at 66, 75 and 83% power for alpha levels of 0.05, 0.10 and 0.20 respectively (Figure III-2a). On average for all 4 sites, a 12% change (ranging from 6 to 18%) in cover was detectable with 80% power at $\alpha = 0.05$ (Figure III-3a).

3.3.2 Barnacles

Under the before-after design, 40% (6 of 15) of comparisons passed the tests of both assumptions (Table III-1a). For all 6 sites a 40% change in the percent cover of barnacles was detectable with 80% power at $\alpha = 0.05$ (Figure III-1a). At $\alpha = 0.10$ and 0.20, at least a 30% change could be detected at this level of power for all comparisons. At one site, a 10% change could be detected with 80% power at $\alpha = 0.05$. On average for all 6 sites, analyses at $\alpha = 0.05$ showed that 10, 20 and 30% changes in the cover of barnacles could be detected with 41, 75 and 94% power, respectively (Figure III-2a). On average for all 6 sites, a 20% change (ranging from 7 to 30%) in barnacle cover is detectable with 80% at $\alpha = 0.05$ and detectable change reduces to18 and 15% at $\alpha = 0.10$ and 0.20, respectively (Figure III-3a).

3.3.3 *Pollicipes polymerus*

Power analyses for *Pollicipes* could be done for 67% of sites because 2 of 3 comparisons passed the tests of both assumptions (Table III-1a). For both sites at least a 20% change in cover was detectable with 80% power at $\alpha = 0.05$, and at one site a 10% change in cover is detectable (Figure III-1a). On average for both sites there was 100% and 78% power to detect 20 and 10% changes in cover at $\alpha = 0.05$, respectively (Figure III-2a). These sites varied considerably, however, in their ability to detect small changes in cover, with power of 55% at one site and 100% at the other site ($\alpha = 0.05$). On average for both sites, a 9% change (ranging from 4 to 14%) in cover was detectable with 80% power at $\alpha = 0.05$ (Figure III-3a).

3.3.4 Mussels

Only 2 of 14 comparisons passed the tests for both assumptions and, consequently, power analyses for mussels could only be done for two sites (Table III-1a). For both sites, there was 100% power to detect a 10% change in the mussel cover at all levels of alpha (Figure III-1a, III-2a). Indeed, at both sites a 3% change in cover was detectable at 80% power at $\alpha = 0.05$ (Figure III-3a).

Figure III-1a: Percentage of before-after comparisons that could detect a 10, 20, 30, 40, or 50% change in cover with 80% power at α = 0.05, 0.10, and 0.20 for each of four target species. Numbers in parentheses indicate the number of replicate sites sampled for that species.

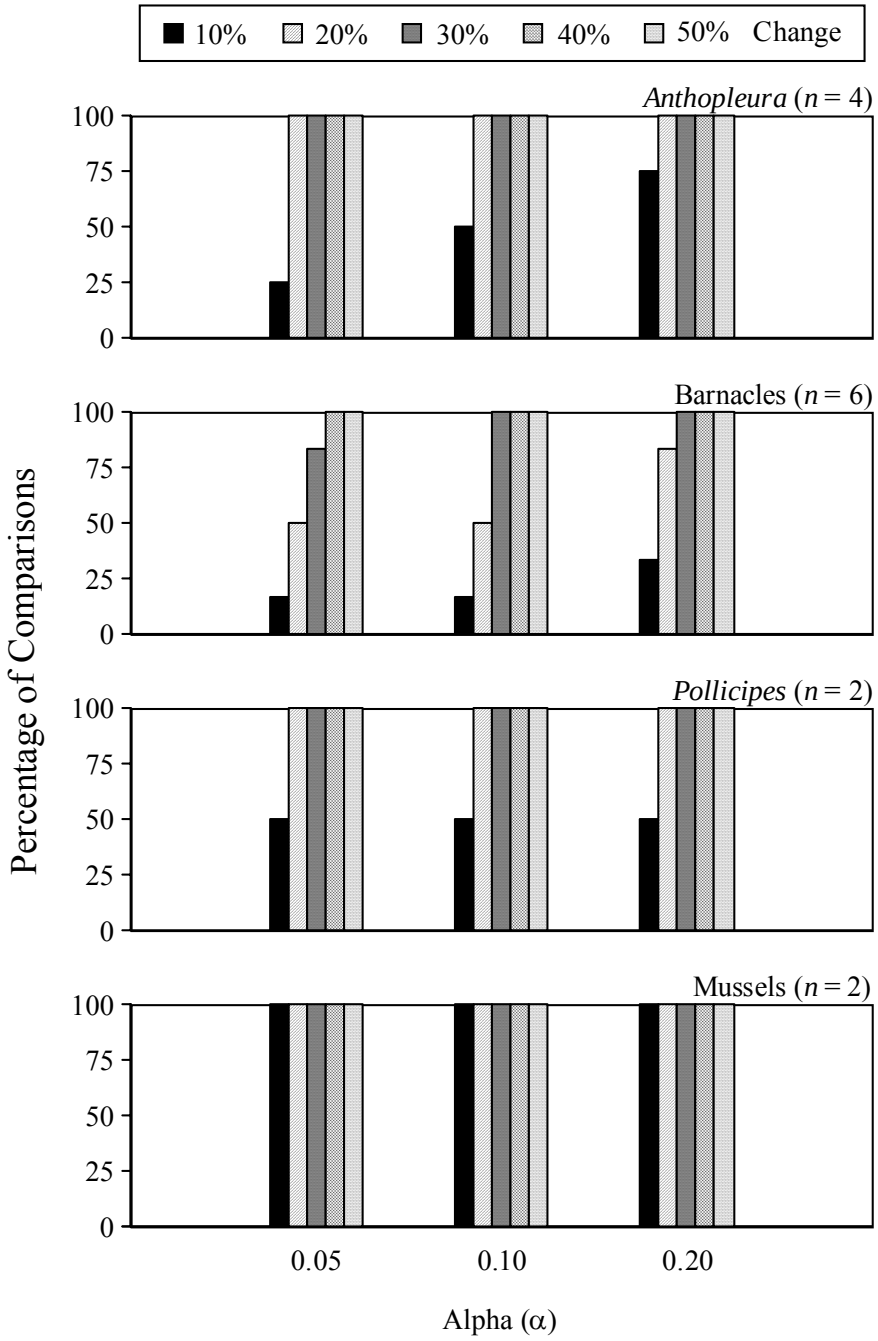


Figure III-2a: Mean (±se) power to detect 10, 20, 30, 40, and 50% changes in cover at α = 0.05, 0.10, and 0.20 for before-after comparisons for each of four target organisms. Numbers in parentheses indicate the number of replicate sites sampled for that organism.

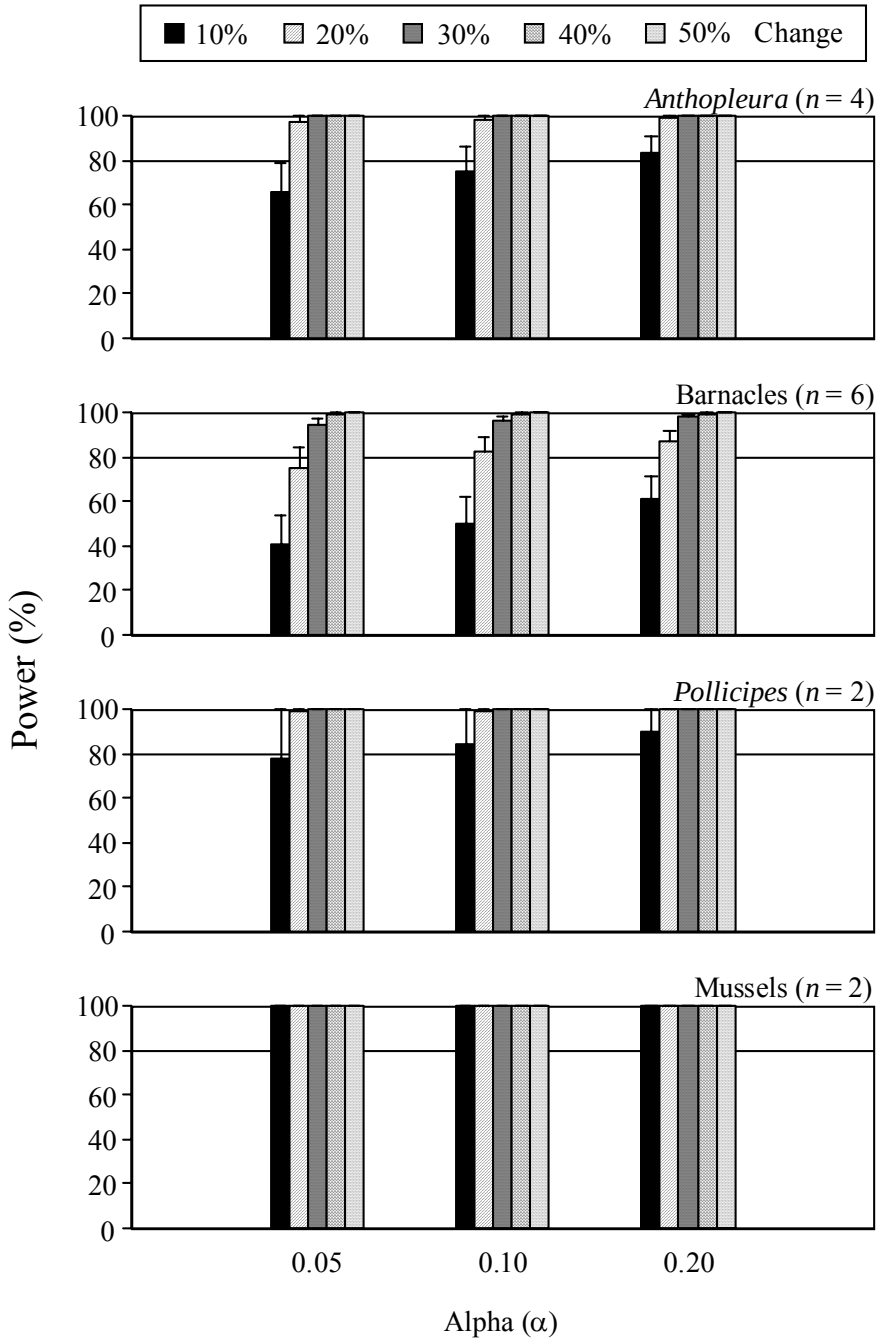
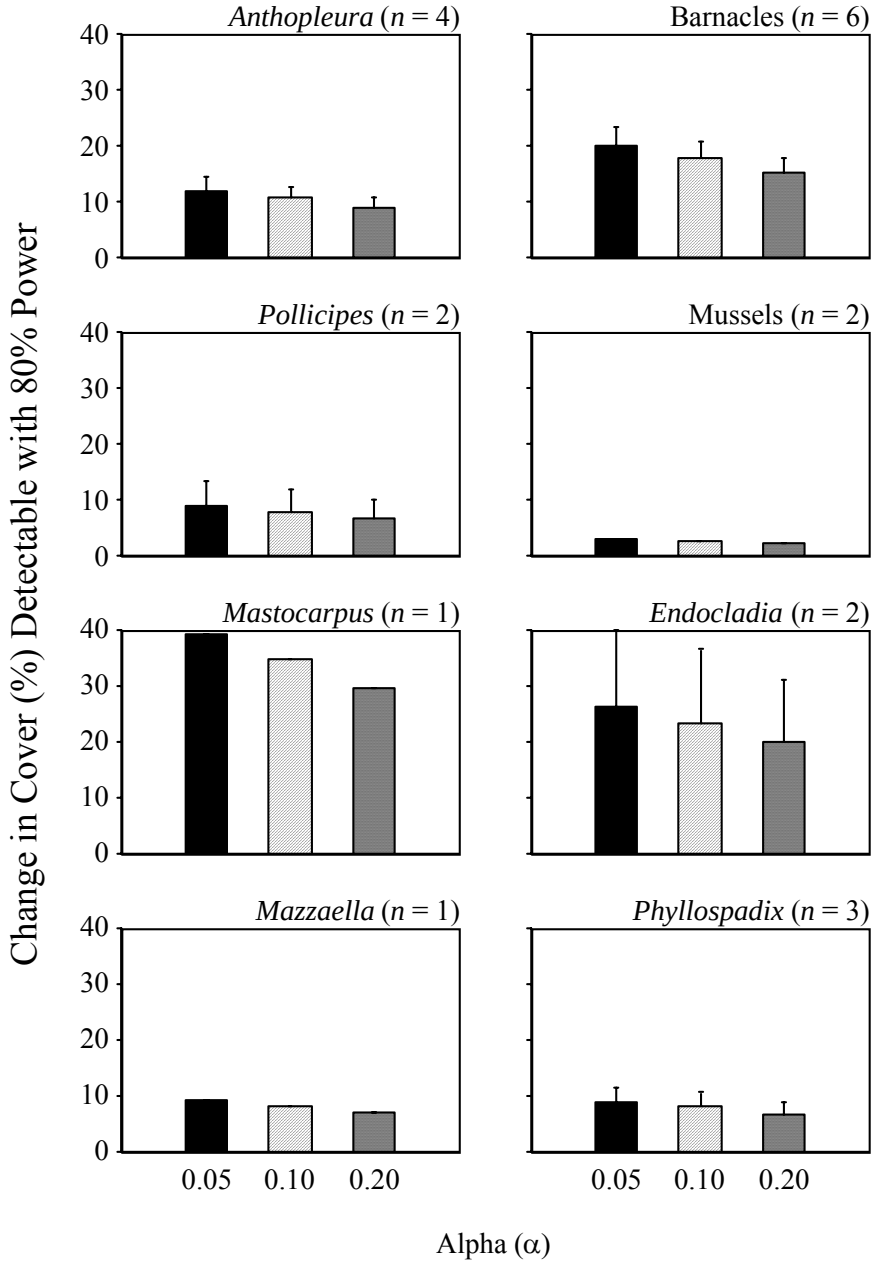


Figure III-3a: Mean (±se) percent change in cover detectable with 80% power at α = 0.05, 0.10, and 0.20 for before-after comparisons for each of eight target species. Numbers in parentheses indicate the number of replicate sites for each target organism.



3.3.5 *Silvetia compressa*

Data for before-after comparisons with *Silvetia* never passed the tests of both assumptions (Table III-1b). Consequently, power analysis was not done for this species.

3.3.6 *Hesperophycus californicus*

Data for before-after comparisons with *Hesperophycus* never passed the tests of both assumptions (Table III-1b). Consequently, power analysis was not done for this species.

3.3.7 *Mastocarpus papillatus*

Power analysis for *Mastocarpus* was only done for 1 of 2 sites, which passed both tests of assumptions (Table III-1b). At this site, a 40% in percent cover was detectable with 82% power at $\alpha = 0.05$ (Figure III-1b, III-2b). Power to detect 10, 20, and 30% changes in cover at $\alpha = 0.05$ was 11, 30 and 58%, respectively (Figure III-2b). At 80% power and $\alpha = 0.05$, analysis could detect a 39% change in the cover of *Mastocarpus* (Figure III-3a).

3.3.8 *Endocladia muricata*

Only 20% (2 of 10) of comparisons under the before-after design passed the tests of both assumptions and, therefore, power analyses were restricted to these two sites (Table III-1b). The two sites differed dramatically in their ability to detect changes in the cover of *Endocladia*. At one site, there was 100% power to detect a 20% change in cover at $\alpha = 0.05$, whereas at the other power to detect a similar change was only 28% (Figure III-1b, III-2b). At 80% power and $\alpha = 0.05$, one site could detect a 12% change in cover whereas the other could only detect a 42% change in cover (Figure III-3a).

3.3.9 *Mazzaella* spp.

Only 1 of 2 sites passed tests of both assumptions (Table III-1b). At this site, there was 86% power to detect a 10% change in cover at $\alpha = 0.05$ (Figure III-1b, III-2b), and 9% cover was detectable with 80% power at $\alpha = 0.05$ (Figure III-3b).

3.3.10 *Phyllospadix scouleri/torreyi*

Under the before-after design, 33% (3 of 9) of comparisons passed both tests of assumptions and, therefore, power analyses were carried out at these three sites (Table III-1b). At 2 of 3 sites, at least a 10% change in cover of the surfgrass *Phyllospadix* could be detected with 80% power at $\alpha = 0.05$ (Figure III-1b). On average for all 3 sites, there was 82% power to detect a 10% change at $\alpha = 0.05$, and almost 100% power to detect changes greater than 20% (Figure III-2b). Indeed, on average for all sites, a 9% change (ranging from 5 to 14%) in cover was detectable with 80% power at $\alpha = 0.05$ (Figure III-3b).

Figure III-1b: Percentage of before-after comparisons that could detect a 10, 20, 30, 40, or 50% change in cover with 80% power at α = 0.05, 0.10, and 0.20 for each of four target species. Numbers in parentheses indicate the number of replicate sites sampled for that species.

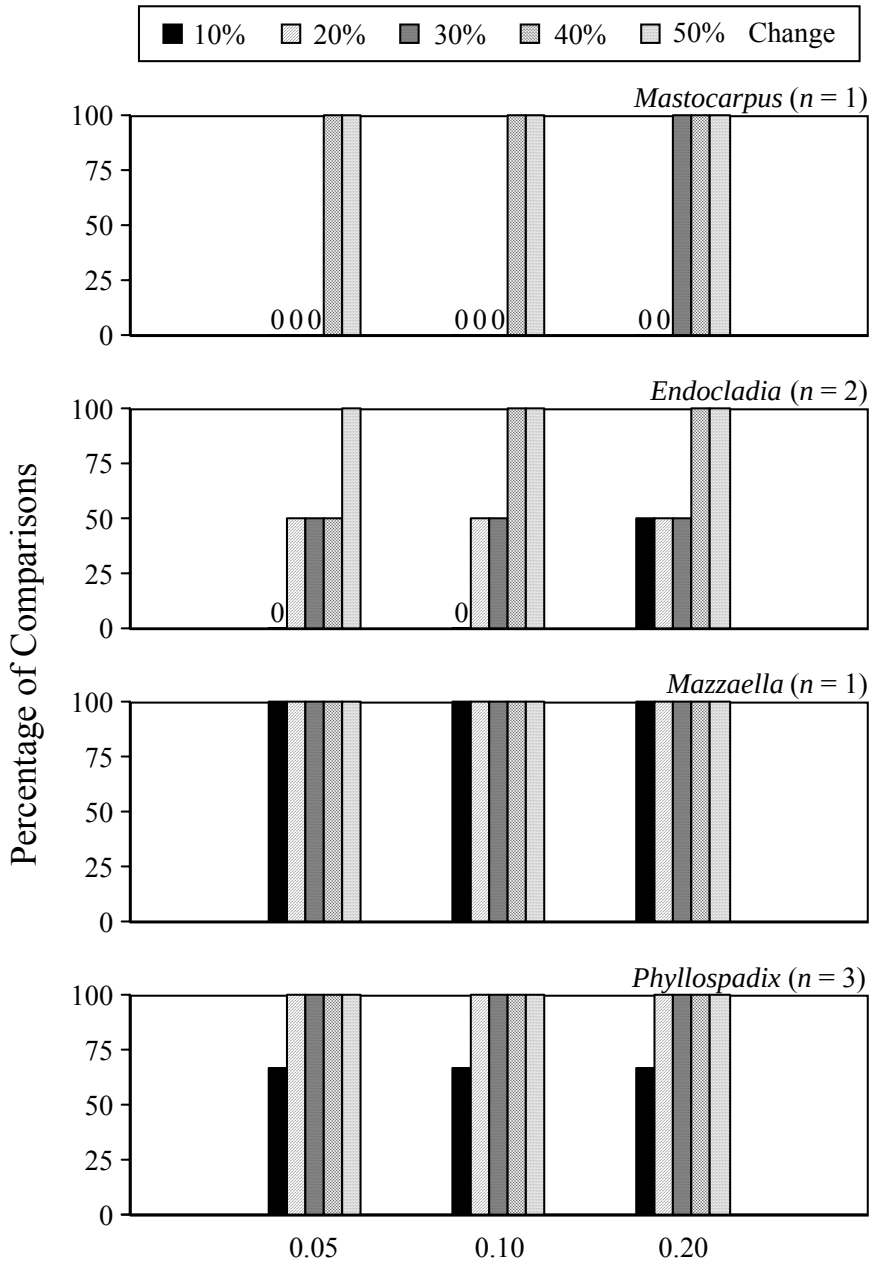


Figure III-2b: Mean (±se) power to detect 10, 20, 30, 40, and 50% changes in cover at α = 0.05, 0.10, and 0.20 for before-after comparisons for each of four target organisms. Numbers in parentheses indicate the number of replicate sites sampled for that organism.

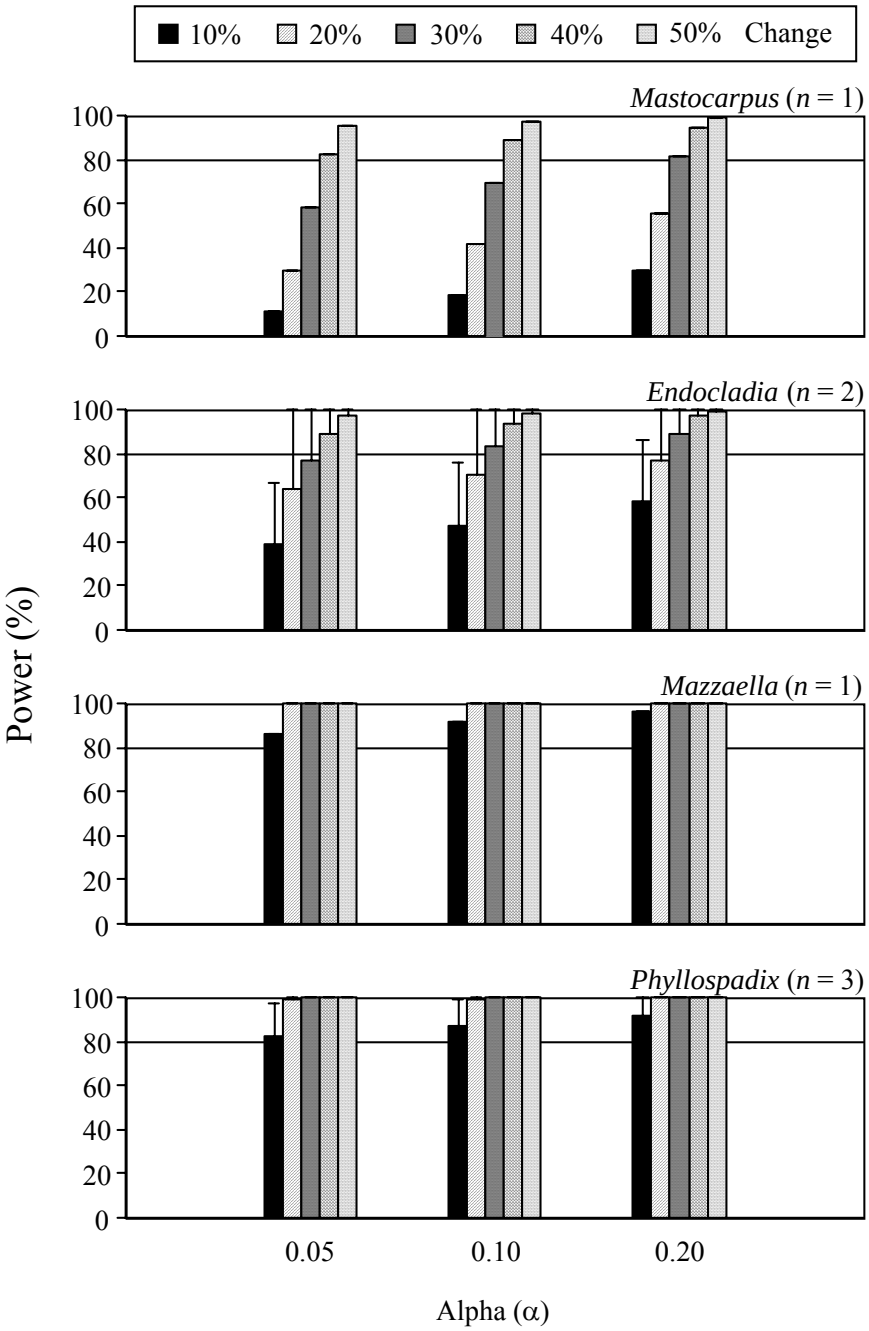
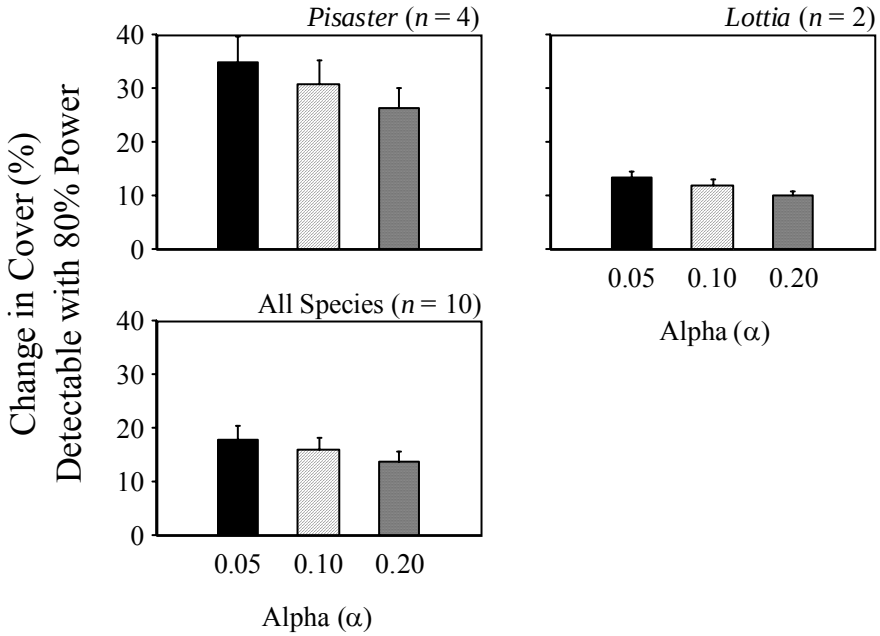


Figure III-3b: Mean (±se) percent change in abundance detectable with 80% power at α = 0.05, 0.10, and 0.20 for each of two and all target species. Numbers in parentheses indicate the number of replicate sites or species sampled.



3.3.11 *Pisaster ochraceus*

Data at 4 of 10 sites passed both tests of assumptions and were therefore used in power analyses (Table III-1c). At all sites, a 50% change in the density of *Pisaster* could be detected with 80% power at α = 0.05 (Figure III-1c). At two of these sites, a 40% change was detectable, but at none of the 4 sites was a 20% change detectable. On average for all sites, there was 87% power to detect a 40% change in abundance at α = 0.05 (Figure III-2c), but power dropped to 69, 41 and 15% to detect changes of 30, 20 and 10%, respectively. On average for all sites, a 35% change (ranging from 23 to 43%) in abundance was detectable with 80% power at α = 0.05 (Figure III-2c).

3.3.12 *Lottia gigantea*

Data at 2 of 6 sites passed tests of both assumptions and were used in power analyses (Table III-1c). For both sites, at least a 20% change in the density of *Lottia* was detectable with 80% at α = 0.05, whereas neither site could detect a 10% change in abundance (Figure III-1c). Indeed, on average for both sites, power to detect a 20% change at α = 0.05 was almost 100%, whereas it was only 55% to detect a 10% change (Figure III-2c). On average for both sites, a 13% change (ranging from 13 to 14%) was detectable with 80% power at α = 0.05 (Figure III-3b).

Figure III-1c: Percentage of before-after comparisons that could detect a 10, 20, 30, 40, or 50% change in cover with 80% power at α = 0.05, 0.10, and 0.20 for each of two and all target species. Numbers in parentheses indicate the number of replicate sites sampled for that species.

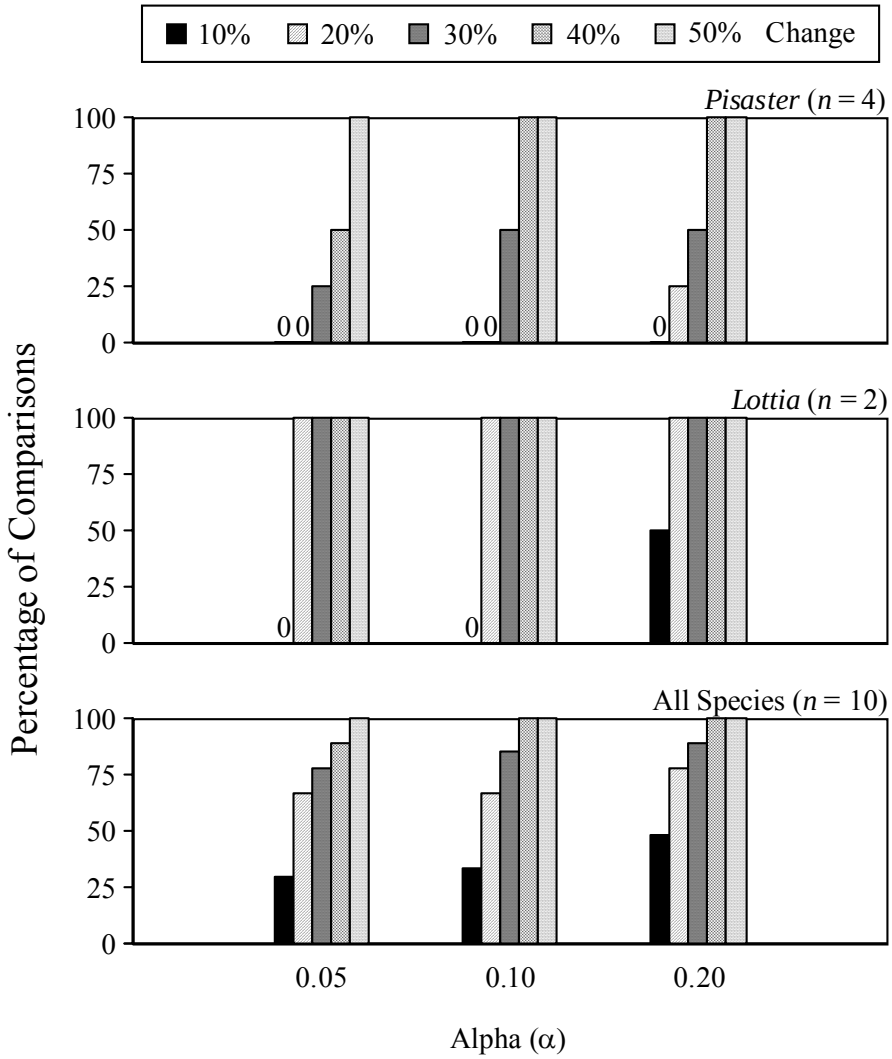
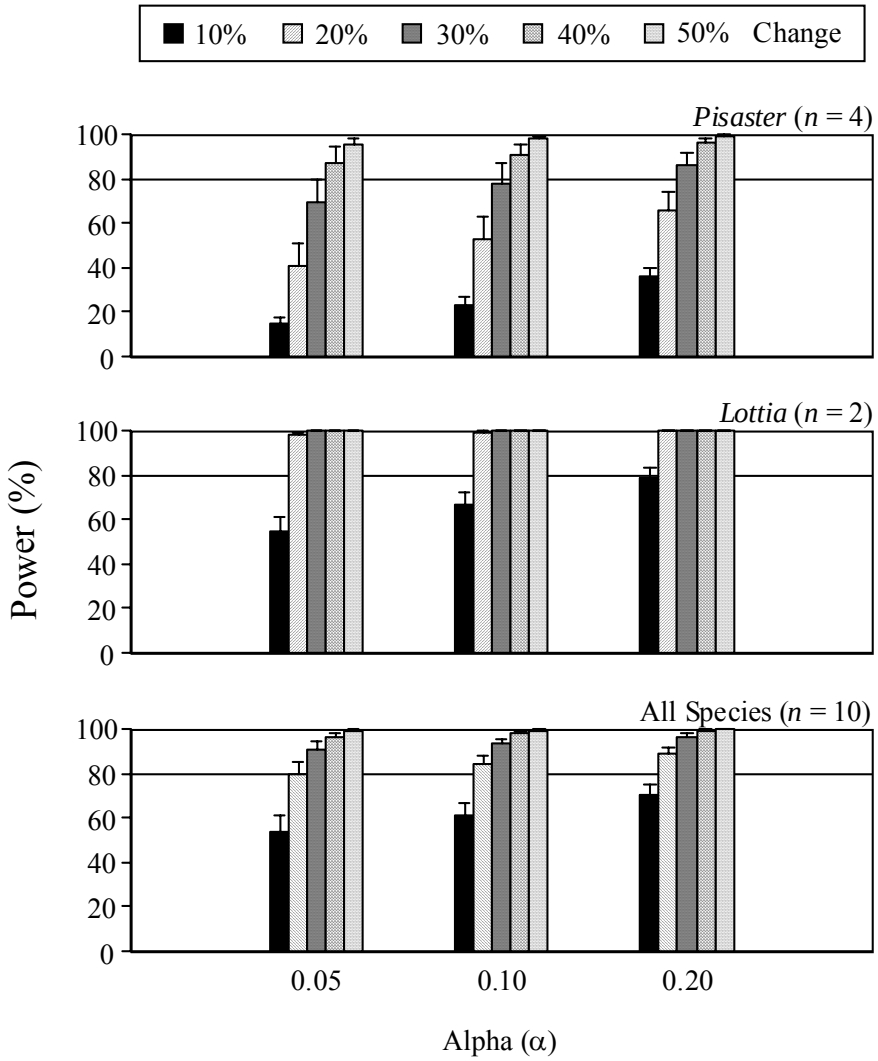


Figure III-2c: Mean (±se) power to detect 10, 20, 30, 40, and 50% changes in abundance at α = 0.05, 0.10, and 0.20 for before-after comparisons for each of two and all target species. Numbers in parentheses indicate the number of replicate sites or species sampled.



3.3.13 *Haliotis cracherodii*

Data for before-after comparisons with *Haliotis* never passed the tests of both assumptions (Table III-1c). Consequently, power analysis was not done for this species.

3.3.14 All Species

For all species, there were 10 that passed both assumptions for at least one site and could be used for power analysis. Although combining across species in this manner may not be justified because there are differences among species in the power to detect changes in abundance, it provides an informative summary. For all species and sites, a 50% change in cover was detectable with 80% power at α = 0.05, and nearly a third (8 of 27) of all species and site combinations could detect a 10% change in abundance (Figure III-1c). Indeed, there is on average 80% power (ranging from 25 to 100%) to detect at least a

20% change in abundance at $\alpha = 0.05$ (Figure III-2c). On average for all species and site combinations, an 18% (ranging from 3 to 43%) change in abundance is detectable with 80% power at $\alpha = 0.05$ (Figure III-3b).

3.4 Discussion

The purpose of this power analysis was to determine whether modifications to the spatial and temporal aspects of the sampling design of the Shoreline Inventory monitoring program are necessary so that future impacts may be reliably detected. To do this, we determined the statistical power to detect changes to the abundances of target organisms following simulated impacts at each of the monitored sites using a before-after design. For all 27 species and site combinations, power analyses revealed that in all cases there was sufficient power to detect 50% changes in abundance with 80% power at $\alpha = 0.05$ (and power was slightly greater at higher levels of alpha). Detecting 50% changes in abundance with 80% power has been proposed as a reasonable goal for monitoring studies (Fairweather 1991). By these criteria, data from the current monitoring program are adequate and no major changes to its sampling design and monitoring protocols are necessary. Indeed, for 67% of comparisons, there was 80% power to detect 20% changes in abundance for all species and site combinations. The levels of acceptable change in abundance and power is, however, something to be determined be the managers of the Shoreline Inventory. Indeed, detection of a 50% reduction in abundance may be insufficient for some species, particularly if they are in decline. A lower level of detectable change, such as 20%, may be a better target for particularly vulnerable and valuable species.

Only two other studies have done identical and extensive analyses under the same before-after sampling design used here. Ambrose (unpublished manuscript) analysed 7 target species across 8 sites in rocky intertidal communities in southern California, including some of the same sites and species as the current analysis, but over a shorter time period. He found that for 30 combinations of species and sites, there was at least 80% power to detect 50% changes in abundance at $\alpha = 0.05$ for about 93% of the comparisons. For 47% of comparisons, there was 80% power to detect 20% changes in abundance. Minchinton and Raimondi (2001) examined 6 target species across 22 sites on rocky shores of the Channel Islands in southern California. Their study included only 8 species and site combinations, but 100% and 88% of these comparisons could detect 50% and 20% changes in abundance, respectively, with 80% power at $\alpha = 0.05$. In this study of 27 species and site combinations, 100% and 67% of comparisons could detect 50 and 20% changes in abundance, respectively, with 80% power at $\alpha = 0.05$. In general, all three studies had substantial ability to detect changes from impacts should they occur. Differences in power among studies might reflect sample sizes, which ranged from 10 sampling times in Ambrose (unpublished manuscript), 12 to 30 sampling dates in Minchinton and Raimondi (2001), and 14 to 15 sample periods in the current study.

Two significant points made by Ambrose (unpublished manuscript) are important to re-iterate here. First, although power analyses under the before-after sampling design in the

above studies suggest that there is great power to detect change, it is important to point out that under this design, the majority of species and site combinations could not satisfy assumptions that would permit power analyses (see similar results in Ambrose unpublished manuscript, Minchinton and Raimondi 2001). Consequently, the majority of the data was excluded. Even under potentially more powerful designs, such as the BACI design, which was not conducted in this study, many of the assumptions prior to power analysis cannot be met (see Schroeter et al. 1993, Ambrose unpublished manuscript, Minchinton and Raimondi 2001). The extremely low pass rate of the tests of assumptions under both designs is probably partially due to the sampling of fixed plots. Sampling fixed plots may lead to greater serial correlation. The many broadscale, long-term trends in the data, which often varied among sites, however, also likely contributed to the high failure rate. This results in some species (*Silvetia*, *Hesperophycus*, *Haliotis*) and sites (e.g. Stairs) not being represented in power analyses. This may be a critical limitation of before-after and BACI analyses. Indeed, it is not surprising that long time-series of data contain trends or non-independence which preclude these analyses. Moreover, some data satisfied assumptions and were included in power analyses despite demonstrating clear trends, particularly a decrease followed by an increase over time, but these were ignored.

The presence of trends may not, however, be a critical limitation if the time-series can be adjusted for known trends (although this will be difficult for the short time series of the current set of data) or times of trends can be eliminated from the data set (although this would have to be clearly justified) (Stewart-Oaten et al. 1986). In the current set of data, it is clear that El Niño events, which sometimes result in a decrease in species abundance, introduces such complexities into the time series of data (Raimondi et al. 1999). Another clear example is the decline of *Haliotis* over time due to withering syndrome (see Altstatt et al. 1996, Raimondi et al. 2002). A final example is the consistent decrease in *Silvetia* over time, which likely occurred because initial plots were chosen to have high cover and thus it is likely that this long-lived species will decline in abundance over time. More sophisticated analyses to detrend the data were beyond the scope of this study, but should be considered as a future task to ensure that all species (and perhaps sites) are represented. Given the scope of the set of data and the ability of the data that met assumptions to detect change, it is likely that more sophisticated analyses, including power analysis under a BACI design, will further demonstrate the adequacy of the current monitoring program to detect impacts. Importantly, regardless of assumptions and designs, all of the data should be used in determining whether an impact has occurred. Indeed, at this stage where few impacts (particularly large-scale impacts) have occurred, it is difficult to appreciate fully the actual, long-term importance of the Shoreline Inventory in space and time. Detection of seasonal variation, responses to El Niño (see Raimondi et al. 1999) and withering disease by abalone (see Raimondi et al. 2002) provides convincing evidence for the maintenance of the current spatial and temporal sampling frequencies. Indeed, the addition of more sites along the coast has been a valuable addition to the monitoring program (see Raimondi et al. 2002).

Second, the initial selection of quadrats with relatively high and consistent abundances and then sampling of these fixed quadrats over time is likely responsible for the relatively

low variability in the data over time and, consequently, the high level of power to detect change, particularly under a before-after design where data from only one site are used. This phenomenon is likely to be accentuated for longer-lived species such as mussels and *Silvetia* relative to shorter-lived species such as barnacles and *Endocladia*. Indeed, this prediction exactly matches what we found here: power to detect impacts was greater for mussels than for barnacles and *Endocladia*.

The limitation of using these permanent quadrats is that conclusions, such as the power to detect an impact, are restricted to these areas and cannot be generalized in space within and among sites. Nevertheless, if one or multiple target species respond the same across multiple sites, then this should be a strong indication that the effect of an impact is real. Importantly, over time the initial starting conditions should become of lesser importance. Moreover, using fixed sites allows a permanent record of specific areas, which may have advantages in the long-term. Discussing the pros and cons of sampling fixed versus random plots is beyond the scope of this report, but maintaining numerous sites for each target species should allow for sound conclusions about spatial generality.

A specific objective of this study was to determine whether the temporal sampling frequency (i.e., number of sample periods) provides adequate statistical power to detect impacts. Based on the ability of the sampling design to detect 50% (and often 20%) changes with 80% power, results clearly indicate that more frequent sampling is unnecessary. With the exception of *Mastocarpus* and *Pisaster*, there was 80% power to detect 20% changes in abundance for all other species for at least one site. Do these results suggest that less frequent sampling is necessary? At less frequent sampling intervals, there will be a greatly reduced ability to detect seasonal and other trends such as those due to El Niño. The cost of losing this information must be weighed against the current sampling effort, but it seems sensible for seasonal sampling to continue.

Despite generally great power to detect impacts for all species, power varied among species. For species where at least 2 sites were sampled, power was greatest for detecting changes in the cover of mussels and least for *Endocladia* and *Pisaster*. Similarly, Ambrose (unpublished manuscript) and Minchinton and Raimondi (2001) also found relatively low power to detect changes in the cover of *Endocladia* relative to other species using before-after and BACI designs. Perhaps not surprisingly, Ambrose (unpublished manuscript), who analysed data from some of the same sites as in this study, also found that there was high power to detect changes in mussel cover. Minchinton and Raimondi (2001), studying sites on the Channel Islands, found the greatest power to detect changes in abundance was for the rockweeds including *Silvetia* and *Hesperophycus*, but in this study, none of the data could satisfy the assumptions before analysis because these species declined significantly over time.

The Shoreline Inventory monitoring program is one of the largest of its kind in terms of both spatial and temporal extent. The duration of the monitoring program meant that there was great power to detect changes in abundance, even though this also allowed only a small subset of the data to be analysed because of the high failure rate of tests of assumptions prior to analysis. Regardless, more sophisticated statistical techniques

would have likely allowed more of the data to be analyzed. The extensive numbers of sites and sampling times and the use of multiple target species means that there will be great flexibility to do statistical analyses to detect future impacts. The greatest benefit of the data may be that they allow graphical interpretation and analysis of an extremely long time series of data for multiple species at multiple sites. Ultimately repeated observations of common trends in space and time may end up being the most powerful and appropriate method to determine correctly whether an impact has occurred or not.

ACKNOWLEDGEMENTS

We thank Jessie Altstatt, the MMS MINT Team, and numerous assistants and volunteers for help in the field. We are grateful to many people for granting access to the study sites. These include Tim King and the Bixby Ranch Corporation and Brad Lendberg and the Cojo-Jalama Ranch (Government Point), the Vandenberg Air Force Base, especially Nancy Reed and the staff at the Recreation Center (Boathouse, Stairs, and Purisima Point), the United States Geological Survey (USGS) (Piedras Blancas), and the Hearst Corporation (Point Sierra Nevada). This work was funded by a Natural Sciences and Engineering Research Council (NSERC) of Canada Postdoctoral Fellowship to Todd E. Minchinton and awards to Peter T. Raimondi from the Mineral Management Service.

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The Department of the Interior Mission

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The Minerals Management Service Mission

As a bureau of the Department of the Interior, the Minerals Management Service's (MMS) primary responsibilities are to manage the mineral resources located on the Nation's Outer Continental Shelf (OCS), collect revenue from the Federal OCS and onshore Federal and Indian lands, and distribute those revenues.

Moreover, in working to meet its responsibilities, the **Offshore Minerals Management Program** administers the OCS competitive leasing program and oversees the safe and environmentally sound exploration and production of our Nation's offshore natural gas, oil and other mineral resources. The MMS **Royalty Management Program** meets its responsibilities by ensuring the efficient, timely and accurate collection and disbursement of revenue from mineral leasing and production due to Indian tribes and allottees, States and the U.S. Treasury.

The MMS strives to fulfill its responsibilities through the general guiding principles of: (1) being responsive to the public's concerns and interests by maintaining a dialogue with all potentially affected parties and (2) carrying out its programs with an emphasis on working to enhance the quality of life for all Americans by lending MMS assistance and expertise to economic development and environmental protection.