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CALIFORNIA FISH AND GAME

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RELATIONSHIP BETWEEN SEA OTTER RANGE EXPANSION AND RED ABALONE ABUNDANCE AND SIZE DISTRIBUTION IN CENTRAL CALIFORNIA

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Red abalone population surveys were conducted near Point Estero, California between 1965 and 1993, before, during, and after reoccupation of the area by sea otters. A decline in abalone density occurred (0.101/m^2 to 0.007/m^2) associated with the reoccupation of the area by sea otters. Evidence is presented suggesting that sea otter predation was responsible for the decline and subsequent stability at densities below those needed to support a viable commercial or recreational fishery.

INTRODUCTION

Sea otters (*Enhydra lutris*) were over-exploited throughout the North Pacific rim during the 18th and 19th century fur trade era (Ogden 1941). Only a few remnant colonies survived that era, with as few as 50 sea otters surviving in California (Kenyon 1969). Many of the colonies are gradually expanding their range under protection of both state and federal law.

Range expansion in California has coincided with the loss of commercial and recreational shellfish fisheries (Wild and Ames 1974, Miller et al. 1975, Wendell et al. 1986). Although sea otter effects on nearshore marine community structure are generally recognized (McLean 1962, Estes and Palmisano 1974, Dayton 1975, Foster et al. 1979, Duggins 1980), their effects on shellfish fisheries have been controversial (Estes and VanBlaricom 1985).

The controversy exists because of the difficulty in assessing the magnitude of the effects of sea otter predation and human harvest in the loss of fisheries. A historical review of shellfish fisheries beyond the sea otter's range emphasized their loss from over-exploitation (Estes and VanBlaricom 1985). The review expressed doubts about the relative effects of human harvest and sea otter predation in the loss of the central California red abalone fishery.

The sport and commercial red abalone fisheries in central California were the first to experience competition with foraging sea otters. Most of the research that focused on the relationship between sea otters and abalone abundance lacked information on pre-reoccupation densities. The earliest research assessed sea otter effects by comparing abalone abundance between areas with and without sea otters (Ebert 1968). Most subsequent research focused on population dynamics over longer time periods within small study areas already occupied by sea otters (Lowry and Pearse 1973, Cooper et al. 1977, Hines and Pearse 1982, Ostfeld 1982).

Here, data on abalone abundance and size distribution are analyzed that were collected intermittently over a long time period that included both pre- and post-reoccupation intervals within a relatively large study area. The original objective of the study was to monitor abalone abundance and population structure in an area important to the commercial fishery. After the loss of the fishery, the objective changed to monitoring the immediate and long-term effects of sea otter predation on abalone abundance and population structure (E. Ebert, California Department of Fish and Game, pers. comm.).

METHODS

The Point Estero study area (lat 35 30'N, long 121 02'30"W) is approximately 20 km north of Morro Bay, California (Fig. 1). The area is characterized by low profile reefs, averaging 2 to 3 m high, aligned nearly perpendicular to the shoreline and spaced about 15 to 60 m apart. Interspersed are gullies or pavement-like substrate strewn with cobbles and boulders. Sand intrusions frequently isolate rocky outcroppings, particularly

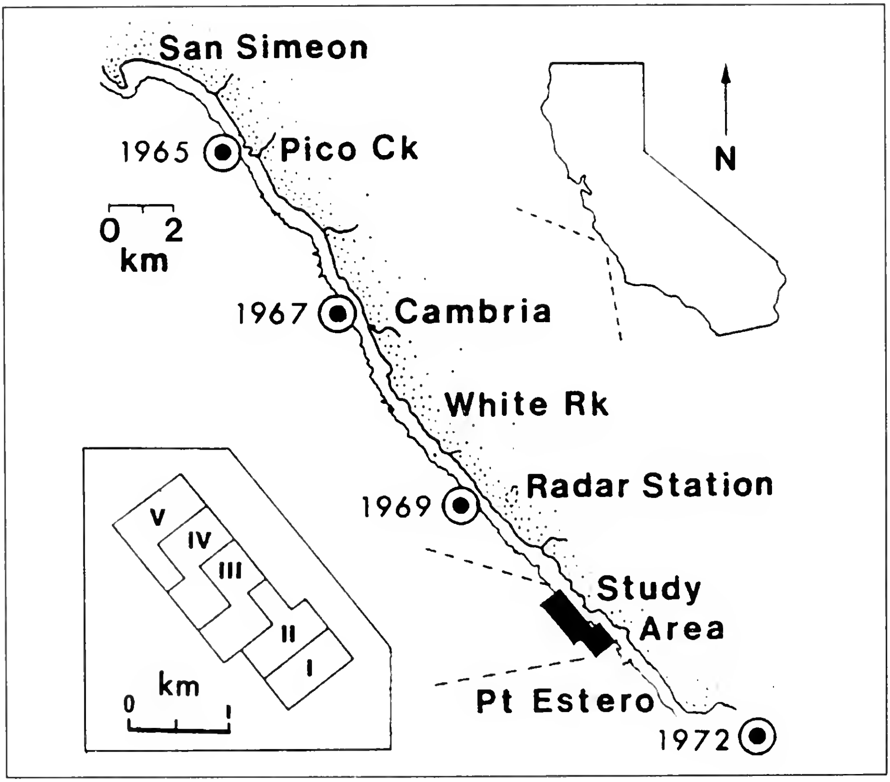

Figure 1. Spatial relationship of the Point Estero red abalone study area (darkened area) and location of the southernmost sea otter rafting site (circles) by selected year. Sampling strata shown in lower left insert.

in shallower depth zones.

Ocean depths within the 2.4 km long by 0.8 km wide study area ranged from 5.5 m to 22.0 m. The area was divided into five strata arranged from south to north (Fig. 1). Strata boundaries were selected to provide comparable depth distributions among strata. Each strata was divided into sampling units (transect areas) measuring 4.6 x 30.5 m each. Four of the strata (II-V) each contained 3,750 sampling units. Strata I contained 3,000 sampling units. Thus, the entire study area was potentially available for sampling.

Sampling effort was evenly distributed among strata and transects were selected randomly within strata. Each potential sampling unit was numbered and a random number generator was used to select those to be sampled. Initially, 15 transects were sampled per annual survey (3/strata). In 1967, the annual sampling effort was increased to 25 transects. Transects were located using triangulation from existing and fabricated shoreline markers and laid on a 290° bearing (i.e., parallel to the shore line).

Abalone were identified by species when possible and counted by SCUBA divers. Some abalone, particularly those observed within narrow crevices, were not identifiable. Species other than red abalone (*Haliotis rufescens*) were present in very small numbers and when identified were excluded from data analysis. Accessible individuals were removed and measured to the nearest millimeter across the longest dimension of the shell and replaced within the transect area. The sizes of inaccessible individuals were visually estimated and assigned to one of three size groups (<102, 102-196, >196 mm).

Population estimates were generated based on a simple random sampling design. Estimates are calculated by multiplying the mean count per transect by the potential number of transects within the survey area.

Sea otter densities and spatial distributions were assessed using aerial survey data. Emphasis, for this paper, was placed on aerial surveys conducted during the period when sea otters expanded their range into the vicinity of the study area. Range expansion in California has typically been defined by changes in the location of the southern- and northern-most large sea otter raft. However, a range boundary identified by this criterion does not recognize that foraging activity occurs beyond the raft location. Foraging activity can be located several miles beyond the raft site (Wild and Ames 1974).

Prey selection by sea otters was assessed using data collected during a food habit study conducted within the Point Estero area in 1971 and 1972 (Wild and Ames 1974). Observations were made from shore using telescopes ranging from 15x to 60x. Food items were identified to the lowest taxon possible.

RESULTS

Surveys were conducted in 1965, 1966, 1967, 1970, 1971, 1973, 1974, 1978, and 1993. A total of 205 transects were sampled during the nine surveys. The proportion of sand on a transect and the water depth were covariates in the study design. For all survey years combined, both showed statistically significant negative correlations with red abalone abundance (Spearman rank (r_s) values of -0.176, $P < 0.05$ and -0.326, $P < 0.01$, respectively). However, no temporal trends were apparent in the distribution of transect substrate types ($r_s = 0.054$, $P > 0.05$) or depths ($r_s = 0.129$, $P > 0.05$) through the study period. The test for heterogeneity of slopes among years in an analysis of covariance showed water depth ($P = 0.0096$) and proportion of sand ($P = 0.0025$) to be heterogeneous among years.

There was a temporal trend in the abundance of red abalone within the study area; the estimated population size declined through time (Table 1). There was a statistically significant negative correlation ($r_s = -0.901$, $P < 0.01$) between median red abalone counts and survey year. The main effect of survey year in the analysis of covariance was highly significant ($P < 0.0005$).

Both mean red abalone count per transect and mean counts adjusted for the covariates declined rapidly starting in 1967 (Fig. 2). The average red abalone density prior to 1967 was 0.100 abalone/m^2, declined to 0.010 abalone/m^2 by 1973, and remained below that level for all subsequent surveys (Table 1). A Kruskal-Wallis multiple pairwise comparison showed counts from all years after 1973 to be significantly different from those that preceded at an experimentwise error rate of 0.15. Some pairs comparing 1971 with earlier years were also significantly different.

The estimated red abalone population size within the study area decreased by 84 percent within six years, and stabilized within eight years at seven percent of the initial (1965) estimate (Fig. 3). The population estimates during 1978 and 1993 suggest that the resource remained relatively unchanged during the 19-year period from 1974 to 1993.

The decline in abundance appears to have occurred progressively from north to south within the study area. The mean count declined below 10 abalone/transect between the 1965 and 1966 surveys within the northernmost strata (V), followed progressively within strata IV (after 1966), strata III (1967), and strata's II and I (1970) (Fig. 4). Although multiple pairwise comparisons between years within strata did demonstrate differences between pairs, the resolution was not sufficient to place

Table 1. Red abalone density (/m^2) and estimated population size within the Point Estero study area by survey year.

Year	Transects sampled	Mean density (S.E.)	Population estimate (2,508,391 m^2)
1965	15	0.1010 (0.0316)	253,350
1966	15	0.0995 (0.0257)	249,600
1967	25	0.0718 (0.0147)	180,100
1970	25	0.0606 (0.0116)	152,000
1971	25	0.0166 (0.0037)	41,650
1973	25	0.0103 (0.0043)	25,850
1974	25	0.0066 (0.0026)	16,550
1978	25	0.0055 (0.0017)	13,800
1993	25	0.0072 (0.0026)	18,050

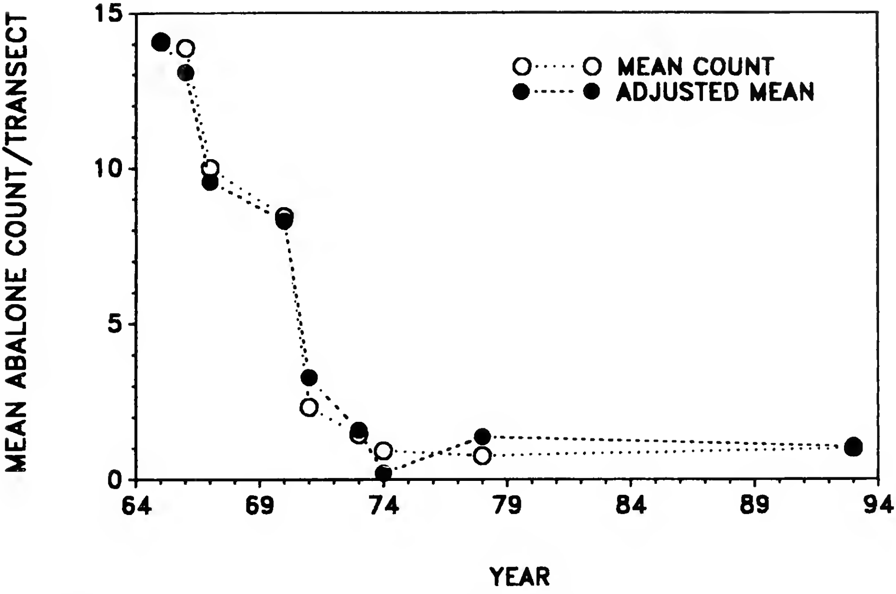

Figure 2. Mean red abalone count per transect and mean count adjusted for covariation between years in distribution of transect depth and substrate type.

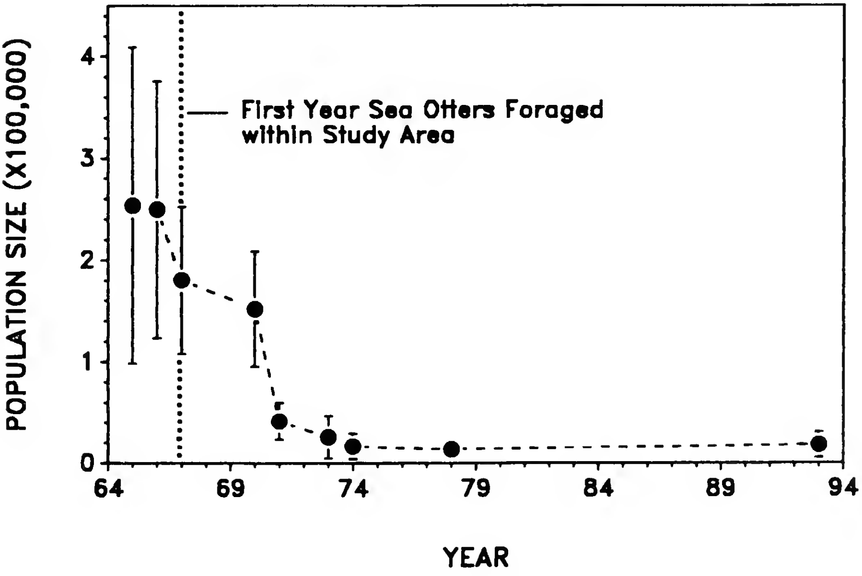

Figure 3. Estimates of red abalone population size (95% CI) within the Point Estero study area through time.

statistical significance to the temporal and geographic patterns within strata.

The size frequency distribution of red abalone within the Point Estero study area remained relatively unchanged through 1978. No abalone were accessible for measuring in 1993. There were no statistically significant differences in the size frequency distributions of measured abalone in pairwise comparisons (Kolmogorov-Smirnov 2-sample tests). The mean sizes of measured abalone ranged from a high of 181 mm in 1971 to a low of 155 mm in 1978 (Fig. 5).

There were also no statistically significant temporal trends in the proportion of abalone within size groups (r_s sm = 0.283, $P > 0.05$; r_s med = -0.217, $P > 0.05$; and r_s lg = -0.333, $P > 0.05$). However, the proportion of abalone in the 1993 survey within the smallest size group (< 102 mm) was higher than on any prior survey and there were proportionally fewer large abalone (> 196 mm).

Reoccupation of the Point Estero region by sea otters occurred through southward range expansion between 1967 and 1971 (Fig. 1). Sea otters were first observed foraging within the study area in 1967 (E. Ebert, CDFG, pers. comm.). The range expanded beyond Point Estero to Cayucos in 1972. In the interim, the peripheral male

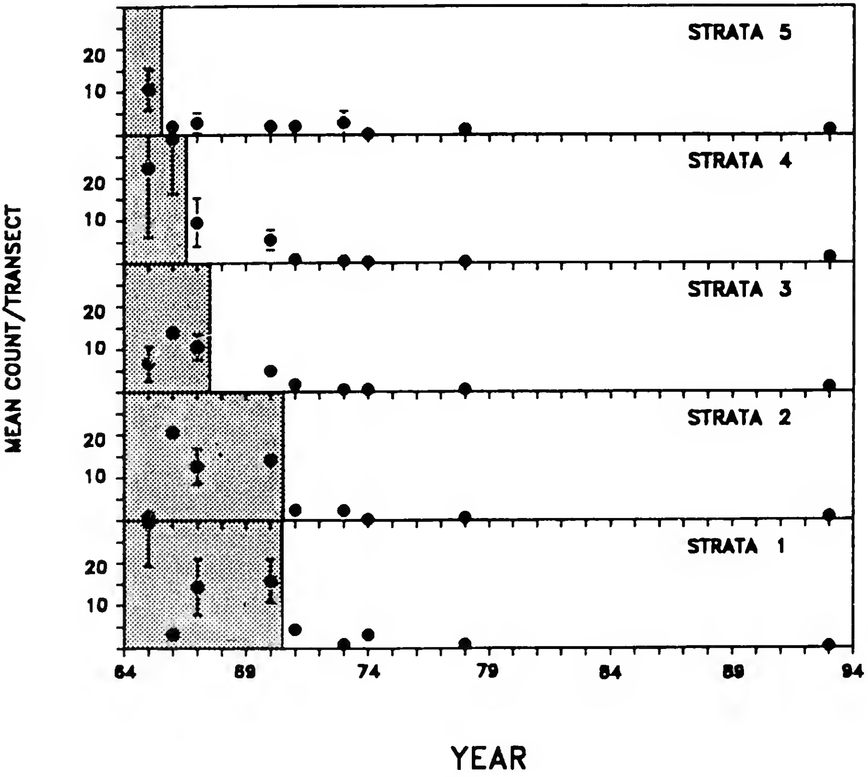

Figure 4. Mean red abalone count per transect (S.E.) by strata and survey year. Shaded area includes surveys with mean counts >10 abalone/transect.

group actively foraged in the vicinity of the Point Estero study area. There are no direct measures of changes in foraging pressure within the study area during this time period.

However, aerial counts of sea otters within the southern peripheral portion of the range between 1968 and 1972 varied considerably (Table 2). The variation in counts followed a seasonal pattern, with high counts between January and July and relatively low counts during the remaining months. The highest count each year ranged from 62 in 1968 to 187 in 1971.

Sea otter food habit observations collected within the vicinity of the Point Estero study area in 1971 and 1972 documented a marked shift to the use of a broad forage base (Wild and Ames 1974). In 1971, red abalone comprised almost 60 percent by number of the food items observed being consumed by sea otters. In 1972, that proportion had declined to less than 2 percent (Fig. 6).

DISCUSSION

The Point Estero surveys documented a precipitous decline in abalone abundance between 1967 and 1971. This decline coincided with the reoccupation of the area by sea otters. This coincidence suggests that sea otters could have caused the decline, particularly since sea otters were actively feeding on abalone through 1971. However, abalone populations have also declined in areas outside of the sea otter's range. Declines in these areas have been attributed to such factors as over-exploitation (Estes and VanBlaricom 1985) and disease (Haaker et al. 1992).

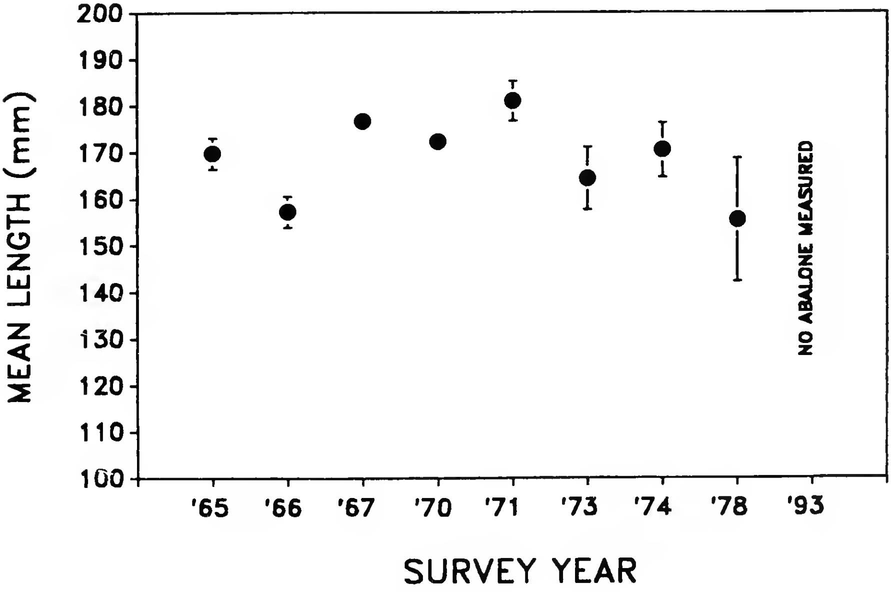

Figure 5. Mean red abalone size (S.E.) by survey.

Table 2. Number of sea otters observed at the southern range periphery by area during aerial surveys, 1968 through 1972. Area 1 = Cambria to White Rock, Area 2 = White Rock to Cambria Radar Station, Area 3 = Radar Station to Point Estero, and Area 4 = South of Point Estero.

Date	Total count	Percent within			
		Area 1	Area 2	Area 3	Area 4
8 Nov 68	62	85	13	2	0
20 Dec 68	34	97	3	0	0
31 Jan 69	117	0	98	2	0
10 Feb 69	100	2	98	0	0
10 Mar 69	129	57	43	0	0
7 Apr 69	73	0	23	77	0
5 May 69	41	5	80	15	0
2 Jun 69	40	10	80	10	0
1 Aug 69	59	5	81	14	0
8 Sep 69	26	58	42	0	0
6 Oct 69	43	2	89	9	0
1 Dec 69	41	2	66	32	0
7 May 70	165	95	0	0	5
17 Sep 70	45	20	78	2	0
13 Feb 71	60	8	10	82	0
16 Apr 71	162	4	94	2	0
1 Jul 71	187	11	58	29	2
5 Oct 71	86	6	62	24	8
4 Jan 72	75	5	13	82	0
19 Apr 72	91	11	6	2	81

Study area located within Area 3.

The commercial red abalone fishery had operated continuously within the Point Estero area since the early 1930's. Morro Bay was the primary port of landing for red abalone taken from this area. Landings there remained relatively stable for decades then decreased rapidly over a short period of time. However, the decline occurred more slowly than that observed within the Point Estero study area (Fig. 7). Red abalone resources in the region supported a commercial harvest through 1978. Fishing effort was gradually concentrated into a smaller area as it shifted southward into less productive habitat south of Morro Bay. Sea otters expanded their range beyond this area into the Pismo Beach area in 1978.

Commercial abalone fishing can be excluded as a probable cause for the precipitous decline in abalone density observed within the Point Estero area for two primary reasons: 1) the decline occurred across all size classes of abalone, even though the minimum commercial red abalone size limit is 197-mm, and 2) the decline appears to have occurred sequentially from north to south in a relatively restricted geographical area compared to the range of operation typical of the fleet. Furthermore, it is improbable that a fishery-caused decline would coincide exactly with the reoccupation of the area by sea otters.

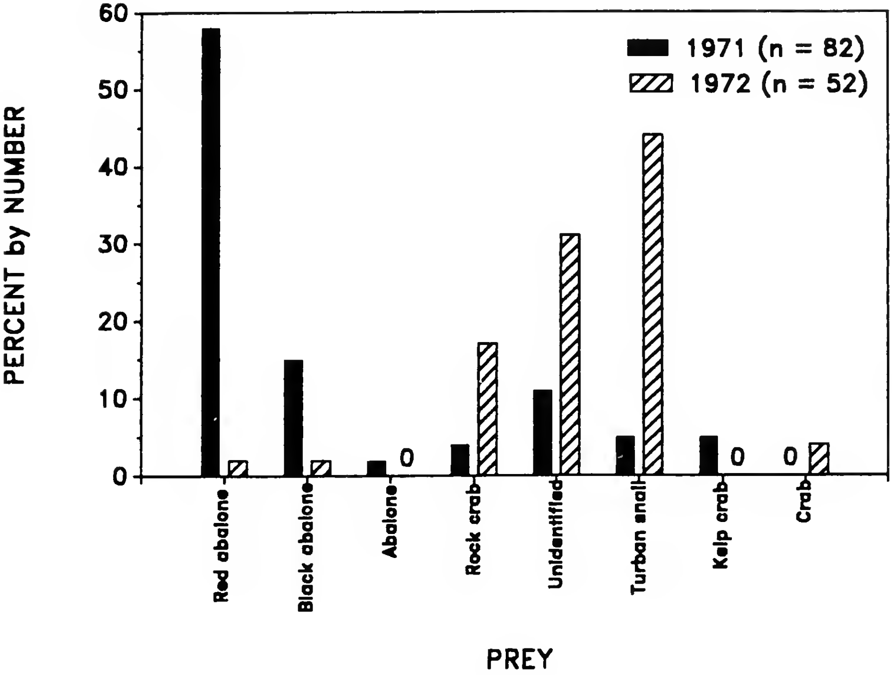

Figure 6. Changes in sea otter prey selection in vicinity of Point Estero between 1971 and 1972.

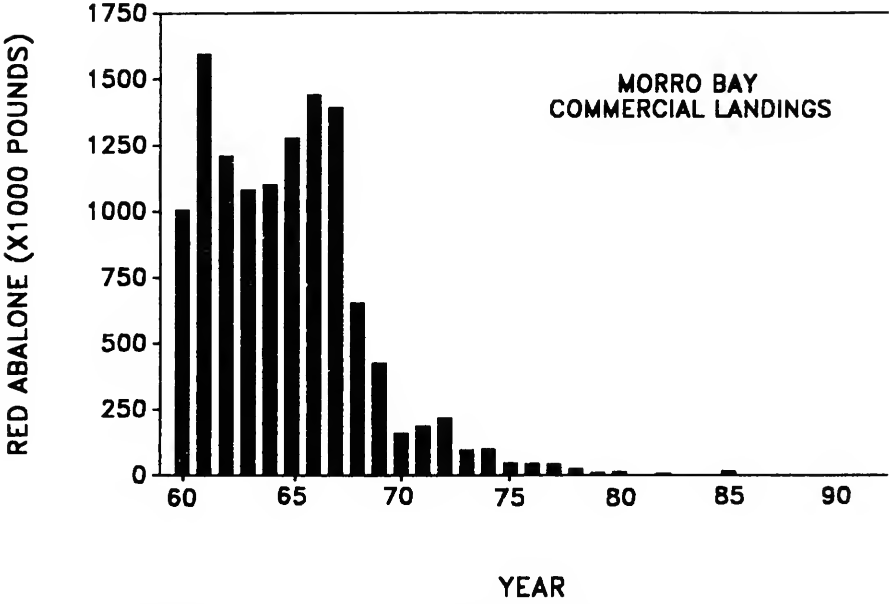

Figure 7. Commercial red abalone landings at Morro Bay from 1960 through 1992.

No evidence exists to suggest that illegal take changed appreciably during this time period despite the focused law enforcement effort generated by the controversy. It also seems probable that abalone densities would have increased in the absence of fishing pressure (especially after 1970, Fig. 2) if that pressure was the causative agent in the initial decline. No such population growth has been observed in the two decades following the collapse of the fishery.

Disease can be excluded as the probable cause for the observed decline because of a lack of evidence given considerable research diving and commercial activity in the area. Moreover, abalone were found only in narrow crevice habitat in every survey conducted after the area was reoccupied by sea otters. It seems improbable that survivors from a widespread disease would be limited to this microhabitat.

Changes in habitat can also be excluded since analysis demonstrated that appreciable habitat modifications or sampling biases had not occurred.

The density of red abalone within the Point Estero study area stabilized within a few years of the initial reoccupation of the area by sea otters (0.007/m^2). This density is similar to the abalone densities reported in several studies conducted within the long-established sea otter range (Ebert 1968, North 1965) and is low enough to preclude commercial effort and limit recreational harvest to a few knowledgeable individuals. Higher abalone densities have been reported within reoccupied habitat, but, they occurred in particularly crevice-rich habitat (Lowry and Pearse 1973, Cooper et al. 1977, Hines and Pearse 1982).

The temporal and geographical pattern of decline in shellfish abundance is comparable to that reported for Pismo clams when sea otters reoccupied beaches with clams in Monterey Bay (Miller et al. 1975, Stephenson 1977) and Pismo Beach (Wendell et al. 1986). The documented loss of shellfish fisheries associated with sea otter reoccupation strongly suggests the pattern can be used to predict future losses wherever sea otter range expansion occurs.

MANAGEMENT IMPLICATIONS

This study provides quantitative documentation of large-scale depletion of abalone stocks directly attributable to the foraging activities of sea otters. This conclusion supports the view that socioeconomic benefits derived from recreational and commercial shellfish fisheries that have developed in the absence of sea otters will largely be lost through sea otter range expansion.

However, there are also other positive effects associated with sea otter range expansion. Algal production has been observed to increase in many areas reoccupied by sea otters due to the reduction in abundance of herbivorous macroinvertebrates (North 1965, Estes and Palmisano 1974, Duggins 1980). Although evidence is generally lacking, these increases presumably can, in turn, influence the abundance of finfish populations that are limited by a lack of habitat or forage provided by algae. Sea otter range expansion also decreases the risks the population faces from human activities, particularly from large-scale oil spills, and increases the opportunities for observing sea otters.

Ongoing debate focuses on the possibility of limiting sea otter distribution while insuring a secure future for sea otters and providing for human use of shellfish resources. If both concerns are to be provided for, the reoccupation of the sea otter's natural range would have to be limited through zonal management.

ACKNOWLEDGMENTS

This study, with a temporal scope covering almost 3 decades, draws upon the research efforts of a great many individuals. I acknowledge the extensive contributions made by the previous principle investigators K. Cox (1964), R. Poole (1965), E. Ebert (1966 - 1967), and R. Burge (1970 - 1978) and thank them for their efforts. I thank D. Gotshall, P. Wild, S. Schultz, J. Duffy, J. Ames, R. Hardy, and C. Pattison for their in-water efforts. I also thank N. Abramson and P. Law for their help in the statistical analysis of the data. P. Wild, R. Heimann, and anonymous referees provided advice and a critical review. To all of the above and those not mentioned, my thanks.

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THE EFFECT OF DIFFERENT HARVEST METHODS ON SEA PALM (*POSTELSIA PALMAEFORMIS*) SPOROPHYLL GROWTH

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Sea palm (*Postelsia palmaeformis*) is an erect annual brown alga occurring along the central and northern coasts of California in high intertidal areas subject to heavy wave shock. Presently, there is no restriction on the amount or method by which sea palm can be commercially harvested. In spring 1989, we initiated a study to compare the effects of three different harvest methods on several growth and development parameters. A factorial experimental design was chosen to examine sporophyll (blade) growth under the three treatments over a 4 month period. Treatments included blade cut in which only blades are taken, branch cut in which the rounded portion of the sporphyll is also removed, and stipe cut in which the the meristem and fruiting portion of the sporophyte is taken, as well as a control group. Monthly incremental blade growth was significantly different among treatments ($P < 0.000$), with the blade cut group exhibiting the fastest growth rate. Mean blade length increased 7.5 times in two months from the start of the experiment for the blade cut group, compared to a 17 percent increase for the control group for the same time period. We conclude that a harvest method similar to the blade cut treatment would have the benefit of allowing multiple yields of sporophylls in a season, ensure spore production, and reduce aesthetic degradation in comparison to the other harvest methods examined.

INTRODUCTION

In the spring of 1989 the potential for large-scale removals of sea palm (*Postelsia palmaeformis*) by commercial kelp harvesters in central and northern California became apparent to the Department of Fish and Game (Department). Sea palm is locally abundant in the middle to high intertidal zone in exposed areas, such as rocky headlands subject to high wave shock, from central California to Vancouver Island (Abbott and Hollenberg 1976). A member of the Laminariales, the sporangial thallus of a sea palm sporophyte can grow to 60 cm (24 in.), standing erect and resembling a miniature version of its terrestrial namesake (Fig. 1). The plant attaches to rocky substrate by means of a hapteron, or holdfast, and is found patchily distributed usually among mussel (*Mytilus* spp.) beds (Dayton 1973). Sea palm is an annual and like all Laminariales exhibits alternating morphological phases, a microscopic gametophye and the large macroscopic sporophyte for which there is an active commercial fishery during the spring to fall months in certain portions of its range. Presently, sea palm may not be taken in California under authorization of a sport fishing license (Title 14, Sec. 30.10), though there is no limitation on the taking of sea palm under a commercial kelp harvesting permit. Resolution of this legal anomaly was a motivating factor for this study.

Preliminary investigation of the local sea palm fishery in Mendocino County, northern California, in 1989 included discussions with a harvester who claimed to harvest sea palm by removal of sporophylls (blades) only, leaving approximately 25 mm of the grooved portion of the blade intact on the plant. This was suggested as an

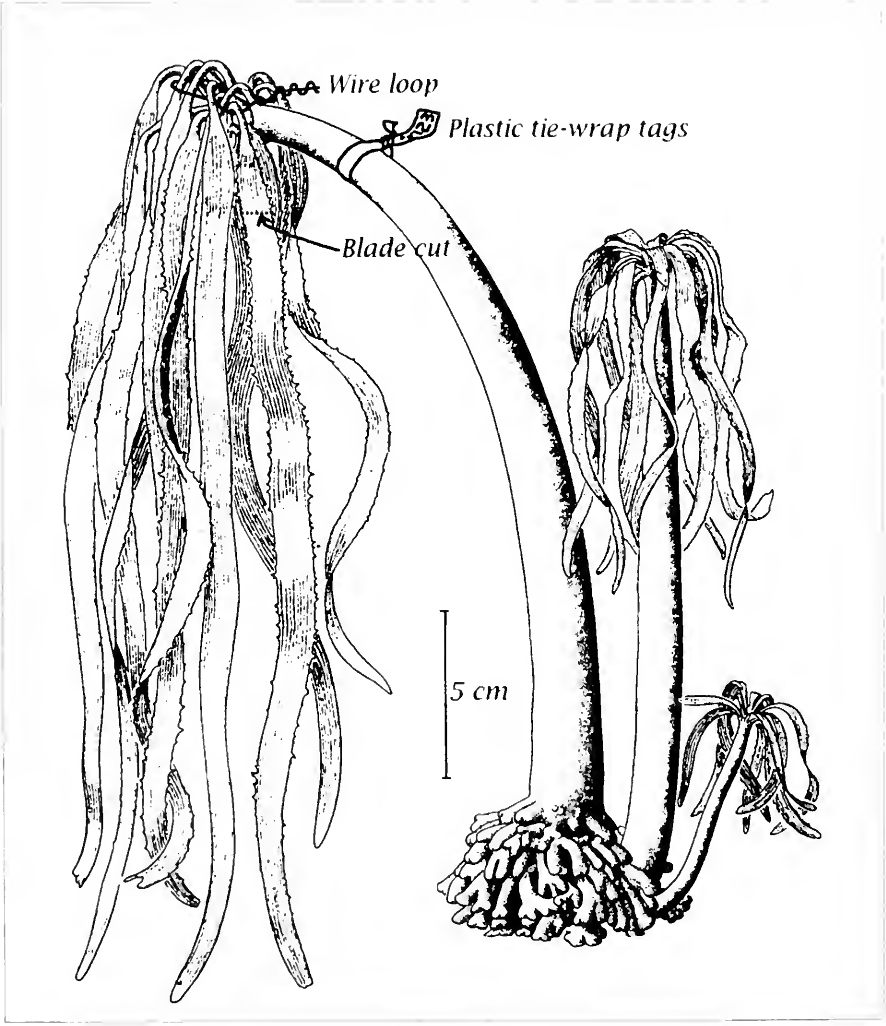

Figure 1. Sea palm (*Postelsia palmaeformis*) showing haptera, stipe, and sporophylls (from Abbott and Hollenberg 1976).

alternative to cutting the stipe, which destroys the thallus, a method which is legal and practiced by some harvesters. The blade cutting method allows the plant to remain alive while allegedly yielding as many as three crops during the spring to early fall growing season. It was unclear whether this method allowed the plant to produce spores for the next generation of gametophytes. Sporangia are first produced in late spring in linear sori lining grooves on the blades, with meiospores then released during low tides onto the adjacent rocky substrate. Blades become eroded after these fruiting phases (Abbott and Hollenberg 1976).

The goal of this study was to determine the impact of the following three alternative harvest methods upon sea palm growth and spore development: 1) complete blade removal; 2) partial blade removal; and 3) stipe cutting to remove the stipe apex and all branches (terete portion of blade) and blades. Impact variables to be measured were change in blade and stipe length over time, sporophyte density, and presence or absence of spore release.

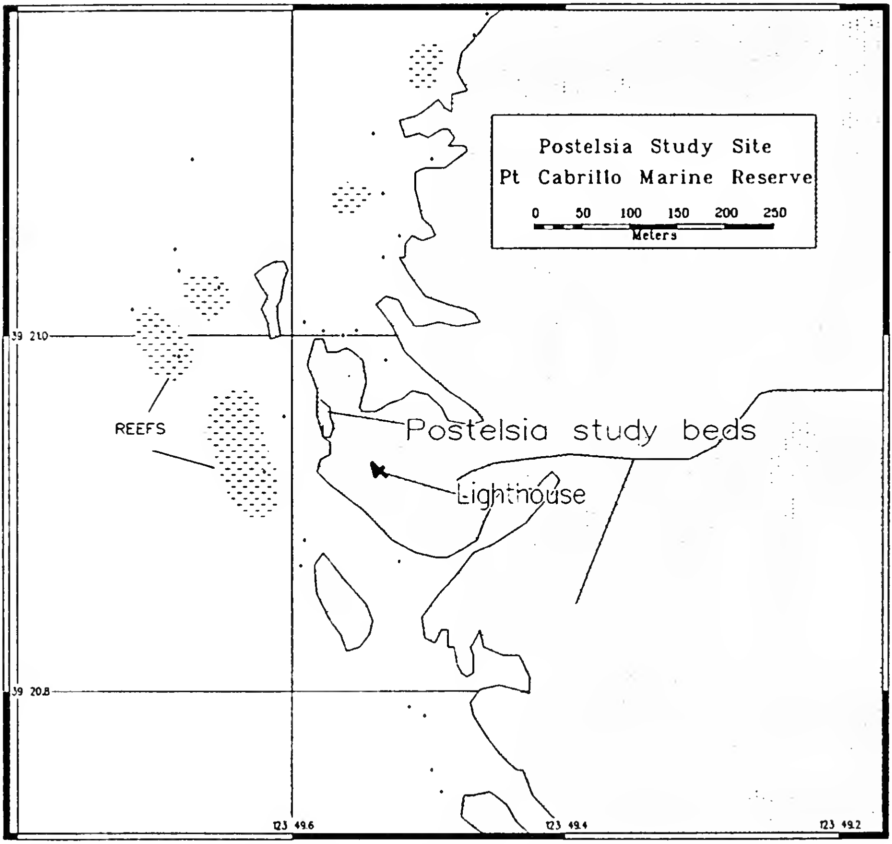

Figure 2. Sea palm study site at Point Cabrillo Marine Reserve near Fort Bragg, California.

METHODS

Beginning in May 1989, a sea palm bed was chosen for study at Point Cabrillo Marine Reserve (PCMR) about 10 kilometers south of Fort Bragg in Mendocino county (Fig. 2). PCMR is on an exposed rocky headland in which all forms of marine life are protected from harvest. The bed we chose is easily accessible from the mainland and though exposed to heavy surf, particularly at high tides, is partially protected by outer reefs. The study area was the northern edge of a main bed about 10 meters in length.

The study was designed as a two-way analysis of variance (ANOVA) with treatments and elapsed time, expressed in monthly intervals, as the main effects. Four plots were chosen in the study area, each representing a portion of a discrete patch of sea palm, within close proximity to each other and subject to similar wave exposure. Plots were 0.5 m by 1.0 m, delineated by means of a removable rectangle of pvc piping, with each plot representative of a separate treatment. The treatments included partial blade removal (blade cut) of all blades at a point approximately 25 mm (1 in.) from the inception of grooves at the proximal end of the blade; complete removal of all blades at the junction of branch and blade (branch cut); partial stipe removal (stipe cut) in which the stipe was cut just below the apex containing all branches and blades; and a control group. All plants in each plot were counted and subjected to one of the four treatments.

Eight plants, chosen at random from mature-appearing plants (all with stipe length > 170 mm) in each plot, were tagged with numbered plastic tie-wraps around the stipe. Approximately 10 blades or branches of each tagged plant were isolated with plastic-coated wire looped through the group and around the plant's stipe so as to ensure that the same blades were measured throughout the experiment, thereby reducing the error variance component in monthly mean blade lengths (Fig. 1). Following treatment of all blades, blade measurements were made to the nearest millimeter, from the inception of grooves at the proximal end of the blade to the blade tip. Stipes were measured from the base at the junction of the hapteron and stipe to the apex immediately below the formation of branches. Stipe length measurements were not made on stipe cut plants during the study and no stipe lengths were recorded on branch cut plants in May. Measurements were made once each month from May through August 1989. Plots were relocated each month with the aid of photographs and pvc rectangles so that plant counts and measurements could be made.

In May and June 1990, sporophylls of five plants in the same bed as in 1989 were tagged, given the blade cut treatment, and subsequently assessed for spore release. Plants were treated in May during a low spring tide and cut portions were retained for laboratory examination. Pieces were wrapped in sea water-soaked paper towels, stored overnight under refrigeration, and subsequently examined with a dissecting microscope for spore release the following day. In June, following a month of growth, sections of new blade were removed from four tagged plants and a control plant and examined as before.

RESULTS

Blade Length

Growth as expressed by elongation of blades was examined in two ways: cumulatively and incrementally. The control group mean bladelength was 238 mm (SD 54) at the start of the study, growing to 298 mm (SD 57) by August: an increase of 25% over the 98 day period (Table 1, Fig. 3). Mean blade cut length in May was 26 mm (SD 7), increased rapidly to 112 mm by June (SD 24), and 195 mm (SD 47) by July. By August only one plant, with a mean blade length of 335 mm (SD 75), remained. The branch cut treatment was expected to hinder growth because the sea palm meristem is intercalary, located between stipe and blade in the region where the branch was incised. Surprisingly, mean blade length was 84 mm (SD 57) in August for this group. By June all remaining sporophytes in the stipe cut group exhibited fractured stipes and no evidence of regrowth. The stipe cut group was not included in the blade length analysis.

Incremental growth was defined as change in mean blade length for each plant between monthly measurements. Negative increments were coded as zero growth. A two-way ANOVA showed that the treatment effect was significant ($P < 0.000$), but month and interaction effects were not significant ($P > 0.18$) (Table 2). The absence of significant interaction between the two main effects (as evidenced by the parallelism of the lines connecting the monthly means for each treatment in Fig. 3) increased the

Table 1. Sea palm blade length (mm) cell means by treatment and month.

	Treatment								
	Control			Blade Cut			Branch Cut		
Month	*n*	$\bar{x}$	SD	*n*	$\bar{x}$	SD	*n*	$\bar{x}$	SD
May	75	238	54	87	26	7	0	-	-
June	76	279	70	83	112	24	29	11	6
July	40	278	76	80	195	47	27	38	28
August	30	298	57	10	335	75	33	84	57

Table 2. Two-way ANOVA of sea palm blade length (mm) increments by treatment and month.

Source of Variation	DF	SS	MS	F	Prob.
TREATMENT	2	20314	10156.98	19.00	0.0001 **
MONTH	2	1917	958.42	1.79	0.1811
TREATMENT*MONTH	4	3311	827.72	1.55	0.2092
Error	36	29222			
Total	44	54764		*R*-Square = 0.649	

usefulness of the other statistics (Nie et al. 1975). Interestingly, though the control group maintained a greater total mean length through most of the summer, the blade cut treatment group exhibited a faster rate of growth in each month (Fig. 4). In two of the three months, the branch cut group growth rate was greater than that of the control group (Fig. 5).

There were several disruptive developments during the experiment. There was an unexpectedly high loss of sporophytes due to stipe breakage or complete removal (presumably from wave shock) as well as the loss of identifying wire for some blade groups. In these cases, blade groups were chosen randomly among remaining undamaged blades. The reduction of numbers of plants during the experiment may have created an unexamined plant effect which could have been responsible for some of the variance in blade growth. Additionally, blade erosion began with the onset of fruiting, somewhat confounding the assessment of growth via blade elongation (Abbot and Hollenberg 1976).

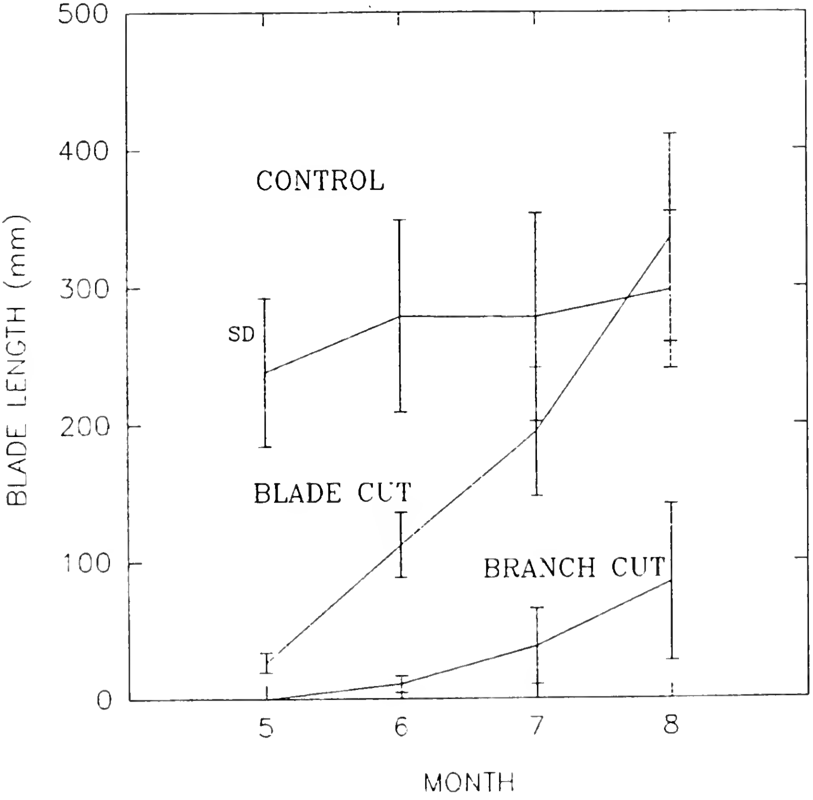

Figure 3. Sea palm mean blade length by treatment and month.

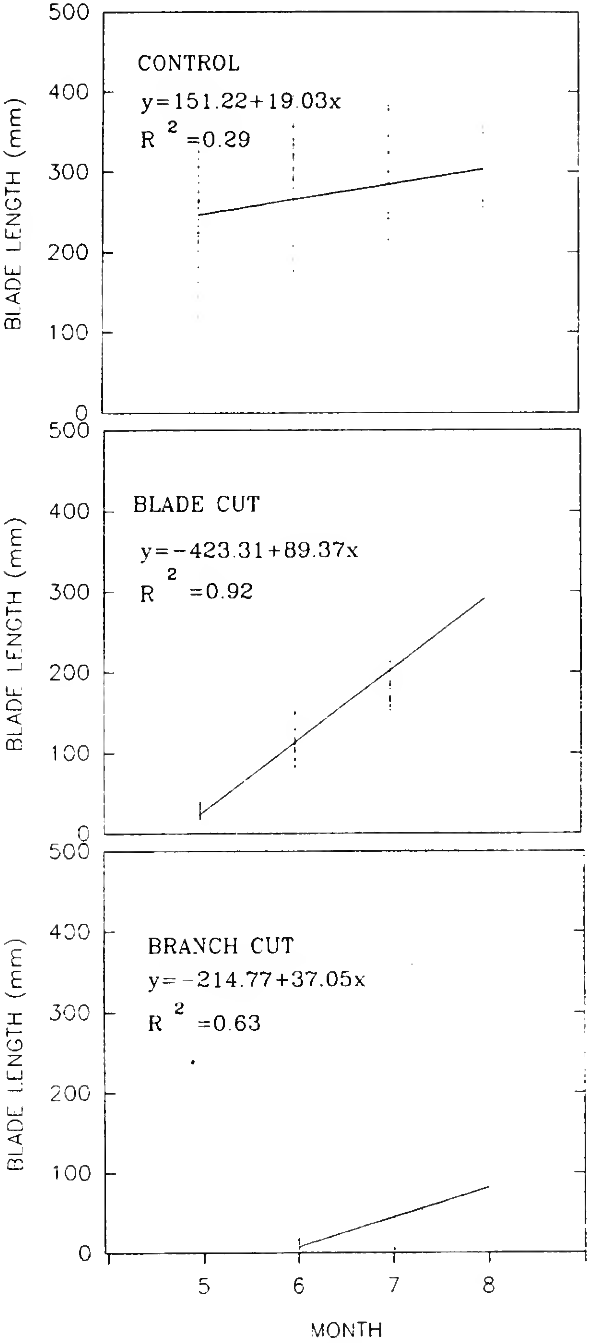

Figure 4. Regressions of sea palm blade length on month by treatment.

The site was revisited in September 1989, but no measurements were made. All treatment plots were partially submerged, with the blade cut group appearing healthy and comparable to the control group in blade length. There was significant growth noted on some of the branch cut sporophytes.

Stipe Length

Stipe elongation was monitored to assess whether blade removals would affect the rate of stipe growth, and evaluated by separate one-way ANOVAs with treatment and month as factors. Stipe lengths were significantly different between treatments ($P <$ 0.0000), however, mean stipe lengths for the branch cut group were initially shorter than the control and blade cut groups (Table 3). *A posteriori* Scheffe tests showed the branch cut group to be significantly different from each of the other two groups ($P <$ 0.0000). There was no significant difference between stipe lengths when examined by the month factor alone ($P = 0.64$).

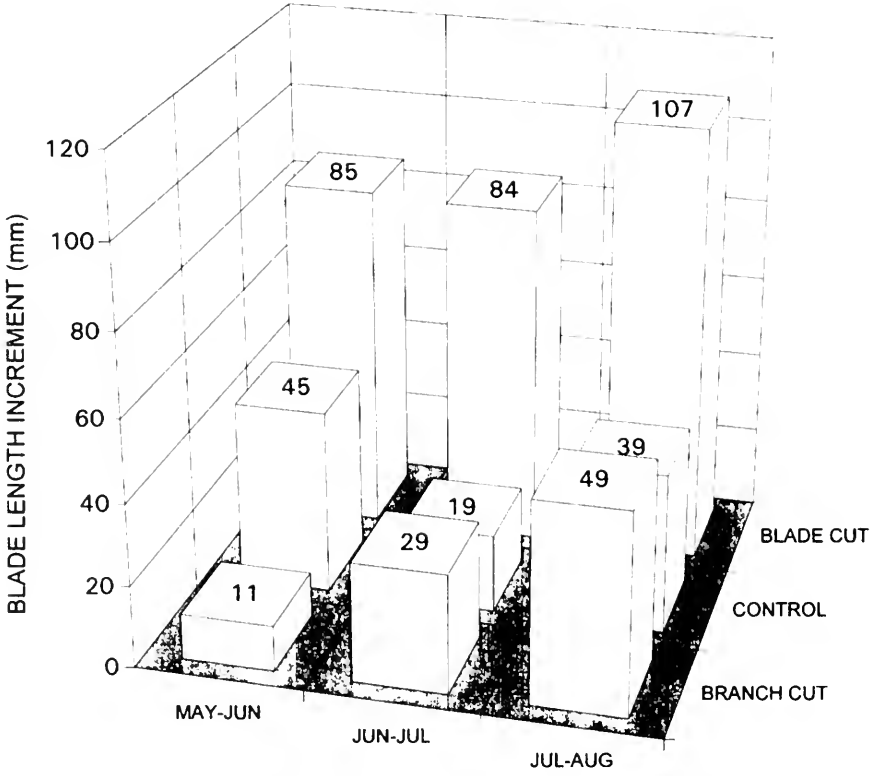

Figure 5. Sea palm mean blade length increments by treatment and month.

Table 3. Sea palm stipe length (mm) cell means by treatment and month.

	Treatment								
	Control			Blade Cut			Branch Cut		
Month	n	$\bar{x}$	SD	n	$\bar{x}$	SD	n	$\bar{x}$	SD
May	8	367	35	8	345	40	0	-	-
June	4	412	16	7	444	28	5	229	42
July	4	431	53	8	434	54	4	248	52
August	3	427	16	1	460	0	4	251	51

Density

Sporophyte counts were made each month within each 0.5 m^2 plot. Overall mean was 53.1 plants or 106.2/m^2. The stipe cut plot suffered the greatest reduction in density - from 128.0 in May to 16.0/m^2 in August, a reduction of 87.5%. The branch cut plot suffered the lowest plant loss at 12.6% and began with the highest density at 172.0 plants/m^2. The control group suffered a 49.0% loss rate compared to 74.0% for the blade cut group. The greatest percentage reduction occurred between July and August for each group other than the control group.

Spore Release

In May 1990, cut sporophylls of the five treated tagged plants were examined by dissecting microscope and zoospore release was noted for all five plants. In June, following a month of growth, sections of new blade were removed from four tagged plants and a control plant and examined as before. The control plant and two of the treated plants appeared to show zoospore release. By October 1990, no tagged plants remained and about 60% of the original bed was gone, presumably due to senescence and wave shock.

DISCUSSION

Results confirmed the anecdotal evidence of local kelp harvesters (Betsy Holliday, pers.comm.) that proper blade cutting allows the production of at least one additional crop in a season, since, by July 20, mean blade length of the blade cut group (195 mm) approached that of the control group (278 mm). Spore production was also evident one month after blade cut treatment in 1990. The blade cut treatment elicited a strong recovery response and capacity for spore production, while branch cutting had a more limiting effect upon blade growth recovery. *Nereocystis luetkeana*, bull kelp, is another annual Laminarian. It exhibits rapid stipe elongation initially, then slows while stipe girth and blade growth accelerates. Blade growth appears to continue at a steady rate well into maturity (Nicholson 1970). Partial blade removal of all but 5 cm of bull kelp blade curtailed growth for the first 2 weeks, then growth increased to a rate at or above that of untreated plants. Nicholson (1970) believed that this method of partial blade removal would permit several blade crops per year from these annual sporophytes.

The sea palm uses a unique strategy to help create patches free of competitors on bare rock, its most suitable substrate. Pioneer sporophytes clear areas by attaching to mussels which can increase the chances of the mussels being torn loose by heavy wave action. In this way sea palm sporophytes can establish themselves in areas otherwise controlled by more dominant species (Dayton 1973).

Sea palm bed boundaries can change within a season and from year to year, with new beds becoming established in suitable habitat (Paine 1988). However, Postelsia seems to have an effective sporophyte colonization distance of only about 3 m from the edge of an existing sea palm patch (Dayton 1973). In a Puget Sound study, only 36% of small patches (1-30 plants) continued to the following year, but all patches with more than 120 individuals persisted (Paine 1988). Local extinctions due to interaction with dominant perennial species such as *Mytilus* are inevitable, but probably transitory (Lawrence and McClintock 1988). Subsequent studies might examine sea palm recruitment success relative to the blade cut treatment while varying such elements as plot size and onset of harvest season.

Based upon the four criteria used to assess harvest impact and our knowledge of *Postelsia* survival strategy, we conclude that the blade cut method is not a significant negative impact upon sporophyll growth and spore production. However, we did not investigate the volume or viability of spores produced following blade cutting. Regarding present management policy, it may be desirable to institute regulations permitting only the blade cut method of harvest, which would allow multiple yields, ensure spore production, and reduce aesthetic degradation in comparison to the existing regulation. Such benefits are offset, at least partially, by reduced harvest efficiency in comparison to the stipe cut method of harvest. However, the stipe cut group suffered the greatest loss of plant density, presumably because holdfasts require the translocation of photoassimilates from the sporophylls for growth (Schmitz and Lobban 1976). In light of the limited spore dispersal of sea palm, clear-cut stipe cutting (removal of all plants over extensive portions of sea palm beds), particularly early in the reproductive season, may create an 'Allee effect', or negative density-dependence (Levitan et al. 1992), in which subsequent sea palm recruitment is reduced in the immediate area, especially where patches are discrete and small.

Clear-cutting in zones, or selective-cutting, may increase harvest efficiency without the negative impact of extensive clear-cutting, but determining the size and number of these zones on a site-specific basis and subsequently enforcing such a system would be impractical. Alternatively, the blade cut-only harvest method could be fairly easily enforced, since possession of any part of the stipe or branches would constitute an infraction. Each kelp harvester's regulation packet could include a copy of the regulation and a diagram depicting the cutting method.

In central and northern California a high percentage of sea palm beds occur within the boundaries of coastal state park lands. While the California Fish and Game Commission has the authority to regulate commercial kelp harvest, the California Department of Parks and Recreation (DPR) has the legal authority to control access across state park property to sea palm beds along state park shores, by virtue of a

California Code of Regulations section empowering DPR to limit commercial operations on park property, thus providing a measure of *de facto* protection to these beds (Dave Bartlett, DPR, pers. comm.). Since access from the sea is not controllable by DPR, it would still be legal under certain circumstances to harvest sea palm along park shores from a vessel.

A comprehensive management scheme for sea palm might include prohibiting commercial and sport take in tidal areas along coastal state parks, while allowing commercial harvest elsewhere under the previously described scenario, and further, opening a limited sport fishery for sea palm requiring the use of the blade cut method and a daily bag limit (by wet weight of blades). In order to ensure that some spore release occurs in a given area, commercial and sport harvest might be prohibited prior to July 1. Also, the problem of one or more harvesters over-cutting an area or returning to previously cut plants too soon could be addressed by a commercial bag limit.

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DUCK AND SHOREBIRD REPRODUCTION IN THE GRASSLANDS OF CENTRAL CALIFORNIA

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We studied the effects of contaminants in agricultural drainwater on the reproductive success of ducks and shorebirds nesting in the Grasslands of western Merced County, California during 1986 and 1987 and at the Mendota Wildlife Area in Fresno County in 1987. Nesting success and egg hatchability varied by species, site, and year, but were, in most cases, higher in the Grasslands than at Mendota. The primary cause of nest failure for ducks and shorebirds at both study sites was predation. Although concentrations of selenium (Se) were found to be elevated in certain Grasslands drainages, embryotoxic effects attributable to environmental contaminants were not observed for any species at either study site.

INTRODUCTION

Wetlands of the Grasslands in Merced County, California (Fig. 1), comprise the largest remaining tract of waterfowl habitat in California's San Joaquin Valley (Gilmer *et al.* 1982). Between 1954 and 1985, managed marsh areas in the Grasslands were flooded in the autumn (September 15-November 1) using a mixture of agricultural drainwater and fresh irrigation water; after November 1, drainwater was used exclusively each year to keep marsh areas filled for duck hunting and other purposes (Grassland Water District and Grassland Water Task Force 1986). In 1984, this water contained an average of 50 μg/L dissolved Se (Presser and Barnes 1985), or about 50 times the concentration in water at a nearby reference site, the Volta Wildlife Area (Saiki and Lowe 1987).

In 1983, the mean concentration of Se in water samples from agricultural drainwater evaporation ponds at the Kesterson National Wildlife Refuge was 122 μg/L (Saiki and Lowe 1987). During 1983-1985, avian reproduction at the Kesterson ponds was impaired (including embryonic deaths and deformities) by elevated concentrations of dietary Se (Ohlendorf *et al.* 1986a, 1986b, 1989; Ohlendorf 1989).

Selenium concentrations in livers and eggs of most birds collected from the South Grasslands in 1984 were intermediate between those from Kesterson Reservoir and the Volta Wildlife Area (Fig. 1) (Ohlendorf *et al.* 1987). Although deformities were not found in the limited numbers of avian embryos examined that year, the concentrations

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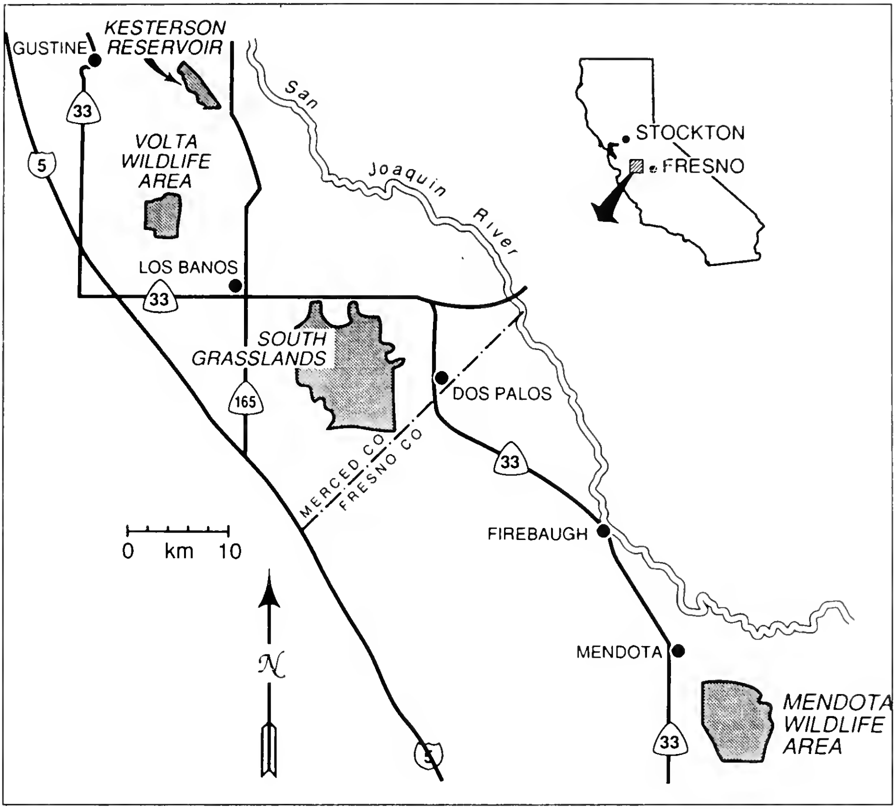

Figure 1. Kesterson Reservoir, Volta Wildlife Area, the South Grasslands, and the Mendota Wildlife Area, California.

of Se in tissues of several species of aquatic birds were sufficiently elevated to cause concern for their health.

In response to this concern, fresh water replaced most agricultural drainwater for marsh management in the Grasslands in 1985 (Grassland Water District and Grassland Water Task Force 1986). Paveglio *et al.* (1992) confirmed that water supplied to the Grasslands during autumn of 1985 and 1987 contained low concentrations of Se (≤ 2 μg/L). We studied ducks and shorebirds nesting in the South Grasslands in 1986 and 1987 and at a reference site, the Mendota Wildlife Area, in 1987 to evaluate the effects of improved water quality in the Grasslands on their reproductive success.

STUDY AREAS

The Grassland Water District, located west of the San Joaquin River in Merced County, includes 20,655 ha of mostly seasonal wetlands. The study was conducted in 1986 and 1987 on privately owned waterfowl hunting clubs in the South Grasslands.

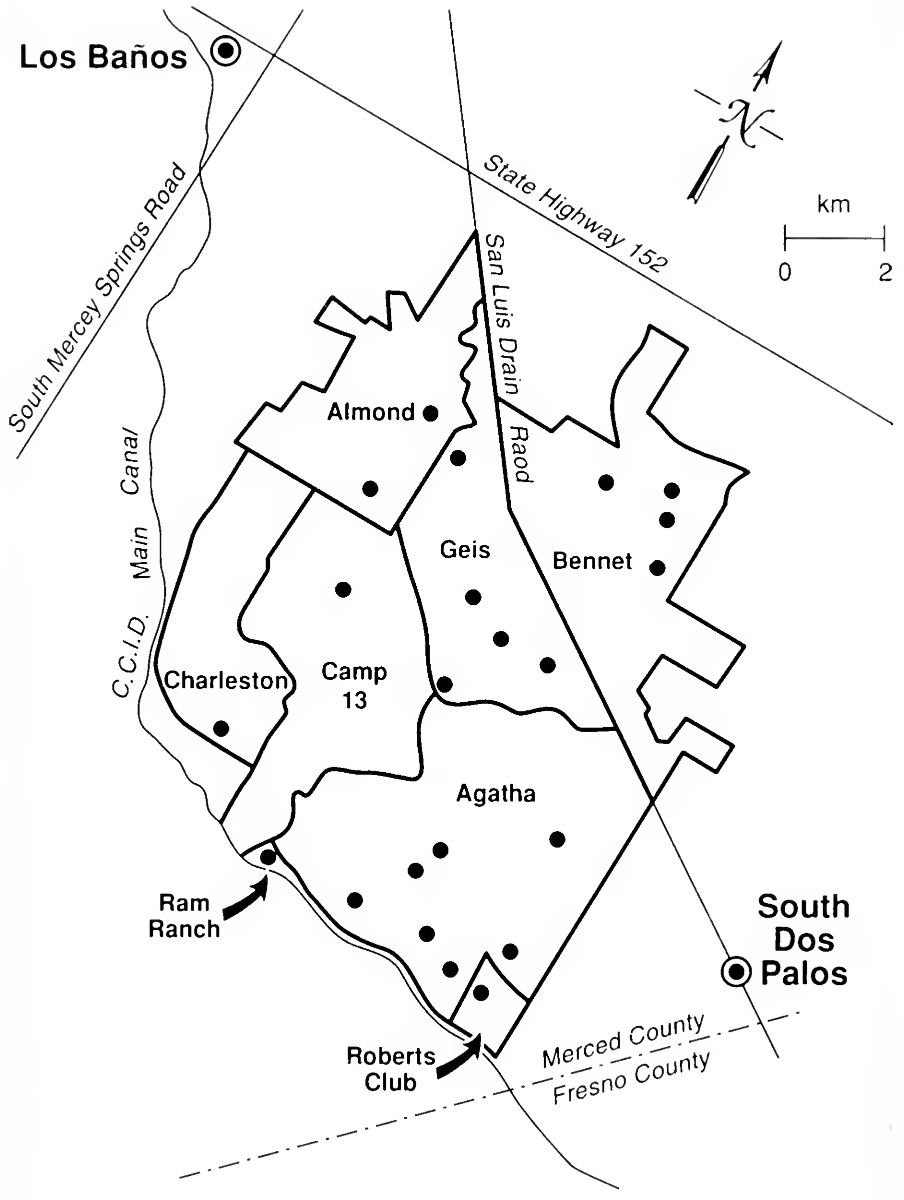

Figure 2. Sampled drainage units in the South Grasslands study area. Dots represent sampled duck clubs.

the southern 8,100 ha of the Grassland Water District (Fig. 1). Most of these clubs were within one of the six major drainage units of the South Grasslands, each defined by its unique source and quality of water: Agatha, Almond, Bennet, Camp 13, Charleston, and Geis (Fig. 2). Paveglio *et al.* (1992) collected birds and other samples for contaminants analyses from these drainages during 1985-1988. Ram Ranch and Roberts Club, two clubs not sampled by Paveglio *et al.*, were included in our study. These clubs, located on the southern periphery of the Grasslands, used uncontaminated water from the Central California Irrigation District (CCID) canal for marsh management and irrigation (F.L. Paveglio, U.S. Fish and Wildlife Service, pers. comm.).

During 1987, in cooperation with the California Waterfowl Association (CWA) and the California Department of Fish and Game, we added a reference site to the study, the Mendota Wildlife Area. Mendota is a 4,900-ha area in Fresno County managed by the California Department of Fish and Game for, among other things, wintering waterfowl and public hunting (Fig. 1). We considered Mendota a reference site because drainwater had not been used there, and previous sampling had not revealed Se contamination. For example, Se was not detected (< 1 μg/L) in one water sample collected from Mendota during September 1985 (Shelton and Miller 1988), and Se concentrations in mosquitofish collected from Mendota in 1984 were similar to those collected at another reference site, the Volta Wildlife Area (Ohlendorf *et al.* 1987).

METHODS

During April-June 1986 and March-June 1987, we spent 2-3 days per week searching for nests in the South Grasslands. We located most duck nests by dragging a chain between two all-terrain vehicles (Higgins *et al.* 1969, Klett *et al.* 1986), but we also found some nests by searching less accessible habitat on foot. Similar methods were used at Mendota in 1987.

We marked nests, and during weekly checks we candled (Weller 1956) or floated (Westerskov 1950) eggs to determine viability and incubation status. We monitored the nests until they hatched or were otherwise terminated. In the Grasslands, we collected one random egg per nest for chemical analysis from as many nests as possible. To reduce nest abandonment and other biases, we sampled only complete clutches; some were not sampled if they were destroyed before completion. At Mendota, we used this procedure to collect random eggs from shorebird nests and from about 25% of the duck nests. When possible, we collected a second random egg from nests at mid- to late-incubation to examine embryos for viability and deformities.

We used the Mayfield method to estimate nesting success (Mayfield 1961, 1975). We calculated exposure days (Klett *et al.* 1986) and calculated rates of nesting success and cause-specific nesting failure (and associated 95% CI) using the computer program MICROMORT (Heisey and Fuller 1985) following Ohlendorf *et al.* (1989). Nests abandoned as a result of the first nest visit were not included in the analyses.

For nests in which all other eggs hatched, we considered that eggs we collected with live, normal embryos, incubated for at least 13 days (the minimum age at which we could recognize gross external deformities) also would have hatched. Based on recent hatchery work, when an egg reaches the half-way point in incubation, the probability that it will hatch is high (>95%) (J.P. Skorupa, U.S. Fish and Wildlife Service, pers. comm.). Therefore, our assumption that late-stage collected eggs would have been successful imparts minimal bias to our hatchability estimates. We used two-tailed standard-normal tests (z-tests) for between-year and between-location comparisons of nesting success and cause-specific nesting failure rates. These tests are discussed by Hensler and Nichols (1981) and Bart and Robson (1982) in relation to statistical comparisons of estimated nesting success by the Mayfield (1961) method. The probability level for comparisons of nesting success rates was $P < 0.05$. We compared cause-specific failure rates by multiple statistical treatments of the same data.

To assess hatchability, we calculated the ratio of hatched eggs to adjusted clutch size for each successful nesting attempt. Following arcsine transformation, we compared hatchability data with two-factor analysis of variance, and, if necessary, Tukey mean separation tests. We calculated adjusted clutch sizes by subtracting from full clutch sizes all randomly collected eggs and eggs that disappeared from nests prior to predicted hatching dates. We rejected two types of eggs from the analyses of hatchability· those with embryos younger than about 13 days which could not be accurately assessed for normality, and those older than 13 days with uncertain viability.

RESULTS

Nest Abundance

We found 126 duck and 83 shorebird nests in the Grasslands in 1986 and 143 duck and 89 shorebird nests in 1987. We found 46 shorebird nests, and the CWA found 210 duck nests at Mendota in 1987 (Table 1).

We searched the same areas both years and found that the species composition of nesting ducks was different in 1987. The percentage of the total nests that were cinnamon teal (*Anas cyanoptera*), for example, declined from about 40% in 1986 to 13% in 1987. The percentage that were mallard (*A. platyrhynchos*) nests, however, increased from 17% in 1986 to 31% in 1987, while the percentage of gadwall (*A. strepera*) nests increased from 33% in 1986 to 54% in 1987. At Mendota, mallard nests (42%) comprised slightly more of the total than did gadwall nests (35%).

Fewer black-necked stilt (*Himantopus mexicanus*) nests were found in the Grasslands in 1987 than in 1986, but numbers of American avocet (*Recurvirostra americana*) nests were about the same each year. About twice as many killdeer (*Charadrius vociferus*) nests were found in the Grasslands in 1987 as in 1986. At Mendota, stilts (19 nests) and avocets (8 nests) nested only during the short time that a few ponds were flooded. Nineteen killdeer nests were found along Mendota access roads and levees.

Table 1. Fates of nests and percent nesting success (95% CI) of aquatic birds in the South Grasslands (SG) and at the Mendota Wildlife Area (MWA), California, 1986-1987.

	Number of nests							
Species[a] Year-site	Found	Incl.[b]	Hatched	Term. by obsrvr.	Failed	Exposure days	Percent nesting success	95% CI
Mallard								
1986-SG	21	18	7	0	11	239.0	19.2	7.2-50.3
1987-SG	45	40	16	1	23	443.5	15.5	7.2-32.9
1987-MWA	87	83	40	1	42	1184.0	28.2	19.2-41.3
Gadwall								
1986-SG	42	42	28	1	13	528.5	41.8*	26.0-67.0
1987-SG	77	75	29	0	46	942.5	17.4	10.4-28.7
1987-MWA	74	70	29	0	41	975.5	22.2	14.0-35.1
Northern Pintail								
1986-SG	8	8	3	0	5	69.5	9.2	1.1-69.8
1987-SG	2	2	2	0	0	27.0	100	
1987-MWA	14	12	1	1	10	117.0	5.7	0.9-3.2
Cinnamon Teal								
1986-SG	51	45	26	0	19	652.5	36.6*	23.2-57.4
1987-SG	18	16	2	1	13	222.0	12.9	4.1-38.5
1987-MWA	34	31	12	1	18	351.0	16.7	7.2-37.8
Killdeer								
1986-SG	20	20	11	0	9	233.5	32.0	15.0-66.7
1987-SG	42	42	24	0	18	583.0	40.3	26.4-61.1
1987-MWA	19	7	2	0	5	78.5	14.8	2.6-75.4
Black-necked Stilt								
1986-SG	28	28	7	0	21	221.0	8.2	2.8-23.5
1987-SG	10	10	1	0	9	49.5	0.6	0-14.5
1987-MWA	19	15	7	0	8	157.5	27.2**	10.8-66.0
American Avocet								
1986-SG	35	35	12	0	23	379.0	19.6*	10.0-37.9
1987-SG	37	33	4	0	29	256.5	4.4	1.4-13.4
1987-MWA	8	7	3	0	4	71.5	22.4	4.9-93.3

[a] Too few northern shoveler nests found to be included in analyses (4 nests SG-1986; 1 nest SG-1987; 1 nest MWA).

[b] Nests already terminated when found, nests destroyed or abandoned due to searching efforts, and nests of unknown fate were excluded from analyses.

* Within species, success rate significantly higher in 1986 than in 1987 in SG ($P < 0.05$).

** Within species success rate significantly higher at MWA than in SG in 1987 ($P < 0.05$).

Nesting Success

Predation was the primary cause of nest failure both in the Grasslands and at Mendota. With the exception of pintails, at least 50% of the duck nests monitored in this study were destroyed by predators. A second major cause of nesting failure, observed only at Mendota, was embryo death. All embryos were found dead in some

Table 2. Cause-specific failure rates (percentages and 95% CI) of duck nesting from the South Grasslands (SG) and the Mendota Wildlife Area (MWA), California, 1986-1987.[a]

Species Year-Site (No. nests)	Predation	Desertion	Flooding	Cattle	Embryo death
Mallard					
1986-SG	73.4	7.3	0	0	0
(18)	(51.6-95.3)	(0-21.2)			
1987-SG	73.5	7.3	0	3.7	0
(40)	(58.0-89.0)	(0-17.1)		(0-10.7)	
1987-MWA	54.7	0	1.7	0	10.3 *
(83)	(42.3-67.0)		(0-5.0)		(2.5-18.0)
Gadwall					
1986-SG	53.7	4.5	0	0	0
(42)	(33.6-73.8)	(0-13.0)			
1987-SG	77.3	3.6	0	1.8	0
(75)	(67.1-87.4)	(0-8.5)		(0-5.3)	
1987-MWA	62.6	3.8	1.9	0	7.6 *
(70)	(50.1-75.1)	(0-8.9)	(0-5.6)		(0.4-14.7)
Northern Pintail[b]					
1986-SG	36.3	0	18.2	36.3	0
(8)	(0-76.1)		(0-50.2)	(0-76.1)	
1987-MWA	75.4	0	0	0	18.8
(12)	(50.7-100)				(0-42.3)
Cinnamon Teal					
1986-SG	50.0	3.3	0	10.0	0
(45)	(32.6-67.5)	(0-9.8)		(0-20.7)	
1987-SG	73.7	0	6.7	6.7	0
(16)	(52.8-94.7)		(0-19.4)	(0-19.4)	
1987-MWA	69.4	4.6	4.6	0	4.6
(31)	(51.0-87.8)	(0-13.5)	(0-13.5)		(0-13.5)

[a] Vehicles and unknown were other infrequent causes of nesting failure.

[b] Only 2 nests found in the South Grasslands in 1987.

* Failure rates at MWA and SG in 1987 differed ($P < 0.01$).

active mallard, northern pintail (*A. acuta*), gadwall, and cinnamon teal nests, and the failure rate due to embryo death was higher at Mendota than in the Grasslands for both mallards ($P = 0.005$) and gadwalls ($P = 0.019$) (Table 2). Other causes of nesting failure that were important for some species of ducks in one or more years were desertion, flooding, and trampling by cattle.

Predation was also the leading cause of failure of shorebird nests. In the South Grasslands more ($P = 0.005$) stilt nests were lost to predation in 1987 (99.3%) than in 1986 (78.7%). In 1987, fewer stilt nests were destroyed by predators at Mendota (63.7%) than in the Grasslands (99.3%), but more killdeer nests were destroyed at Mendota (85.2%) than in the Grasslands (46.5%). Predation on avocet nests was high

Table 3. Egg hatchability in successful nests in the South Grasslands (SG) and at Mendota Wildlife Area (MWA), California, 1986-1987.

Species		Hatchability	
Year-Site	Nests	$\bar{x}$	SE
Mallard			
1986-SG	7	94.6	5.4
1987-SG	16	96.1*	3.2
1987-MWA	39[a]	79.9	4.6
Gadwall			
1986-SG	28	93.3**	2.2
1987-SG	29	77.5*	5.1
1987-MWA	29	63.7	4.8
Northern Pintail			
1986-SG	3	89.7	5.2
1987-SG	2	83.3	6.7
1987-MWA	1	75.0	
Cinnamon Teal			
1986-SG	26	96.7	1.8
1987-SG	2	83.3	16.7
1987-MWA	12	75.1	8.6

[a] Even though 40 nests were successful, $n = 39$ because total eggs that hatched in one nest was unknown, and hatchability could not be calculated.

* Hatchability higher ($P < 0.05$) than at MWA.

** Hatchability in SG higher ($P < 0.05$) in 1986 than in 1987.

in the Grasslands in both 1986 (80.3%) and 1987 (85.7%) and at Mendota (77.6%). A second major source of mortality for killdeer was motor vehicle traffic, because most nesting occurred along roads where nests were especially vulnerable to destruction by vehicles.

In the Grasslands, the nesting success of gadwalls, cinnamon teals, and avocets was lower ($P < 0.05$) in 1987 than in 1986 (Table 1). The only species which had poorer ($P < 0.05$) nesting success in the Grasslands than at Mendota in 1987 was the black-necked stilt. Too few northern pintail and northern shoveler (*A. clypeata*) nests were found to allow statistical comparisons.

Hatching Success

We found no abnormalities in any late-stage embryos from 79 duck and 32 shorebird eggs collected from monitored nests in the Grasslands in 1986, from 93 duck and 29 shorebird eggs from nests in the Grasslands in 1987, and from 29 duck and 21 shorebird eggs from nests at Mendota in 1987.

In the Grasslands, hatchability of gadwalls, but not mallards or killdeer, was lower ($P = 0.012$) in 1987 than in 1986 (Table 3). Comparisons were not made for other species because too few nesting attempts were successful in one or both years. In 1987, hatchability was higher in the Grasslands than at Mendota for mallards ($P = 0.003$) and gadwalls ($P = 0.004$), the only species with sufficient sample sizes for statistical comparisons (Table 3).

DISCUSSION

In this study, we found that, overall, the most common species of duck nesting in the Grasslands was the gadwall, followed by cinnamon teals and mallards. Mallard, followed by gadwall nests, were most abundant at Mendota in 1987. As in previous studies (Anderson 1956, Gray and Schultze 1977), nests of northern shovelers and northern pintails were rarely found (<3% of all nests).

The greatest observed change in species composition was the decrease in the relative numbers of cinnamon teal nests. Drier conditions during 1987 than during 1986 may have reduced the quality of nesting cover, thus contributing to the observed differences in species compositions between years. In addition, during 1987, ranchers put cattle on the waterfowl hunting clubs earlier than in 1986, and grazing and trampling seemed to reduce the amount and quality of saltgrass (*Distichlis* spp.), a preferred habitat for nesting cinnamon teals.

As in previous studies in the Grasslands (Anderson 1956, Gray and Schultze 1977), predation was the major cause of nesting failure for ducks. In this study, predation rates in the Grasslands were about 75% for mallards both years and for gadwalls in 1987. The increased predation rates (about 50% higher) for gadwalls and cinnamon teals in 1987 may have been related to poorer habitat quality that year.

Predation rates on stilt and avocet nests in the South Grasslands were high both years of this study, but they were especially high for stilts in 1987. Stilts nested almost exclusively on duck-blind islands, and dry conditions in 1987 exposed gravel walkways to these islands earlier in the nesting season than in 1986. Tracks indicated that mammalian predators, primarily coyotes (*Canis latrans*), used these walkways to reach the nests.

Despite the presence of elevated concentrations of Se in eggs from certain drainages in the Grasslands (Hothem and Welsh 1994), deformities and other toxic effects were not seen in avian embryos during either year of our study. Most mean Se concentrations in eggs from the Grasslands were above levels observed at reference sites but below the 13-24 μg/g threshold for teratogenesis suggested by Skorupa and Ohlendorf (1991). However, the presence of elevated concentrations of Se in eggs (Hothem and Welsh 1994) and adult birds (Paveglio *et al.* 1992) suggests that certain areas in the Grasslands were still contaminated with Se during 1986 and 1987, despite the change to uncontaminated water for irrigation and autumn flooding in 1985 (Grassland Water District and Grassland Water Task Force 1986).

Previously reported rates of nesting success of ducks in the Grasslands ranged from 9.4% (Anderson 1956) to 20% (Gray and Schultze 1977). Nesting success in our study was within this range or slightly higher. However, because traditional methods render higher estimates of nesting success than the Mayfield method (Mayfield 1961, Johnson 1979), our estimates were more conservative than the estimates from previous studies.

In 1986, hatchability of duck eggs in the Grasslands (Table 3) was similar to, or better than, the rates in previous studies of waterfowl in California (Anderson 1956, 1957, 1960; Hunt and Naylor 1955; Miller and Collins 1954). Mean hatchability at Mendota, however, did not exceed 80% for any duck species, which was lower than has been reported for most studies of nesting ducks in California. Of 746 monitored duck nests during previous studies in the Grasslands, only one clutch contained all eggs with dead embryos (Anderson 1956). None of the clutches in the Grasslands during our study had 100% dead embryos, but four species of ducks at Mendota had from 4.6 to 18.8% of their clutches in this category. Causes for this reduced hatchability at Mendota are not known, but chemical analyses of eggs indicated that mortality was not related to contamination by arsenic, boron, or selenium (Hothem and Welsh 1994). Further study of waterfowl nesting at Mendota is needed to determine the cause of this mortality.

Our results suggest that relatively high rates of nesting success and egg hatchability may be found even when concentrations of Se are on the lower end of the range believed to cause harm. Fledging success, however, may be affected, and it should be evaluated in future studies of areas with medium to low concentrations of Se.

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ROOSEVELT ELK DIETARY QUALITY IN NORTHERN COASTAL CALIFORNIA

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We investigated crude protein (CP) levels in two herds of Roosevelt elk (*Cervus elaphus roosevelti*) at Prairie Creek Redwoods State Park (PCRSP), Humboldt Co., California, during 1987 in response to concerns that the elk were nutritionally stressed. Both herds showed the same annual pattern of fecal CP levels with highest values in the spring months and lowest values in the fall months. Crude protein levels indicate that neither herd was nutritionally stressed relative to protein nutrition.

INTRODUCTION

Roosevelt elk (*Cervus elaphus roosevelti*) have occupied the area currently within the boundaries of Prairie Creek Redwoods State Park (PCRSP), Humboldt Co., California, continuously since at least 1937 (Harper et al. 1967). Distinct herds of elk have been recognized occupying the Boyes Prairie and Gold Bluffs Beach sections of the park (Franklin and Lieb 1979). Elk within the park are not harvested or manipulated in any way although in the past they have been used as a source for translocations elsewhere in northern California (Harper et al. 1967, D. A. Jessup, California Department of Fish & Game, pers. commun.). Establishment of Redwood National Park immediately adjacent to PCRSP in 1968 further buffered elk from direct manipulation.

California Department of Parks and Recreation personnel became concerned that poor body conformation and low cow:calf ratios (Mandel and Kitchen 1979) were indicative of nutritional stress. We opted to use crude protein (CP) levels in fecal samples as a rapid, nonintrusive means of assessing annual dietary quality of elk. Protein is essential for body maintenance and growth, reproduction, and lactation (Nelson and Leege 1982). Dietary CP levels reflect other indices of dietary quality such as in vitro dry matter digestibility and levels of dietary phosphorous (Leslie and Starkey 1984, 1987, Kie 1988).

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STUDY AREA

Prairie Creek Redwoods State Park lies along the northern California coastline approximately 75 km north of Eureka. Vegetation of the PCRSP is classified as grand fir-sitka spruce forest (Kuchler 1977). Elk herds using the Gold Bluffs Beach and Boyes Prairie portions of the park occupy distinct, nonoverlapping ranges (Franklin and Lieb 1979). The Gold Bluffs Beach herd increased from 15 animals in 1965 (Franklin and Lieb 1979) to 50 by 1979 (Mandel and Kitchen 1979) while the Boyes Prairie herd increased from 25 to 70 over the same time period (Franklin and Lieb 1979, Mandel and Kitchen 1979).

METHODS

About 20 elk fecal pellets were collected from each of five different fresh defecations at each of the two sample areas on a monthly basis from January 1987 through January 1988. Fresh samples were placed in plastic bags and frozen for up to six months until analyzed. Samples from each area were pooled by month through June 1987 for Gold Bluffs Beach and through July 1987 for Prairie Creek but were analyzed individually for both sites thereafter. Fecal nitrogen (FN) content was determined by the Kjeldahl method (Williams 1984). Crude protein was determined by multiplying the percent FN content by 6.25 (Nelson and Leege 1982). Ninety-five percent confidence limits were calculated for the mean of all estimates produced by analyzing individual samples.

RESULTS

The percent CP in both herds showed the same annual pattern with relatively high values (> 19%) in the spring months and relatively low values (< 15%) during the fall months (Fig. 1). The highest (20%) and lowest (9%) values were recorded for the Gold Bluffs Beach herd in April and November, respectively. At no time did the CP level drop to 4 to 7% or less. The only significant difference in fecal CP between herds occurred in November 1987, when the Gold Bluff value was 9% and the Boyes Prairie value was 14%.

DISCUSSION

Fecal CP levels in free-ranging elk reflect changes in forage quality (Leslie et al. 1984), and analysis of fecal CP is an appropriate tool for comparisons of seasonal forage quality between herds utilizing similar habitat types (Leslie and Starkey 1987). Fecal CP levels can be a misleading indicator of dietary quality when values are elevated by high levels of tannins in ingested plants, primarily forbs and shrubs (Hobbs 1987). This concern may be allayed in the case of Roosevelt elk at PCRSP as their diet consists of 58-76% grasses throughout the year (Harper et al. 1967). Differences in estimates of fecal CP levels from individual or composite samples have been found to be minimal (Jenks et al. 1989). The annual pattern of fecal CP levels for both Roosevelt elk herds at PCRSP is comparable to levels for tule elk at Point Reyes National Seashore (PRNS), Marin Co., with spring season peaks of approximately 19% and fall lows of approximately 13% (Gogan and Barrett, unpubl. data). This was achieved on diets of primarily grasses and forbs at both locales (Harper et al. 1967, Gogan 1986). The seasonal CP levels for both elk subspecies in the Mediterranean-like climate of northern coastal California contrast with those for Roosevelt elk in the temperate climate-influenced Olympic National Park (ONP), Washington, where seasonal CP levels drop to as low as 8% in winter and peak at 20% in summer (Leslie and Starkey 1985:Table 1). There, the elk diet consists predominantly of trees and ferns in winter and forbs, grasses, and ferns in summer (Leslie et al. 1984). Crude protein levels for fecal samples from Roosevelt elk at PCRSP reveal no differences in seasonal patterns between elk herds at Gold Bluffs Beach or Prairie Creek, except that the fecal CP level was significantly greater for the Boyes Prairie herd in November, 1987. Neither herd was nutritionally stressed relative to protein nutrition.

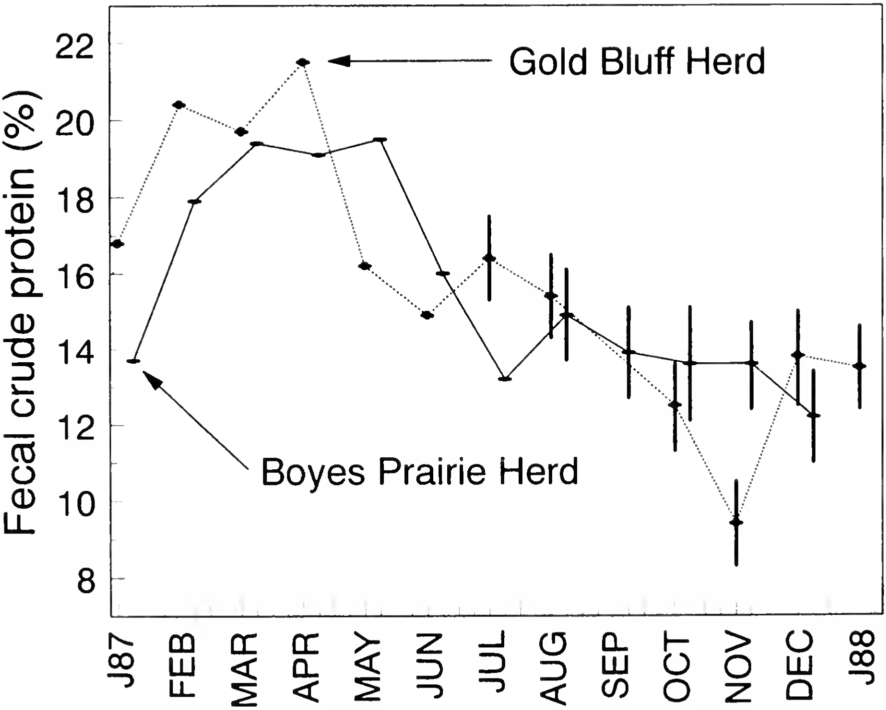

Figure 1. Monthly fecal crude protein (CP) values (6.25 x ppm fecal nitrogen) from two herds of Roosevelt elk at Prairie Creek Redwoods State Park, California. Month (1) is January 1987 and month (13) is January 1988. Vertical lines on some values indicate 95% confidence limits (*n* = 5). The overall mean monthly CP value is indicated by the trend line.

Indeed, lower CP levels have been recorded for tule elk at PRNS (Gogan and Barrett, unpubl. data) and Roosevelt elk at ONP (Leslie et al. 1984). The extent to which elk at PCRSP may be limited by other nutrient requirements remains unknown.

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A PORTABLE FIELD SAMPLING TABLE FOR DOCK-SIDE SAMPLING OF FISH

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In 1977, the California Department of Fish and Game and the National Marine Fisheries Service initiated a joint fisheries monitoring program known as the Cooperative Rockfish Survey. The program involves gathering biological data from commercial fishery landings at unloading sites in ports along the California coast. Data collected consist of species length, sex, and female gonadal condition. Otoliths are removed for age determination analysis.

To accomplish these tasks, the field sampler needs a suitable place to set up sampling equipment consisting of a measuring sheet holder, plastic measuring sheets, otolith trays, knives, scalpels, forceps, plastic bins, and clean up gear. A portable sampling table with wide applicability to field use, meeting the field samplers needs, is discussed.

This portable sampling table is not limited to dock-side sampling of fish, but is adaptable to many situations where a field sampling table is needed. The sampling table may be used on small research boats, rocky shore and beach sites, river banks, skiff launch areas, wharfs, jettys, and piers. It also can be used as a work station for sampling other species such as birds, small animals, and plants. It is ideally suited for on-the-spot examination, measuring, and dissection of most small specimens.

The sampling table was designed with the following criteria in mind: 1) it should be easy to carry to and from sampling sites, 2) quickly and easily set up out of the way of the receiver's operations while still allowing the sampler to view and monitor unloading, 3) it should be stable and self-contained, 4) simple and easy to clean, and 5) easily disassembled and packed up for transport.

The sampling table consists of a "brief case" style box that measures 31-inches x 3½-inches x 15½-inches (Fig. 1). With the exception of the top, all wooden parts of the sampling table are made of Douglas fir. The four sides and bottom are made of ½-inch thick boards. I used boards instead of plywood because I prefer the workability of boards to plywood. The front and right sides (as viewed from the front) are hinged to swing down and out of the way. The top cover is made of Masonite board, ¼ -inch x 30¼-inches x 14¾-inches, and slides into ¼-inch-wide grooves, cut ¼ inch down from the top edge of the left, back, and right sides. A ¾-inch-diameter hole drilled near the front of the top cover facilitates its removal. The bottom boards are butt jointed together. Attached to the top surface are two wooden rails made of ¼-inch x ¾-inch x 30-inch wooden slats (Fig. 2). A routed-out, 1/16-inch x 1/16-inch area the length of each slat forms a lip along one edge of each slat. These face each other creating a holding channel to slide plastic measuring sheets into. A sampling table of this size

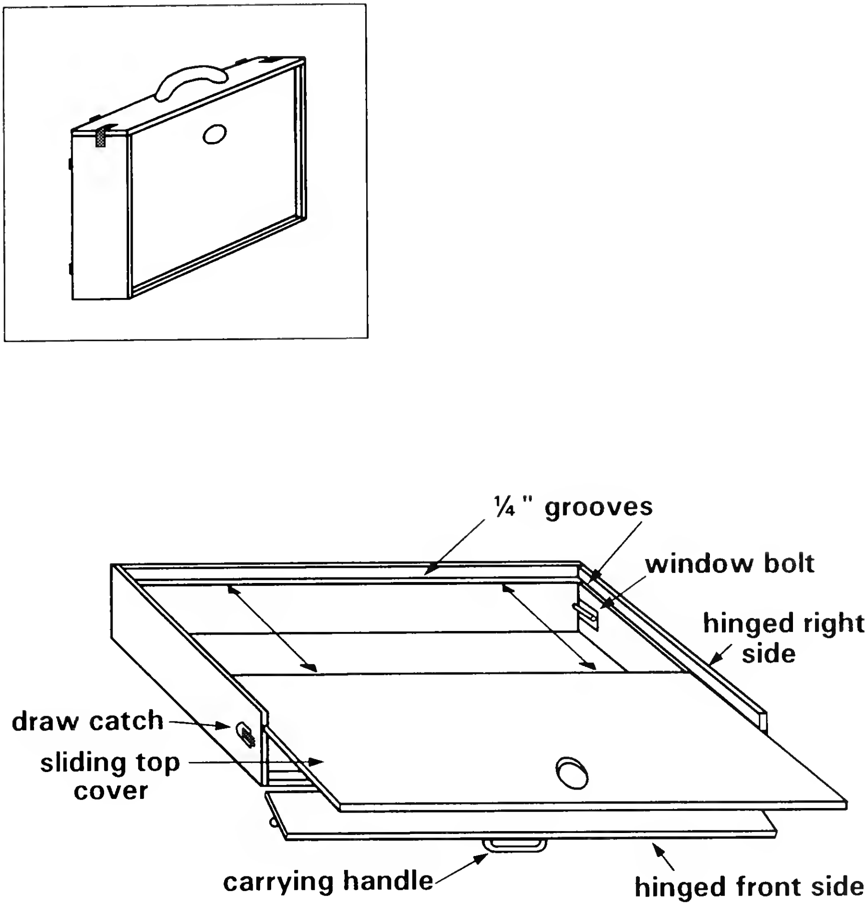

Figure 1. Sampling table in the closed, "brief case" position (insert), and shown partially open.

accommodates plastic measuring sheets approximately 30 inches in length. The distance between the slats is set depending upon the width of the measuring sheets. The slat edges opposite from the lips are bevel cut for appearance and ease of cleaning. Draw catches and hinges attached in appropriate positions allow the front and right sides to swing out of the way or to be locked into place as necessary (Fig. 2). The right side is also locked to the back side with a window bolt. Four 3/8-inch x 5/8-inch wooden slats attached along the edges of the underside of the box create a frame for the scissor-type legs to open up and wedge against. The leg base is made of two sets of 1½-inch x ¾-inch x 35-inch wooden slats (Fig. 2). Each set of legs is fastened together with a bolt, washer, and wing nut through a hole drilled approximately one-third the distance down from the top of the legs. Four 5/8-inch diameter wooden

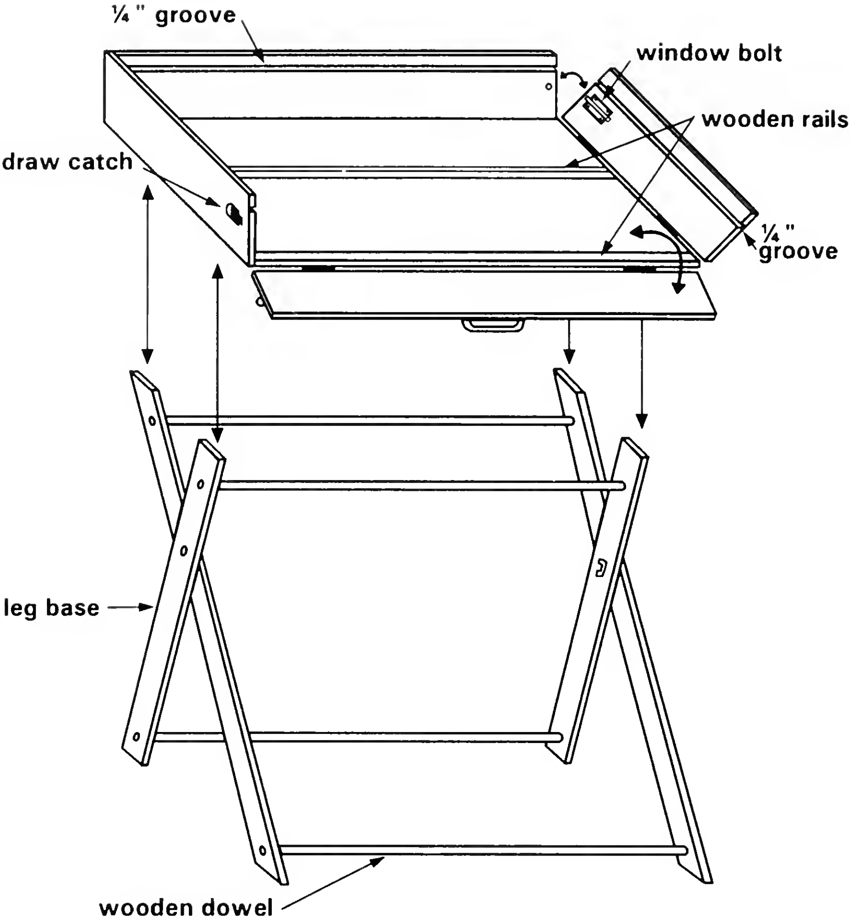

Figure 2. Sampling table in the open position showing its position relative to the leg base.

dowels connect the legs and serve as support cross pieces. Both ends of each leg are angle-cut so that the tops fit under the table and wedge into place, and the bottoms set flat on the ground when the legs are fully opened. Leg length may be varied, depending upon the height of the person using the table. Brass hardware and marine glue were used throughout the unit for assembly. The entire unit was painted with a heavy duty marine paint.

The sampler sets up the unit by first opening the scissor leg base to the approximate width of the underside of the table and loosely locks the legs into position using the wing nuts at the leg junctions. The table is placed on the leg base with the handle facing the sampler. The legs are then adjusted for a snug fit and locked into place using the wing nut arrangement. Unsnapping the draw catches on the left and right sides allows the sampler to swing the front side down and out of the way. The top cover is slid out and placed out of the way, uncovering the sampling equipment stored inside the unit. The sampler now can arrange this equipment to facilitate sampling. Additional plastic measuring sheets are kept inside the unit during storage. A new measuring sheet is installed by opening the right side, inserting the sheet, then closing and locking that side with the window bolt. With the right side and a new measuring sheet locked in place, the sampler has a convenient semi-enclosed table to measure and cut fish. The front side is usually left in the open position, hanging down and out of the way during sampling. Sampling tools are within easy reach and are less apt to be misplaced, dropped, or lost.

When sampling is completed, the entire unit and sampling equipment are washed, brushed and allowed to drip dry for a few minutes. All the equipment is then placed inside the unit and the right side is closed and locked. The top cover is slid into place and the front side is closed and locked. The sampler then lifts the table off the scissor leg base, loosens the leg wing nuts, folds up the legs, and carries the unit away.

The sampling table has been in use for about 2 years. Other than normal wear and tear only one minor problem occurred during that time. The Masonite top became frayed and soft at the corners. This problem could probably be avoided by using a thin sheet of marine plywood instead of Masonite.

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