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FACTORS AFFECTING REGENERATION OF MORRO MANZANITA (*ARCTOSTAPHYLOS MORROENSIS*): REPRODUCTIVE BIOLOGY AND RESPONSE TO PRESCRIBED BURNING

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I) Abstract

Regeneration of the rare Morro manzanita (*Arctostaphylos morroensis*) is dependent on fire. This plant is an obligate seeder; that is, individuals do not resprout after fire, and the species maintains itself solely by reproducing from seed. Because fire in stands of *A. morroensis* will be controlled by land management decisions, it is imperative to consider the fire regime that will best maintain this rare shrub. In order to gain information about the factors affecting regeneration of this species, we conducted studies on its reproductive biology and response to prescribed burning. This report presents results of our studies conducted 1998 - 2000.

We completed studies on the reproductive biology and seed predation of *A. morroensis*. Our main findings were: 1) bumblebees were the most common pollinator, although these were not abundant; 2) *A. morroensis* does not self-pollinate; 3) plants produced an average of 57 flowers and 8 fruits per stem (on stems selected haphazardly), with a mean 18% fruit set; 4) we found no difference in flower or fruit characters among sites; 5) fruit predation was occurring at all 3 sites, and was especially high at the Elfin Forest (77% fruit lost within 3.5 weeks), and 6) the number of seeds added to the seed bank in 1999 ranged from 248 to 1008 per m^2. These studies confirm that a long fire interval is needed for seed to accumulate to the levels present in moderate-aged stands.

To examine regeneration of *A. morroensis* following fire, we conducted studies in a prescribed burn. The burn was carried out in a ten-acre parcel in Montaña de Oro in the fall of 1998. Soil heating during the burn varied considerably throughout the site. This, along with variation in pre-burn seed densities was likely responsible for the differences observed in seedling emergence among sample plots. Seedling recruitment overall was extremely low, and was not adequate to replace the pre-burn adult population. We discuss the importance of a long fire interval to the accumulation of a sufficient soil seed bank that will ensure successful post-fire regeneration.

We compared seed viabilities and germinabilities in litter vs. soil-stored seed to identify the best sources of seed for use in restoration efforts. We found that simulated burning enhanced the germination of both types of seed. The percent of intact seeds that germinated was lower for seeds collected in the litter compared to soil-stored seed, but the litter may be still provide a useful source of seed for restoration projects.

We conclude this report with a preliminary geographic information systems (GIS) analysis of the species distribution. Our results indicate that the present ~350 Ha of maritime chaparral and coastal scrub in which *A. morroensis* occurs are about 1/3 the estimated historical extent.

II) Introduction

Morro manzanita *(Arctostaphylos morroensis)* is a distinctive shrub with a very restricted distribution. It is endemic to Baywood fine sandy loam soil near the southern portion of Morro Bay, San Luis Obispo County (Fig. 1). Here, an area totaling ~350 Ha (865 acres) supports from 1 - 100% cover of the shrub amid maritime chaparral (Fig. 2). We estimate this to be about 1/3 of the historical extent of *A. morroensis*. The reduction in this species' extent has occurred with decades of habitat elimination as a result of suburban growth, which remains the main threat to *A. morroensis*. The expansion of suburbs not only displaces habitat, it fragments populations, constrains management options, and increases the likelihood of accidental or arson fire. Mechanical disturbances and exotic species invasion are also byproducts of increased human presence. Concern about the long-term viability of *A. morroensis* prompted the California Native Plant Society to place this species on its "rare, threatened or endangered" list.

Unlike most shrubs, *A. morroensis* cannot resprout from its root crown. In a previous study (Tyler and Odion 1996), we found very few young *A. morroensis*, and no evidence of seedling establishment in the absence of fire. Dendrochronological evidence indicated that stands of the manzanita were even-aged, dating back to the previous fire. Thus, *A. morroensis* is an obligate-seeder, a relatively uncommon life history type found mostly among shrubs in semi-arid areas in California, Australia, and South Africa (Bond and van Wilgen 1996). Populations of these non-sprouters are maintained by growth of seeds that germinate after the adult generation is consumed by fire. Because obligate-seeders evolved the scheduling of death and reproduction with fire, and because they are phylogenetically-advanced, their life history is considered to be an evolutionary response to fire (Whelan 1995).

Since fire in stands of *A. morroensis* will be controlled by land management decisions, it is imperative to consider what fire regime will best maintain the rare shrub. The obligate-seeding life history can only be successful if the interval between fires is long enough for seeds to accumulate the quantity required to replace the parent generation. The amount needed may be great because seed mortality has been found to be substantial in chaparral burns (Bullock 1982, Zammit and Zedler 1988, Odion and Davis 2000). In addition, most of the individuals that survive and germinate succumb to moisture stress as young seedlings during the long summer drought (Frazer and Davis 1988, Tyler and D'Antonio 1995). Because seed populations may not accumulate to adequate sizes if fire-intervals are relatively short, the obligate seeding life history is considered to be selectively favored by intermediate to long fire-intervals (Keeley and Zedler 1978).

How long does it take for stands of *A. morroensis* to accumulate seed populations of sufficient size to maintain populations through fire? We address this question here with data from our studies on the reproductive biology of *A. morroensis* on seed

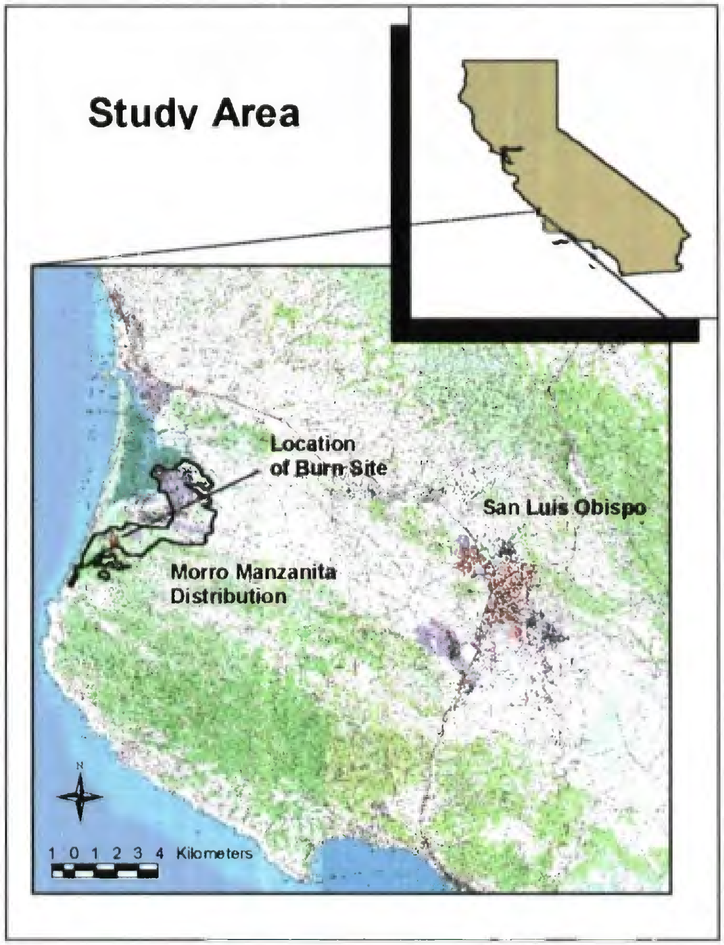

Figure 1. Study area and location of prescribed burn site in Montaña de Oro State Park

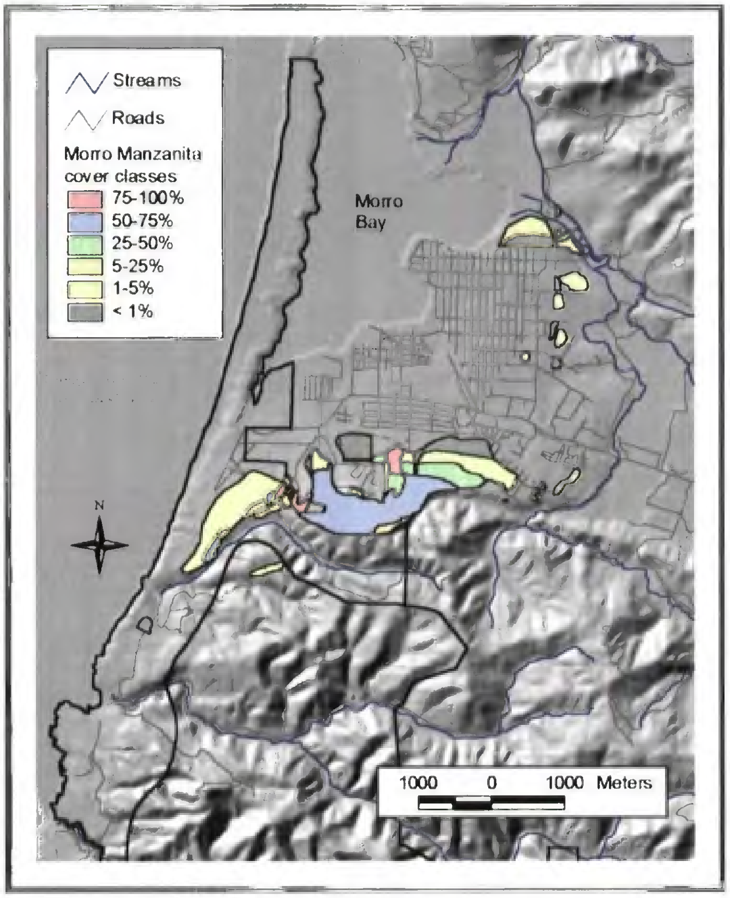

Figure 2. Distribution and cover classes of *A. morroensis*

production, seed predation, and seed bank size as a function of stand age. In studies conducted in a prescribed burn we determined seed and seedling mortality following fire. We evaluate these data in the context of fire management in the landscape around Morro Bay. We also tested whether seed banks from burned stands could be useful in establishing new populations of *A. morroensis*. In addition, we compare seed viabilities and germinabilities in litter vs. soil-stored seed to identify the best sources of seed for use in restoration efforts. Based on previous research (Keeley and Fotheringham 1998), we were curious whether soil-storage increases the potential for seed to germinate in the field.

Strategies for fire management and restoration of *A. morroensis* need to be developed considering many features of the landscape that presently and historically supported this species. Thus, we conclude this report with a preliminary geographic information systems (GIS) analysis of the species distribution, and we provide recommendations for additional GIS analyses that would be beneficial to conduct in the future.

III) Reproductive biology of *A. morroensis*

The goal of our studies on the reproductive biology of *A. morroenis* was to examine its reproductive output and rates of seed accumulation, and to determine the factors that influence recruitment of individuals in extant populations and that contribute to differences in reproductive output among stands. These studies were conducted in the spring and summer of 1998 and 1999. Detailed results for 1998 are provided in Tyler, Odion, and Meade (1998). We examined pollinators, self-fertilization, fruit set, seed set, numbers of mature fruits that reach the soil, and the effect of predation on fruit survival.

Study sites

We examined aspects of reproductive biology in three sites. The first site was the Elfin Forest, north of Santa Ysabel Ave. and west of South Bay Blvd. This area is a mix of maritime chaparral, coastal scrub, and oaks (*Quercus agrifolia*). The percent cover of *A. morroensis* is low, ranging from <1% to 5-25%. The manzanita stands at this site are older than 47 years, and some individual manzanitas appear to be very old. Previous studies (Tyler and Odion 1996) found that the seeds in the soil seed bank here had particularly low seed densities and low viabilities, suggesting that potential reproductive stress is occurring in this stand.

The second site was the dunes area west of Pecho Valley road in Montaña de Oro State Park, where the stand age was estimated at 40 years in 1999. This site is a mix of maritime chaparral and coastal scrub. The percent cover of *A. morroensis* in this site (according to Mullany 1990) ranged from 25-50% to 75-100%. This site was adjacent to (south of) the prescribed burn described later in this report.

The third site was north of Hazard Canyon, east of Pecho Valley Road in Montaña de Oro State Park. This site is a mix of *Quercus agrifolia* and dense *A. morroensis*. The percent cover of *A. morroensis* ranged from 50-75% to 75-100%. This stand of manzanita is intermediate in age between sites 1 and 2. It is located relatively close to site 2 (<400 m), but upslope and farther from the ocean.

Pollinators

Pollinators were observed during February, March, April, and May 1999. Seventeen different observation periods were conducted on six different days for a total of 890 minutes. Counts of pollinators visiting *A. morroensis* flowers were recorded for each site.

Similar to our observations in the previous year, during 1999 the most abundant pollinators observed visiting *A. morroensis* flowers were yellow-faced bumblebees (*Bombus vosnesenskii*). The only other bee observed was the common Anthoporid bee, *Anthophora urbana*. One Monarch butterfly (*Danus plexxipus*), and several Bee flies (family Bombyliidae, *Bombylius* sp.) were observed feeding on *A. morroensis* flowers (Table 1). No flower flies (Syrphidae) were observed in 1999 on *A. morroensis* flowers. As in 1998, the most striking observation in 1999 was the absence of abundant pollinators, even on warm, sunny days.

Self-fertilization

To determine whether *A. morroensis* is capable of self-fertilization, inflorescences were bagged to exclude all pollinators. Open flowers were removed from the inflorescence prior to bagging and closed buds were counted. Bags were left in place until all of the flowers within the bag had senesced. The stems were then examined for fruit.

A total of 226 flowers on 15 plants were bagged to exclude cross pollination. Inflorescences were covered beginning on February 24, 1999, until all of their flowers dropped their corolla. None of the bagged flowers set fruit. This result supports our conclusion from 1998 that *A. morroensis* can not self pollinate. In our 1998 study, 8 out of 1243 flowers bagged did set fruit. We hypothesized that these 8 flowers had likely already opened and been pollinated before being bagged. Thus, during this year's experiment flowers were selected at an earlier developmental stage to insure that all of the bagged flowers did not receive pollen before being enclosed. The result of zero fruit set supports our suggestion that the fruit set in the 1998 trial resulted from including flowers that had already received pollen.

Table 1. Pollinators observed during 1999.

Location	Observation period (minutes)	Date	Species and number observed
Elfin	120	2/5/99	0
Hazard Cyn.	40	2/5/99	*Bombus vosnesenskii* – 1
Dune	40	2/5/99	*Bombus vosnesenskii* – 1
Dune	35	2/20/99	0
Dune	15	2/24/99	*Bombus vosnesenskii* – 1
Elfin	120	3/23/99	*Anthophora urbana* – 4 *Bombylius* sp. – 3
Dune	90	3/23/99	0
Hazard Cyn.	25	3/23/99	0
Hazard Cyn.	80	4/14/99	*Bombus vosnesenskii* – 1
Hazard Cyn.	15	4/14/99	*Bombus californicus* – 1 *Danaus plexxipus* – 1 *Bombus* sp. - 1
Dune	45	4/14/99	0
Elfin	60	4/14/99	0
Hazard Cyn.	5	4/14/99	0
Hazard Cyn.	5	4/14/99	0
Hazard Cyn.	90	5/28/99	0
Dune	60	5/28/99	0
Elfin	45	5/28/99	*Bombus vosnesenskii* – 1
Total	890		

Fruit set

During 1999, to examine fruit set, ten plants were selected haphazardly within each site and ten stems were selected and banded for identification on each plant. Stems were selected that were new in bud, and these were followed weekly to determine the full count of flowers produced by the stem from February through April. Buds were marked or noted and new buds were added to the total when they occurred. Fruits were counted on stems from March through September. The counts of flowers and fruit on the ten stems on each plant were used to determine a mean value for flower number and fruit number per stem for each plant. The ratio of fruit number to flower number was calculated to determine the average fruit set per plant. We used one-way analysis of variance (ANOVA) to detect statistically significant differences in flower production, fruit production and fruit set among study populations.

Results

The mean number of flowers produced per stem per plant ranged from 50 to 65 (Table 2). We detected no statistically significant differences among the sites (ANOVA: P = 0.567, n = 30). The number of flowers per stem produced in 1999 was roughly half that produced in the previous year (range 80 – 135).

Table 2. Mean number of flowers per stem on ten plants at each study site, and results of analysis of variance to detect differences among sites.

Means Table for mean no. of flowers per stem, per plant
Effect: location

	Count	Mean	Std. Dev.	Std. Err.
Elfin forest	10	54.2	29.6	9.4
Dune	10	65.7	36.1	11.4
Hazard Cyn	10	51.8	26.1	8.3

ANOVA Table for mean flowers per stem

	DF	Sum of Squares	Mean Square	F-Value	P-Value
location	2	1106.3	553.1	0.580	0.567
Residual	27	25753.0	953.8		

Fruit production averaged between 5 - 10 fruits per stem (Table 3). Highest numbers of fruits produced per stem were found in the Hazard Canyon population, but this difference was not statistically significant (Table 3).

Table 3. Mean number of fruits per stem on ten plants at each study site, and results of analysis of variance to detect differences among sites.

Means Table for mean no. fruit per stem per plant

Effect: location

	Count	Mean	Std. Dev.	Std. Err.
Elfin forest	10	5.2	3.8	1.2
Dune	10	9.3	7.8	2.5
Hazard Cyn	10	10.4	7.9	2.5

ANOVA Table for mean no. fruit per stem per plant

	DF	Sum of Squares	Mean Square	F-Value	P-Value
location	2	149.3	74.7	1.615	0.218
Residual	27	1248.2	46.2		

Fruit set over all study populations averaged 18% (Fig. 3). The Hazard Canyon site had twice the fruit set of the Elfin Forest site, although this difference was not statistically significant (ANOVA: P = 0.102, n = 30). Interestingly, the patterns observed differed between the two years. Overall, fruit set was almost twice as high in 1999 compared to 1998, even though flower production was lower in 1999. Also, the pattern of fruit set of the different sites varied between the years: Hazard Canyon plants had lowest fruit set in 1998, but the highest in 1999. Given that *A. morroensis* is dependent on insect pollinators, our results indicate that the abundance or activity of pollinating bees varies among sites and years, and that one site is not consistently more pollinator-limited than another.

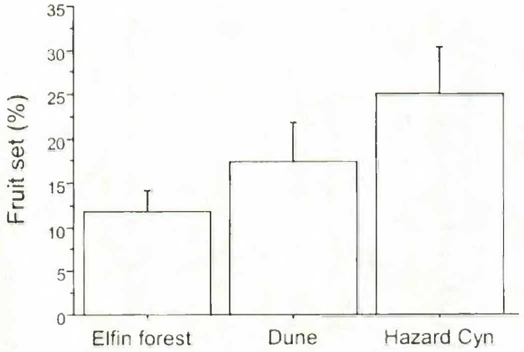

Figure 3. Differences in fruit set among study populations. Data are means of 10 plants per site, + 1 s.e.

Estimate of mature fruit reaching the soil

The average number of fruits falling from a plant was estimated by placing trays under each of the six plants at each site (total number of trays placed in the field = 70). When possible, four trays were placed under the outer canopy edge of each plant in each of the compass quadrants. Trays were 25 cm x 25 cm. They were constructed of wood frames, wire screen bottoms, and hardware cloth tops that were slightly v-shaped; the top was contoured in this way so that fruits hitting the cage would roll toward the center and fall into the tray. Thus, the trays excluded rodents from the fruit once the fruit entered the tray. Fruit falling into the trays was used to estimate the number of fruits that reached soil per area, under each study plant. Fruits in the trays were counted and emptied throughout the spring and summer until no fruit remained on the plant.

The majority of fruits fell from the plants between August and early October (Figure 4). Of the total 808 fruits collected at all sites (as of October 12, 1999), only 369, or 46%, were collected by August 12th. Fruit drop occurred much later than the previous year, when the majority of fruits fell before July.

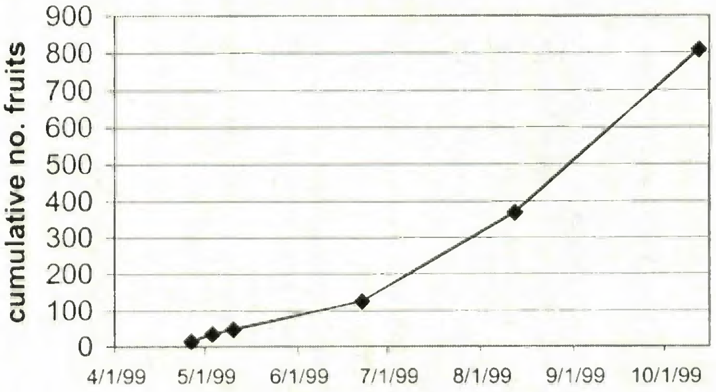

Figure 4. Cumulative number of fruits collected in trays (April - October 1999). Data are total numbers for all sites, all trays (70) combined. Total ground area covered = 4.4 m^2.

Combining all sites, shrubs dropped an average of 12 fruits per tray (area = 0.0625 m^2), or 185 per 1 m^2. There was no significant difference in the number of fruits reaching the ground among sites, although the Hazard Canyon site, above the road in Montaña de Oro had the highest number of fruits (Table 4). This is not surprising, given that this site also had the greatest number of fruits produced per stem (Table 3). In contrast, in the previous year this site had the lowest number of fruits produced and dropped.

Table 4. Mean number of fruits dropped, and results of analysis of variance to detect differences among sites.

Means Table for Total fruit dropped per tray (25 cm x 25 cm)
Effect: site

	Count	Mean	Std. Dev.	Std. Err.
Elfin Forest	24	8.5	14.3	3.0
Dune	24	9.9	12.1	2.5
Hazard Cyn	24	15.3	13.3	2.7

ANOVA Table for Total fruit dropped per tray (25 cm x 25 cm)

	DF	Sum of Squares	Mean Square	F- Value	P-Value
site	2	618.9	309.4	1.750	0.181
Residual	69	12201.6	176.8		

In 1998 we found that there was a significant difference in the amount of fruit that reached the ground on different sides of the plant. The northeast half of the plants had significantly fewer fruits fall than did the southwest half of the plants. We examined this pattern in 1999, but found no differences among the quadrants (Figure 5).

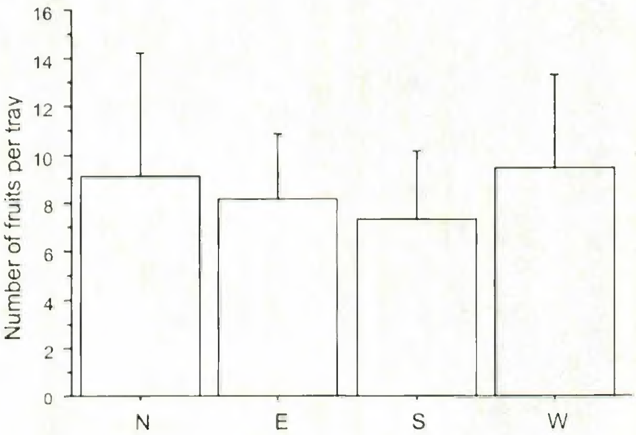

Figure 5. Numbers of fruit that reached the ground from plants at all three study sites when divided by the side of the plant according to compass direction. Data are means (+ 1 s.e.) of total numbers of fruits per tray (25cm x 25cm), all sites combined.

Fruit and seed predation

The effect of vertebrate predators on fruit survival was examined by experiment. We used 3 treatments to determine the rate of fruit removal from trays in which known numbers of fruit were placed. These treatments were: a) under the plant canopy and screened to prevent vertebrate predator access, b) under the plant canopy and open to predators, and c) away from the plant canopy (1 - 3 m from canopy edge) and open to predators. The first treatment served as a control to determine whether fruits were lost due to a factor other than vertebrate predation. The trays were 10 cm by 20 cm and 3 cm high, made of a wood frame with a screen bottom, and those that prevented predators had hardware mesh tops. Ten mature fruits were placed in each tray; trays were placed in association with each of the ten study plants at each of the three sites. The number of fruit and seeds remaining were counted at intervals for four weeks. We looked for differences among treatments in the number of fruits remaining to determine whether vertebrate predators affect the number of fruits that contribute to the seed bank, and whether predation is more intense beneath the cover of a plant, or outside of the cover of the plant.

Results

We found that predators removed a significant number of fruits over the period of the study (Table 5). Combining all sites, after only 26 days (20 days sooner than in 1998), there were an average of 4 fruits remaining per tray in both open treatments (i.e., predators present) compared to 10 fruits in the screened controls (ANOVA, $P < 0.001$; posthoc Scheffe test indicate that both open treatments were significantly lower than the control, but not different from each other). This difference between open and the screened control treatments was significant as early as 5 days after the initiation of the study (ANOVA, $P<0.01$). No loss of fruit in the screened control trays verifies that the seed loss in the other treatments was in fact due to active seed removal by vertebrates.

Table 5. Mean number of fruits remaining (out of 10) in trays after 26 days in three experimental treatments.

Means Table for Fruit remaining out of 10 - 5/10/99
Effect: Treatment

	Count	Mean	Std. Dev.	Std. Err.
Open/under plant	30	3.2	3.3	0.6
Open/ not under plant	29	3.3	3.6	0.7
Screened/under plant	30	9.7	2.1	0.4

The finding that open treatments did not differ from each other indicates that the seed predators are not confined to the shrub canopy, and are able to remove fruits that fall 1 to 3 m away from the canopy edge.

We conducted two-way analyses of variance to examine both the effects of site and treatment on fruit predation after 26 days (Figure 6, Table 6). We found a significant treatment effect, as discussed above. There was also a significant difference among sites. The number of fruit lost to predation was significantly higher at the Elfin forest and Hazard Canyon sites (predation rates = 77% and 68%, respectively) than at the Dune site (predation rate = 20%). There was no significant difference between the Elfin forest and Hazard Canyon sites. Predation rates at the Dune site were much higher in the previous year (47%) than in 1999. This site is located below and adjacent to the prescribed burn area, so one possible explanation for the low predation rates in 1999 is that seed and fruit predators such as woodrats and brush rabbits may have been less abundant in the area following the burn. We found no location by site interaction indicating that patterns of seed predation were similar among all sites.

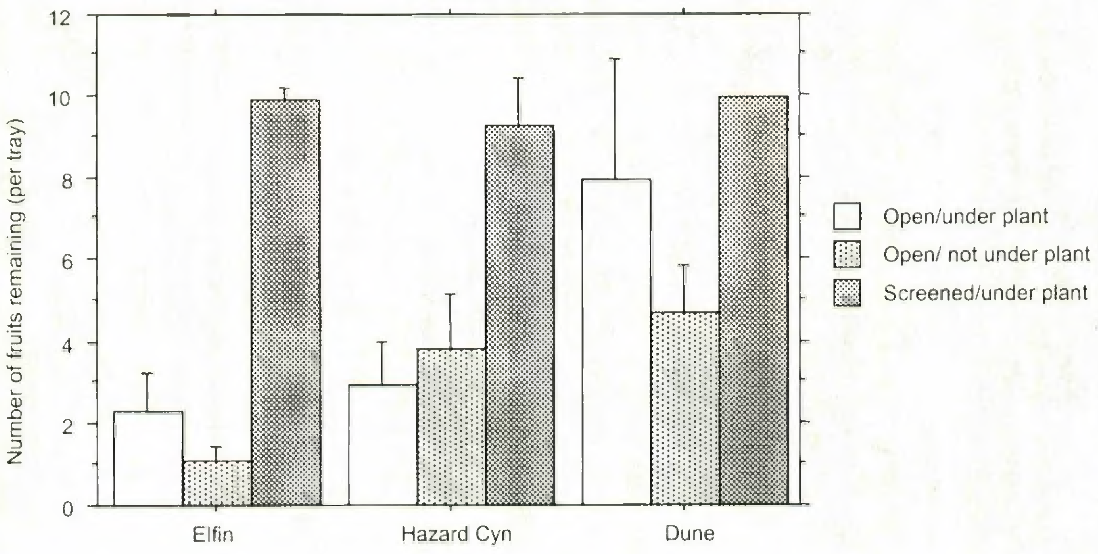

Figure 6. Numbers of fruit remaining (out of 10) in open and screened control trays at 3 sites. Data are means (+ 1 s.e.) on May 10, 1999 (26 days after study initiation.)

Table 6. Results of analysis of variance to detect differences among sites and treatments in fruits remaining after 26 days.

ANOVA Table for Fruit remaining 5/10/99

	DF	Sum of Squares	Mean Square	F-Value	P-Value
Treatment	2	717.362	358.681	21.099	<.0001
Site	2	151.950	75.975	4.469	.0145
Treatment * Site	4	109.669	27.417	1.613	.1791
Residual	80	1359.977	17.000		

One case was omitted due to missing values.

By measuring reproductive characters and rates of seed removal over two years, we found that flower production, fruit set, and predation rates varied considerably among sites and years. Thus, addition of seed in the soil seed bank also varied. We used our data on fruit production, fruit predation, and previous data on numbers of seeds per fruit (Tyler and Odion 1996) to estimate the 1999 contribution to the soil seed bank (Table 9). This estimate was derived with the following formula: number of fruits per m^2 minus loss due to predation times number of seeds per fruit. We calculated that the number of seeds added to the Elfin Forest was lowest (248 seeds/m^2), moderate at the Hazard Canyon site (624 seeds/m^2), and highest at the Dune site (1008 seeds/m^2). This relative order was the same in 1998, but all sites had at least 20% fewer seeds added to the seed bank in 1999 than in the previous year.

Table 7. Estimates of numbers of seeds added to the seed bank in 1999 at three study sites. All data from the present report, except # seeds per fruit, and total # seeds in seed bank; the latter are from Tyler and Odion 1996.

site	# fruits per m^2	% loss due to predation	# fruits per m^2 remaining	# seeds per fruit	# seeds per m^2 added in 1999	total # seeds per m^2 in seed bank	# yrs to accumulate present seed bank
Elfin Forest	136	77	31	8	248	18,131	73
Dune	158	20	126	8	1008	15,224	15
Hazard Cyn	245	68	78	8	624	42,569	68

Our estimates for the number of years required for seeds to accumulate the present seed bank (given this year's fruit production and loss rates) are likely underestimates. Seed predation rates are conservative given that they were based on a short time period (26 days). These findings corroborate our conclusions from 1998 that a long fire interval (i.e., greater than 60 years) is needed for seed to accumulate to the levels of moderately aged stands.

IV) Prescribed Burn

Understanding the response of *A. morroensis* to fire is of critical importance to managing the species, because stands will eventually burn, either during wild or prescribed fires. Prescribed (i.e., controlled, intentional) burns could be deemed necessary to initiate seedling regeneration, diversify stand ages, or to reduce fire hazards at the urban interface. Although laboratory studies on the response of seeds to simulated burning may provide useful information to predict patterns of postfire seedling establishment, it is imperative to test these predictions in the field. In order to determine whether post-fire seed germination and subsequent seedling survival of *A. morroensis* is sufficient to maintain populations of this species, we studied the fate of seeds and seedlings in an area where a prescribed burn was conducted.

Site description

The burn was conducted in a ten-acre parcel located east of Pecho Valley Road in Montaña de Oro State Park (Figure 1, Photos 1, 2). The vegetation at the site was approximately 40% *A. morroensis*, 5% *Q. agrifolia* and the remainder a mixture of open chaparral dominated, by *Ceanothus cuneatus* and chamise (*Adenostoma fasciculatum*), coastal sage scrub, and veldt grass (*Ehrharta calycina*) grassland. The area was previously cleared and burned in 1958 by the U. S. Army Corps of Engineers during ordnance removal (Tyler and Odion 1996). Thus, based on air photos, age of the *A. morroensis* stand was 40 years at the time of the burn in 1998. We also confirmed stand age by analyzing rings on cross-sections of shrubs cut to enlarge a fuelbreak.

This site, which was identified with the help of State Park Ecologist, Vince Cicero, was ideal for our studies for several reasons. It included pure stands of *A. morroensis* that appeared very representative in size, and density, and we had previously studied seed banks in these stands. From a logistical perspective the site was relatively safe and easy to burn. The parcel also contained areas degraded by veldt grass invasion; this provided an opportunity to test the efficacy of adding soil-stored seed from burned chaparral as a procedure for restoring such degraded sites which are common in the area. Finally, the chaparral in the burn area was the youngest age class found in the range of *A. morroensis* (~40 years). If seed populations in this stand proved adequate to ensure stand

Photo 1. Prescribed burn site in Montaña de Oro State Park prior to the burn. View looking to north. Photo from 1996.

Photo 2. Prescribed burn site in Montaña de Oro State Park prior to the burn. View looking to south. Photo from 1996.

Photo 3. Prescribed burn, Montaña de Oro State Park, November 4, 1998. View looking to south.

replacement of *A. morroensis*, then remaining stands should not be at risk due to their "immaturity" (Zedler 1995).

Preburn vegetation composition

Prior to the burn we established 24 permanent monitoring plots. These were marked with rebar to facilitate locating them following the burn. The plots were established within the proposed burn boundaries in: a) dense monospecific stands of *A. morroensis*, b) under isolated *A. morroensis* individuals, and c) in adjacent vegetation. We conducted preburn vegetation sampling on September 30, 1998, recording the percent cover of all species present in 12.6 m^2 circular plots (2 m radius around center stake).

Table 8. Preburn vegetation cover in permanent monitoring plots (n = 24).

species	mean % cover
Arctostaphylos morroensis	57
Ehrharta calycina	13
Ceanothus cuneatus	7
Lotus scoparius	4
Artemisia californica	3
Ericameria ericoides	2
Quercus agrifolia	2
Baccharis pilularis	1
Adenostoma fasciculatum	1
Eriogonum parvifolium	1
Croton californicus	1
Mimulus aurantiacus	<1
Dudleya lanceolata	<1
Salvia mellifera	<1
Lessingia filaginifolia	<1
Horkelia cuneata	<1
Eriophyllum confertiflorum	<1
Cardionema ramosissimum	<1
Quercus agrifolia seedling	<1
Gnaphalium bicolor	<1
Carpobrotus spp.	<1
Bromus madritensis rubens	<1
bare ground	8

On average, plots had over 50% cover of *A. morroensis* (Table 8.) The other shrub that was relatively common was *Ceanothus cuneatus*, also an obligate seeder. Veldt grass, *Ehrharta calycina*, was also relatively abundant.

Burn characteristics

Light rainfall (approximately 0.5") occurred 10 days prior to the burn on October 25, 1998. We did not measure fuel moisture, but it was undoubtedly near its seasonal low as chaparral shrubs typically do not recover moisture levels with small amounts of rainfall after the long summer drought (Pyne 1984). In addition, soils under *A. morroensis* probably remained dry because the deep litter layer would have absorbed virtually all the moisture. This is significant because if the soils were moistened prior to fire, soil-stored seeds of *A. morroensis* could have imbibed water and lost their heat tolerance (Sweeney 1956). Our previous germination study found significantly reduced germination of seed that had been soaked for 24 hours prior to simulated burning (Tyler, Odion, and Meade 1998).

The prescribed burn was conducted November 4, 1998. California Department of Forestry and Fire Protection, in cooperation with State Parks, conducted the burn. Ignition occurred at approximately 10 a.m., and the burn was concluded by about 2 p.m. Although coastal fog had cleared at the start of the burn, the sky remained somewhat hazy, and fog persisted over the nearby ocean. The temperature remained about 60^0 F through the duration of the burn, while relative humidities ranged from 50-60%. Although fire behavior was subdued by these mild atmospheric conditions, there were still flame lengths as great as ~50' in the tall, dense patch of *A. morroensis* in the burn area (Photo 3.). However, only the vegetated areas where woody fuels were contiguous carried fire. When flames reached areas lacking shrubs, or where the shrub canopy was sparse, combustion ceased. Where flaming combustion was intense, it was followed by smoldering combustion of deep layers of litter and other oxygen-deprived fuels, particularly where *A. morroensis* had occurred. This continued for several days in large fuel particles, gradually decreasing over time. However, we observed no collapse and combustion of *A. morroensis* canopy on the ground, as has been observed in mixed chaparral (Craddock 1929), and chamise chaparral (Odion and Davis 2000). Twenty-four hours after the burn, soils were still warm to hot where there had been smoldering litter at the surface (i.e., most of the area where *A. morroensis* had been). In some spots, litter was still smoldering at this time. Based on our observations, as well as previous research (Hartford and Fransden 1992), it appears most of the soil heating that occurred resulted from the prolonged smoldering of the deep litter under *A. morroensis*. Following the burn, portions of *A. morroensis* bigger than~1/2"-1" in diameter remained, so fuel consumed was primarily leaves, twigs, and litter. Burned shrub skeletons with interlocking branches persisted where dense patches of *A. morroenis* had occurred.

Estimates of burn severity and soil heating

To assess soil heating during the burn, we set out "water cans" at each permanent plot just before the fire (i.e., the morning of the burn). Standard one-quart paint cans were filled with 800 ml of deionized water; a 1 cm hole was drilled in the top/lid of the can, and tape placed over the hole. Relative soil heating with fire is determined by amount of water loss from the cans (Moreno and Oechel 1989, Odion and Davis 2000). Three cans per plot were placed 80 cm from the center stake. The morning after the fire, we collected the cans. In the laboratory we measured and recorded the amount of water remaining in each.

After the burn, at each plot we also described the fire severity qualitatively, employing the descriptions used in Ryan and Noste (1985). The ratings used were: unburned, light, scorched, moderate, and heavy. We also derived a burn severity rating by assigning each class of burn severity a value (i.e., unburned = 1, light = 2, scorched = 3, moderate = 4, and heavy = 5) and adding the proportion of each found in the plots. For example, a plot that was 50% unburned, and 50% moderate, received a rating of 2.5 (0.5 x 1 + 0.5 x 4); one that was 100% unburned was 1, and 100% heavily burned was 5.

Results

Soil heating, as measured by water loss from cans, varied considerably among plots. The amount of water lost per can ranged from 0 to 105 ml, with an average of 20 ml lost. Burn severity varied from 100% unburned to 100% heavily burned. As expected there was a positive relationship between water lost in cans and burn severity rating (Figure 7). We found that measurable soil heating (i.e., water loss) occurred for the most part in plots that were 100% heavily burned. Yet, within the severity rating of "100%

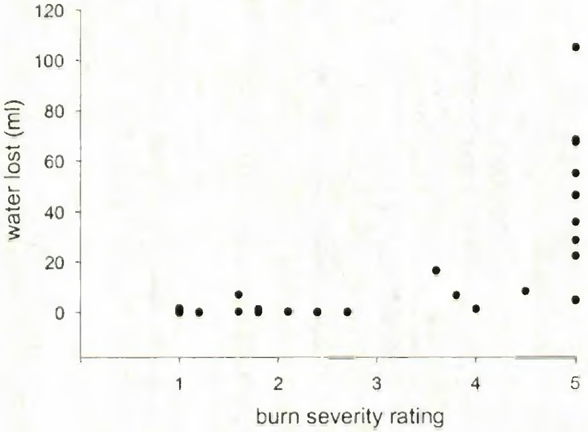

Figure 7. Relationship of burn severity rating (1 = 100% unburned, 5 = 100% heavily burned) to soil heating, as estimated by water lost from cans placed in the burn.

heavily burned," there was variation in water loss, suggesting that our measure of soil heating may be a more sensitive indication of fire effects on soil seed banks than the qualitative descriptions.

We also examined the relationship between preburn vegetation (% cover) and soil heating during the burn, as estimated by water lost from cans. Soil heating was detected primarily in plots that had 100% cover of vegetation prior to the burn (Figure 8).

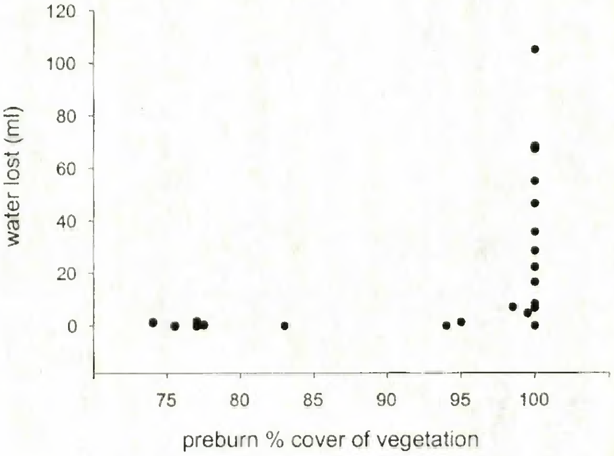

Figure 8. Relationship of preburn vegetation cover to soil heating during the burn, as measured by water lost from cans.

Pre- and post-burn seed bank samples: seed viability and germinability

We collected preburn seed bank samples in eight plots with manzanita present. At each of these plots, we collected 4 cores (corer = 7.1 cm in diameter, 10.3 cm deep). We kept separate the top and bottom halves of these cores, to determine the relative seed densities and viabilities at these 2 different soil depths. One week following the burn we collected postburn seed bank samples from the same plots.

In the laboratory, postburn samples were sifted, and seeds manually separated from the soil. Seeds were placed on moist filter paper in growth chambers, kept at a constant temperature of 20°C, and monitored for germination. This study was terminated after 95 days.

After completing the germination trial, we assessed viability of these postburn seed samples, and viability of preburn seed bank samples. We counted and cut open all seeds and examined the chamber for viable embryos using a dissecting microscope.

Embryos that were white and fleshy were counted as viable. Chambers lacking these were either completely empty, or contained dried residue of embryonic tissue, so viability was not underestimated. In our previous study (Tyler, Odion, and Meade 1998), seeds that were subjected to a fire treatment had a germination percentage equal to the percent viability measured by this method. This suggests that all white and fleshy embryos were in fact viable, and that our methods do not overestimate viability.

Results

In previous germination trials with *A. morroensis* we found that some seeds germinated within 28 days, and most seeds that were viable germinated within 36 days. Although we monitored germination trials with postburn seeds for a considerable length of time, no seeds germinated.

The total number of preburn seeds cut open was 1292. There were 39 viable seeds. The total number of postburn seeds cut open was 1041; surprisingly, although none of the seeds germinated, 12 were viable.

The number of viable seeds per sample varied in preburn samples from 1 – 11 in the top sections and 0 – 3 in the bottom; in postburn samples viable seed number ranged from 0 – 2 in the top and 0 – 1 in the bottom (Table 9). In a previous study (Tyler and Odion 1996) we found that the majority of total and viable seed was in the upper 5 cm of soil. However, the number of viable seeds was greatly reduced following fire in the top samples (Table 9, Figure 9). Number of viable seeds in bottom samples was the same in pre- and postburn samples.

Table 9. Results of seed bank sampling in the prescribed burn area. Given are mean values (+ 1 s.e.) per sample (n = 8) of total number of seeds, number of viable seeds, and percent viability in samples from the top 5 cm of soil and lower 5 cm of soil. Volume of each sample was 800 cm^3 (158 cm^2 x 5 cm.) Also given is the estimated number of viable seeds in $2m^2$ - the area in which seedling surveys were done.

	#seeds top	#viable top	%viable top	#seeds bottom	#viable bottom	%viable bottom	total # viable seeds	est. total # viable seeds per 2 m sq
preburn	120.5	3.5	4.3	41.0	1.4	3.7	4.9	616
	(27.1)	(1.2)	(1.6)	(10.0)	(0.5)	(1.5)	(1.3)	(164)
postburn	113.4	0.9	1.1	16.8	0.6	3.9	1.5	189
	(16.7)	(0.2)	(0.4)	(3.9)	(0.2)	(1.4)	(0.3)	(41)

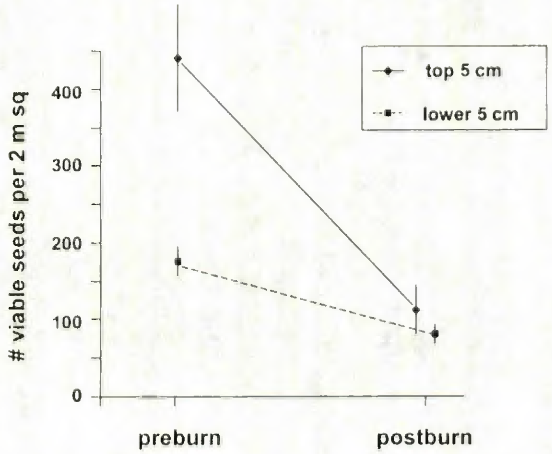

Figure 9. Change in viable seed number in top and lower soil samples following the burn. Values are means ± 1 s.e. (n = 8), estimated for 2 m^2 (based on 158 cm^2 samples).

Although the reduction in viable seed populations may have been a result of seed predation occurring between the collections of pre- and post-burn samples, the predominant cause was probably heat-induced mortality of seeds. We did not measure seed mortality directly in this burn (i.e., identify individual seeds that were killed). However, we found a significant negative relationship between soil heating and postburn seed viability in the top seed bank samples (Figure 10.). Based on previous research, even the most heat tolerant chaparral seeds would not have been able to withstand the maximum temperatures that would have occurred at the soil surface and deeper, where there is prolonged combustion of litter (reviewed by Borchert and Odion 1995). Even though seeds of many chaparral plants have remarkable heat tolerance, a temperature of 170°C for 5 minutes is generally lethal. The depth of lethal heat would have penetrated deeper in the soil where total surface heating, and thus water loss from cans, were also greater, accounting for the negative relationship with numbers of viable seed. However, lethal heat apparently did not penetrate to the depth of the lower samples (~5 cm), as the same relationship with soil heating was not found. This is because soil is such an effective insulator. In a previous study in similarly sandy soils, Odion and Davis (2000) found that temperatures at 5 cm only reached about 75°C during a fire where collapsed canopy burned on the soil surface, heating thermocouples there to 900-1000°C.

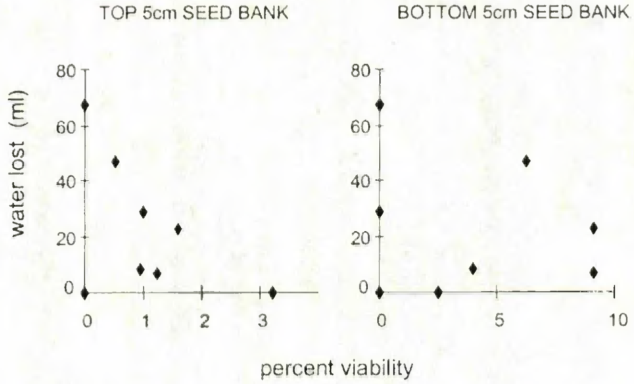

Figure 10. Relationship between % viability of seeds from postburn samples and relative soil heating, as measured from water lost from cans placed in the burn.

Postfire seedling recruitment

We conducted a survey of our plots at the burn site to document seedling regeneration. We used a smaller plot size, $2m^2$, to avoid sampling locations near the outside edges of the plots where seed bank samples were taken. The surveys were done on three dates: March 18, 1999 to determine number of seedlings emerging; May 10, 1999 to determine rates of seedling establishment; and April 1, 2000 to determine rates of seedling establishment in year 2, as well as seedling survivorship from the first year.

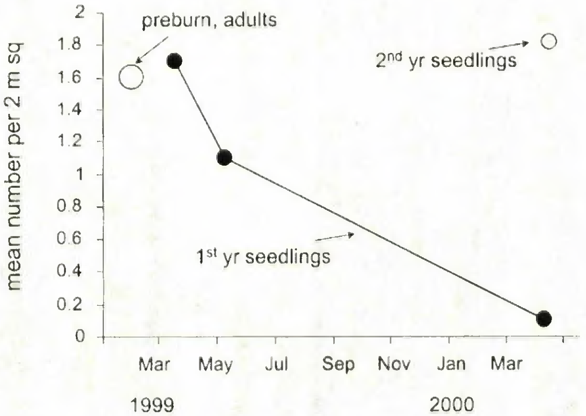

Figure 11. Mean density of first year *A. morroensis* seedlings in prescribed burn (n = 24). Also shown are preburn adult, and 2^{nd} year seedling densities.

A. morroensis recruits must outnumber the adult population for the pre-burn population to be replaced in number. Based on past studies (Sampson 1944, Horton and Kraebel 1955, Keeley and Zedler 1978, Tyler and D'Antonio 1995, and Odion and Davis 2000) seedlings of obligate seeders will suffer considerable mortality. Most seedlings die the first year due to water stress from the initial summer drought they must endure (Frazer and Davis 1988, Tyler and D'Antonio 1995). In maritime chaparral, seedling survival to the 5th year after fire for the obligate-seeder *A. purissima* was found to be between 11 and 15 percent in different locations depending on its initial density and the density of other seedlings (Odion and Davis 2000). Seedling recruitment for *A. morroensis* the first year after fire only produced about the same number of individuals found in the previous generation (Figure 11). However, in our postburn seed bank analyses, we documented that there were viable seed remaining in the soil that did not germinate the first year. Dormant seed appeared to be abundant enough to allow second year germination equal to the first year's. This was affirmed when we re-sampled our plots in year 2 (Figure 11).

Little or no further postfire germination of *A. morroensis* is expected because the seed bank is nearly or completely exhausted. To our knowledge, germination of obligate seeders in the third growing season after fire has not been observed. If there are any remaining seed in the soil at the burn site, they will probably remain dormant until another fire occurs. Based on the number of viable seeds minus measured germination, it is possible that there is viable seed remaining. However, it is also possible that some germination that occurred in the field was undetected before early seedling mortality.

Mortality of first year seedlings has been high, nearly 95% (Figure 11.) Only 2 (out of 41) year-1 seedlings survived to the second year sampling date. One month after our second year sampling, field reconnaissance indicated that many of the second year germinants had died as well. Thus, total postfire seedling recruitment in the study burn was not adequate to replace the pre-fire population. If mortality of the new cohort of seedlings is comparable to that found for *A. purissima* (i.e., much lower than found among 1st year seedlings), the population at the study burn would decrease to only about 15 percent of that found before fire. In fact, it appears that net mortality is even greater. Thus, it appears that the new population size will be only a fraction of the former.

Germination in year one was observed in several plots that were located relatively far away from pre-burn patches of *A. morroensis* (Figure 12, plots 1, 6, 8). This was unexpected given that very little *A. morroensis* seed is found in the soil away from adult shrubs (Tyler and Odion 1996). However, none of these germinants survived to expand the extent of *A. morroensis* in the burn area. Germination in plots located within pre-burn patches of dense *A. morroensis* was very patchy. Several plots were devoid of any seedlings. There were nine plots with seedlings the first year. In year 2, germination occurred in seven plots, all within pre-burn patches of *A. morroensis*. In three of these plots there had been no germination detected year 1. The net result of these patterns of postfire germination and survival is that the distribution of seedlings remaining as of April 2000 (solid and hatched bars in Figure 12) was strongly heterogeneous. Seedlings remained in 8 plots, with one plot (no. 11) containing almost half.

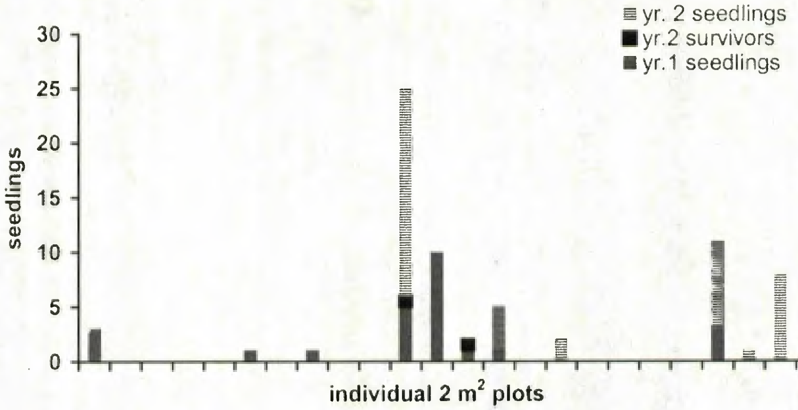

Figure 12. Distribution of *A. morroensis* seedlings in permanent plots.

Addition of burned seedbank samples to unburned plots

In plots where *A. morroensis* was absent, particularly where veldt grass was present, we added soil from underneath burned manzanita shrubs. This allowed us to examine the effects of exotic plants on the recruitment of *A. morroensis*, as well as test the efficacy of moving seed banks to restore degraded sites back to manzanita chaparral. In the burned area we collected soil cores (diameter = 9.8 cm; depth = 10.3 cm) from under shrubs that had burned and where other post-burn seedbank sampling was done. We selected 10 1-m^2 plots within the burned area that had not burned and marked these with rebar stakes; these plots all contained veldt grass. Two soil cores were added to each plot and the soil lightly raked to shallowly cover the seed. Soil cores were collected and distributed to plots the same day, one week following the burn.

Plots were monitored at three dates (March 18, 1999; May 10, 1999; and April 1, 2000) to record the presence of manzanita seedlings. No seedlings were present in 1999. In April 2000, three plots had one seedling each. These results are not encouraging, especially given the low probability of these seedlings surviving amongst the veldt grass. Larger samples, preferably from seed banks containing greater numbers, would need to be used to establish significant numbers of seedlings.

Postburn vegetation composition

Permanent plots were re-sampled April 2000 recording the percent cover of all species present in 12.6 m^2 circular plots. Vegetation cover was still very low in the second year after the fire, and bare ground was still nearly 70% on average (Table 10.) The dominant vegetation present were resprouts or adults that had survived the burn. Very few new seedlings of any species were present.

Table 10. Second year postburn vegetation cover in permanent monitoring plots (n = 24).

species	mean % cover
Ehrharta calycina	8
Arctostaphylos morroensis old	4
Ceanothus cuneatus old	4
Quercus agrifolia	3
Cardionema ramosissimum	2
Ericameria ericoides old	1
Lotus scoparius old	1
Adenostoma fasciculatum old	1
Claytonia perfoliata	1
Ericameria ericoides yr. 2	1
Eriogonum parvifolium	1
Amsinkia spectabilis	1
bare ground	68

Implications for management

The most likely causes for the failure of *A. morroensis* to reestablish pre-burn populations following the prescribed burn were: low numbers of viable seed in the soil, and relatively high mortality of both germinants and young seedlings.

Prior to the prescribed burn we determined that there were ~300 viable seeds/m^2 in the seed bank at the site (Fig 9., and Tyler, Odion, and Meade 1998). This quantity was reduced in size considerably by fire (Fig. 9). Nonetheless, there was an order of magnitude more viable seeds in the soil than seedlings present at the time of our spring sampling. Some seeds undoubtedly failed to germinate because of inadequate moisture. There is evidence that this may be important from research on the closely-related *Arctostaphylos purissima*. Germinants from well watered post-burn seed bank samples numbered 55.9/m^2, while in the field 14.4/ seedlings/m^2 were found to emerge (Odion and Davis 2000). In addition, many *A. morroensis* seeds probably germinated and failed to emerge, which may occur if they are too deeply buried (Auld 1986), or if the germinants are attacked by pathogens. Surviving seed tended to be relatively deeply buried (Fig. 9).

As a consequence of these factors, and possibly post-burn seed predation, the numbers of seed in the soil before fire must accumulate to seemingly astronomical numbers to ensure that populations of obligate-seeders are maintained across fire cycles. Far greater numbers of *A. morroensis* seed in the soil prior to the study burn would have been needed to compensate for the poor transition from seed to seedling that we documented. Our studies have estimated conservatively that from 9-15 years are all that would have been required for the stand of *A. morroensis* at the burn site to accumulate the seed bank present at the time of the burn. Thus, if the stand could have accumulated seeds for several more decades, the potential for seedling regeneration would have been greatly improved.

A. morroensis apparently evolved in an environment where its seeds could accumulate for long periods between fires. With Santa Ana winds absent, and a lower frequency of ignitions, coastal environments have likely supported less frequent fire. Evidence suggests that fires probably occurred relatively infrequently in shrublands on the coast of Central California (reviewed by Greenlee and Langenheim 1980, Hickson 1988, and Odion, Hickson and D'Antonio 1992). Our evidence suggests that this species may require a relatively long fire-free interval, probably longer than obligate-seeders from inland locations even though these also are thought to be selectively favored by a long fire-interval (Keeley and Zedler 1978). Therefore, we recommend that stands should not be burned until they are much older than 40 years. In fact, considering the regeneration failure following our study burn, we believe land-managers should avoid burning *A. morroensis* stands altogether unless it can be unequivocally demonstrated that this will be beneficial to populations overall. In addition, stands of *A. morroensis* should be protected to the maximum extent possible during wildfire suppression. There is the possibility that successful exclusion of fire from stands of *A. morroensis* could eventually lead to a reduction in seed populations as input decreases due to shrub senescence. However, Zedler (1995) reviewed the evidence for "senescence risk" in obligate-seeders in chaparral, and could find none. Even species such as *Ceanothus greggii*, which decrease in cover in older stands, regenerate well after fire in old stands. Moreover, cover of *A. morroensis* may increase with time since fire as occurs with two other manzanitas endemic to Central Coast maritime chaparral (*A. purissima* and *A. rudis*). Nonetheless, stands should be monitored for shrub mortality. Presently, this is occurring among isolated old shrubs at the Elfin forest. It is not clear what the best prescription is for this area. Our previous report discusses how regeneration under dead *A. morroensis* individuals is likely to be poor (Tyler, Odion, and Meade 1998).

Mortality of 1st year seedlings at our burn site was exceptionally high compared to what has been reported in the literature. Mortality of 2nd year seedlings was observed to have begun early in their initial drought season, and therefore will probably be high as well. Does *A. morroensis* typically exhibit such high rates of seedling mortality? If so, seedlings emerging after fire would have to be unusually abundant. For example, a total of 1.7 seedlings/m^2 were recruited into the prescribed burn area in years 1 and 2. With 96 percent mortality of seedlings in their first year as we measured, ~38 seedlings/m^2 would have been required for .8 individuals/m^2, the pre-burn population, to reestablish. The number of recruits would actually have to be greater than 38/m^2 because mortality will also occur after seedlings are more than a year old. This is such an exceptionally high density of seedlings compared to what has been reported in the literature for obligate-seeders. We suspect that mortality will typically be lower among seedlings of *A. morroensis* after fire. Mortality may have been greater in our prescribed burn because of the small burn size. Mills (1983) has reported elevated rates of herbivory in small chaparral burn areas because of the proximity of unburned chaparral, which provides cover for mammals that prefer the young forage available in recent chaparral burns. Mortality could also have been unusually high due to passers-by trampling young seedlings. Herbivory and trampling could lead to high rates of mortality in future burns in stands of *A. morroensis* because they will likely be of limited size and located where they will be susceptible to trampling. Our results suggest that land managers should take steps to protect *A. morroensis* burn areas from foot traffic, and perhaps even herbivores to ensure greater seedling survival.

V) Comparison of seed viability and germinability in litter vs. soil-stored seed

We were interested in comparing the viability of the seeds in the litter to that of soil-stored seed, to determine whether the relatively "fresher" seed found in the litter would have significantly higher viability and thus be a better source of seed for restoration. In addition, research has documented that seed of some chaparral species with deep dormancy requires soil storage to become germinable (Keeley and Fotheringham 1998). This raises the question of whether *A. morroensis* seed in the litter layer would have sufficient germination to be useful as a seed source for reestablishing or enhancing *A. morroensis* populations. We performed the following test to answer these questions.

Methods

We collected seed (May 1999) from the litter layer and from the soil seed bank (top 5.1 cm of soil) under three mature shrubs, in the stand south of the prescribed burn in Montaña de Oro State Park. Samples were sifted, and the seeds and fruits were manually removed. Only intact seeds that appeared to be "healthy" were used for this study. Seeds were counted and divided between two treatments: simulated burning and controls. All seeds were placed on filter paper (Whatman #1, 110mm circles), on top of untreated sponges, in trays filled with approximately 1" of deionized water. Water was added as needed to maintain this level, allowing the filter paper to remain moist for the duration of the germination trial. Trays were kept inside dark growth chambers programmed to alternate between cold and warm cycles: 12 hours at 5°C followed by 12 hours at 20°C. Seeds in simulated burned treatments were heated at 100° C for 5 minutes and then received 0.2g charate (charred wood/ash) spread evenly on the germination medium (filter paper). We placed the seeds on top of the treated paper. Charate was collected in the field from chaparral in the Santa Monica Mountains that had burned in a wildfire; several weeks after the fire, charred wood was scraped directly from chamise stems (*Adenostoma fasciculatum*) that remained.

We monitored these seeds every 2 days to record germination. Germinated seeds were counted and removed. Water was added to the trays, and fungicide (Physan 0.4% solution) applied as needed. After three weeks, the temperature in the growth chambers was changed to be a constant 21°C for 24 hours because no further cold stratification was needed, and embryo growth was expected to be more rapid with warmer temperatures. First germination was recorded February 14, 2000, 28 days from the start of the experiment. Most germination (85%) occurred within the first 49 days. The study was terminated after 73 days.

At the completion of the germination experiment, all seeds were cracked open and inspected under the microscope for the presence of a viable embryo. The number of viable seeds found was summed with the number that germinated to establish the total number of viable seeds from each sample. These methods allow comparison of the proportion of viable seed that germinated in each sample as well as treatment.

Results

Similar to our findings in previous studies (Tyler, Odion, and Meade 1998, Tyler and Odion 1996), seed viability was very low (Table 11), only 4.7% overall. Contrary to our expectations, we found that the percentage of total viable seed from the litter (3.6%) was lower than from soil-stored seed (5.8%). This relationship could differ depending on the time samples are collected or other factors, such as seed predation rates in a given year. However, our results indicate that the litter is not necessarily a better source of viable seed.

Seed germination varied with the source of seed, as well as with treatment. The difference in the total percentage of seeds to germinate between soil-stored seed and that in the litter was likely due to the difference in viability; within treatments the proportion of viable seed that germinated was similar for litter, and soil-stored seed (Fig. 13, and Table 11). The control treatment had a relatively high proportion of live seed germinating (>40%). This corroborates our previous finding that untreated seed had higher than expected germination rates (Tyler, Odion, and Meade 1998). Also as found previously, heat and charate significantly enhanced germination (Table 11); close to 80% of viable seed germinated with this treatment. Soil-stored seed appeared to respond better to this treatment, but this could only be confirmed if additional studies are undertaken with a large sample size collected under many shrubs.

Table 11. Results of germination study comparing soil stored seed to that found in the litter layer. Data are totals for all shrubs sampled.

	CONTROL		HEAT + CHARATE	
	soil	litter	soil	litter
% viable	6.0	3.4	5.5	3.8
% germinable	2.5	1.5	4.3	2.3
% of viable seed that germinated	42	44	78	60
Total # seeds/trt	434	476	418	391

Although fewer seeds germinated from litter samples, our results clearly indicate that the thick litter layer under *A. morroensis*, could be utilized as a resource for Morro manzanita revegetation because it contains readily germinable seed. Although germination would be enhanced by a fire treatment, forsaking this endeavor, which would be impractical on a large scale, could still result in considerable germination.

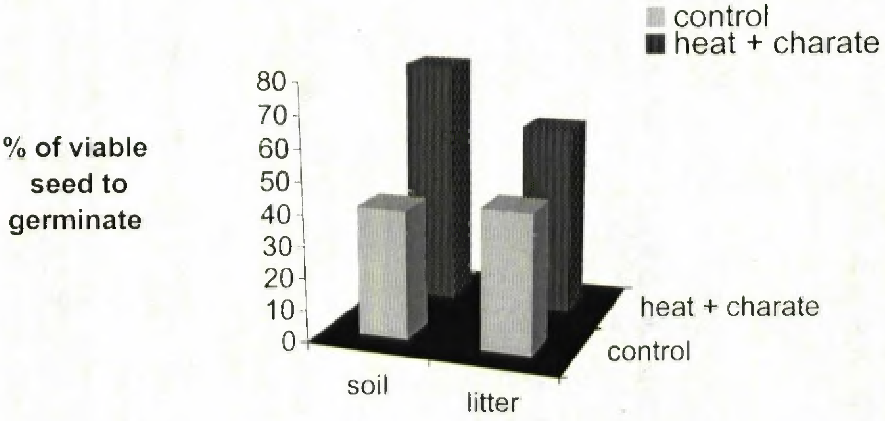

Figure 13. Comparison of germination behavior of seeds in the soil vs. seeds in the litter layer.

An additional finding of note was that seed viability varied considerably among samples from under different shrubs. Samples from one shrub yielded only 6 live seeds out of 776 intact seed: <1% viability. Samples collected from under the other 2 shrubs had 6% and 11% seed viability. As described previously, patterns of seedling recruitment after the prescribed fire were very patchy. One explanation, in addition to variation in soil heating, may be low number of viable seeds under certain shrubs. Our results also indicate that if seed were collected for restoration purposes, it would be desirable to collect samples from a range of different individual shrubs; it may also be useful to dissect a sub-sample of seeds collected to estimate viability and thus potential germinability.

VII) Preliminary GIS analysis of the distribution of *Arctostaphylos morroensis*.

As a preliminary analysis of the distribution of *Arctostaphylos morroensis* using geographic information systems (GIS), we produced the following:

- map and digital coverage of existing distribution of *A. morroensis;*
- map and digital coverage of Baywood fine sands by slope class;
- map and digital coverage of presumed historic distribution of *A. morroensis;*
- analysis of relationship between *A. morroensis* distribution and soil slope classes, and losses of *A. morroensis* over time;

Here we describe the methods used to prepare the coverages, maps, and analyses, and discuss our findings. We also suggest further work that would be possible with this approach.

Methods

The primary map source for most work on *A. morroensis* is from Mullany (1990). Attributes of this map include the following cover classes for the shrub: <1%, 1-5%, 5-25%, 25-50%, 50-75%, 75-100%. Previously, we established plots in a stratified random distribution that captured the range of variation in vegetation within each of these cover classes (Tyler and Odion 1996). The association analysis that was undertaken with the plot data complements this GIS analysis, and it is incorporated into the discussion below. We were provided with a distribution map (paper) based on Mullany (1990), with updated information provided from field reconnaissance by Diana Hickson and Deb Hillyard from The California Department of Fish and Game. From the Land Conservancy of San Luis Obispo County we obtained a digital version of the *A. morroensis* distribution map found in Mullany's 1990 report. We used these data to create the map shown in Figure 2. This digital version of *A. morroensis* distribution appeared to be slightly misregistered (i.e., possibly due to incorrect specification of datum). If this coverage is used in future work, this issue should be investigated and corrected. However, a ground-truthed and digitized version of the updated distribution map would be optimal.

For the "pre-modern" view of *A. morroensis*, we digitized polygons representing the presumed historical distribution map created by Diana Hickson and Deb Hillyard. These polygons are estimates, based on the existing distribution and the presumed relationship to soils and slope classes.

A digital coverage of the Soil Conservation Service soils map of San Luis Obispo County, Coastal Part (Survey Area ID CA664) was obtained from the Landscape Architecture Department at Cal Poly. Many of the common attributes found in soil surveys are included in this digital soils coverage. Notably, percent slope classes are present, which, as we demonstrate, are useful in evaluating the distribution of *A. morroensis*. Interestingly, when we compared the digital map to the soil survey document we identified errors in the soil type classes of the digital version, which we corrected. This highlights the importance of at least a perfunctory quality-check of GIS data one receives, even if it is the official version distributed by a government agency.

Our GIS analyses included an assessment of *A. morroensis* losses, in terms of area, by the spatial overlay and "subtraction" of the present distribution from the historical distribution. By quantifying the spatial coincidence of vegetation and soils coverages, we also investigated the relationship between slope classes of Baywood fine sands and both the historical and current distributions of *A. morroensis*. This GIS undertaking should be considered a first step in the spatial analysis of relationships between *A. morroensis* and its environment. The digital coverages and custom maps we prepared are provided with this report for future use by California Department of Fish and Game, or under their auspices.

Results

A. morroensis is confined to Baywood fine sands. Thus, the historical distribution of the shrub is estimated to be about that of this distinctive substratum, within a perimeter defined by where shrub is known to have occurred (Fig. 14). The area of this estimated historical distribution is ~1100-1200 Ha. About 45% of this area is relatively level (2-9% slopes, Fig. 15). On level terrain, *A. morroensis* is mixed with many other species in a diverse association we have referred to as Morro Bay maritime chaparral (Tyler and Odion 1996). Other shrubs in this association are *Mimulus aurantiacus*, *Ceanothus cuneatus*, *Adenostoma fasciculatum*, *Salvia mellifera*, and *Ericameria ericoides*.

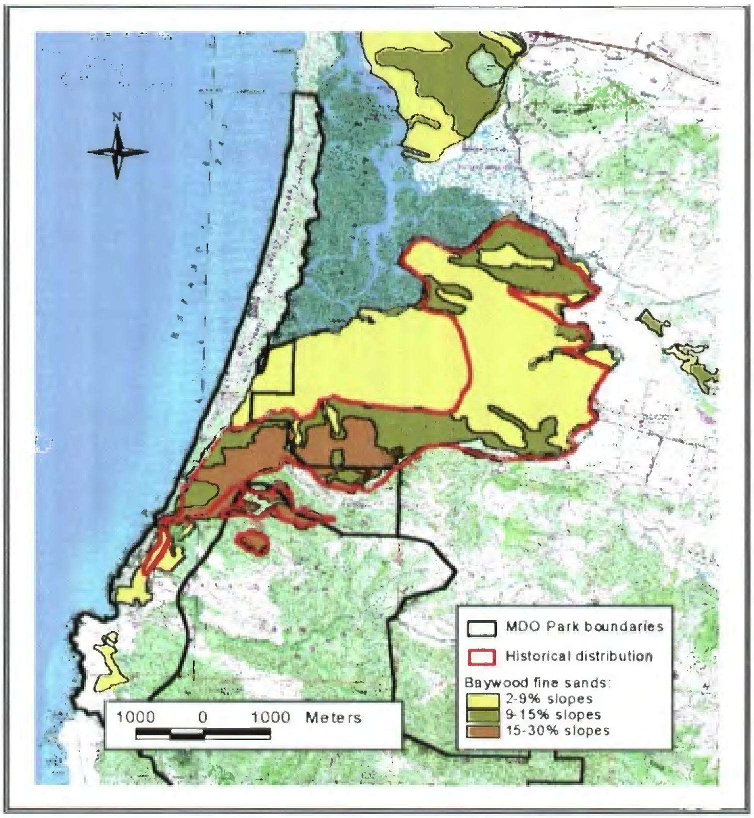

Figure 14. Estimated historical distribution of *A. morroensis* vs. local soil types.

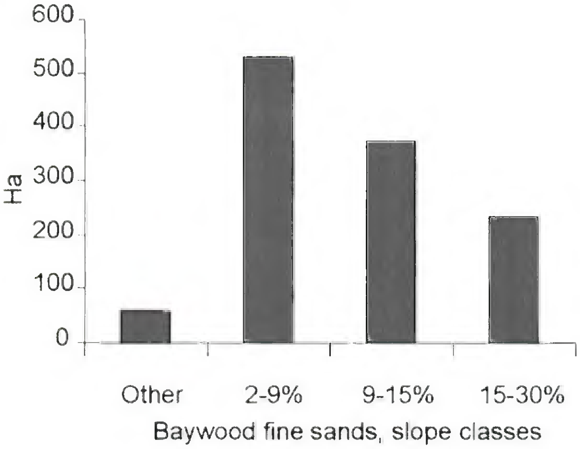

Figure 15. Estimated historical distribution (in hectares) of *A. morroensis* in various slope classes of Baywood fine sands.

The current distribution of *A. morroensis* amounts to ~350 Ha, or about 1/3 of its former extent (Fig. 16). The relatively level terrain supporting the diverse Morro Bay maritime chaparral has been reduced in extent more than chaparral habitat on relatively steeper slopes. Almost half of the current distribution occurs on slopes between 9 and 15% (Fig 16).

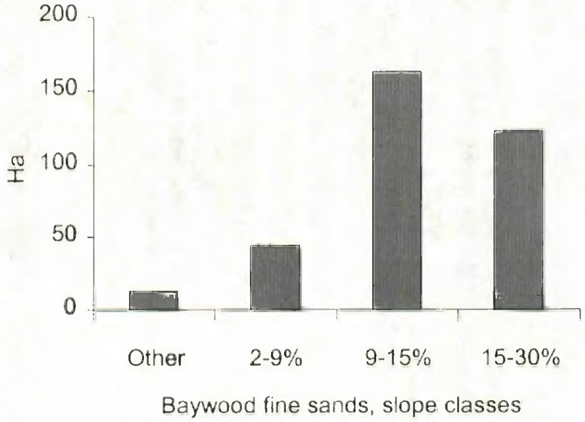

Figure 16. Current distribution (in hectares) of *A. morroensis* in various slope classes of Baywood fine sands.

Arctostaphylos morroensis is currently found mostly either as a scattered shrub (1-5 % cover, ~110 Ha), or in rather dense stands (50-75% cover, ~100 Ha) (Fig. 17).

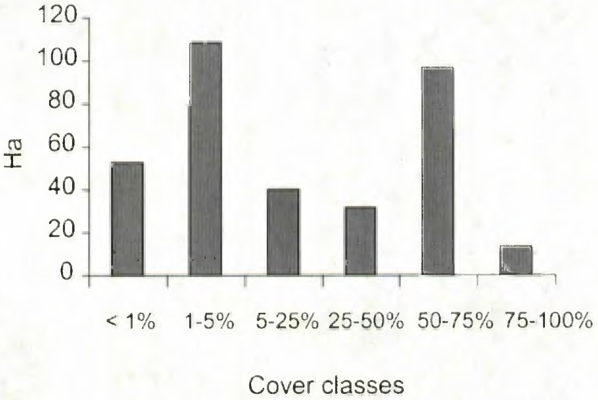

Figure 17. Current area (in hectares) of *A. morroensis* in various cover classes.

The preponderance of stands with relatively low cover presently found is, in part, because this abundance class occurs on all slopes (Fig. 18). Past clearing of chaparral in areas of sloping terrain followed by regeneration of a low percent cover of *A. morroensis* has contributed to this, as is evident from historical aerial photography. Without such clearing, chaparral with relatively high cover of *A. morroensis* occupies steeper slopes on Baywood sands. (There are also patches of *Quercus agrifolia* on north-facing aspects). Dense stands of *A. morroensis* are noted for supporting few other species (occasional *Ceanothus cuneatus*).

There is a good representation of chaparral with 1-5% cover of *A. morroensis* in all slope classes (Fig. 18). The relatively high proportion of stands with 50-75% cover is a reflection of the high percentage of habitat remaining on moderate (9-15%) slopes. On the relatively steeper terrain, the cover of *A. morroensis* tends to increase, and it often forms pure stands on slopes in the 15-30% cover class. However, dense, pure stands of *A. morroensis* (75-100%) are the rarest type; only about 10 Ha remain (Fig. 17). These are about evenly divided between areas with slopes of 9-15% and those with 15-30% slopes (Fig. 18) even though the steeper slopes are less common. Stands of pure *A. morroensis* are lacking on level terrain (Fig. 18).

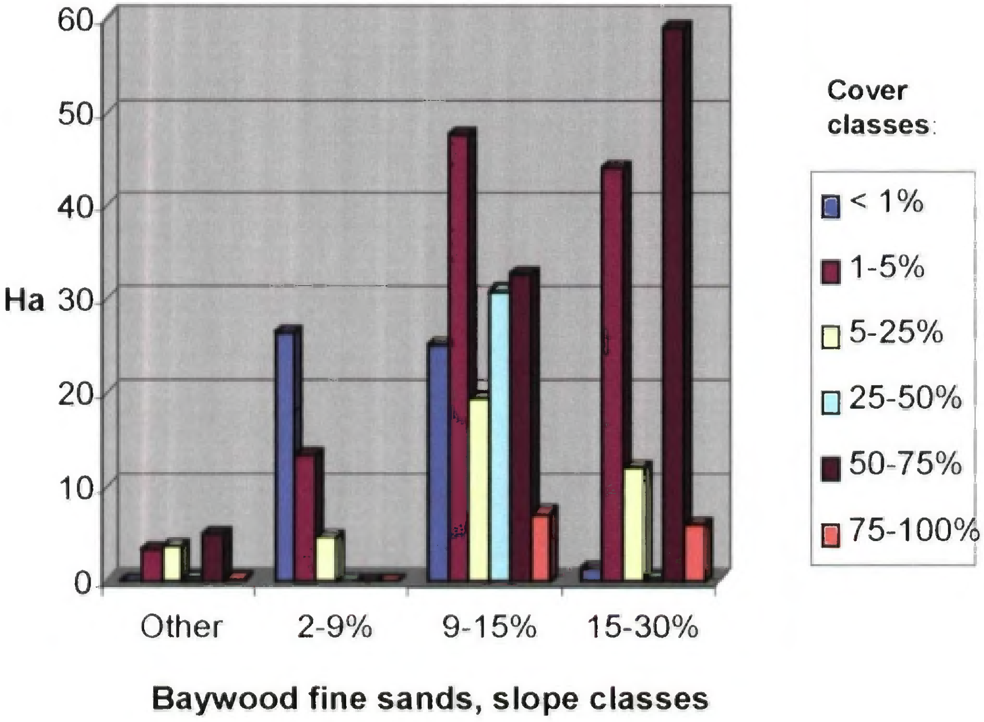

Figure 18. Current area of *A. morroensis* by cover class and slope.

There are additional GIS analyses that could prove valuable in the management of *A. morroensis*. For example, maps of the vegetation of the region could be converted to a fuels coverage and used in concert with topographic data to model potential fire spread under different weather scenarios. This might identify specific vegetation management actions that could be taken to prevent spread of fire where it is ecologically undesirable (e.g., into younger stands). To minimize suppression impacts, parcels through which fireline construction and bulldozer activity could be avoided may also be identified. The environment supporting *A. morroensis* could also be better quantified. For example, potential solar radiation, coastal influence, substratum age, and disturbance history could be mapped and analyzed. Finally, relationships between *A. morroensis* and associated species and/or vegetation could be investigated. Of particular concern is the invasion of *A. morroensis* chaparral by exotic pest plants such as *Carpobrotus edulis*, *Conocosia pugioniformis*, and *Erhrharta calycina*. Analyzing the spatial aspects and distribution patterns of these species could identify areas that would be the most effective to treat, in order to thwart exotic invasions. For example, D'Antonio, Odion and Tyler (1993) found that deer and rabbits disperse seed of *Carpobrotus* into recently burned maritime chaparral. Overlaying a map of *Carpobrotus* distribution with the *A. morroensis* distribution map could identify critical areas to eradicate this exotic species.

Finally, the following additional GIS products could be useful in land use planning, and for management of *A. morroensis*:

- map and digital coverage of *A. morroensis* distribution as of 1949, based on historical photo;
- map and digital coverage of current land ownership classes (i.e., public-protected, private-protected, and private-unprotected);
- map and digital coverage of land planning and zoning designations;
- map and digital coverage of *Eucalyptus* stands within *A. morroensis* distribution;
- distribution and level of threat (based on ownership or zoning) to remaining *A. morroensis* habitat.

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