National Park Service
U.S. Department of the Interior

Point Reyes National Seashore
California

Vision Fire

Lessons Learned from the October 1995 Fire

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U.S. Department of the Interior
Washington, DC

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GRATON

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SKY CAMP
· CAMPING BY PERMIT ONLY
· NO WOOD FIRES
· PACK OUT ALL TRASH
⇦HORSES USE HITCHRAIL

Introduction

by Sarah G. Allen & William Shook, National Park Service

On October 3, 1995 a wildfire ignited on Mount Vision and became what is now known as the Vision Fire. The Vision Fire was the most devastating wildfire at Point Reyes in sixty years, with more than 12,000 acres of state, federal and private lands burned. The origin of the fire was an illegal campfire on State Park lands that was improperly extinguished. The smoldering embers were later propelled by hot, dry 50-mph winds, and the fire spread rapidly through several decadent vegetation communities extending from Bishop pine/Douglas fir stands along the Inverness Ridge to beach grass assemblages in the sand dunes along the Pacific Ocean. At its peak, the fire spread at a rate of 3,100 acres per hour.

By the third day, the fire had consumed over 12,000 acres from mountain ridges to the sea, including 45 privately owned structures in the wildland urban interface along the Inverness Ridge. Although news agencies reported that 20% of the fire was contained on the second day, a discerning eye could easily distinguish from a map of the fire that the only containment was the Pacific Ocean. The fire continued to flare up over the next 10 days, re-threatening homes along the park boundary. Not until October 17, 14 days after ignition, was the fire officially declared controlled. Over 95% of the fire burned within the Point Reyes National Seashore. Within the Seashore, the Vision Fire burned a total of 11,410 acres, consuming 25% of the park's designated wilderness. Other areas burned included 370 acres of Tomales Bay State Park and 386 acres of private lands.

At the height of the fire suppression campaign, 2,164 personnel including 74 hand crews, 27 bulldozers, 7 air tankers, 7 helicopters and 196 fire engines were involved. Over 429,00 gallons of fire retardant and 1 million gallons of water were dropped over the land to control the fire. Park Headquarters were converted into a self-contained city overnight with personnel from several agencies National Park Service (NPS), California Department of Forestry (CDF), U.S. Forest Service, California Department of Corrections, California State Parks, U.S. Weather Service, Bureau of Indian Affairs, and fire departments extending from northern and central California and organizations (Salvation Army, American Red Cross, etc.) forming an encampment on park lands (BAER 1995).

The fire and fire suppression actions together had the potential for long-lasting effects to the natural resources of the Seashore. Fire suppression actions associated with containing and controlling the Vision Fire relied heavily upon direct and indirect mechanized bulldozer fireline construction. Much of the bulldozer line construction occurred in the upper reaches of watersheds with extremely steep and unstable slopes and with cascading effects to existing trails. Bulldozer lines in some areas traversed locations of known invasive nonnative plants, increasing the potential for their spread. Down slope of the line construction were numerous riparian areas, wetlands and estuaries that could be affected by siltation and runoff.

In the midst of the fire fighting, several eager researchers sought to methodically study the effects of the fire on the environment. Many of the researchers had pre-exiting sample sites that they wanted to examine while the fire was still smoldering. Others came with new ideas and proposals for study to record the effects of the fire over the short and long-term. The purpose of this document is to tell the story of the research that grew out of this fire just as quickly as the plants that sprouted amidst the

Cattle on Home Ranch during the Vision Fire.

Fog along flank of Inverness Ridge

Setting

Point Reyes National Seashore is located in Mann County, California, approximately 40 miles northwest of San Francisco. Established by Congress on September 13, 1962, the enabling legislation stated that park designation was

"to save and preserve, for the purpose of public recreation, benefit, and inspiration, a portion of the diminishing seashore of the United States that remains undeveloped" (PL 87-657).

The Wilderness Act of 1976 (P.L. 95-544) established 33,373 acres of wilderness in the Point Reyes National Seashore, thereby adding special protection. Today, nearly half of the Seashore is included in the National Wilderness Preservation System. Encompassing 71,046 acres of coastal dunes, coastal prairies, marine terraces, coastal scrub and forests, this geologically unique peninsula has rightly been called an "Island in Time".

Point Reyes National Seashore shares boundaries with many entities. Marine boundaries are shared with the Gulf of the Farallones National Marine Sanctuary; terrestrial boundaries are shared with Golden Gate National Recreation Area (GGNRA, another National Park), Tomales Bay State Park, The Nature Conservancy, and many private landowners. In 1988, UNESCO's Man in the Biosphere program designated the Golden Gate Biosphere Reserve (GGBR) under the International Biosphere Program; GGBR includes the Seashore, Golden Gate National Recreation Area and other public lands in the region. Much of the eastern border of the park is shared with private landowners. Within the park boundaries, there are several dairy and cattle ranches, which have continued operation after park acquisition, U.S. Coast Guard facilities, and some private in-holdings.

The Mediterranean climate produces heavy summer fog and moderate winter rains (average of 30 inches/year). Fall tends to be hot and dry and is the period of highest fire danger when vegetation is desiccated, and humidity is low. The year-round ambient temperatures are moderate, around 65°F; the difference in the monthly temperatures is only around 6.5°F. Coastal areas have a more moderate climate than the interior and can receive significant moisture from fog in summer. The inland areas receive about half the rainfall as the coastal range. With this variability, many microclimates occur. For example, Point Reyes Headland can be 55°F with fog and wind in the summer, contrasted with Olema Valley, just 15 miles distance, with temperatures above 80°F with no wind.

Point Reyes is part of the Mediterranean division of eco-regions of California and is distinguished by alternate wet and dry seasons (Bailey 1995). The area is notable as a transition zone between the dry west coastal desert and the wet west coast. Mediterranean-type ecosystems host a disproportionate share in the number of plant species worldwide in both the number of species and the number of rare or locally endemic species.

The Coast Ranges are gently to steeply sloping low mountains or marine terraces underlain by shale, sandstone, and igneous and volcanic rocks. Overlain on this topography, the major biomes of the park include forests, grasslands, savannahs, and several types of aquatic environments. Hard leaved evergreen trees and shrubs called sclerophyll forests, which can withstand severe drought and evaporation in the summer, typically dominate the vegetation. The pattern of plant community distribution consistently has forest on north facing slopes and on wetter sites, chaparral/scrub on south facing slopes and drier sites, and riparian corridors between ridges and along valleys. Additionally, the plant communities vary with distance from the marine influence, temperature, and elevation.

Fire History

Many of the plants of Point Reyes have evolved with fire, either requiring fire for regeneration such as Bishop pine, or resprouting rapidly after fire such as coyote bush. The ignition of fires, though, is rarely from lightening strikes at Point Reyes, accounting for only 2% of fires recorded over the past 50 years. he earliest human inhabitants of Point Reyes, the Coast Miwok, used fire to manage the landscape for food and fiber. They concentrated burning in coastal grasslands and oak woodlands to support grazing ungulates such as deer and to force oak trees to produce higher yields of acorns. Mexican ranchers, on the other hand, burned mostly brushlands, saving the grasslands for introduced livestock forage.

Researchers who have studied the fire history of the Point Reyes Peninsula determined from the scarring of old trees and from core samples that there were an average of 2-12 fires per year over the past 300 years (Brown et al. 1999). This fire frequency likely reflects the regular burning efforts of the Coast Miwok and Mexican

A staging area at Sky Camp.

ranchers. Over the past century, though, several fires occurred in Bishop pine stands on Inverness Ridge. The Santa Barbara Weekly Independent reported a forest fire raging near Point Reyes in October 1887. Another fire that started on the west slope of Mount Vision burned through the Bishop pine forest to within two miles of Inverness in October 1927. Dr. Millard Ottinger who purchased a large portion of land on the northern end of Inverness Ridge set numerous fires between 1930 and 1960 in order to clear brush and improve forage production for livestock and deer.

Only one major wildfire, though, has occurred at Point Reyes since the Seashore was established in 1962. The Kelham Beach fire in June of 1976 was started by campers in brush on coastal bluffs in the southern area of the wilderness. The fire burned 325 acres in brush and Douglas fir before it was finally brought under control. All other fires over the past forty years were small and did not alter the landscape.

Fires have occurred at all times of the year at Point Reyes, but the vegetation is more prone to burn beginning in early June, after rains cease in April and May. Typically, a persistent coastal fog forms by early July, following the stabilization of the Pacific high over central California. From July through early September, the fog moves inland and back out to sea in a 3-4 day cycle in response to heating and cooling in California's Central Valley. In September and October, the fog pattern changes as temperatures drop in the Central Valley. Strong foehn winds, also called Santa Ana winds in southern California and Mono winds in central California, may develop if there is a low-pressure trough off the coast. These winds bring warm, dry air to Point Reyes and cause rapid drying of fuels, live and dead. These episodes usually last 1-2 days causing extreme fire danger. The combination of prolonged drought, low relative humidity and a peak in dry vegetation often causes fire danger to be high through October. In addition, almost one-fifth of the area's annual lightning storms occur at this time. The period from September through October is considered the most critical time of fire danger for the Seashore.

Vision Fire Evaluation and Restoration

The Vision Fire met the criteria for generating a major wildfire in Point Reyes: it began in early October when vegetation was desiccated after summer months of drought; and foehn winds blew intensely at 40 to 50 mph along the Inverness Ridge. Ignited by an unattended campfire at around 1300 hr on Mt. Vision, within the Tomales Bay State Park, winds accelerated the spread of the fire through steep terrain and in volatile Bishop pine forests.

Shortly after the fire began, and before it was contained, a Department of Interior multi-disciplinary team, the Burn Area Emergency Rehabilitation (BAER) team, was assembled to assess the fire effects. The BAER team was made up of resource specialists with expertise in plants, animals, soils, water resources, cultural resources, structures, roads and trails. Agencies represented on the team included the U.S. Forest Service, National Park Service, U.S. Fish and Wildlife Service, U.S. Bureau of Land Management, and Bureau of Indian Affairs. The primary task of the BAER team was to produce a report documenting the fire effects on the resources of the park and adjacent lands, analyzing the fire suppression effects on park resources, and presenting recommendations for mitigation actions, including actions to restore the integrity of the wilderness area portion of the Point Reyes National Seashore in accordance with the Wilderness Act, National Park Service Management Policies.

The BAER team identified several areas of additional research necessary to guide mitigation recommendation. Simultaneously, a number of scientists who were conducting independent research within the park were presented with an opportunity to evaluate the effects of the wildfire upon their studies. Other scientists were eager to initiate research to understand the effects and implications of the wildfire on a myriad of the environmental subjects.

Burn intensity was one of several factors that the BAER team and the researchers analyzed to understand the behavior and results of the fire. Burn intensity on the landscape was identified based on several measures such as soil friability, soil moisture content, cover left, and quality of ash. Approximately 11% of the fire area was burned by high intensity fire, 19% by moderate intensity and 70% by low intensity. Burn intensity varied significantly across the landscape because of slope, aspect and plant species. Plant communities were affected in various ways as burn intensities fluctuated across the landscape, and in turn, burn intensities influenced the recovery of ecosystems.

Plant communities that burned included marshland, coastal prairie, coastal grasslands, riparian, coastal dune, northern coastal scrub, mixed hardwood conifer, Bishop pine forest, and Douglas fir forest. Each of these com-

munities has associated species that are unique to California, and in some cases, to the world. Within the burn perimeter, many species of plants (23), mammals (8), birds (24), insects (8), amphibians (4), reptiles (2) and fish (4) were identified as sensitive or endemic to the park. Several species had special recognition under the U.S. Endangered Species Act and the California Endangered Species Act. Of particular note were the federally listed species such as the steelhead trout (*Oncorhvnchus mykiss*) and northern spotted owl (Strix occidentalis), but also species unique to Point Reyes such as the Point Reyes mountain beaver (*Aplodontia rufa yhaea*) and the Point Reyes jumping mouse (*Zapus trinotatus orarius*; see BAER report 1995).

Eighteen watersheds within the perimeter of the Vision Fire also were affected at various stages based upon fire intensity, slope and soils. Soils on slopes >11% are considered highly erosible and over 80% of the area burned had slopes with this grade. Soil associations within the burn were not only highly prone to erosion but were also hydrophobic (repel water) following exposure to moderate to intense fire intensities. The waxy surface on the leaves of many plant species such as coast live oak was deposited on the soils when the plants were burned, contributing to the hydrophobic soil condition. Consequently, erosion potential was very high within portions of the burn area due to this combination of factors and to the potentially high annual winter rainfall. Of these watersheds, eight were known to contain populations of the threatened steelhead trout and potentially contained populations of the Coho salmon. The Seashore was concerned that impacts from increased sedimentation, ash and carbon input, and alterations of water chemistry could have adversely affect these species.

The BAER team ultimately recommended numerous restoration projects to mitigate the effects of the fire and fire suppression actions on the resources of the park. The projects included 1) restore 23 miles of bulldozer line, 2) restore 6.4 miles of fire fighter hand lines, 3) rebuild 5 miles offence line, 4) remove 13 helicopter landing spots, 5) mitigate 1,100 hazardous trees, 6) restore 10 safety zones, 7) assess 31 historic sites, 8) monitor the spread of non-native weeds, and 9) monitor 10 threatened, endangered or sensitive species.

Research Emphasis

The rapid momentum of events propelled the Seashore to begin restoration actions of natural resources with limited information, and consequently, the Seashore decided to initiate and integrate several studies. The purpose of the research was to evaluate fire and fire suppression effects on the integrity of abiotic resources and plant and animal communities. The projects would encompass analyses of physical, chemical and biological changes to the ecosystems in different areas of bum intensity. The studies were designed to monitor treatments and affected resources to determine the effectiveness of measures taken to mitigate suppression and rehabilitation actions. Feedback from the

Rehabilitation of a burned area for erosion control.

studies could then be utilized to improve future treatments.

The studies also were designed to evaluate the short and long-term post-fire effects on the natural resources of the Seashore. Changes in ash and soil would occur with each rain and resprouting of plants would begin immediately, and so there was an urgency to begin some of the studies while the fire was still burning. Changes in the landscape and populations of some species, though, might continue for several years.

On the first year anniversary of the Vision Fire, the Seashore hosted a symposium giving the researchers an opportunity to tell their stories to the community. This report is a record of some of those fire stories.

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Literature Cited

BAER Report. 1995. Mount Vision fire incident burned area emergency rehabilitation plan. Department of the Interior, BAER Team, North Zone.

Bailey, R.G. 1995. Description of the ecoregions of the United States. USDA, Forest Service. Misc. Publication No. 1391.

Brown, P.M., M.W. Kaye, and D. Buckley. 1999. Fire history in Douglas-fir and coast redwood forests at Point Reyes National Seashore, California. Northwest Science 73(3): 205-216.

Mount Vision in the distance at the mouth of Drakes Estero and Drakes Head.

Fire Effects on the Point Reyes Mountain Beaver (*Aplodontia rufa phaea*)

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Abstract

In October of 1995, the Vision Fire burned 5,000 ha (12,354 acres) on the Point Reyes peninsula. In most of the non-forested areas, the fire burned the vegetation and associated leaf litter down to mineral soil. This effectively cleared the ground and revealed thousands of mountain beaver (*Aplodontia rufa*) burrow openings that had been well hidden and difficult to find prior to the fire. In the first six months after the fire, we surveyed 730 ha (1,800 acres) of burned coastal scrub and riparian habitat to 1) count the number of burrow openings that existed at the time of the fire, and 2) evaluate whether there were signs of post-fire mountain beaver activity. Based on our counts of burrow openings and assumptions about the number created by each individual, we estimated that only 0.4 - 1.2% of the 5,000 mountain beavers within the burn area survived the fire.

Mountain beaver populations could recover either by immigration from outside the fire area, or by growth of the small populations that survived within the burn area. To evaluate recovery within the burned area, we monitored mountain beaver activity for five years at eight sites where mountain beavers survived. By the end of the first year, one of these sites no longer had any mountain beaver activity. In another two sites, the numbers of active burrows remained low; we believe that both of those sites were occupied by only one or a few non-reproducing individuals. In four other sites, the number of active burrows returned to approximately pre-fire levels by the third year, and the remaining site recovered by the fourth year.

To evaluate immigration from surrounding areas, we repeatedly surveyed three sites where mountain beavers disappeared, two near the edge of the burn and one 1.0 km inside the burn. One at the edge had a few active burrows in the third, fourth, and fifth years post-fire, but the other two sites showed no sign of activity after five years. We estimate that it will take 15-20 years post-fire for the mountain beaver population to recover to the pre-fire estimate of approximately 5,000 individuals.

Mountain beaver in burrow. (USGS)

Introduction

Mountain beavers (*Aplodontia rufa*) are an unusual and primitive species of rodent. They are about the size of a muskrat (27-30 cm long), but they have a short (1 cm) tail. Mountain beavers feed on a wide variety of vegetation including coyote brush (*Baccharis pilularis*), sword fern (*Polystichum munitum*), cow parsnip (*Heracleum lanatum*), blackberries (*Rubus ursinus*), poison oak (*Toxicodendron diversilobum*), California nettle (*Urtica dioica*), foxglove (*Digitalis purpurea*), and thistle (*Cirsium* sp.) (Steele, 1989; pers. obs.). They live in underground burrows that are typically dug in dense thickets or in forest openings. The presence of 15-18 cm diameter burrow openings (Grinnell and Storer, 1924) is often the most conspicuous evidence of mountain beaver activity. Typically, there are multiple openings in an area of only about 14-16 m^2. Camp (1918) described the burrow system:

"Wherever the aplodontia lives it digs extensive underground tunnels that in a populous colony form a network of passages a few inches beneath the surface of the ground. Each burrow system has many openings to the surface, but excavated dirt and rubbish is pushed out usually at only a few of these holes."

A mountain beaver needs 1/3 of its body weight in water every day (Nungesser and Pfeifer, 1965) because its kidneys are simple and inefficient at conserving water (Sperber, 1944). An adult needs to consume 295-450 ml of water each day, by drinking or from food. Because of this, mountain beavers are restricted to areas near water or with extensive summer fog along the Pacific coast.

Mountain beavers range from the southwest corner of British Columbia south through western Washington and Oregon. In California, their range extends through the Sierra Nevada Mountains and barely into Nevada. Along

the California coast, mountain beavers are found south to near Cape Mendocino. Further south, small, isolated coastal populations occur at both Point Arena (Mendocino County) and Point Reyes (Marin County).

The Point Reyes mountain beaver (*A. rufa phaea*) is only known to occur in western Marin County, almost entirely within Point Reyes National Seashore where it is found on cool, moist, north-facing slopes in moderately dense coastal scrub. This scrub vegetation typically includes coyote brush as well as sword fern, bracken fern (Pteridium aquilinum), poison oak, California nettle, and cow parsnip, which tend to grow in the moister areas.

Most of the area occupied by the Point Reyes mountain beaver was regularly burned by Coast Miwok Indians who once occupied the Point Reyes peninsula. In the last 100 years, however, fires have been far less frequent and routinely suppressed. This fire control has resulted in a buildup of highly combustible fuels (Sugnet and Martin, 1985).

The Vision Fire of October 3-12, 1995 burned 5,000 ha (12,354 acres) 94% of the burn area was within Point Reyes National Seashore (Fig. 1.). The fire consumed mostly coastal scrub, but also some Bishop Pine (*Pinus muricata*) and Douglas fir (*Pseudotsuga menziesii*) forest, grassland, and riparian habitats. The fire burned 40% of the known range of the Point Reyes mountain beaver, including the majority of what was believed to be prime habitat.

Study Area

Point Reyes National Seashore is a 28,757 ha (71,059 acre) natural area located 50 km NW of San Francisco. The park includes 10,767 ha (25,370 acres) of officially designated wilderness. Major plant communities include Douglas fir forest, Bishop pine forest, coastal scrub, and grasslands.

Methods

Pre-Fire Surveys (1984 - 1994)

Prior to the Vision Fire, mountain beaver habitat within Point Reyes National Seashore was systematically surveyed to evaluate the distribution and habitat preference of mountain beavers within the Seashore. Thickets were surveyed by walking the perimeter and exploring natural openings and indentations in the vegetation. A stout wooden pole was used to move aside vegetation and make burrow openings more obvious.

When burrows were not detected, thickets were traversed at approximately 25 m intervals. Many thickets were so dense that it was not possible to penetrate them (without cutting a swath through the vegetation); these thickets were examined only around the perimeter. After the Vision fire, we could see that burrow openings were scattered throughout the thickets and were not always observable from the perimeter. It is clear that both burrow openings and entire populations had been missed during the pre-fire surveys.

Openings in forests were surveyed by traversing the entire opening at approximately 20 m intervals, making sure to inspect areas that appeared to be particularly suitable for mountain beavers.

In addition to their distinctive size, mountain beaver burrow openings are always found in groups or clusters (Lyon, 1907; Camp, 1918). We assumed that mountain beavers were present when we found > 5 suitably sized burrow openings in an area of < 25 m^2. We counted the number of openings observed, but it was not practical to quantify population size.

We use the term site for each group of mountain beaver burrows that are separated from other burrows by at least 50 m, a distance not regularly traversed by this species (Martin, 1971).

Population refers to the mountain beaver(s) occupying a site. A population can range from a single individual to hundreds of animals.

Initial Post-Fire Surveys (1995 - 1996)

Post-fire surveys were conducted primarily in coastal scrub. In this habitat, the fire reduced the vegetation to a few charred skeletons of the larger bushes, and the charred bases of sword ferns. Mountain beaver burrow openings were fully exposed, allowing us to map distribution and estimate population size. Additionally, a layer of ash covered the ground, including the burrow openings and dirt piles associated with burrows that pre-dated the fire. Burrow openings utilized post-fire were conspicuously ash-free and easily detected.

Between December 6, 1995 and April 25, 1996, we spent 20 person-days surveying 730 ha (1,800 acres) in the northwest part of the burn area (Fig. 1). This survey area included all of the Home Ranch Creek and Whitegate Valley watersheds upstream from the Muddy Hollow Road, and most of the Glenbrook Creek watershed above the same road. We also surveyed

the site of one well-known mountain beaver population on the Laguna Ranch, near the center of the burn.

We estimated the area occupied by mountain beaver at each site by pacing, and counted or estimated the number of burrow openings. Areas surveyed and the location of burrow openings were plotted in the field on 7.5' USGS topographic maps. By late April 1996 (6 months after the fire) the regrowth of vegetation made it impractical to continue mapping.

Photo Documentation

We confirmed that fresh digging at burrows was due to mountain beavers by deploying infrared-triggered still and video cameras at sites with fresh dirt at burrow openings. Cameras were a modified Olympus Mini DLX, triggered by a Trailmaster 1500 unit (Goodson & Associates, Lenexa, KS 66215). The Trailmaster system utilized a transmitter and receiver. The transmitter was situated so the infrared beam was about 3" above the ground. A picture was taken when beam was broken for at least 0.15 sec. We configured the Trailmaster so that the camera would not be triggered again for at least one minute. The camera was set to take pictures 24 hours per day. We checked the cameras every two weeks to replace film and batteries, as needed. Sony video cameras in waterproof housings (CompuTech, Bend, OR 97708) were deployed in a similar fashion except that the camera was triggered by a passive infrared sensor that detected motion in a general area (similar to a burglar alarm). Once triggered, the camera recorded 60 sec of video.

Burrow Temperature

To evaluate whether the loss of vegetation affected burrow temperature, we monitored temperature at two burrow systems, one at Muddy Hollow within the burn area and the other outside at Home Ranch. The burrows were both on north-facing 40-45% slopes, where coyote brush, coffee berry (*Rhamnus californica*), and poison oak were the predominant pre-fire vegetation. The burrows were carefully matched for similarity in slope, exposure, and pre-fire vegetation. One burrow was at Muddy Hollow (burned) and the other at Home Ranch (unburned). We deployed a temperature logger (Hobo, Onset Computer, Pocasset, MA 02559) at the entrance and another 1.5 m deep at each burrow. Temperature was recorded at noon every day for a year beginning April 19, 1996.

Monitoring (1996 - 2000)

We surveyed in late spring and early summer since that is when we have observed the highest level of mountain beaver activity (e.g., fresh burrowing, cut vegetation) at Point Reyes. In November 1996 (13 months after the fire) and in the late spring and early summer of 1997, 1998, 1999, and 2000, we resurveyed 11 sites that had been surveyed immediately after the fire. At eight sites we had observed active mountain beaver burrows in the initial post-fire surveys (A-H, Table 3, Fig. 1). We used these eight sites to evaluate post-fire persistence and population growth. The three other sites (X-Z) were places where there had been pre-fire populations, but where we found no mountain beaver activity immediately after the fire. Two of these three sites were located at the edge of the burn and

Vision Fire viewed from Limantour Road.

the other was 1.0 km inside the fire area. These three sites were used to evaluate if, and how rapidly, mountain beavers immigrated from outside the burn area.

Complete resurveys of the 11 focal areas were not possible since newly grown vegetation obscured many of the burrows. Hence, we surveyed 1) in the immediate vicinity of all active burrows found during the initial post-fire field work, and 2) along transects spaced at 10 m intervals across the entire site. In the 1996 and 1997 surveys, we noted only the presence or absence of active burrow openings and the general locations where burrow openings were found. In 1998, 1999, and 2000, we counted active burrow openings and extrapolated to get the estimated total number of openings.

Results

Pre-Fire Surveys (1984-1994)

Surveys were conducted on 69 days between July 6, 1984 and March 10, 1994. We visually searched for burrow openings and collapsed tunnels. Mountain beaver burrows were observed at 74 survey sites prior to the Vision Fire (Fig. 2, Table 1); 70 of these represented sites not previously recorded (Dale Steele, personal communication; California Natural Diversity Data Base). Forty-six of these were outside the Vision fire area, while 28 were within the burn.

Initial Post-Fire Surveys (1995 - 1996)

We estimated that in the 730 ha (1,800 acre) area surveyed (15% of the burn area), 46,300 burrow openings were present at 107 sites at the time of the fire (Table 1). These sites were primarily on north-facing slopes, and encompassed 60 ha (149 acres), 8% of the total area surveyed. Within the 107 sites, there was a mean of 8 burrow openings per 100 m^2 (Table 2). If we make assumptions about the number of burrow openings associated with each burrow system and the number of individuals that occupy each burrow system, it is possible to estimate the total number of animals represented by the openings we observed. In 1918, Camp trapped mountain beavers at Point Reyes and calculated that each burrow system had nine openings. Aside from a short time when young are born, each burrow system is normally occupied by a single individual (Pfeiffer, 1954), though Grinnell and Storer (1924) found two adults in a burrow system in the Yosemite area. Using these numbers (9 openings and one animal per burrow), the 46,300 burrow openings we observed would have represented about 5,144 individuals.

The largest site extended 2,200 m along the north-facing slope of a drainage of Home Ranch Creek. In places, burrows were scattered across the entire 100 m distance from the creek to the top of the slope. Elsewhere, the burrows were limited to a band as narrow as 20 m, along only the upper slope. Despite its large size, we considered this a single site since there were no gaps >50 m between adjacent burrows (Martin, 1971).

In the first six months after the fire, we found 110 burrow openings that had been excavated post-fire. These burrows were at 10 different sites, located in widely separated areas. One site had seven distinct clusters of burrows, another site had three clusters, and one site had two clusters. If each cluster was made by a different individual, then the 10 sites would have supported 19 individuals. These 19 animals would represent a 0.4% survival rate, based on our calculations of a pre-fire population of 5,144 individuals. No other pre-fire sites showed any sign of activity after the fire.

The average number of burrows per individual is 6.9, somewhat below the nine reported by Camp (1918), but these were new burrow systems in a highly modified habitat that my not be typical.

Photo Documentation

We photographed mountain beavers at seven sites within the burn area. In addition, we photographed five species that appeared to be sharing the burrows with mountain beavers, at least on occasion. These were brush rabbit (*Sylvilagus bachmani*), deer mouse (*Peromyscus maniculatus*), California vole (*Microtus californicus*), long-tailed weasel (*Mustela frenata*), and spotted skunk (*Spilogale gracilis*).

Burrow Temperature

Temperatures at the entrance of the burned and unburned burrows were essentially the same (13.9 versus 13.8 C, $t = 0.43$, df = 364, $p=0.66$). Temperatures measured 1.5 m into the burrow were significantly different (12.2 versus 11.4 C, $t = 5.93$, $p<0.001$). The results were similar whether data were analyzed for the entire year (as above), or for only the first 30 days when there was little vegetative regrowth.

Monitoring (1996 - 2000)

Areas With Post-Fire Burrows

Among the 10 sites that had active mountain beaver burrows in the initial post-fire survey, we resurveyed eight of these for the following five years. There were three patterns of

recovery: strong recovery (five sites), limited recovery (two sites), and extirpation (one site) (Fig. 1, Table 3). In the five sites with strong recovery (sites A - E), the number of active burrows was back to pre-fire levels within five years or less.

In the two sites with limited recovery (F and G), not only was the number of active burrow openings in 2000 far below the pre-fire levels, but there also appeared to be declines between 1998 and 1999, although there was some recovery in 1999 - 2000.

In less than a year post-fire, there were no active burrow openings at site H (Fig. 1, Table 3). We observed that the four active burrow openings found in the spring of 1996 showed no signs of activity within only a few weeks, perhaps because this site was atypical in being on an exposed south-facing slope that became too hot, or retained insufficient moisture.

Sites Without Mountain Beaver Activity in the Initial Post-Fire Surveys

Of the three sites that had no active mountain beaver burrows immediately post-fire and were surveyed annually for the next five years, one at the edge of the burn had 40 active burrow openings (probably 4 - 5 animals) in 1998 and 1999 (site X, Table 4), presumably as a result of mountain beaver immigration from outside the burn area. By 2000, there were 88 burrows; this increase could have been from immigration, or from reproduction by the individuals in the previous two years. This site was 150 m long and 20 m wide, situated on the north-facing bank of the headwaters of Home Ranch Creek. All active burrows were in dense vegetation (primarily poison oak) along the stream. The upper slope, which had hundreds of burrows before the fire, had no active burrows in 2000.

The other two monitored sites with no active burrows post-fire (sites Y and Z, Table 3) still lacked activity through 2000, even thought one site was right at the edge of the burn (site Z), and the other was only 1.0 km inside.

Discussion

The Vision Fire of October 1995 had a devastating effect on the mountain beavers living within the 5,000 ha burn area. Our pre-fire surveys (1984-94) showed that mountain beavers were common and widespread in suitable habitat at Point Reyes. Since most of the burrows were in densely vegetated areas, however, it was not possible to determine population size using our survey techniques. After the fire, much of the vegetation was reduced to ashes allowing us to make population estimates for the first time. We counted 46,300 burrow openings in 107 sites encompassing 60.2 ha. There are two studies that provide information that might be used to calculate how many individual animals occupied this area prior to the fire.

Camp (1918) reported that mountain beaver at Point Reyes had an average of nine burrow openings per mountain beaver. Using that number, the pre-fire population in the area we surveyed would have been 5,1444 individuals, and a post fire survival rate of 0.4%. There are no other data for the Point Reyes subspecies (*A. rufa phaea*), but Lovejoy and Black (1979) provide data on mountain beaver (*A. rufa pacifica*) densities in coastal Oregon. Over a two-year period, Lovejoy and Black captured 150 individual mountain beaver in a 320 x 181 m area. This would result in an average of 25.9 mountain beavers per ha. Using this density to estimate the pre-fire population in the 60.2 ha we surveyed results in an estimated population size of 1,559 mountain beaver, and a 1.2% survival rate. We are unable to determine which estimate of pre-fire population size is more accurate, but whichever number is used, it is clear that post-fire survival was extremely low (0.4 – 1.2 %).

The fire caused major changes in the habitat, reducing coastal scrub to charred sword fern bases and blackened skeletons of coyote brush. As relatively sedentary, burrowing herbivores with an unusually high daily requirement for water, those few mountain beavers that survived the fire found themselves in a rather inhospitable environment. Our temperature data showed the expected increase in temperature 1.5 m inside the burrow opening, but the increase was small (0.8 C) and probably not biologically important. More importantly, there was a loss of available food, an overall drying of the local habitat, and a general lack of moisture.

Surveys for fresh dirt outside burrow openings and photographs from remote-triggered cameras suggested that only 19 mountain beavers survived within the surveyed fire area. This number represents only 0.4 – 1.2% of the population that we estimate had previously inhabited the surveyed area. While this number would be low if we had overlooked active burrows in our initial post-fire surveys, it could also be high. At sites D, E, and F, there were three, seven, and two distinct clusters of active

Long-tailed weasel.

burrows, which we assumed had been made by 12 different individuals. If some of the individuals were responsible for digging at more than one cluster of burrows, our estimates of survival would be too high.

Actual causes of death for most of the mountain beavers are difficult to assess. In spite of a great deal of field work by ourselves and other biologists, only one dead mountain beaver was found after the fire. We believe that most mountain beavers died in their burrows, probably from excess heat, smoke, or lack of water.

Mountain beavers reach sexual maturity in their second year and give birth to 2-3 (rarely four) young in the spring (Pfeiffer, 1958). Longevity has been estimated at 5 – 6 years (Lovejoy, 1972). Given an average of 2.5 young per litter, a pair of mountain beavers would give rise to 20 descendants after three years (with complete survival). The population would then double every year. If we assume that one pair of mountain beavers survived in each of the five strong recovery sites (A, B, C, D, and E; Table 4), this rate of increase would explain the observed population growth.

Interestingly, in three of the sites with strong recovery, the number of active burrows five years post-fire was about twice the number prior to the fire (A, C, and D; Table 4). In these three somewhat limited areas (6,000 and 8,000 m^2), it appears that habitat changes occurred during the first four years post-fire that must have improved conditions for mountain beavers. While we have not been able to evaluate exactly what the key changes might have been, we suspect it was a stimulation of favorable vegetation (especially cow parsnip, *Heracleum lanatum*) that has flourished in only a few sites.

The failure of mountain beavers to repopulate at sites F and G would be expected if these areas had single individuals, same-sex individuals, or individuals that failed to reproduce for some other reason. The total loss of mountain beaver activity at site H was expected since it was on an exposed south-facing slope, completely atypical of most mountain beaver habitat at Point Reyes. After the fire, the area became completely unsuitable for mountain beavers.

There are two additional factors that would tend to retard population recovery, 1) slow dispersal of mountain beavers between suitable sites and 2) post-fire changes in vegetation that could make some areas unsuitable for mountain beavers. While the extremely limited immigration was a bit surprising, it appears that post-fire changes in vegetation have played a more significant role in slowing the recovery of mountain beavers.

The lack of recovery is likely related to shifts in both plant species composition and the physical structure of thickets. We have no pre-fire vegetation descriptions, but we can determine at least the dominant species. For example, at site Z, the vegetation has undergone a significant change from a high density of sword ferns (which are common in areas inhabited by mountain beavers at Point Reyes) to thimbleberry and blackberry. Post-fire, sites E and Y were largely overgrown with blue-blossom ceanothus (*Ceanothus thrysiflorus*), a species not typically associated with mountain beavers, and thus unlikely to have been the dominant plant before the fire.

Though it is difficult to make exact predictions, we estimate that it will take 15 – 20 years for mountain beavers to reach the pre-fire estimate of 5,000 individuals. The primary factor limiting recovery seems to be vegetation. We expect it will take 10-15 years or more for sufficient successional changes to take place. As the vegetation becomes more suitable, an increase in the mountain beavers will likely occur due to population growth and, to a lesser extent, immigration from outside the burn area.

Management Recommendations

Wildfires have a strong, negative impact on mountain beaver. This is of particular concern for the two small, geographically isolated populations along the California coast, both of which are distinct subspecies. The Point Arena mountain beaver (*A. r. phaea*) is Federally listed as Endangered (U.S. Fish and Wildlife Service, 1991). Its entire range encompasses approximately 60 km^2. While some of the habitat at Point Arena would not easily carry a fire, many populations occupy coyote brush thickets are similar to those that we studied at Point Reyes. We recommend that fires in the vicinity of Point Arena or Point Reyes mountain beaver not be allowed to burn substantial portions of areas occupied by mountain beaver. Periodic small fires would allow for normal changes in mountain beaver habitat by mimicking what was probably the natural fire regime with which these animals evolved.

Acknowledgements

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Literature Cited

Camp, C. L. 1918. Excavations of burrows of the rodent *Aplodontia*, with observations on the habits of the animal. Univ. Calif. Publ. Zool. 17 (18): 517-536.

Grinnell, J. and T. Storer. 1924. Animal life in the Yosemite. Univ. Calif. Publ. Berkeley, CA. 752+xvii Pp.

Lovejoy, B. P. 1972. The capture-recapture analysis of a mountain beaver population in western Oregon. Ph.D. Thesis. Oregon St. Univ., Eugene.

Lovejoy, B. P. and H. C. Black. 1979. Movements and home range of the Pacific mountain beaver, *Aplodontia rufa pacifica*. American Midland Naturalist. 10:393-402.

Lyon, M. W. Jr. 1907. Notes on mammals collected at Mount Rainier, Wash. Smithsonian Misc. Coll. 50: 89-92.

Martin, P. 1971. Movements and activities of the mountain beaver (*Aplodontia rufa*). J. Mamm. 52(4): 717 – 723.

Nungesser, W. and E. Pfeiffer. 1965. Water balance and the maximum concentrating capacity in the primitive rodent, *Aplodontia rufa*. Comp. Biochem. Physiol. 14: 289-297.

Pfeiffer, E. 1954. Reproduction in a primitive rodent, *Aplodontia rufa*. Unpubl. Ph.D. Thesis. Univ. Calif., Berkeley, CA.

Pfeiffer, E. 1958. The reproductive cycle of the female mountain beaver. J. Mamm. 39(2): 223-235.

Sperber, I. 1944. Studies on the mammalian kidney. Zoologiska Bidrag Fran Uppsalla. 22: 249-431.

Steele, Dale T. 1989. An ecological survey of endemic mountain beaver (*Aplodontia rufa*) in California, 1979-83. Wildlife Manage. Div. Admin. Report 89-1, Calif. Fish and Game. 58+iv Pp.

Sugnet, Paul W. and Robert E. Martin. 1985. Fire History and Post-Fire Stand Dynamics of the Inverness Bishop Pine at Point Reyes National Seashore. Unpubl. report to the National Park Service.

Recovery of Lepidoptera (Moths and Butterflies) Following a Wildfire at Inverness Ridge in Central Coastal California

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Abstract

In numbers of species, Lepidoptera (butterflies and moths) make up the largest group of plant-feeding animals in North America. Caterpillars of nearly all species feed on plants, and most of them are specialists on one or a few kinds of plants. Therefore they are liable to be severely affected by wildfires, and secondarily, their parasites and predators, including birds, bats, lizards, and rodents suffer losses of a major food resource. In October 1995, a wildfire swept over part of Point Reyes National Seashore, burning more than 12,300 acres (5,000 hectares) of public and private land, following a fire-free period of several decades. I tracked survival and recolonization by moths and butterflies during the subsequent five seasons. I made daytime searches for adults and caterpillars approximately monthly from March through October and collected blacklight trap samples, mostly in May and September-October. More than 600 species of Lepidoptera have been recorded in the Inverness Ridge area, and about 375 of them were recorded during the post-fire survey, including larvae of 31% of them. Plants in a Bishop pine forest higher on the ridge, where the fire was most intense, accumulated their caterpillar faunas slowly, while Lepidoptera feeding on plants typical of riparian woods in the lower canyons reestablished sooner and more completely. Recolonization varied markedly among different plant species,

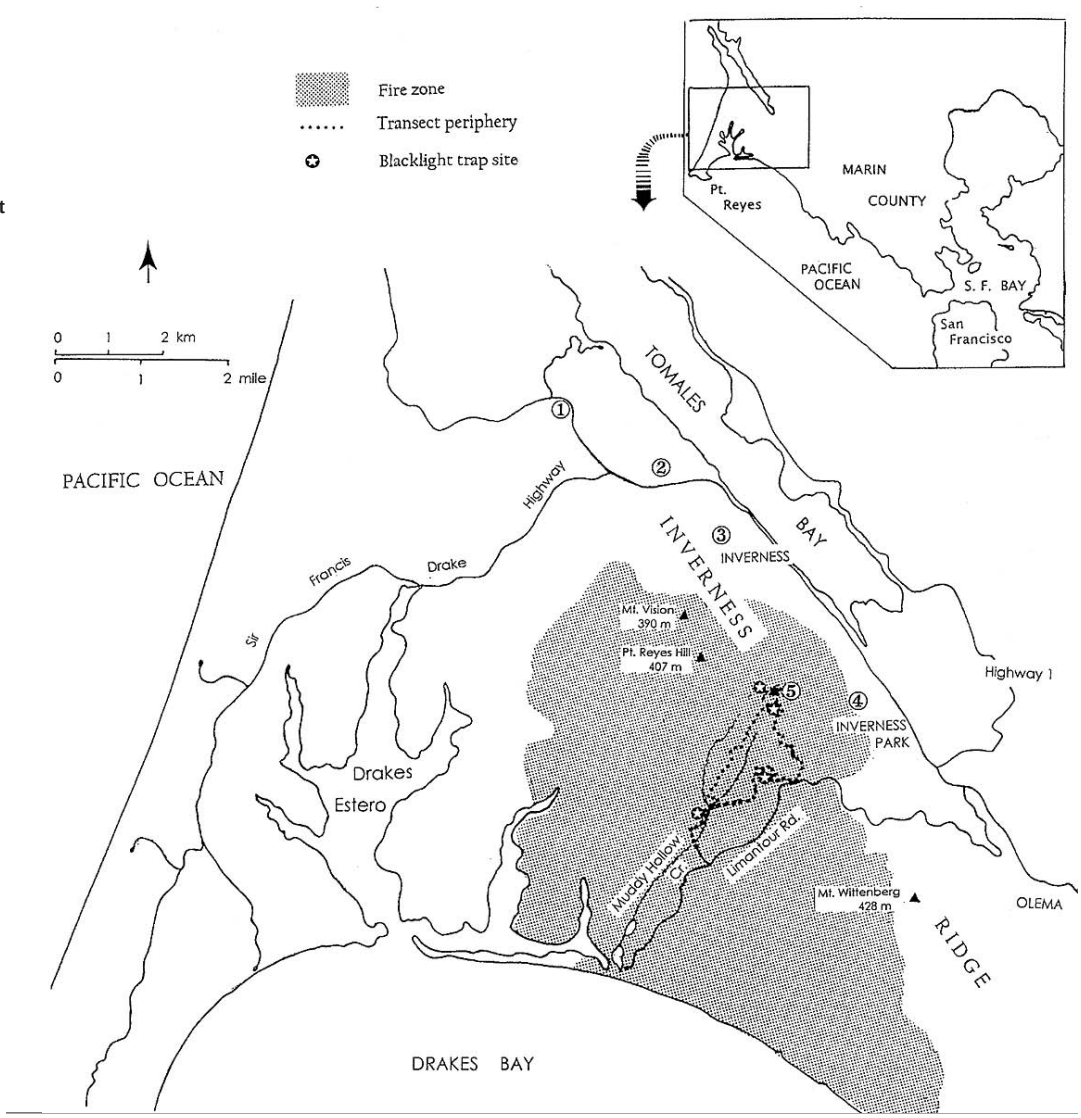

Fig. 1
Map of Inverness Ridge and lower Tomales Bay, 30 miles northwest of San Francisco. Shaded area depicts the extent of the October 1995 fire, which burned 12,300 acres along the crest and southwest slopes of the ridge in Pt. Reyes National Seashore and into private land in Inverness Park. Historical and recent sites of Lepidoptera inventory are indicated by circled numbers:

1) **Pierce Pt. Road, 3 miles west of Inverness (Bauer and Buckett, 1940s -1965;**
2) **Sea Haven district (Powell, 1994-95);**
3) **Inverness (Bauer & Buckett, 1940s-50s; Patterson, 1950s-present; Toschi, 1950s- 1963; Powell 2000);**
4) **Inverness Park (O'Brien, 1959-61, Powell 1995-99);**
5) **Summit area at terminus of Drakes View Rd. (U.C. Berkeley Field Entomology classes, 1970-82).**

in marked contrast to generalizations about effects of fire on arthropods derived from fire management of grasslands.

Introduction

A wildfire in early October 1995, at the end of California's long dry season, burned an extensive portion of Inverness Ridge, from Mt. Vision eastward into private land, where more than 40 homes were destroyed the first day, then southward to Drakes Bay (Fig. 1). Its effects were most dramatic higher on the ridge, where the pine forest and its understory vegetation were destroyed, and ground litter was burned to bare soil. The fire was less intense down slope, where vegetation in chaparral and grassland burned, but most plants were not killed, diminishing to an understory fire in the canyons, where alder- and willow-dominated riparian woods partially survived.

There is a wealth of literature on effects of fire on insects and spiders, but most studies deal with managed fires in prairie and other grasslands habitats where rotational plots can be compared — that is, they assess communities of insects that are fire-tolerant and persist where periodic burning has been practiced. Most have involved short term monitoring, i. e., a few months or a year, and often the insects have been recorded by biomass or order and family level taxa, not species. No study has comprehensively assessed Lepidoptera, the major group of plant feeding insects, recording species of all kinds, from tiny leaf miners to large moths and butterflies, and none has attempted to correlate colonization by larval host plants. I proposed a 3-year study to document the recovery of all Lepidoptera and their hosts and extended it to 5 seasons.

Background

The fire history on Inverness Ridge is not documented in detail. There had been no fires in the area I monitored for 30 years prior to 1995, after ownership of the land shifted from private ranchers to the National Park Service. Earlier records are fragmentary. Large fires occurred in the area in the 1880's and 1927, and ranchers routinely used fire management to suppress brush in grazed grasslands (Sugnet 1985, based on newspaper articles and other sources). Robert Ornduff estimated Bishop pines killed in the 1995 fire to be 30-45 years old, from counts of tree rings in stumps on private land (Ornduff and Norris 1997). However, there was a mature pine forest in 1970, when I first visited and photographed that area, indicating no major fire for at least 25-30 years before that time. Probably the Bishop pine community was at least 50 to 68 years old (dating to the 1927 fire). The chaparral community and grassland of the lower slopes and riparian woods along Muddy Hollow Creek were fire free for 30 years or more, in a mosaic pattern.

There was no comprehensive baseline inventory of the moths and butterflies that lived in the Pt. Reyes National Seashore. Pre-fire insect survey of the ridge top and seaward slopes of Inverness Ridge that burned was limited to that conducted during annual 2-day visits by a field entomology class from the University of California, Berkeley, in late April or May during the 1970s, in the area of the western terminus of Drakes View Road. Hence, a post-fire census of species could not be compared directly to a pre-fire community, and I tracked recolonization of selected plants, especially woody shrubs and trees, which harbor the richest communities of Lepidoptera.

Methods

Inventory of insects presents several problems: most species are present in life cycle stages that are easily sampled for only short seasonal periods; many populations undergo enormous fluctuations in individual numbers so may be overlooked or considered rare in a given year; and the success of most sampling methods is dependent upon insect activity and therefore local weather, especially temperature. Predicting weather conditions at Pt. Reyes is inexact, so sampling consistently in ideal conditions was not possible. Strong, cold onshore winds are prevalent, with coastal fog persisting unpredictably on the seaward, burned side of the ridge, especially in spring and summer. Warm conditions in the region, even at Inverness on the east side of Inverness Ridge, often give way to cold, blowing fog on the ridge top and west slope.

Area sampled

The area selected is defined on the west side by the Drakes View trail, on the south by Bayview trail from Inverness Ridge trail to Muddy Hollow Road, and to the east along Inverness Ridge trail from Limantour Road to Drakes View Road, a perimeter distance of about 6 miles (9.6 km) (Fig. 1). The triangulate plot thus defined provides a transect from the more severely burned pine forest at 1,000 ft. (300 m) elevation on the ridge top, descending through chaparral and grassland, to riparian gallery forests along Muddy Hollow Creek at 100 ft.

(30 m). Most of the sampling after two years following the fire was done along the perimeter trails, because much of the area became impenetrable thickets of dense vegetation growing over fallen trees, dominated by pine, blue blossom, blackberry, and poison oak on the ridges, and nettle and blackberry in the canyons. By years 4 and 5 some chaparral and canyon areas became more accessible, with willow saplings or shrubs replacing the dense low thickets, but on the ridges incredibly dense stands of young pine, often 12-18 trees per square m, which had grown to 4-4.5 m (12-14 ft.) high, rendered many areas inaccessible and crowded out more varied post fire vegetation.

Sampling techniques

Monitoring was designed to maximize observations of diversity rather than to monitor single sites or standardized transects. The goal was to record colonists of plant species as vegetation reappeared and grew to stabilized structure, the diversity, occurrence, and succession of which could not be predicted. Observations and collections were made by three approaches: diurnally, either 1) by observing or netting adults, and 2) visual search and beating sheet sampling of selected plants for caterpillars and larval mines or galls; and 3) by nocturnal attraction of adult moths to 8- or 15-watt ultraviolet lights ("blacklight") powered by small rechargeable batteries. Adult butterflies were recorded mostly by sight, adult moths mostly by net or blacklight trap collection. Caterpillars were transported to Berkeley for lab rearing.

Abandoned leaf mines were pressed and dried. If host plant identifications were required, small samples were pressed and submitted to the University of California Herbarium. In general, plant feeding insects occur in greater species richness on woody plants of high architecture, i. e., trees, larger shrubs, then successively fewer on smaller perennials, herbs, and monocots (Lawton 1983), although grasses as a group harbor many Lepidoptera. Our sampling emphasized trees and shrubs, including pine (*Pinus*), alder (*Alnus*), willow (*Salix*), blackberry and thimbleberry (*Rubus*), currant and gooseberry (*Ribes*), coyote brush (*Baccharis*), oak (*Quercus*), madrone (*Arbutus*), manzanita (*Arctostaphylos*), huckleberry (*Vaccinium*), blue blossom (*Ceanothus*), coffeeberry (*Rhamnus*), and poison oak (*Toxicodendron*).

We recorded Lepidoptera on 82 dates from March 14, 1996 to November 1, 2000, during which we made 45 diurnal visits, 263 larval and leaf mine collections, and 54 blacklight samples on 45 dates. Daytime survey, usually of 2.5-6.5 hours, was conducted approximately monthly from mid March to late October, with sporadic additional visits, mostly in May and October. After initial trials, I used four light trap sites at different elevations, situated 0.8-1.2 miles (1.3 to 2.0 km) inside the perimeter of the fire zone, at spots sheltered from sight of unburned areas by intervening hills (Fig. 1). Additionally, in 1999 M. Hart, made six evening visits to collect at light sheets, mostly during January to April, when we had few records. Blacklight samples were initiated one year after the fire and were made in all months except December and June, 75% of them in May or September-October.

Estimating expected species richness

To provide a basis for estimating potential caterpillar guilds of individual plants, I compiled a list of lepidopteran species that had been collected historically and recently on Inverness Ridge, including areas outside the fire. In addition to our collections with entomology students in the 1970s, sporadic collections had been made by amateur collectors and university students from 1940 to present, in and near Inverness (see Acknowledgments and Fig. 1). I sampled moths at lights on about 175 dates between July 1994 and November 2000, mostly in Inverness Park (Fig. 1). In total, there have been moth collections at lights in all months of the year, more extensively in May-October, but there had been few larval collections prior to my post-fire survey. From the collection data, I abstracted lists of Lepidoptera known to be larval host plant specialists of selected plant taxa. As the study progressed I added specialist species that had not been recorded in the Inverness area previously, as well as generalists that were found feeding on each plant in the burn zone, to derive potential lepidopteran communities for each plant. For example, I found larvae of 6 species feeding on nettle (*Urtica*), 4 specialists and 2 generalists, adults of one additional *Urtica* specialist were trapped at blacklight in the fire zone, and another specialist was collected in Inverness Park = total expected guild of 8. Lists of species recorded at Inverness Ridge and their larval host plants are posted on our Essig Museum website (http://essig.berkeley.edu/doc/invernessleps.htm.

Results

In early October most insects are dormant, in mines or larval galleries in plants, or buried in ground litter or soil. Therefore, many species were not affected by temporary loss of larval foods, and some were protected, but where the

fire was most intense, as in the pine forest, even ground dwellers were destroyed. As might be expected, few Lepidoptera survived or quickly colonized the fire zone — we found adults or larvae of only 16 species during our first visit in March 1996, five months following the fire. As native vegetation returned, many moths and butterflies reappeared, and numbers of lepidopteran species observed steadily climbed during five seasons (Fig. 2). We recorded 376 species, 33 butterflies, 343 moths (+ ca. 10 species of uncertain taxonomic status), of which we found caterpillars or larval leaf mines of 118 (31%). Among species observed only as adults, 121 are known to be larval host plant specialists (32% of the total), so larval food associations were projected for 63% of the species recorded. These feed on 65 plant taxa. Most of the remainder are generalist plant feeders, fungivores, or detritivores, or their larval habits are unknown.

The observed species accumulation was continuous, enhanced by increased blacklight sampling effort in years 4 and 5; 50% of the 5-year total were recorded by May 1998 (24 sampling dates) and 75% by September 1999 (58 dates), nearly four years after the fire (Fig. 2). Butterflies were recorded sooner than moths, 48% by the third survey date, 64% by the end of year 1 (6 dates), and 75% within 18 months following the fire (9 dates) (Fig. 3). Leaf mining and gall causing species, all of which are tiny moths believed to be of limited dispersal tendencies, were recorded at a faster rate than larger, presumably stronger flying moths: 50% by the end of year 2 (20 dates) and 75% by the end of year 3 (38 dates) (Fig. 2).

The species can be grouped in three cohorts: 1) *Transitional species*, those that colonize early vegetational components, then disappear or become rare as plant succession proceeds; 2) *Long term residents*, species that colonize later stage plants, especially woody shrubs and trees; and 3) *Vagrants*, non resident individuals that migrate through or immigrate as temporary residents or casual visitors.

Transitional species

Characteristically in California, herbaceous vegetation appears profusely in the first spring following a fire, diminishing in succeeding years. This flora has been documented by several authors (e.g. Cosy and Mooney 1978, Keeley and Zedler 1978). In chaparral communities the first year post-fire flora attains a higher species richness than normally found in other post-disturbance, such as flooding or landslides. This richness begins to decline by the second year and does not increase to a later maximum, shown by post-fire studies in some other vegetation types (Smith and James 1978). In spring following the fire at Inverness Ridge, there was an impressive flush of new growth, especially herbaceous vegetation, dominated by weedy grasses, vetch (*Vicia*), poison hemlock (*Conium*) and other exotics, as well as cow parsnip (*Heracleum*) and other native plants. This set the stage for opportunistic colonization and abundance of some Lepidoptera

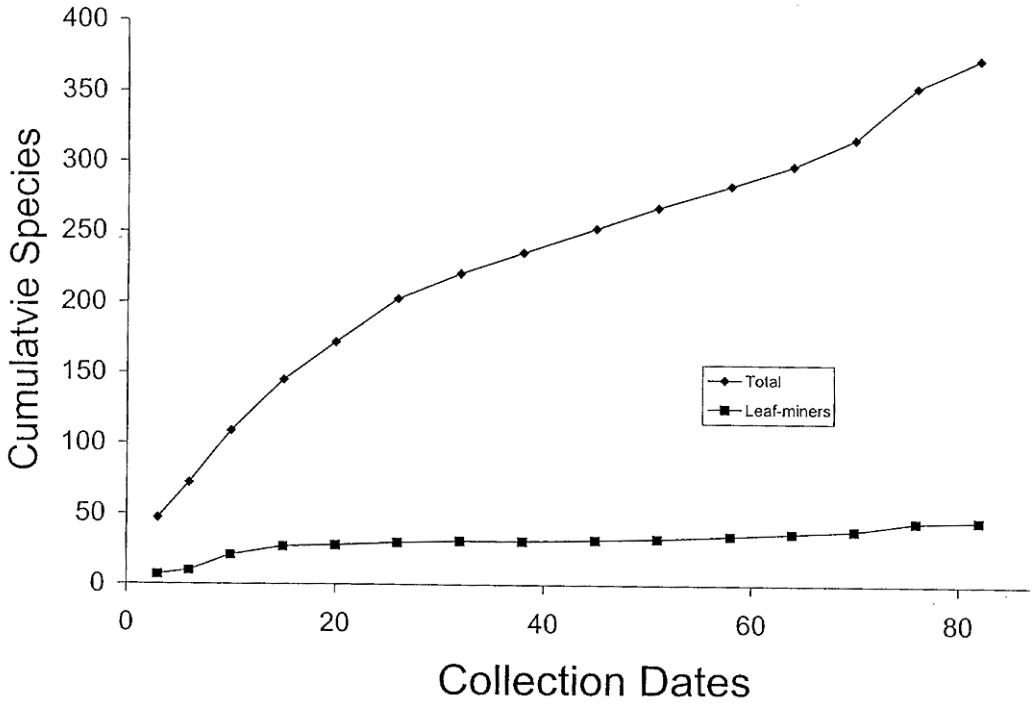

Fig. 2
Species accumulation during 1996-2000 in the fire zone at Inverness Ridge. Upper graph: all Lepidoptera (n=376); lower: leaf-mining species only (n=46)

during the first two or three years, until this community of rapid growth plants was succeeded by the more slowly developing shrubs and trees.

The most spectacular examples of short-lived, fire-adapted plants at Inverness Ridge were several legumes, including a bush lupine and extensive mats of bird's-foot trefoils (*Lotus*) and clovers (*Trifolium*), mixed with broom-rose (*Helianthemum scoparium,* Cistaceae). All of these grew from seeds that evidently had been dormant in the soil for several decades, and in a "Big Bang" kind of reproductive strategy, the legumes flowered for one or two seasons, leaving a replenished seed bank, then disappeared.

The bush lupine was judged by Ornduff (1998) to be a form of *Lupinus arboreus*, although it differs from nearby beach dune populations of that species in having blue flowers rather than yellow, short inflorescence stems, and by its fire-adapted life history. Although this lupine had been absent or very rare in recent decades (Ornduff & Norris 1997, Ornduff 1998), it appeared in an almost continuous stands in deeper soils of the former pine forest and much of the subtending chaparral slopes, becoming 1-1.5 m (4-5 ft.) high shrubs by the end of the first year, and flowering in unison in the second spring in a tremendous display that painted the hills in vivid blue, after which every plant that set seed died. A few plants or parts of plants that failed to bloom did so in year 3 then died, leaving only scattered new plants, mostly in disturbed spots along the trails.

The legume mats, determined by Ornduff to consist of 5 species of *Lotus* and 2 of *Trifolium*, occupied all the ridge areas characterized by poor soils. *Lotus hiermanni* var. *orbiculatus*, which was judged to be the most abundant species, had not been known in the National Seashore (Fellers et al. 1990). Except for sporadic *L. junceus* in disturbed soil along the paths, all the native *Lotus* disappeared after the second year.

Colonization by Lepidoptera quickly followed. Two species of widespread native butterflies, the orange sulphur (*Colias eurytheme*) and the acmon blue (*Plebejus acmon*), appeared abundantly in association with the *Lotus* by the end of year 1, along with several inconspicuous moth species, all of which declined in numbers as the colonies of *Lotus* were replaced by pine and manzanita thickets in later years (Table 1). *Acmon* persists, dependent upon the introduced *Lotus* corniculatus that grows along the disturbed trail edges. Similarly, nettle (*Urtica*) was abundant in the canyons the first two years following the fire, then declined as it was crowded out by young alders and other riparian growth. Four species of Lepidoptera that feed on nettle were numerous in early years of succession but rare in later years (Table 1).

The most remarkable example of such succession is *Helianthemum* and a tiny black moth of the genus *Mompha*. This plant, which had been listed of doubtful occurrence in the National Seashore (Fellers et al. 1990), formed extensive turf-like patches by the thousands, interspersed

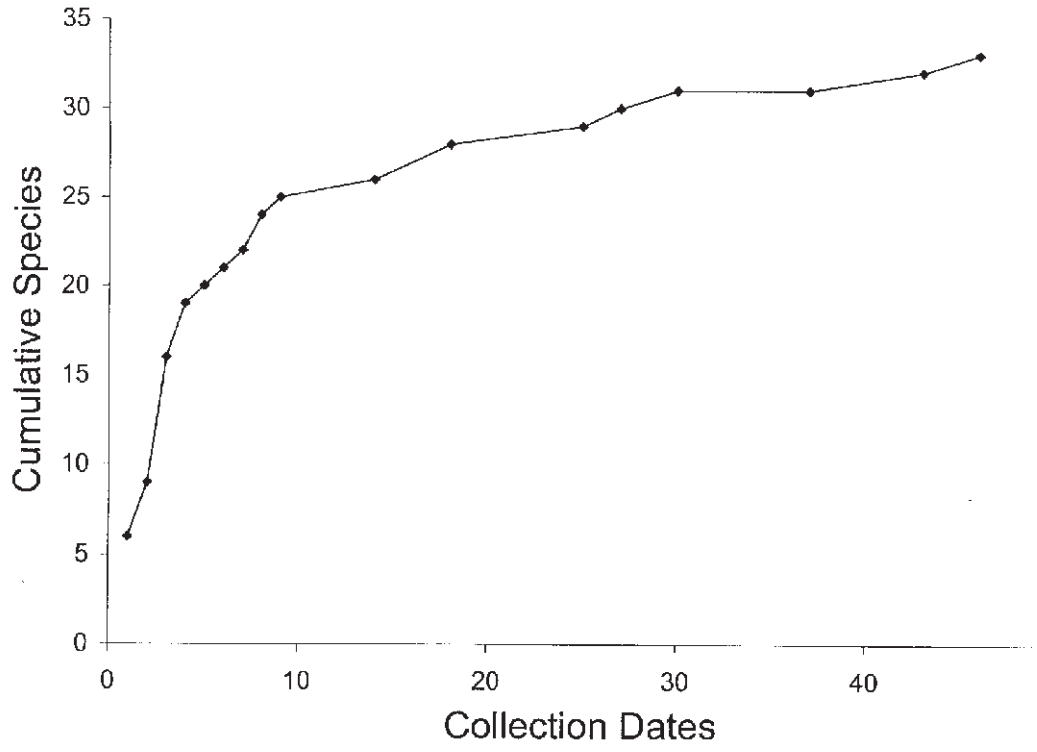

Fig. 3
Accumulation of butterfly species (n= 33) during 1996-2000 in the fire zone at Inverness Ridge.

with *Lotus*, and after five years persisted on decomposed sandstone soils where not supplanted by manzanita and pine. In mid 1997 I began to find the *Mompha* associated with this plant and discovered its larvae feeding in the seeds. I had not seen this moth before, and a specialist has determined it to be an undescribed species. Its populations occurred in enormous numbers, sometimes 300-600 specimens in a single 8-watt blacklight trap, and did not diminish appreciably through year 5. Based on past behavior of *Helianthemum* (Howell 1949) following large fires, in time the plant and its seed predator will become very rare again.

Numerous other lepidopteran species passed through stages of abundance then became rare in later years (Table 1), either because they used weedy early succession plants (e.g. the noctuid moth *Heliothis phloxiphaga*, which is a generalist often feeding in flowers of herbs), or their native plant host was crowded out. For example, the mustard white butterfly, *Pieris napi*, feeding on milk maids, *Cardamine*, which was abundant in year 1, and the butterflies flourished through three or more generations, flying until October, highly unusual for this species. But its habitat changed as dense ground cover replaced the shaded understory of the riparian alder-willow woods, and butterfly numbers declined abruptly to only a few seen in years 4 and 5 (Table 1). In some instances the initial abundance may have resulted from a release from natural parasites and predators that normally regulate insect populations. Later host moths diminished in abundance biotic controls reestablished . Examples include the gracillariid leaf miners, *Caloptilia* and *Phyllonorycter* on *Ribes sanguineum*, and *Epermenia* on cow parsnip (*Heracleum*). Tiny larvae of *Epermenia*, which feed gregariously on undersides of the leaves and create conspicuous skeletonized patches, were already abundant in March 1996, five months after the fire, and remained so for two seasons, then gradually became scarce (Table 1), although the host plant remained abundant.

Long term resident species

Recovery of species believed to be long term residents was tracked by searching selected plant species for larvae. Collection of adults of species known to be host specific on those plant species provided additional evidence of colonization. Plants that support guilds of numerous caterpillar species, emphasized during this survey, are grouped as characteristic of three communities: *Riparian*, *Chaparral*, and *Bishop pine forest* (Tables 2-4).

Table 1. Transitional Species — moths (m) and butterflies (b) that were observed multiple times during years 1, 2, or 3 then became rare in years 3-5, 4-5, or 5. Numbers designate number of dates observed; * designates numerous individuals.

	'96	'97	'98	'99	'00
Lotus, *Lupinus* feeders:					
Aristotelia sp. (m)	2	0	1	1	0
Syncopacma sp. (m)	2	5	0	0	2
Epinotia infuscana (m)	-	4	4	1	2
Uresiphita reversalis (m)	-	2	0	0	0
Semiothisa californiata (m)	-	9	6	0	0
Colias eurytheme (b)	5*	8*	5*	0	0
Plebejus acmon (b)	5*	9*	6	7	3
Urtica feeders:					
Anthophila alpinella (m)	2	6*	0	1	0
Hypena californica (m)	2	4	2	3	1
Polygonia satyrus (b)	5	7	3	4	1
Vanessa atalanta (b)	4*	5	7	2	1
Brassicaceae feeders:					
Euchloe ausonides (b)	-	3	1	0	0
Pieris napi (b)	4*	3	2	2	1
Pieris rapae (b)	1	2	5	0	0
Heracleum feeder:					
Epermenia californica (m)	4	3	1	2	1
Generalists:					
Amblyptila pica (m)	2	0	1	0	0
Heliothis phloxiphaga (m)	2	3	0	0	0

The Riparian Community along Muddy Hollow Creek experienced a comprehensive burn, but its intensity was considerably lower than on the ridges and was mosaic in distribution. Hence, there was an understory fire in some areas, with most buckeye (*Aesculus*), bay trees (*Umbellularia*), and some alders surviving, while willows and most low-growing vegetation burned to the ground. The former shaded, riparian corridor mostly disappeared, and a rich mixture of herbaceous vegetation, including stump sprouted willows, alder seedlings, blackberry and other species of *Rubus*, nettle, cow parsnip, and adventive weeds such as poison hemlock flourished.

Regrowth of willow and alder to nearly full stature was much more rapid than for trees and shrubs of the ridges, such as the pine, oak, and manzanita. As a result, many Lepidoptera adapted to canyon plants survived or quickly recolonized (Table 2). Nearly all expected nettle feeders were established by the end of year 1, and species associated with other riparian plants reestablished relatively quickly, 67% of the expected total by year 3, 87% by year 4, and 92% after 5 years (Table 2).

Plants of the intermediate elevation Chaparral Community are mostly shared with the pine or riparian communities (e.g. *Baccharis, Rhamnus, Lotus, Rubus ursinus, Toxicodendron*); Lepidoptera feeding on those that are most characteristic of the dry, open westerly facing slopes are summarized in Table 3. These species were slower to colonize than those on riparian plants. Particularly surprising was the *Baccharis*-feeding guild because coyote brush grew rapidly from stump sprouts and is dominant over large areas, forming a seemingly ideal island for recolonization. Yet none of its numerous Lepidoptera was observed in year 1, and fewer than half the expected total until year 4. One-third of potential grass (*Poaceae*)-feeding species were not recorded. Possibly following 30+ fire- and grazing-free years of brush intrusion the grassland is too limited in size, species richness, or habitat diversity, to support more of the region's grass feeders. Overall, nearly half the expected caterpillar fauna of the chaparral and grassland plants had established by the end of year 3, and 76% after year 5 (Table 3).

Reestablishment was even slower in the Bishop pine community. As noted, much of the formerly forested ridge crest was covered at first by legumes growing from seed banks. Later stage plants are adapted to survive fire in different ways, and most either grew from stump sprouts, e.g., coffeeberry (*Rhamnus*), huckleberry (*Vaccinium*), madrone (*Arbutus*), tanbark oak (*Lithocarpus*), or the mature plants were replaced by fire-adapted seed germination, notably Bishop pine (*Pinus muricata*), and Marin manzanita (*Arctostaphylos virgata*). These require much longer growth periods to reach stages suitable for support of insect populations. For example, *Arctostaphylos* seedlings were not in evidence until late spring 1996, when many other plants already had been colonized by caterpillars, and did not achieve appreciable standing as shrubs until year 3. Coast live oak (*Quercus agrifolia*) survived the fire in another way. Entire trees burned, but the trunks and larger limbs survived, giving rise to epicormic vegetative growth. The new foliage was subject to mildew during the first season, and trees did not develop appreciable canopy foliage until the 2nd or 3rd year. Therefore, reestablishment of the caterpillar guilds of the ridge plants lagged behind that of other communities: only 12% of the expected fauna were recorded in year 1, 47% by year 3, and 61% after 5 years (Table 4).

Vagrants and short-term resident species

Census of Lepidoptera, many of which possess strong dispersal tendencies, inevitably encounters individuals that are not resident in the local area. Transients often are difficult to identify as such because one or a few records may result from migrants or vagrants, or they

Table 2. Post fire recovery of Lepidoptera guilds on Riparian plants: Cumulative larval host plant use, based on larvae, mines, and projected from adults of specialist species.

Year:	1	2	3	4	5	Expected
Alnus	2	7	7	10	10	11+
Salix	5	9	11	14	17	17+
Ribes	0	3	4	5	5	7
Rubus	1	3	4	5	5	5+
Urtica	6	6	6	7	7	5
Heracleum	1	3	3	4	4	4
Total	**15**				**48**	**52+**
	27%				**86%**	

may be resident species that escape notice because their seasonal activity is primarily restricted to times other than the survey effort, or they may have small populations located away from transects sampled. In general, I regarded all species for which I found larval activity as resident, as well as those seen as adults on multiple dates, and smaller microlepidoptera were assumed to be non-migrant. Larger moths and butterflies recorded only once are suspect vagrants, but those seen only in year 4 or 5 may be new colonists. At least 15 of the 376 species recorded likely were non-resident.

Discussion

Recovery of plant-feeding insect populations following an extensive wildfire is a function of several factors, especially intensity of the fire, its seasonal timing, and the size of the area burned in relation to availability of nearby plant/insect communities that can serve as recolonization sources. If the fire is intense, occurs during a season when many insect species are most vulnerable as adults or larvae, many species likely will be eradicated from the site. If the burn area is sufficiently large to support insect populations that depend on plants that are unique in the region (i.e., no access to colonization sources nearby), the time to recovery may be very long. Conversely, insects may survive in local spots if the fire is variable in intensity and affects the habitats in a mosaic pattern, as is typical of wildfires in diverse topography, and/or it occurs at a season when most insects are dormant (e.g., as pupae on the ground). If the burn zone is adjacent to or surrounded by similar habitat harboring colonies of the insects, recovery likely will occur relatively rapidly through immigration and colonization.

The fire at Inverness Ridge, while moderately large (5,000 ha) and extremely intense on the higher ridges, occurred with attributes generally favorable for recovery of its herbivorous insects. As the fire proceeded downslope during days 2-5, it diminished in intensity and varied, becoming only an understory burn in places, and many trees and larger shrubs survived. Moreover, the area is bordered on all sides except along Drakes Bay by extensive areas of native vegetation, including all major communities in the burn area (Fig. 1). Hence, the situation was favorable for recovery of vegetation and associated insect populations.

Recovery of the pre-fire community

Except for butterflies, species accumulation did not stabilize after 5 years (Figs. 2, 3), although recorded recovery rates are partly artifacts of sampling error. Butterflies are more easily observed than moths, so they are detected more readily. Inventory of larger moths is primarily dependent upon attraction of adults to lights, and effectiveness of sampling varies with seasonal flight period, temperature and wind conditions, moon phase, positioning of traps, and activity of vertebrate predators at the traps. Adults of some microlepidoptera are diurnal, and those of tiny nocturnal species are not reliably sampled by light traps. However, larvae of shelter makers in foliage are readily discovered, and leaf mines and plant galls induced by some microlepidoptera can be recorded by experienced observers because they persist throughout the season and can be recognized long after the larval feeding period.

Recolonization of particular plant species by larval Lepidoptera was more rapid in less intensely burned habitats. Species feeding on plants in the riparian community recovered sooner and more completely, compared to their projected caterpillar guilds (92%), than in chaparral and grassland (76%), and the more severely burned bishop pine community (61%) (Tables 2, 3, 4). Pooling data from these three plant lists yields an estimate of 74% of the expected total Lepidoptera fauna as a baseline estimate of the proportion of pre-fire species that colonized within 5 years.

Table 3. Post-fire recovery of Lepidoptera Guilds on Chaparral and Grassland plants: Cumulative larval host plant use, based on larvae, mines, and projected from adults of specialist species

Year:	1	2	3	4	5	Expected
Anaphalis/Gnaphalium	3	5	7	7	7	7
Rhamnus	3	5	5	6	6	7
Baccharis	-	4	5	9	11	14
Toxicodendron	-	-	-	-	3	4
Poaceae	10	11	13	17	20	30
Total	**16**				**47**	**62**
	26%				**76%**	

A total of 602 species of Lepidoptera, 40 butterflies and 562 moths (+ about 30 of uncertain species status) have been recorded during the past 60 years in the Inverness Ridge area, whereas 376 (+10 ?) species were found in the fire zone during the 5 seasons. Obviously not all the species in the region are expected to have lived in the area sampled because small portions of any area harbor fewer habitats and species than the whole area. For example, the east slope of the ridge has a dense, shaded, and, more diverse forest type, dominated by Douglas-fir (*Pseudotsuga*) and tanbark oak (*Lithocarpus*), both of which were rare in the burn area I monitored. If we assume butterflies are completely censussed, we can project a percent recovery based on them. Of the total in the region, 34 species likely are resident on Inverness Ridge (several others live in adjacent marsh and beach habitats), and 28 of 33 species observed in the fire zone are assumed to be resident (82% of the resident ridge fauna). If moth diversity is comparably distributed, 480 species (80% of 600) might have comprised the pre-fire community, and the observed 376 species recorded make up 77% of the expect fauna (Table 5). Probably this is an optimistic projection because inventory of the region as a whole is incomplete — there has been minimal effort to collect and rear larvae other than in the fire zone — 650-700 species is a more realistic estimate of the fauna. If true, 80% of the total projects to 520-560 species expected in the fire zone, of which 60-65% were recovered.

Post-fire recovery of Lepidoptera contrasted with insects in other habitats

This study demonstrates dramatic differences in the effects of fire on insect herbivores among plant species and vegetation types within complex plant communities. Contrasting the recovery at Inverness Ridge with other post-fire studies of arthropods is problematic because none has attempted to comprehensively assess moth species, which make up 95% of the Lepidoptera. Traditional fire management for various uses has been contentious (e.g., in prairie grasslands in the midwestern U.S., pine plantations in the southeast, heaths in Europe, eucalypt forests in Australia), and there have been many studies of the effects of prescribed burning on arthropods (reviews by Warren et al. 1987, Folkerts et al. 1993, Swengel 2001, Panzer 2002). Such studies usually compare recently or annually burned with unburned and/or less frequently burned, comparable small plots of habitat in areas that have been subjected historically to periodic burning. Therefore, they assess recovery response among arthropod communities that tolerate periodic fires and exist in spite of or even because of them. Typically, standard sampling methods have been used in order to make statistical comparisons between burned and unburned plots, primarily by pitfall traps, flight intercept traps, or sweep net samples along fixed transects. Most have identified insects only to order or family level. Those that have listed insect species usually focus on Orthoptera (grasshoppers), Hemiptera (true bugs, leafhoppers) or butterflies as herbivores, and spiders or carabid beetles as predators. Evaluations often assess short (0-2 months) or intermediate term (2-12 months) effects.

In general, the pattern revealed by numerous such studies is high species richness and abundance, or biomass (often dominated by grasshoppers), during the first season following prescribed fires, as compared to unburned plots or second and third year post-fire samples. Because prescribed burning aims to suppress invasive shrub and understory growth, short-term studies following them measure primarily the rapidly developing cohort I have termed Transitional Species. Managed grasslands, savanna, and pine plantations are prevented from evolving in stages to shrub dominated communities and wood lots, which would support more diverse, long-term Lepidoptera communities. As might be expected, these sam-

Table 4. Post-fire recovery of Lepidoptera Guilds on Bishop Pine forest plants: Cumulative larval host plant use, based on larvae, mines, and projected from adults of specialist species.

Year:	1	2	3	4	5	Expected
Pinus	3	3	5	5	6	15+
Arbutus	-	2	3	3	3	5
Arctostaphylos	-	1	7	9	10	14
Vaccinium	1	3	5	5	5	5
Quercus	4	11	16	19	22	37+
Ceanothus	4	11	13	15	16	18+
Total	**12**				**62**	**94+**
	13%				**66%**	

pling regimes recover insignificant numbers of Lepidoptera relative to other arthropods tallied (e.g., 2% of all arthropods sampled by Nagel 1973 and Siemann et al. 1997).

Nonetheless, generalizations are offered for arthropods as a whole. For example, Nagel (1973) found significantly greater numbers and biomass of herbivore arthropods on a burned site than on unburned, but the difference in Lepidoptera was insignificant and 9% higher in the unburned plot. Siemann et al. (1997) concluded that burns necessary to maintain grassland and oak savanna do not appear to be harmful or to greatly impact the arthropod fauna; yet a visual count averaged 47 butterflies per unburned and infrequently burned vs. 18 per frequently burned plot, and light traps produced 22 moth species/sample in unburned and infrequently burned vs. 16/sample in frequently burned plots.

Moreover, there are contradictory results from some studies of grassland-,soil-, and litter-dwelling invertebrates. Buffington (1967) found fewer taxa and individuals of soil arthropods one year after a wildfire in New Jersey pine barrens, and litter arthropods studied by Metz and Farrier (1973) were less abundant in annually burned loblolly pine woods in South Carolina than in unburned or less frequently burned plots. Research by Nekola (2002) clearly showed that fire management causes significant reduction in land snail richness and abundance and most significantly impacts the rarest species. Panzer (2002) assessed effects of prescribed burning in 21 tallgrass prairie reserves in Indiana, Illinois and Wisconsin. He found negative post fire responses in 40% of 151 species representing 33 families of insects, positive responses in 26%. Among negatively affected populations, all recovered within two years. Butterflies and *Papaipema* (Noctuidae) moth species, which are stem borers in herbs, comprised an appreciable proportion of the remnant-dependent species examined.

Force (1981) tracked insect succession in a chaparral community in southern California for 3 seasons following a wildfire in November. He conducted observational transects along a 1.5 km trail, similar to my sampling approach, several times monthly, augmented by thorough census of 10 plots annually. He found insect family richness and diversity and species richness to be highest in year 1, declining in years 2 and 3, while species diversity was highest in year 1, lowest in year 2. He did not differentiate among detritivore, herbivore, parasitoid, and predator taxa in the analysis and did not report the insect families tallied, so the representation of Lepidoptera is unknown. Force believed the large food resource available to foliage-, sap-, and pollen and nectar-feeding insects attracted many vagrants from outside the burn.

Implications for fire-management policy

A fallacy in assuming that diverse and lush post-fire vegetation and the accompanying richness of some insects represents richness of all insects lies in the nature of the early post fire vegetation. Much of it consists of weedy herbs and native annuals, which serve as larval food for few Lepidoptera, whereas the later succession shrubs and trees provide resources for the great majority of moth larvae. Although post-fire vegetation was lush at Inverness Ridge initially, Lepidoptera were low in species richness and diversity in year 1, then gradually increased as the long-term, high-architecture vegetation became established. Lepidoptera make up the most diverse group of plant-feeding insects, and more state and federally protected species in the U.S. are Lepidoptera than any other insects. The adults and caterpillars provide an important food resource for many vertebrates, including lizards, rodents, bats, and birds. The fate of these herbivores should not be subsumed within management decisions made on the basis of generalists (e.g. grasshoppers) and other early succession insects (leafhoppers, spittlebugs), or carabid beetles, spiders, and soil arthropods in fire-adapted communities. In addition, at Inverness, butterflies were recovered appreciably sooner than leaf mining taxa or Lepidoptera as a whole (Figs. 2, 3), which suggests that they comprise a group that

Table 5. Summary of Lepidoptera recovery five years after the Mt. Vision Fire.

	Species Observed	% of Area total	% of Projected
microlep	148	67	84
pyraloid	28	51	65
macro moth	167	55	70
butterfly	33	80	100
Total	**376**	**60**	**77**

includes Transitional Species and is easily censussed but is not an indicator of typical rates of recolonization by other plant-feeding insects.

Caution should be exercised in application of generalizations produced from study of postfire recovery in periodically burned grasslands and savannas, based on pooled data from diverse arthropod taxa (detritivores, herbivores, predators). Policy decisions concerning use of prescribed burning for weed suppression or other management, such as in national wildlife refuges, national parks, need careful consideration of long-term goals and effects on the native insect fauna.

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Literature Cited

Buffington, J. D. 1967. Soil arthropod populations of the New Jersey pine barrens as affected by fire. Annals of the Entomological Society of America 60: 530-535

Cody, M. L. and H. A. Mooney 1978. Convergence vs. nonconvergence in Mediterranean Ecosystems. Annual Review of Ecology and Systematics 9: 265-321.

Fellers, G. M., V. Norris, and W. Follette 1990. Point Reyes National Seashore Plant Checklist. Pt. Reyes National Seashore Association, Pt. Reyes, CA; 41 pp.

Folkerts, G. W., M.. A. Deyrup, and D. C. Sisson 1993. Arthropods associated with xeric longleaf pine habitats in the southeastern United States: A brief overview. Proceedings of the Tall Timbers Fire Ecology Conference, 18: 159-192.

Force, D. C. 1981. Postfire insect succession in southern California chaparral. American Naturalist 117:575-582.

Howell, J. T. 1949. Marin Flora. University of California Press, Berkeley and Los Angeles; vii + 323 pp.

Keeley, J. E. and P. H. Zedler 1978. Reproduction of chaparral shrubs after fire: a comparison of sprouting and seeding strategies. American Midland Naturalist 99: 142-161.

Lawton, J. H. 1983. Plant architecture and the diversity of phytophagous insects. Annual Review of Entomology 28: 23-39.

Metz, L. J. and M. H. Farrier 1973. Prescribed burning and populations of soil mesofauna. Environmental Entomology 2: 433-440.

Nagel, H. G. 1973. Effect of spring prairie burning on herbivorous and non-herbivorous arthropod populations. Journal of the Kansas Entomological Society 46: 485-496.

Nekola, J. C. 2002. Effects of fire management on the richness and abundance of central North American grassland land snail faunas. Animal Biodiversity and Conservation 25: 53-66.

Ornduff, R. 1998. Three years after the Vision Fire in Point Reyes. Fremontia 26(3): 26-27.

Ornduff, R. and V. Norris 1997. Rebirth of a Bishop Pine forest: First year after the Mount Vision fire. Fremontia 25(3): 22-28.

Panzer, R. 2002. Compatibility of prescribed burning with the conservation of insects in small, isolated prairie reserves. Conservation Biology 16: 1296-1307

Powell, J. A. 1995. Lepidoptera inventories in the continental United States. pp. 168-170 in: LaRoe, E. et al., (eds.) Our Living Resources.

A Report to the Nation on the Distribution, Abundance, and Health of U. S. Plants, Animals, and Ecosystems. U. S. Department of Interior, National Biological Service, Washington, D.C.

Siemann, E., J. Haarstad, and D. Tilman 1997. Short-term and long-term effects of burning on oak savanna arthropods. American Midland Naturalist 137: 349-361.

Smith, D. W. and T. D. W. James 1978. Changes in the shrub and herb layers of vegetation after prescribed burning in *Populus tremuloides* woodland in southern Ontario. Canadian Journal of Botany 56: 1792-1797.

Sugnet, P. W. 1985. Fire history and post-fire stand dynamics of Inverness Bishop Pine populations. M. S. Thesis, University of California, Berkeley; v + 82 pp., tables.

Swengel, A. B. 2001. A literature review of insect responses to fire, compared to other conservation managements of open habitat. Biodiversity & Conservation 19:1141-1169.

Warren, S. D., C. J. Scifres, and P. D. Teel 1987. Response of grassland arthropods to burning: a review. Agriculture, Ecosystems, and Environment 19: 105-130.

Community Dynamics of Ectomycorrhizal Fungi Following the Vision Fire

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Introduction

Many people that read this chapter may be aware of mammals, birds, and maybe even plant species that are common at Point Reyes, and may have witnessed or have been aware of the ways that these organisms responded to the Vision fire of 1995. Few will be familiar with fungal species at Point Reyes, and even fewer may have noticed their response to the fire. This is unfortunate, because fungi play many critical roles in structuring and maintaining plant (and therefore also animal) communities at Point Reyes. It is understandable, however, because even those of us that study fungi for a living know less about them than plant or animal biologists typically know about their organisms. There are several reasons for our collective ignorance. Fungi can be difficult to identify, so reliable species records are rare. For most of their lives, fungi are microscopic and hidden in soil or plant parts; this makes it difficult to observe their basic life histories and ecology. Finally, relatively few people study fungi, especially when one considers that there are roughly six species of fungi for every plant species on the planet (Hawksworth 1990). The good news is that there is much to learn, and the Vision fire provided an exceptional opportunity to do exactly that.

Getting to the root of Point Reyes plant communities

The goal of this chapter is to heighten awareness of the role of ectomycorrhizal fungi and to relate what we have learned about this group from the Vision Fire. To start this process it is important for the reader to understand that in nature fine roots of most plants are not simply plants. Instead, they are colonized by fungi in mutualistic interactions known as mycorrhizae. This symbiosis is a way that plants "contract-out" the specialized function of collecting mineral nutrients from soil. They pay their fungal partners with sugar, and in return fungi provide phosphorus, nitrogen, and other mineral nutrients to plants. The vast majority of land plants are normally mycorrhizal, and some plants, such as pines and oaks, require these fungi for normal growth. Similarly, most fungi involved in mycorrhizal symbioses, require plants.

At Point Reyes, and elsewhere, there are three main types of mycorrhizal interactions. Different groups of fungi are involved with each, and plants are generally restricted to one of these three groups (Table 1). For a pine seedling to grow and compete, it must encounter an ectomycorrhizal fungus within the first few months. There are two basic forms of fungi that an uncolonized pine root can encounter: spores and sclerotia, or mycelium. Spores and sclerotia are roughly microscopic equivalent of seeds and tubers, respectively, in plants, and they can be dispersed in various ways to new locations. Mycelium (Fig 1a) is the growing body of a fungus. It is made up of thread-like hyphae. Mycelium can spread locally from a colonized root to a new uncolonized root of a seedling. Note that this latter type of colonization can only work if living roots of a plant, that uses the same general type of mycorrhizal fungi, are in the immediate neighborhood. For example, fungi found on roots of coyote bush and pines are different, and, therefore, pine cannot rely on coyote bush as a reservoir for its fungi. In

Table 1: The three main Mycorrhizal types found at Point Reyes

Mycorrhizal Type	Examples of Plants	Examples of Fungi
Ectomycorrhizae (EM or Ecto)	bishop pine, Douglas-fir, tanbark oak, live oak, manzanita, madrone, alder	Basidiomycetes: mushrooms, boletes, chanterelles, false- truffles, crusts; and Ascomycetes: cup fungi, and true truffles.
Ericoid mycorrhizae	huckleberry, salal	Ascomycetes: particularly *Hymenoscyphus ericae*, and close relatives.
Arbuscular mycorrhizae (AM, VAM, or endo)	Most plants: coyote bush, California lilac, poison oak, bay laurel, grasses, ferns, herbs	Glomalean fungi: microfungi (e.g. *Glomus, Acaulospora, Gigaspora*)

contrast, Douglas-fir and pine, or coyote bush and poison oak, share many of the same fungi, and so one plant may inadvertently supply its neighbor with fungi.

There is a genetic difference between spores and mycelium. Spores are produced in fruiting bodies such as mushrooms or truffles, and they are usually the product of a sexual recombination (i.e., they are products of meiosis); therefore each spore is genetically unique. This is different from mycelial growth or sclerotia which are vegetatively (i.e. mitotically) produced and therefore result in the spread of identical fungal genotypes. Understanding this point is important, because later in this chapter we use knowledge of fungal genotypes to infer whether fungi spread primarily by spores or vegetative growth.

The pre-fire ectomycorrhizal community

We have studied mycorrhizal fungi associated with Bishop pine in the Limantour road area of Point Reyes since 1991. We have done this in two ways. First, we have collected mushrooms, truffles, and other fruiting bodies of fungi of interest. Second, we have examined roots of Bishop pine trees or seedlings and used molecular-based identification methods to catalogue fungi present and to quantify their frequencies and abundance (Table 2).

This work has revealed several interesting features of mycorrhizal community associated with Bishop pine. One of the most unexpected findings of our early work was that some species that exhibit the most abundant fruiting were rare or low abundance species on roots, and conversely, some species that were dominant colonizers of roots, appeared to be rare fruiters. For example, one of the most abundant fruiters was *Suillus pungens* (Fig 1d), but in mature forests, we only found it colonizing roots when we looked directly below its mushrooms, and even then it was not the most common or abundant species. One of the most dominant species on roots of bishop pine was *Tomentella sublilacina*. It took us about four years to identify this species, which we initially knew only from DNA sequences obtained directly from pine roots. We had missed its fruiting bodies because they consist of tiny brown crusts that form on woody litter on the forest floor (Fig 1b). Once we learned this, we were able to find it fruiting fairly often, but biomass (i.e., the collective weight) of its fruiting is still substantially lower that *Suillus pungens*. Other species that dominated the root community in mature Bishop pine forests included *Russula* and *Lactarius* species (Fig 1c). These can be common fruiters at Point Reyes, but were never as abundant as *Suillus*. Several *Amanita* species, especially *A. francheti, A. muscaria*, and *A. gemmata* were common, and sometimes abundant prefire fruiters, and were often co-dominant species on roots (Gardes and Bruns 1996; Horton and Bruns 1998; Taylor and Bruns 1999).

The abundant fruiting of *Suillus pungens* allowed us to examine how it spread in pre-fire forests. We did this by collecting and mapping its mushrooms in a 50 X 30 meter plot and by using molecular genetic methods to determine the genotypes present. As explained above, this information could tell us whether this fungus spreads primarily by spore, or by vegetative growth. What we found was that a single genotype, termed a "genet", covered over 360 sq. meters of this area (Bonello et al. 1998). This was an area large enough to include about 30 mature pine trees. That meant that this species was very good at spreading vegetatively and moving from root to root across a pine forest. Parallel strategies among plants such as bracken fern and blackberries, are well known by botanists. Using the age of the forest we could estimate a minimum growth rate of a half a meter/year for the species. We thought this was likely to be a conservative estimate, because position of mushrooms delimited only the minimum area that was occupied by this genet (Bonello et al. 1998). However, there were two ways that we could have overestimated the spread rate. First, if our molecular methods were not sensitive enough to differentiate closely related genotypes, then we would mistakenly conclude that the different individuals were a single genet. This turned out to be an unlikely error because we were able to show that our methods could separate single-spore isolates from a single mushroom. This is approximately the equivalent of separating full-sibs (i.e., brother and sisters in the animal world). Another way we might have overestimated growth rate is if the genet we observed was older than the forest. This would only be possible if *Suillus pungens* could survive periodic fires that establish new bishop pine forests. At the time, we had no way to test this idea.

One final prefire study gave us a crucial piece of information: it showed us that in addition to the species that were common on roots, and that fruited frequently, there were species that had dormant spores and sclerotia stockpiled in the soil (Taylor and Bruns 1999). In other chapters of this volume, reference is made

Table 2. Ectomycorrhizal fungi observed with bishop pine in Limantour Rd area

	Pre-fire Fruiting	Pre-fire EM	Pre-fire Bioassay	Post-fire EM	Post-fire Fruiting
Amanitaceae					
Amanita pachycolea	I			I-1	
A. francheti	A	F			I-5
A. gemmata	F	F			F-2
A. muscaria	F	F			I-2
A. magnaverrucata[1]	I				
Boletales					
Chalciporus piperatoides	I				
Chroogomphus vinicolor	F				A-2
Suillus brevipes	I				
S. pungens	A	I		I-1	A-2
S. tomentosus	I				
Boletus edulis	F				
B. subtomentosus	I				
B. chrysenteron	F				I-2
Boletoid spp.		F			
Rhizopogon salabrosus[2]	F	F	D	A-2	F-4
R. occidentalis[3]	I		A	A-2	A-5
R. smithii			F		A-5
R. olivaceotinctus			F	A-2	?
R. rubescens			F		?
R. vulgaris			?		A-5
unidentified suilloid (probably *Rhizopogon*)				A-2	
Cortinariaceae					
Dermocybe spp.	F				
Cortinarius spp.	F				
Hebeloma cf. crustuliniforme	I				F-2
Inocybe spp	F				F-5
Inocybe subocracea	F				?
Thelephoraceae					
Tomentella sublilacina	F?	D			
Tomentella spp.		F			
Thelephora cf. terrestris	I				F-5
Russulaceae					
Lactarius rufus	F	A			
L. fragilis var. rubidus	I				
Russula brevipes	F	A		R-1	
R. amoenolens	I	D			
R. rosacea	F				
R. cf. xerompelina	F	A			
R. sp.					
Cantharellaceae					
Cantharellus cibarius s.l.	I				
Clavulina sp. [4]	F?	F		F-1	
Corticiaceae					
Amphinema byssoides	?				
Hymenochaetaceae					
Coltricia perennis					
Tricholomataceae					
Laccaria amethysteo-occidentalis	F	I			
L. proxima	I				I-3
Ascomycota					
Tuber californicum	?			F	?
Tuber spp.	?			A	I?-5
Wilcoxina mikolae	?			A	?
Wilcoxina sp.	?		F	A	?
Cenococcum geophilum	NA	F		F	NA
Phialophora sp.	NA		F	A-1	

I infrequent, F frequent; A abundant; D dominant (usually abundant at every site); for post-fire species the season when it was first observed is given. ? may have been missed for various technical reasons. [1] the Limantour road area is the type location for the species; [2] referred to as *R. subcaerulescens* in cited publications; [3] referred to as *R. ochraceorubens* in cited publications; [4] referred to as Cantharoid 1 in publications from Point Reyes cited.

to a soil "seedbank" for plants. What we are talking about here is the fungal equivalent: a soil **sporebank**. We assayed the sporebank by removing soil, air-drying it, and then planting bishop pine seedlings into it. After about a year the roots of the seeding were examined to see which fungi had colonized them. We call this approach a "**bioassay**", because we are using pine seedlings to assay soil for compatible mycorrhizal fungi contained in it. It turned out that the species we found in our bioassays were almost entirely different from those that were present on roots of mature trees that had been present in the very same soil samples. The only species that was dominant on roots of mature trees and also common in bioassays was *Tomentella sublilacina*, but the species that dominated bioassays were from two different groups of fungi: *Rhizopogon* species, and Ascomycetes, such as *Wilcoxina* and *Tuber*. In fact, spores of the Rhizopogon species were so abundant that even when the soil was diluted 100-fold with sterile soil, most seedlings were colonized by a *Rhizopogon* species (Taylor and Bruns 1999). We have now found that *Rhizopogon* sporebanks are widespread features throughout California pine forests (Kjøller and Bruns 2003).

Post-fire Ectomycorrhizal Communities

Within about three months after the fire Bishop pine seedlings were starting to establish. The first fungi to colonize their roots were "dark-septate" fungi, probably related to *Phialophora* sp. (Horton et al. 1998). The role of these fungi is unclear. They are not strictly ectomycorrhizal in their morphology, as they often penetrate root cells, and prior work has given conflicting evidence for their mutualistic or parasitic interactions. Our earlier bioassay work had shown that *Phialophora*-like fungi were present in the sporebank (Taylor and Bruns 1999). Arbuscular mycorrhizal fungi were also among the first colonists of young pine roots, but as mentioned above, these are not normally thought to be important symbionts with pine (Table 1), so again their role in the post-fire community is unclear (Horton et al. 1998). Ectomycorrhizal associations were found in some one-month-old seedlings, and their frequency increased with time, such that about 85% of five-month-old seedlings contained at least some ectomycorrhizal roots. Among surviving 1-year and 2-year old seedlings, all were ectomycorrhizal (Baar et al. 1999; Grogan et al. 2000).

Composition of post-fire ectomycorrhizal communities had a strong resemblance to the composition of the pre-fire sporebank community, and this resemblance appears to be caused by survival of the soil sporebank (Baar et al. 1999). Immediately following the fire we collected soil samples from sites that we had studied prior to the fire, and when we bioassayed these samples we found the same fungi were still present. Thus the sporebank survived the fire. For *Rhizopogon* and *Tuber* species, which were very abundant at some sites, it is likely that the sporebank alone may account for their post-fire success. New dispersal is unlikely to explain the observed pattern for several reasons. Both genera fruit underground as truffles, and spores contained in these fruiting bodies either stay put in the soil (Miller et al. 1994), or are eaten by mammals, such as mice, squirrels, and deer, and dispersed to new locations via their scat (Maser et al. 1978). For these reasons, post-fire dispersal from unburned areas would be unlikely to provide the high frequency of truffle inoculum necessary to account for the abundance of these fungi observed on post-fire seedlings. Fruiting within the burn area is also an unlikely source, as it was uncommon and essentially restricted to areas around the few surviving Douglas-firs in the first year following the fire. Most *Rhizopogon* species, and perhaps *Tuber* species as well, have narrow host ranges, and would not associate with both Douglas-fir and pine (Massicotte et al. 1994; Molina and Trappe 1994).

Some fungi that colonized post-fire seedlings, and have wind-borne spores, probably did disperse into the burned area in the first few wet seasons following the fire. *Hebeloma cf. crustuliniforme* is the best example of this strategy. It was not common prior to the fire, and was not found in the bioassays, but it was common in year one and year two samples after the fire (Baar et al. 1999). *Hebeloma* species are known from other studies to be early colonizers of disturbed sites (Gryta et al. 1997; Gryta et al. 2000). *Wilcoxina* species may be an example of a mixed strategy. They were over-represented in post-fire seedlings relative to their apparently low abundance in the sporebank. This suggests that additional dispersal or perhaps fire-activation of the sporebank was important for these fungi. Prior studies show that they are common in post-fire settings, and are capable of wind dispersal (Egger et al. 1991).

The fungi that were dominant members of the pre-fire community were not eliminated by the fire, only reduced in their dominance. In fact, they were found at the very earliest sampling of post-fire seedling and at all subsequent sampling times (Horton et al. 1998; Baar et al. 1999;

Grogan et al. 2000). One of the pre-fire dominants, *Tomentella sublilacina*, was represented in the sporebank community, as determined from our bioassays, and its frequency in the sporebank was very similar to its frequency on the post-fire seedlings (Baar et al. 1999). In addition, we have since shown that its spores are an effective inoculum for pine seedlings under laboratory conditions (Lilleskov and Bruns 2003). Thus, for this species spore survival could account for its post-fire occurrence.

For all other pre-fire dominants and co-dominants, such as species in the Russulaceae and Amanitaceae, the manner in which they survived is less clear. The fact that they were present within the first few months, suggests that new dispersal alone may not account for their presence. The two most obvious ways for survival are 1) as spores and 2) as mycelium or colonized root tips. Superficially the spore option seems unlikely, because we have never recovered any species in the Russulaceae or the Amanitaceae in our bioassays, and studies elsewhere suggest that spores are not very effective inoculum under laboratory conditions (Deacon and Fleming 1992). The "under laboratory conditions" caveat may be important however, because field conditions are different, and difficult to replicate. Recent work on genet sizes in the Russulaceae and Amanitaceae at Point Reyes and in neighboring areas shows that spore establishment must be an ongoing process in undisturbed forests (Redecker et al. 2001); therefore it could also be important in post-fire environments. Survival of mycelium on dying root tips is possible, as it has been observed in other types of disturbance such as logging (Hagerman et al. 1999), and circumstantial evidence suggests that it is common in ground fires where overstory trees are not killed (Jonsson et al. 1999; Stendell et al. 1999). We have made two observations that suggest that mycelial survival might have occurred in the Vision fire. First, we observed what appeared to be dead *Russula*-colonized root tips of mature trees within the same soil core as new *Russula* root tips on a seedling. Second, we observed that new *Russula* tips tended to be deeper in the soil, where mycelial survival might be expected to be highest. However, neither observation is particularly convincing, as spore inoculum might also follow the same patterns. The best evidence would be to demonstrate that the same genotype of a particular fungus was found both before and after the fire. Unfortunately, it was not possible to test for genet survival with any of the dominant species, as we did not have prefire genetic data from the same locations where we observed post-fire species occurrence.

Suillus pungens, which was an abundant fruiter but a rare mycorrhizal type in prefire forests, is the only species in which we have been able to test directly for post-fire survival. This was possible because we had extensive pre-fire genetic data for it from one location where it recolonized after the fire. Our results showed that all of the post-fire *Suillus pungens* genotypes were new; thus, spore colonization appears to be the most important process for this species (Bruns et al. 2003). This answer was disappointing to us in one sense, because we had hoped to find a solid example of mycelial survival. It was good news in a different way; it showed that our previous assumption that clonal spread starts with establishment of a new forest was a good one, and therefore, estimates of minimum growth rate for *Suillus pungens* remain conservative.

Spatial patterning of fungal species in the post-fire environment provided additional clues as to how they might have survived. Grogan et al. (2000) showed that the pattern of species occurrence appeared to be random with respect to either space between seedlings or occurrence on different parts of the root system of individual seedlings. In other words, inoculum for individual species behaved like point-sources. This patterning does not resolve the spore versus mycelium question, but it does suggest that if mycelium was involved it was very limited in spatial coverage. Thus, widespread mycelial survival appears unlikely, but rare transfer from individual dying root tips to newly established seedling cannot be eliminated.

The fact that *Rhizopogon* species were found on all pine seedlings sampled within the former coastal scrub community, shows that these were dispersed spores that probably have laid dormant for many years (Horton et al. 1998). These conclusions are based on the fact that fruiting of *Rhizopogon*, like many other ectomycorrhizal fungi in the area, would only occur in conjunction with pine or Douglas-fir, and neither tree existed in the scrub community prior to the fire. Thus, spores of *Rhizopogon* must have been dispersed there at some point in the past. Recall that these are animal dispersed spores, which therefore might be expected to have a rather spotty spatial distribution. To achieve the observed uniform distribution, would roughly require a *Rhizopogon*-containing rodent or deer dropping every 100 square centimeters. This becomes feasible if the spatial

distribution results from dispersal over many years followed by a long dormancy.

Most *Rhizopogon* species appear to be early successional species. In fact, the only pine-associated *Rhizopogon* that we found as root colonist in the pre-fire forest was *R. salebrosus*. Other species were occasionally found fruiting, but primarily in disturbed settings, such as parking lots, or former quarry sites. In addition there is evidence that these *Rhizopogon* species may be relatively weak competitors. Our soil bioassays showed that they are the most frequent colonists when air-dried soil is extensively diluted with sterile soil, yet their colonization frequency drops in undiluted soil as the frequency of competitors increase. Similarly, they are less abundant in fire sites that were previously parts of large forested tracts, such as those studied by Horton et al. (1998) and Grogan et al. (Grogan et al. 2000), and they are most abundant on seedlings that established in the former scrub community and in small patches of former forest such as the site studied by Baar et al. (1999).

Fruiting in post-fire forests

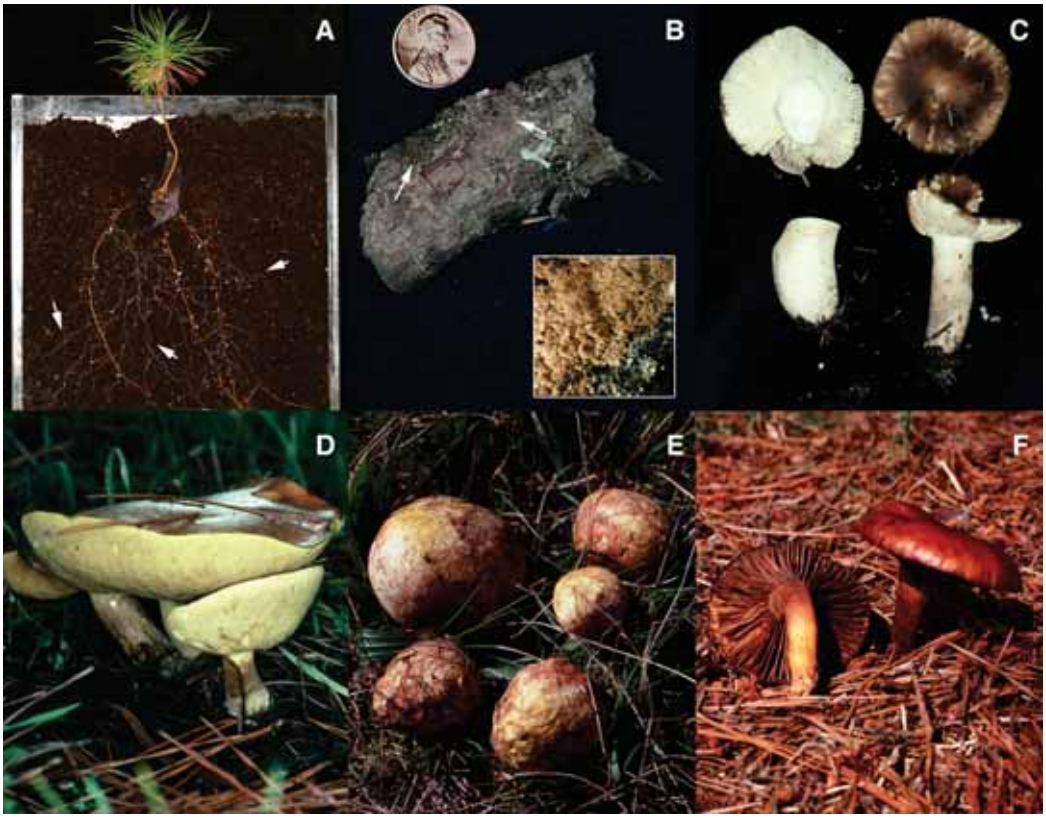
Most of our efforts have been focused on root colonization rather than fruiting. As a result, our records on mushroom and truffle production are only qualitative, and are prone to under-recording of infrequent species (Table 2). Our data for truffles, such as *Rhizopogon* and *Tuber* species, and small cup fungi, such as *Wilcoxina* spp., are particularly sparse, since truffles fruit below-ground and require directed effort to find, and tiny cup fungi are easily overlooked. Nevertheless, some patterns are obvious, and transcend these sampling deficiencies.

Fruiting usually occurs in the rainy season, October through April. The first such season following the fire (i.e., winter 1995-96) we observed no fruiting of any ectomycorrhizal species except near the few surviving Douglas-fir trees. This is probably an accurate observation, as pines, shrubs, and herbs were not very dense and so above-ground fruiting, when it occurred, was obvious. The following year, winter 1996-7) we observed the first few mushrooms of ectomycorrhizal fungi in areas that contained only pine seedlings. Fruiting was not abundant in this year, but *Hebeloma cf. crustuliniforme*, and *Coltricia perennis* were among the most common fruiters. Neither was common prior to the fire, and both are known to be early successional ectomycorrhizal fungi. Other species that fruited in the second season, such as *Suillus pungens*, *Chroogomphus vinicolor*, *Boletus chrysenteron*, *Amanita muscaria*, and *A. gemmata*, were species that had been common fruiters in the pre-fire forest.

Notably absent until the 2000-2001 wet season were any species in the Russulaceae, despite the presence of some species on seedling roots in the first few months after the fire (Horton et al. 1998). In some ways this lack of correspondence between fruiting and mycorrhizae fits the pre-fire pattern, where species in the Russulaceae were often dominant on the roots, but were minor components in the above-ground fruiting record (Gardes and Bruns 1996). In the post-fire community, they lost their dominant position below-ground and were essentially absent in the fruiting record. However, by the winter of 2002-3, *Russula amoenolens* became a dominant fruiter; this was the first time in either pre or post fire settings where we observed abundant fruiting of this species.

By year four 1998-99, *Suillus pungens* and *Chroogomphus vinicolor* began to become truly abundant fruiters. In fact, it was hard to walk through the young pine forests without noticing these two species. In winter 2000-2001 season we found that fruiting of several *Rhizopogon* species was very abundant during late February and early March. We also found ample evidence of rodent and deer feeding on these truffles. This is a necessary step for dispersal of *Rhizopogon*, and it underlines one of the ways that these fungi connect plant and animal communities at Point Reyes.

Suillus, Chroogomphus, and *Rhizopogon* are members of the Suilloid lineage within the Boletales (Bruns et al. 1998), and so their similar post-fire behavior may be a result of evolutionarily shared characters. In particular spores of all three genera may be long-lived residents of the sporebank. As discussed above, this is likely the case with *Rhizopogon*; the situation in the other two genera is less clear. *Suillus* occasionally shows up in bioassays (unpubl. results), but not nearly as commonly as *Rhizopogon*. Thus, if post-fire abundance of *Suillus* is due to its sporebank, there must be additional factors that function to activate its spores in the natural setting. It has become clear in recent years that members of the Gomphidiaceae, such as *Chroogomphus*, may primarily be parasitic on *Suillus* and *Rhizopogon* species, and their hyphae are often found within mycorrhizae of *Suillus* and *Rhizopogon* (Agerer 1990; Olsson et al. 2000). This behavior could explain why *Chroogomphus vinicolor* is a common fruiter and yet we have never found its mycorrhizae. It



Six important ectomycorrhizal fungi associated with bishop pine.

A: A bishop pine seedling grown in the laboratory and colonized by *Rhizopogon salebrosus* is shown. Arrows point toward the white mycelium of the fungus.

B: *Tomentella sublilacina* was one of the most abundant colonizers of bishop pine roots before the fire. Its fruiting bodies are small crusts, which are often formed on pieces of bark or wood on the forest floor. The arrows show such crusts on a small piece of bark collected from the pre-fire forest. The insert shows a 40X magnification of a *Tomentella sublilacina* fruiting body. This small crust would be densely packed with spores.

C: *Russula amoenolens* was another dominant fungus in the pre-fire forest, but its mushrooms, which are about the size of a fist, were rare.

D: In contast, mushrooms of *Suillus pungens* were abundant in the prefire forest, but it colonized relatively few roots. *Suillus pungens* is an even more abundant fruiter in the post-fire forest. Its mushrooms are large and often weigh over 100 grams.

E: *Rhizopogon occidentalis* and other *Rhizopogon* spp. were abundant colonizers of the post-fire seedings. Their fruiting bodies, which are shown here, are false-truffles. They are about marble to golf-ball sized and fruit below ground. Rodents and deer dig them up and disperse the spores in their scat.

F: *Chroogomphus vinicolor* is the most abundant post-fire mushroom in the five-year old post-fire forest. It appears to be a parasite of *Suillus* and *Rhizopogon* species.

would also mean that their spores would easily be missed in a laboratory bioassay.

Conclusions

The fungal ectomycorrhizal community associated with bishop pine changed quantitatively with the fire. It went from a community dominated by *Tomentella sublilacina*, *Russula,* and *Lactarius* species to one dominated by *Rhizopogon, Wilcoxina*, and *Tuber* species. Colonization of these post-fire dominants, especially *Rhizopogon* and *Tuber*, was the result of an extensive soil sporebank that survived the fire and provided inoculum for bishop pine seedlings. This sporebank was probably established in the years immediately after earlier fires when abundant fruiting and dispersal of these taxa were likely to have occurred. We expect most of *Rhizopogon*, *Tuber*, and *Wilcoxina* species will be eventually replaced by the taxa that previously dominated the mature forest. How quickly this will occur, and whether other species will dominate at intermediate times, remains unknown. These questions and others remain, and Point Reyes continues to be a great place to look for the answers.

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Literature Cited

Agerer, R. (1990). "Studies on ectomycorrhizae. 24. Ectomycorrhizae of *Chroogomphus helveticus* and *C. rutilus* (Gomphidiaceae, Basidiomyetes) and their relationship to those of *Suillus* and *Rhizopogon*." Nova Hedwigia 50: 1-63.

Baar, J., T. R. Horton, A. Kretzer and T. D. Bruns (1999). "Mycorrhizal recolonization of P*inus muricata* from resistant propagules after a stand-replacing wildfire." New Phytol. 143: 409-418.

Bonello, P., T. D. Bruns and M. Gardes (1998). "Genetic structure of a natural population of the ectomycorrhizal fungus *Suillus pungens*." New Phytol. 138: 533-542.

Bruns, T. D., T. M. Szaro, M. Gardes, K. W. Cullings, J. Pan, D. L. Taylor and Y. Li (1998). "A sequence database for the identification of ectomycorrhizal fungi by sequence analysis." Mol. Ecol. 7(3): 257-272.

Bruns T.D., Tan J., Bidartondo M.I., Szaro T.I., and D. Redecker. 2002. Survival of *Suillus pungens* and *Amanita francheti* ectomycorrhizal

genets was rare or absent after a stand-replacing wildfire. New Phytol. 155:517-523

Deacon, J. W. and L. V. Fleming (1992). Interactions of ectomycorrhizal fungi. Mycorrhizal functioning an integrative plant-fungal process. M. F. Allen. New York, Chapman and Hall: 249-300.

Egger, K. N., R. M. Danielson and J. A. Fortin (1991). "Taxonomy and population structure of E-strain mycorrhizal fungi inferred from ribosomal and mitochondrial DNA polymorphisms." Mycol. Res. 95(7): 866-872.

Gardes, M. and T. D. Bruns (1996). "Community structure of ectomycorrhizal fungi in a *Pinus muricata* forest: Above- and below-ground views." Can. J. Bot. 74(10): 1572-1583.

Grogan, P., J. Baar and T. D. Bruns (2000). "Below-ground ectomycorrhizal community structure in a recently burned bishop pine forest." J. Ecol. 88(6): 1051-1062.

Gryta, H., J. C. Debaud, A. Effosse, G. Gay and R. Marmeisse (1997). "Fine-scale structure of populations of the ectomycorrhizal fungus *Hebeloma cylindrosporum* in coastal sand dune forest ecosystems." Mol. Ecol. 6(4): 353-364.

Gryta, H., J. C. Debaud and R. Marmiesse (2000). "Population dynamics of the symbiotic mushroom *Hebeloma cylindrosporum*: mycelial persistence and inbreeding." Heredity 84(3): 294-302.

Hagerman, S. M., M. D. Jones, G. E. Bradfield, M. Gillespie and D. M. Durall (1999). "Effects of clear-cut logging on the diversity and persistence of ectomycorrhizae at a subalpine forest." Can. J. For. Res. 29(1): 124-134.

Hawksworth, D. L. (1990). "The fungal dimension of biodiversity: magnitude, significance, and conservation." Mycol. Res. 95: 641-655.

Horton, T.R. and T.D. Bruns (1998). "Multiple-host fungi are the most frequent and abundant ectomycorrhizal types in a mixed stand of Douglas-fir (*Pseudotsuga menziesii*) and bishop pine (*Pinus muricata*)." New Phytol. 139: 331-339.

Horton, T.R., E. Cazares and T.D. Bruns (1998). "Ectomycorrhizal, vescular-arbuscular and dark septat fungal colonization of bishop pine (*Pinus muricata*) seedlings in the first 5 months of growth after wildfire." Mycorrhiza 8: 11-18.

Jonsson, L., A. Dahlberg, M. C. Nilsson, O. Karen and O. Zackrisson (1999). "Continuity of ectomycorrhizal fungi in self-regenerating boreal *Pinus sylvestris* forests studied by comparing mycobiont diversity on seedlings and mature trees." New Phytol. 142(1): 151-162.

Kjøller R. and T. D. Bruns. 2003. *Rhizopogon* spore bank communities: within and among Californian pine forests. Mycologia. 95:603-613

Lilleskov E.A. and T.D. Bruns. 2003. Root colonization dynamics of two ectomycorrhizal fungi of contrasting life histories is mediated by organic nutrient addition. New Phytol. 159: 141-151

Maser, C., J. M. Trappe and R. A. Nussbaum (1978). "Fungal-small mammal interreletionships with emphasis on Oregon coniferous forests." Ecology 59: 799-809.

Massicotte, H. B., R. Molina, D. L. Luoma and J. E. Smith (1994). "Biology of the ectomycorrhizal genus, *Rhizopogon* II. Patterns of host-fungus specificity following spore inoculation of diverse hosts grown in monoculture and dual culture." New Phytol. 126: 677-690.

Miller, S. L., P. Torres and T. M. McClean (1994). "Persistence of basidiospores and sclerotia of ectomycorrhizal fungi and *Morchella* in soil." Mycologia 86(1): 89-95.

Molina, R. and J. M. Trappe (1994). "Biology of the ectomycorrhizal genus, *Rhizopogon* I. Host associations, host-specificity and pure culture syntheses." New Phytol. 126: 653-675.

Olsson, P. A., B. Munzenberger, S. Mahmood and S. Erland (2000). "Molecular and anatomical evidence for a three-way association between *Pinus sylvestris* and the ectomycorrhizal fungi *Suillus bovinus* and *Gomphidius roseus*." Mycol. Res. 104: 1372-1378.

Redecker D, T. M. Szaro, J.R. Bowman, T. D. Bruns. 2001. Small genets of *Lactarius xanthogalactus*, *Russula cremoricolor* and *Amanita francheti* in late stage ectomycorrhizal successions. Mol. Ecol. 10: 1025-34

Stendell, E., T. R. Horton and T. D. Bruns (1999). "Short-term effects of a ground fire on the ectomycorrhizal community." Mycol. Res. 103: 1353-1359.

Taylor, D. L. and T. D. Bruns (1999). "Community structure of ectomycorrhizal fungi in a *Pinus muricata* forest: minimal overlap between the mature forest and resistant propagule communities." Mol. Ecol. 8: 1837-1850

After the Vision Fire: Monitoring Landbirds in the Riparian Zone

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Abstract

We monitored the riparian bird community for two years following the 1995 Vision Fire in the Point Reyes National Seashore (PRNS), Marin County, California. We compare bird and vegetation data on burned and unburned sites within PRNS and the Golden Gate National Recreation Area. Bird abundance, as indexed by fixed-radius point counts, was higher for several shrub-nesting species at burned sites than at unburned sites. For some tree nesting species, however, abundance was lower at burned sites. A comparison of vegetation data collected at the bird survey points showed that burned riparian had significantly less tree cover and had greater shrub and herb cover than unburned sites. Reproductive success and survival may serve as better indicators of habitat quality and population health than abundance of individuals. Therefore, we located and monitored nests of the Song Sparrow (*Melospiza melodia*) – a species whose abundance was greater in the recently burned riparian habitat – to estimate nest survival at one burned site (Muddy Hollow) and two unburned sites (Lagunitas and Olema Creeks). We then built population growth rate models (λ) to determine the relative "source-sink" status of this species by site, to index local population health. Nest survival was significantly higher on the burned site than the unburned sites, and the λ values suggested that the burned site was a population source while the unburned sites were sinks. The fact that abundance and reproductive success were higher at the burned site is intriguing because it suggests that high Song Sparrow abundance can be used to infer habitat quality in coastal riparian areas, but more rigorous studies are needed. Our results provide evidence that fires in riparian zones may benefit breeding populations of songbirds (given the right conditions and timing) and highlight the utility of monitoring programs to land managers.

Song Sparrow. Peter LaTourrette North American Bird Photo Gallery: http://birdphotography.com/

Introduction

On 3 October 1995, an illegal campfire erupted into a wildfire (Vision Fire) that, over the next 12 days, burned more than 12,000 acres of the Point Reyes National Seashore (PRNS) and surrounding land-holdings. Fueled by atypically dry 50-mph winds, the rate of spread of the fire reached 3,100 acres per hour. The fire was classified as low to moderate intensity and several different habitat types burned. We used standardized landbird monitoring methods to assess whether the Vision Fire had any impact on the riparian-breeding songbird community.

We compared bird and vegetation data on burned and unburned sites to: (1) Assess whether the fire influenced abundance of riparian-breeding landbirds. We predicted that species would respond differently to the fire depending upon their particular life-history characteristics. (2) Characterize habitat changes caused by the fire and qualitatively relate them to bird response. For example, a ground-nesting bird may respond differently than a tree-nesting bird due to changes in vegetation caused by the fire. (3) Estimate site-specific population "health" for the Song Sparrow (*Melospiza melodia*) in burned versus unburned sites. We indexed habitat quality by building source-sink models (Pulliam 1988). A *source* population is one in which the production of young surviving to breeding age exceeds adult mortality. These populations will sustain themselves and/or export excess individuals to other populations. A sink population is one in which young are not produced in sufficient numbers to compensate for adult mortality. Sink populations will either go locally extinct or be maintained by individuals produced elsewhere.

Source and sink populations often form networks or landscapes of populations joined by immigrants and emigrants, and a single source population may effectively maintain many sink populations (Pulliam 1988). Thus, population dynamics cannot be understood at the single population level. Sink populations have the potential to drain source populations that, in turn, could lead to widespread population declines, though sinks are not without value (see Howe et al. 1991).

Methods

Study Sites

We used data from an on-going monitoring

program to investigate songbird response to the Vision Fire. All study sites were located within the Point Reyes National Seashore and the Golden Gate National Recreation Area, coastal Marin County, California. A total of six riparian forests made up our study areas, three burned and four unburned (Table 1). Intensity of the fire at our study sites was classified as "low" with the exception of a portion of Muddy Hollow that was "moderate".

Vegetation at all study sites was similar. Trees were mostly Red Alder (*Alnus rubra*) and willow (*Salix* spp.) with lesser amounts of California bay (*Umbellularia californica*), coast live oak (*Quercus agrifolia*), and California buckeye (*Aesculus californica*). The understory included saplings of the tree species as well as California blackberry (*Rubus ursinus*), rush (*Juncus* spp.), hedge nettle (*Stachys* sp.), stinging nettle (*Urtica californica*), and a wide variety of other herbs and grasses. The area surrounding our study sites is mostly a mosaic of coastal scrub and/or mixed evergreen forest typical of the region (Shuford and Timossi 1989).

Relative Abundance

We estimated relative abundance of birds using the point count method (Ralph et al. 1993, 1995). At each study site we established a series of point count survey stations at least 200 m apart. Point count stations were surveyed three times by trained observers, at least 10 days apart, during the summers of 1997 and 1998. The duration of each count was 5 minutes, and all birds seen or heard were recorded. We used only detections within 50 m of the observer and made the assumption that detection probabilities were similar between treatments. All counts began around local sunrise and continued for no more than four hours to restrict the census to peak singing hours. A total of 38 burned points and 49 unburned points (Olema Creek only used for site-specific comparisions, see below) were used in our analyses (Table 1).

Vegetation

We used a 50 m radius plot, or relevé (Ralph et al. 1993) to sample vegetation in burned and unburned forests in summer, 1997. We visually estimated the percent cover of the plot within each of three vegetation strata; tree, shrub and herb. Herbs consisted of both forbs and grasses. Shrubs and trees were distinguished by structure as opposed to by species. Thus a single species could be present as both shrubs and trees depending on size (≥ 5 m = tree). We noted the percent cover of dominant plant species by strata, determined the maximum and minimum heights for each vegetation layer, and the maximum and minimum diameter at breast height of trees within the 50 m radius relevé. Additionally, we noted the width of the riparian zone, slope, aspect (direction of slope), percent cover of closed canopy, running water, bare ground, litter, and wetland vegetation.

Statistical Analyses of Bird Abundance and Vegetation Data

We (1) compared habitat characteristics between burned and unburned vegetation plots and (2) compared bird abundance between burned and unburned sites for 12 bird species that occurred on at least 10% of the point count stations. We also compared Song Sparrow abundance among the nest monitoring sites; Muddy Hollow, Lagunitas Creek, and Olema Creek in 1997 only. For our abundance index, we used the total detections across 3 visits; each survey point was treated as an independent sampling unit. To account for potential site-specific differences unrelated to fire effects we used 3-way ANOVA that included site as a covariate in all analyses. Data were transformed when necessary (log or square-root) to meet the assumptions of parametric analyses. For one species (Bewick's Wren), we grouped count data into three categories of abundance after transformations failed to normalize residuals of ungrouped data. Significance was assumed at the $P \leq 0.05$ level. All statistical analyses were carried out using the program STATA (STATA Corp. 1997).

Table 1. Names, locations, and number of point count stations in burned and unburned riparian forests. The Olema Creek site was used only in the site-specific comparisons of Song Sparrow abundance (1997).

Study Site	Latitude (N)	Longitude (W)	Number of Point Counts Stations
Burned			
Muddy Hollow	38°02'	122°48'	17
Coast Trail/Laguna	38°02'	122°52'	13
Coast Camp	38°02'	122°85'	8
Unburned			
Bear Valley	38°03'	122°48'	16
Redwood Creek	37°51'	122°34'	20
Lagunitas Creek	38°02'	122°45'	13
Olema Creek	38°06'	122°80'	13

Source-Sink Model

The source-sink model (λ) employed here determines if a population can maintain itself without immigration or emigration (closed population) by estimating a population growth rate (Pulliam 1988). Components (demographic parameters) of the model are adult survival (probability that an adult will survive from one breeding season to the next), juvenile survival (survival from fledgling age to breeding adult), and reproductive success (probability of a nest surviving from egg laying to fledging x number of nesting attempts x average number of young produced per successful nest). We can write the equation for sources and sinks as:

λ = adult survival + (juvenile survival x reproductive success)

A value of 1 indicates that a population is exactly replacing itself, > 1 indicates that a population is increasing or is a source, < 1 indicates that a population is declining or is a sink.

To obtain data necessary to estimate nest survival we located and monitored nests in 1997 at Muddy Hollow, Lagunitas Creek, and Olema Creek. We attempted to replicate the burned treatment by monitoring nests on Laguna Creek, a drainage that runs parallel to Muddy Hollow, but sample sizes were too low at this site to warrant analysis.

Nest searching and monitoring followed standardized procedures described by Martin and Geupel (1993). We attempted to determine the fate of each nest by checking nest status every 1 - 4 days. A nest attempt was considered successful if it fledged at least one Song Sparrow young as evidenced by the presence of one or more of the following; fledglings outside of the nest, nest condition and timing suggested fledging (e.g., fecal matter on and around nest), adults carrying food in immediate vicinity of nest after expected fledge date, young fledged in response to the nest being checked by field biologists. A nest was considered to have failed if it was damaged and inactive, fledged only a cowbird, eggs were damaged, nestling(s) found dead, and nest contents disappeared during egg laying, incubation, or the early nestling stage before fledging would have been possible. Nests where success or failure may have both been possibilities, but none of the criteria for fledge or fail assessment was met, were considered of uncertain fate.

We calculated nest survival using the Mayfield (1975) method as recommended by Johnson (1979) for computing standard errors. We calculated the nest termination date following Manolis et al. (2000). We computed daily nest mortality by dividing total number of nests that failed by total number of exposure days within each site. The number of exposure days at each site were 220.0 from Muddy Hollow, 259.5 from Lagunitas Creek and 147.5 from Olema Creek. Daily nest survival was thus calculated as 1 - daily mortality, and overall nest survival estimates were calculated as a function of this daily survival rate raised to the length of the average nesting cycle (24 days from Chase 2002). For the other components of reproductive success, number of nestlings per successful

Bewick's Wren. Peter LaTourrette North American Bird Photo Gallery: http://birdphotography.com/

Pacific-slope Flycatcher. Peter LaTourrette, North American Bird Photo Gallery: http://birdphotography.com/

nest and number of nesting attempts, we used averages of 2.51 and 2.2, respectively (PRBO, unpublished data). We compared daily nest survival among study sites using the program CONTRAST (Sauer and Williams 1989).

We used an adult survival estimate of 59.9% for Song Sparrow based on capture-recapture analysis using 14 years of mist-net data from the nearby Palomarin Field Station (Nur et al. 2000). Similarly, we used an estimate of Song Sparrow juvenile survival of 38.4% from the Palomarin Field Station (Chase 2001). Survival estimates are probabilities that birds will survive from one year to the next, whether or not they were recaptured.

Results

Abundance varied for several species between burned and unburned sites (Table 2). There were fewer Pacific-slope Flycatchers and Chestnut-backed Chickadees in burned sites, and more Song Sparrows, Allen's Hummingbirds, Orange-crowned Warblers, and Bewick's Wrens in burned sites (see Table 2 for scientific names of species).

Several habitat characteristics differed between burned and unburned vegetation plots (Table 3). Burned plots had significantly greater shrub and herb cover than unburned plots. Unburned plots, however, had significantly greater tree cover, and the tree and shrub communities contained more species of plants than the burned plots (Table 3).

We located and monitored 22 Song Sparrow nests at Muddy Hollow, 23 at Lagunitas Creek, and 15 at Olema Creek (Table 4). Nest survival was 0.64 at Muddy Hollow, 0.29 at Lagunitas Creek and 0.19 at Olema Creek. We found significant differences in nest survival among all sites ($\lambda^2 = 7.20$, df = 2, P = 0.03). Between-site comparisons revealed that Muddy Hollow had higher nest survival than Lagunitas ($\lambda^2 = 3.85$, df = 1, P = 0.05) and Olema ($\lambda^2 = 4.80$, df = 1, P = 0.03) but Lagunitas and Olema did not differ ($\lambda^2 = 0.45$, df = 1, P = 0.5).

In 1997, Song Sparrow abundance was greater at Muddy Hollow than at Lagunitas Creek (F = 14.54, df = 1, P = 0.006) and Olema Creek (F = 7.87, df = 1, P = 0.009), but there were no differences in abundance between Lagunitas and Olema (P = 0.63).

Estimates of λ using the observed nest survival estimate were 1.28 for Muddy Hollow and remained above 1.0 even when using our lowest and most conservative nest survival estimate (i.e., lower bounds of the 95% CI). Estimates of λ were below 1.0 for Lagunitas Creek (0.91) and Olema Creek (0.80). However, when we used our highest estimates (upper bounds of the 95% CI) both Lagunitas and Olema were > 1.0.

Discussion

Variation in abundance between several species in burned versus unburned sites may be attributable to habitat changes caused by the fire. Song Sparrows, Allen's Hummingbirds, Orange-crowned Warblers, Bewick's Wrens, Wilson's Warblers, and American Goldfinches were more abundant at sites that had burned than at the unburned sites (Table 2). These species have an affinity for areas with moderately dense understory for nesting and foraging (reviewed in Shuford 1993) and burned sites had a richer and more abundant shrub layer than did unburned (Table 3). The Song Sparrow, for example, primarily places its nests low (< 1.5 m) in shrubs and herbs and thus may have benefited from the greater amount of understory created by the fire. The American Goldfinch and the

Table 2. Comparison of bird relative abundance (mean number of detections [within 50 m] per survey station) between burned (n = 38) and unburned (n = 49) riparian sites, 1998. Alpha (P) values (burn effect term) are from three-way ANOVA with transect (study site) included as a covariate. NS = non-significant. Table taken from Gardali et al. (2003).

Species	Burned (± SE)	95% CI	Unburned (± SE)	95% CI	P value
Allen's Hummingbird (*Selasphorus sasin*)	1.37 (± 0.19)	0.99-1.75	0.88 (± 0.15)	0.57-1.19	0.03
Pacific-slope Flycatcher (*Empidonax difficilis*)	0.55 (± 0.12)	0.30-0.80	0.61 (± 0.14)	0.33-0.90	0.06
Warbling Vireo (*Vireo gilvus*)	0.37 (± 0.13)	0.11-0.63	1.18 (± 0.17)	0.84-1.53	NS
Chestnut-backed Chickadee (*Poecile rufescens*)	0.47 (± 0.12)	0.22-0.72	1.51 (± 0.24)	1.03-1.99	0.09
Bewick's Wren (*Thryomanes bewickii*)	0.84 (± 0.19)	0.45-1.24	0.24 (± 0.09)	0.07-0.42	< 0.001
Swainson's Thrush (*Catharus ustulatus*)	1.53 (± 0.21)	1.11-1.94	2.14 (± 0.23)	1.69-2.60	NS
American Robin (*Turdus migratorius*)	0.55 (± 0.12)	0.30-0.80	0.49 (± 0.14)	0.21-0.77	NS
Orange-crowned Warbler (*Vermivora celata*)	1.13 (± 0.23)	0.67-1.59	0.31 (± 0.08)	0.15-0.46	0.03
Wilson's Warbler (*Wilsonia pusilla*)	2.5 (± 0.21)	2.08-2.92	2.08 (± 0.24)	1.60-2.56	NS
Song Sparrow (*Melospiza melodia*)	5.29 (± 0.33)	4.61-5.96	3.53 (± 0.29)	2.95-4.11	0.004
Black-headed Grosbeak (*Pheuticus melanocephalus*)	0.55 (± 0.14)	0.26-0.85	0.86 (± 0.13)	0.59-1.13	NS
American Goldfinch (*Carduelis tristis*)	0.87 (± 0.21)	0.45-1.29	0.24 (± 0.10)	0.05-0.44	NS

Allen's Hummingbird, generally placing their nests in understory vegetation, show greater flexibility in nest height placement but may have benefited from an increased supply of their preferred foods. The American Goldfinch has a diet of mostly seeds and, uncommon for non-carduline songbirds, even feeds its nestlings regurgitated seeds (Middleton 1993). Seeds produced by the abundant and rich herb layer probably provided ample foraging opportunities for the goldfinch. Similarly, the Allen's Hummingbird likely benefited from abundant sources of nectar from several plant species common after the fire (e.g., hedge nettle and bee plant [*Srophulania californica*]). For other species however, whose nesting and foraging requirements are met primarily in trees and tree canopy (reviewed in Shuford 1993), abundance was lower at burned sites (Table 2). Loss of trees most likely resulted in a reduction in numbers for most of these species. Other studies have noted reduced abundance of tree and canopy nesting species following a fire (Raphael et al. 1987, Dieni and Anderson 1999).

Abundance data may not act as a proxy to population health (e.g., source-sink status) and it has been suggested that density may in fact be a misleading indicator of habitat quality (Van Horne 1983, Brawn and Robinson 1996, but see Bock and Jones 2004). Does the presence of more Song Sparrows at burned sites equate to higher quality habitat? If so, some effect of the Vision Fire may have been responsible. We inferred habitat quality by estimating reproductive success and then putting it into the context of population health. The demographic components for the source-sink models differed only in reproductive success; specifically nest survival (Table 4). Reproductive success (even our lowest estimate) at Muddy Hollow was sufficiently high enough to offset adult mortality indicating that this site was a population source for the Song Sparrow. By contrast, only the highest estimates of nest survival at Lagunitas and Olema would characterize those sites as population sources (λ values > 1.0).

Populations of sources and sinks are difficult to identify because estimates for the demographic components are generally lacking and difficult to measure (see below). In our case, however, relatively robust estimates (adult and juvenile survival) were available from the long-term monitoring project at the nearby Palomarin Field Station. The parameters used in the model are assumed to be constant (i.e., remain unchanged over time) which is likely an incorrect assumption from an ecological standpoint, as demographic parameters will vary as the environment fluctuates (Nur and Sydeman 1999). Thus, populations may fluctuate between source and sink status. Despite these limitations, we believe that modeling source-sink status is useful as a comparative measure of population health.

The Vision Fire may have created conditions that increased the probability of nest survival at Muddy Hollow. The fire, for example, may have decreased the number of nest predators. Indeed, preliminary results indicate that small mammal numbers were reduced following the Vision Fire (G. Fellers pers. comm.). Additionally, the density of shrubs and herbs may have provided increased nest concealment from predators thereby resulting in higher success. Others have reported higher nest success in relatively well-concealed Song Sparrow nests (Nice 1937:93-94, Chase 2001). We know of only two studies that examined the effect of fire on the reproductive success of songbirds and both found results opposite ours; reproductive success was reduced in burned areas (Rohrbaugh et al. 1999, Aquilani et al. 2000). These studies, however, were conducted in habitats that may not be comparable; grasslands in Oklahoma and oak-hickory forests in Indiana. With prescribed burning and "let-burn" policies growing in popularity, studies examining the effects of fire

Table 3. Comparison of habitat characteristics between burned (n = 38) and unburned (n = 49) riparian sites, 1997. Alpha (P) values from ANOVA with transect (study site) included as a covariate. NS = non-significant. Table taken from Gardali et al. (2003).

Habitat Variable	Burned	Unburned	P value
Shrub cover (%)	52.6	24.9	< 0.001
Herbaceous cover (%)	42.8	28.6	0.001
Tree cover (%)	30.3	47.3	< 0.001
Tree species richness (#)	1.6	3.4	0.003
Shrub species richness (#)	3.8	4.5	< 0.001
Difference in shrub layer height (m)	3.3	3.5	0.003
Maximum tree dbh (cm)	31.7	49.9	NS
High shrub layer height (m)	3.2	3.7	NS
High tree height (m)	21.0	18.2	NS

on reproductive success of songbirds in various habitats and regions are sorely needed.

We acknowledge that a three-site comparison does not a definitive study make. However, our estimate of nest survival from Muddy Hollow (64%) was far higher than those reported for coastal scrub breeding populations at the nearby Palomarin Field Station (23.2 – 32.5%; Chase 2001) as well as three sites in coastal British Columbia (15-42%; Rogers et al. 1997). The question of whether Song Sparrow density was an indicator of habitat quality cannot be answered from our study. Still, it is interesting to note that Muddy Hollow had significantly more Song Sparrows than the unburned sites and also had significantly higher nest survival. Additionally, in a more formal test of the assumption that higher abundance infers greater habitat quality, Sallabanks and Quinn (2000) suggested that habitat quality could be reliably inferred from high abundance for Song Sparrows in Washington. Additionally, a recent literature review concluded that "birds are ususally more abundant in habitats where reproduction is highest, confirming the legitimacy of using bird counts as indicators of breeding habitat quality and as a basis for management decisions" (Bock and Jones 2004).

Conclusions

The impact of the Vision Fire on riparian-breeding landbirds varies by species as expected; abundance for some species was higher on burned sites, lower for others. These results were in concordance with the habitat associations of the individual species. In general, the fire did not appear to strike a catastrophic blow to the birds breeding in the riparian habitat at Muddy Hollow. In fact, the fire may have created conditions that enhanced reproductive success for Song Sparrow and perhaps other species as well.

Long-term monitoring is essential to understanding the effects of episodic events like fire (O'Conner 1991). It is unfortunate that we had no standardized pre-fire data, as these would have provided a more robust picture of the effects of the Vision Fire on landbirds. Still, our results highlight the utility and importance of monitoring programs to inform land managers.

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Literature Cited

Aquilani, S. M., D. C. LeBlanc, and T. E. Morrell. 2000. Effects of prescribed surface fires on ground- and shrub-nesting neotropical migratory birds in a mature Indiana oak forest, USA. Natural Areas Journal 20:317-324.

Bock, C. E., and Z. F. Jones. 2004. Avian habitat evaluation: should counting birds count? Frontiers in Ecology and the Environment 2: 403-410.

Brawn, J. D., and S. K. Robinson. 1996. Source-sink population dynamics may complicate the interpretation of long-term census data. Ecology 77:3-12.

Chase, M. K. 2001. Reproductive success, natal dispersal, and recruitment in a Song Sparrow (*Melospiza melodia*) population breeding in

Table 4. Number of Song Sparrow nests monitored, total days of observation, number of nests that failed, were successful, or of uncertain fate, daily nest survival, and cumulative (total) nest survival by site for the entire nesting period[a] (24 days from Chase 2002), 1997.

Site	n	Observer days	Failed	Successful	Uncertain	Daily survival ± SE	Total survival (95% CI)
Muddy Hollow	22	220.0	4	16	2	0.982 ± 0.009	0.64 (0.42-0.99)
Lagunitas Creek	23	259.5	13	9	1	0.950 ± 0.014	0.29 (0.15-0.56)
Olema Creek	15	147.5	10	4	1	0.932 ± 0.029	0.19 (0.06-0.52)

[a] Total nest survival = (daily survival)^{24}

California coastal scrub. PhD. Dissertation, University of California, Riverside.

Chase, M. K. 2002. Nest site selection and nest success in a Song Sparrow population: the significance of spatial variation. Condor 104: 103-116.

Dieni, J. S., and S. H. Anderson. 1999. Effects of recent burning on breeding bird community structure in aspen forests. Journal of Field Ornithology 70:491-503.

Gardali, T., G. R. Geupel, A. L. Holmes, and D. L. Humple. 2003. Songbird response to wildfire in a coastal riparian system. In California Riparian Systems: Processes and Floodplain Management, Ecology, and Restoration (Faber, P.M. ed.). 2001 Riparian Habitat and Floodplains Conference Proceedings, Riparian Habitat Joint Venture, Sacramento, CA.

Howe, R. W., G. J. Davis, and V. Mosca. 1991. The demographic significance of 'sink' populations. Biological Conservation 57:239-255.

Johnson, D. H. 1979. Estimating nest success: the Mayfield method and an alternative. Auk 96: 651-661.

Manolis, J. C., D. E. Anderson, and F. J. Cuthbert. 2000. Uncertain nest fates in songbird studies and variation in Mayfield estimation. Auk 117: 615-626.

Martin, T. E. and G. R. Geupel. 1993. Nest monitoring plots: methods for locating nests and monitoring success. Journal of Field Ornithology 64:507-519.

Mayfield, H. 1975. Suggestions for calculating nest success. Wilson Bulletin 87:456-466.

Middleton, A. L. A. 1993. American Goldfinch (*Carduelis tristis*). In The Birds of North America, No. 80, A. Poole and F. Gill (Editors). Philadelphia: The Academy of Natural Sciences; Washington D.C.: The American Ornithologists Union.

Nice, M. M. 1937. Studies in the life history of the Song Sparrow. Trans. Linn. Soc. NY 4:1-246.

Nur, N., Geupel, G. R., and Ballard, G. 2000. The use of constant-effort mist-netting to monitor demographic processes in passerine birds: annual variation in survival, productivity, and floaters, in Strategies for bird conservation: The partners in flight planning process. Proceedings of 3rd Partners in Flight Workshop, Oct. 1-5 1995, Cape May, NJ. Proceedings RMRS-P-16 (R. Bonney, D. N. Pashley, R. J. Cooper, and L. Niles, eds.), p. 185-194. U.S. Dept. Agri., Forest Serv., Rocky Mountain Research Station, Ogden, UT.

Nur, N., and W. J. Sydeman. 1999. Demographic processes and population dynamic models of seabirds: implications for conservation restoration. Current Ornithology 15:149-188.

O'Conner, R. J. 1991. Long-term population studies in the United States. Ibis 133:36-48.

Pulliam, H. R. 1988. Sources, sinks, and population regulation. American Naturalist 132: 652-661.

Ralph, C. J., G. R. Geupel, P. Pyle, T. E. Martin, and D. F. DeSante. 1993. Field Methods for Monitoring Landbirds. General Technical Report PSW-GTR 144. USDA Forest Service.

Ralph, C. J., J. R. Sauer, and S. Droege. 1995. Monitoring Bird Populations by Point Counts. General Technical Report PSW-GTR 149. USDA Forest Service. 181 pp.

Raphael, M. G., M. L. Morrison, and M. P. Yoder-Williams. 1987. Breeding bird populations during twenty-five years of post fire succession in the Sierra Nevada. Condor 89:614-626.

Rogers, C. M., M. J. Taitt, J. N. M. Smith, and G. Jongejan. 1997. Nest predation and cowbird parasitism create a demographic sink in wetland-breeding Song Sparrows. Condor 99:622-633.

Rohrbaugh, Jr. R. W., D. L. Reinking, D. H. Wolfe, S. K. Sherrod, and M. A. Jenkins. 1999. Effects of prescribed burning and grazing on nesting and reproductive success of three grassland passerine species in tallgrass prairie. Studies in Avian Biology 19:165-170.

Sallabanks, R., and T.Quinn. 2000. Can we infer habitat quality from the results of wildlife surveys? Final report to Washington State Department of Natural Resources, Olympia, WA.

Sauer, J. R., and B. K. Williams. 1989. Generalized procedures for testing hypotheses about survival or recovery rates. Journal of Wildlife Management 53:137-142.

Shuford, W. D. 1993. The Marin County Breeding Bird Atlas: a Distributional and Natural History of Coastal California Birds. California Avifauna Series 1. Bushtit Books, Bolinas, CA.

Shuford, W. D., and I. C. Timossi. 1989. Plant communities of Marin County. California Native Plant Society. Special Publication 10.

Stata Corp. 1997. STATA, Release 5.0 Stata Corp., College Station, TX.

Van Horne, B. 1983. Density as a misleading indicator of habitat quality. Journal of Wildlife Management 47: 893-901.

Post-Fire Vegetation Response in the Bishop Pine Forest at Point Reyes National Seashore

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Abstract

The effect of the 1995 fire on the regeneration, composition and cover of the Bishop Pine (*Pinus muricata*) forest at Point Reyes National Seashore is examined. The human-caused fire burned over 12,000 acres in and around the National Seashore engulfing most of the bishop pine forest along Inverness Ridge, where fire intensity was the greatest. Bishop pine is a closed-cone pine that typically needs fire in order to regenerate. Past fire suppression policies created a forest of mostly senescent trees with little regeneration. Vegetation data were collected in 1996 and 1997 by sampling transects and quadrats noting species composition, cover, and number of seedlings present. Bishop pine seedling densities after the fire were high, yet variable. Overall seedling density decreased over the two years of study, but remained high. Significant increases in blue blossom (*Ceanothus thyrsiflorus*), fireweed (*Erechtites minima*), lupine (*Lupinus arboreus*), and huckleberry (*Vaccinium ovatum*) were observed over the sampling time. Species diversity increased over time. An inverse relationship between fireweed (*E. minima*) (an exotic species) and seedling survival and density was detected, although long-term effects appear minimal. These data provide a baseline for long-term fire recovery with this National Seashore surrounded by an urban environment.

Introduction

In October 1995, a wildfire burned over 12,000 acres of the Point Reyes National Seashore. The Vision Fire, named after its ignition point, engulfed most of the bishop pine (*Pinus muricata* D. Don) forest along Inverness Ridge, where fire intensity was greatest.

Prior to the Vision Fire there was some concern about the regeneration of the bishop pine forest. Fire suppression policies in most National Park units had left a legacy of heavy fuel buildup leading to extreme crown fires (Kilgore 1976a). The importance of fire was finally recognized in 1968, when the National Park Service adopted a new administrative policy. Natural fire was recognized as one of the ecological factors that contributed to the viability of plants and animals native to a given habitat (Kilgore 1976a). Monitoring the effects of such fires is one step toward understanding the complexity of the relationships in these unique plant communities.

This study examines the response of vegetation after fire in the bishop pine community. To investigate these responses, thirty permanent plots were established and monitored as part

Chinook helicopter dropping water on fire.

of a long-term study. This information will provide a biological inventory that will assist resource managers in making informed decisions about managing the vegetation community and its response to fire.

Regeneration in the bishop pine community is one focus of this study. Bishop pine seedling density in relation to fire intensity within the forest area is examined. Species composition also provides an important basis for the ecological understanding of fire effects upon vegetation. The change in vegetative cover during the first few years following the fire gives an indication of the progression of vegetation change over time. The associations found between and among species during the early regenerative period provide a baseline for future studies. Since the Vision fire was an intense crown fire, and Park Service policy is to minimize such conflagrations, this study will provide a baseline for comparison with vegetation recovery under planned controlled fires. In this manner ecological understanding of species composition and plant community survival under different fire regimes can grow.

Study Area

Bishop pines form a dense forest of approximately 5,000 acres along the northern portion of Inverness Ridge on the Point Reyes Peninsula, with occasional small islands of trees scattered along the ridge toward Drakes Bay. Fossil records indicate that this forest is a relict of a much greater forest that dates back to the Pleistocene era (Peattie 1981; Sugnet 1984; Vogl et al. 1988). The Inverness Ridge bishop pine population is but one of several discontinuous related populations found scattered in the coastal areas from northern Baja California, Mexico to the California-Oregon border (Forde and Blight 1964; Wehtje 1991). The distribution of bishop pine on Inverness Ridge adheres closely to the granitic rock type (Galloway 1977). The south end of the ridge bears a forest of Douglas fir (*Pseudotsuga menziesii Mirbel*) growing in association with the Monterey Shale substrate.

The Point Reyes Peninsula supports several rare plant species. Within the bishop pine forest, the Vision ceanothus (*Ceanothus gloriosus var. porrectus* J. Howell) occurs. This species is listed as rare by the California Native Plant Society (CNPS) (Hickman 1993). Marin manzanita (*Arctostaphylos virgata*), another rare species, is also found within the bishop pine forest (USDI 1995).

Trees growing in association with bishop pine are: madrone (*Arbutus menziesii*), coast-live oak, (*Quercus agrifolia*), wax myrtle (*Myrica californica*), bay laurel (*Umbellularia californica*), tan oak (*Lithocarpus densiflorus*) and golden chinquapin (*Chrysolepis chrysophylla*). Shrub species include: California coffeeberry (*Rhamnus californica*), huckleberry (*Vaccinium ovatum*), salal (*Gaultheria shallon*), manzanita (*Arctostaphylos* species), bush lupine (*Lupinus arboreous*), and blue blossom (*Ceanothus thyrsiflorous*). Herbaceous species include: morning-glory (*Calystegia purpurata*), peak rush-rose (*Helianthemum scoparium*), hedge nettle (*Stachys ajugoides*), and blackberry (*Rubus ursinus*).

The Vision Fire

The Vision fire began in the afternoon of October 3, 1995 from the flare up of an unat-

Inspecting area for hotspots.

tended illegal campfire. Fueled by high winds (40-50 mph) and a buildup of late summer forest fuels, the fire quickly spread. Pushed by warm offshore winds with low relative humidity, the fire consumed 12,350 acres by the time it was contained on October 7, 1995. Of the total acreage, 11,600 acres are located within Point Reyes National Seashore. Hot spots continued to flare up after containment with full control of the fire declared at midnight October 16, 1995. Forty-four homes were destroyed in the private community adjacent to Point Reyes National Seashore (PRNS 1997).

Approximately 1,300 acres (11%) of the burn area were listed as high intensity burn, 2,290 acres (19%) were listed as moderate intensity burn, and 8,340 acres (70%) were listed as low intensity burn (USDI 1995).[1] Most of the high intensity burn occurred in the northern third of the fire range in the steep granitic terrain along Inverness Ridge. This area is largely populated by bishop pine forest. Burn intensity for the Bayview Trail area was "moderate", even though destructiveness seemed to equal the Inverness Ridge Trail upon post-fire observations. Total acreage of bishop pine forest burned was 1,040 acres (USDI 1995).

Natural History of The Bishop Pine Community

Bishop pines are a member of the closed-cone pine group, which includes Monterey pine (*Pinus radiata*), knobcone pine (*P. attenuata*), and two subspecies of *Pinus contorta* - beach pine (*P. contorta* ssp. *contorta*) and pygmy pine (*P. contorta* ssp. *bolanderi*) (Vogl et al. 1988). The bishop pines possess two needles per fascicle in the adult stage (Schoenherr 1992). The Point Reyes population exhibits a three-needle seedling stage. The needles can vary from a bluish-green form in the northern populations to a more distinct green in southern populations. The Point Reyes population displays the southern green needle color (Millar and Critchfield 1988). The cones of the tree are born in whorls held tight to the tree stem (Peattie 1981). Cones are approximately two inches long with prickly spines (Schoenherr 1992). Cone production was thought to occur by about ten years of age (Krueger 1916) but cone production was evident five years after the fire in the bishop pine along Inverness Ridge Trail. The tree grows tall but twisted in Point Reyes, whereas in the Mendocino pygmy forest the bishop pine is often dwarfed (Duffield 1951). The morphological plasticity of bishop pine throughout its range pays tribute to its climatic and edaphic adaptability.

Serotinous bishop pine cone opening during fire.

Fire and the Bishop Pine Forest Community

Most plant species in fire-dependent ecosystems have developed structures, mechanisms, and functions that depend on fire for their regeneration (Vogl 1980). The plant community of the bishop pine forest at Point Reyes National Seashore is no exception; it is a population that is periodically subjected to fire as part of its natural history (Sugnet 1984). Bishop pines bear tightly closed cones that open only after fire or extreme heat (Vogl et al. 1988; Johnson 1994). These pines produce copious quantities of seeds that prefer the freshly mineralized soil after a fire for germination.

Most Point Reyes' bishop pine stands have come in as a result of fire (Krueger 1916). The last evidence of a large stand-replacing fire, like the recent Vision fire, was in October 1927 (Sugnet 1984). Since bishop pine is a stand-replacing species, another fire before cone production commences could have a severe impact on the bishop pine population.

Plant species have evolved various mechanisms of survival in fire, and a high intensity fire will favor certain species. Seedlings of fire dependant species after a high intensity fire are usually more abundant than resprouting species (Horton and Kraebel 1955). Different fire intensities determine germination of different species (Whelan 1995). Low intensity fires uncover seed banks in only the topmost layers of soil, while higher intensity fires uncover deeper seed banks that have been stored much longer.

Fire frequency is also seen as a determinant of species composition (Keeley 1995); too frequent a fire interval can cause local extinctions by destroying plants before they are able to reproduce. This difference in species maintenance with different fire regimes points to the problem of trying to use fire in a prescribed fashion. Application of only low intensity fire regimes may eliminate certain species that depend upon higher intensity fires. A small test burn on Mount Vision in the 1970's demonstrated that even fires of moderate intensity can destroy stands of mature bishop pine, throwing into doubt the effectiveness of prescribed fire for protecting these adult trees (USDI 1993). Hence, fire prescription for this type of area is difficult, given the chaotic mix nature seems to prefer.

Other species associated with the bishop pine forest, such as wax myrtle (*Myrica californica*) and huckleberry (*Vaccinium ovatum*)

reproduce from sprouts after fire. Resprouting strategies employed by many shrub species, as well as abundant production of seeds that need fire to germinate, virtually ensures the continuation of fire-adapted species (Keeley and Zedler 1978). Ceanothus seeds are stimulated to germination by the heat of fire, and provide nitrogen-fixing qualities to the soil (Daubenmire 1968, Quick and Quick 1961; Biswell 1989; Schoenherr 1992). Another nitrogen-fixing plant, the lupine germinates only after intense fires (Whelan 1995).

Methods

Data on plant community response to the Vision fire were collected during the summers of 1996 and 1997. Currently a five-year assessment (1995-2000) is underway. Areas under study are located on Inverness Ridge and Bayview Trails. The Inverness Ridge Trail area had been represented in Burn Area Emergency Rehabilitation Plan (USDI 1995) maps as a high-intensity burn, while the Bayview Trail area was listed as a medium-intensity burn. Consequent reconnaissance in early February 1996 revealed little visible difference in the two areas.

Thirty 50-meter line transects were laid out systematically, in random directions, off Inverness Ridge Trail and portions along Bayview Trail. Along Inverness Ridge Trail transects were placed about every 25-30 paces for approximately one mile. These sites were all located approximately 20 to 30 meters off the trail to aid in relocation and uniformity, while preventing disturbance. Bayview Trail sites were located within a large island of burned adult bishop pine trees. Transect direction selection were deliberately placed away from the trail. Topography varied from protected canyon to windswept ridge top, while elevation varied little.

The line-point intercept method was used for sampling vegetation. Seedling density was noted using quadrat sampling along the transect line. General slope, aspect, and elevation data were also recorded for each transect.

Transect beginning and end points were marked with a permanent three-foot rebar, metal tagged with the site number, then located using a Trimble Geo-Explorer II GPS unit (Trimble Navigation 1991).

Thirty 50 meter line transects were sampled for species presence and seedling density and were recorded along with general site information. Species at point intercept were recorded at every meter along a transect line. Species occurrence and height of the tallest species were recorded at each point for a total of 1500 point samples. If no species were found at the point sampled, the substrate of bare ground or litter was recorded. Bishop pine seedlings were counted in 20 square meter sections staggered along each side of the line transect at 5-meter intervals. This provided a total area of 600 square meters (30 x 20m^2) of seedling samples. This method was chosen to allow a more accurate view seedling density to be able to document natural thinning in later years.

Data Analysis

General vegetation and site characteristics were recorded for each transect. These field observations are useful for understanding the general picture of succession. Species lists were tallied for each site for each year sampled. Percent cover and seedling density were obtained.

Paired sample t-tests were performed to determine the significance of changes in species frequencies between 1996 and 1997. Analysis of Variation (ANOVA) was used to find the variance levels of seedling densities. Correlations were performed by transect to show relationships among species, as well as among species and environmental factors. Statistical analyses were performed using SPSS 7.5 (Norusis 1997).

Results and Discussion

Species present during each of the two years sampled were recorded. Since vegetation was sampled during summer months, many annual species had already died and were not recorded. Consequently, this sample reflects only the occurrence of trees, shrubs, perennials, and hardy annuals.

Overall species numbers were established during the first year, with very little change from the first year to the second, although most transects showed an increase in species diversity. Twenty-eight species were found during the first year, increasing to 32 species the second year. Fewer than half of the total species were represented in any one transect. Also, while species diversity increased very little overall, the distribution of species diversity shifted with passage of time. Fewer species were represented on each site the first year, while 1997 showed a shift toward greater diversity per study area. Inverness Ridge sites that were protected represented the greatest diversity of species both years (26 and 28 respectively), but also contained the largest

[1] Burn intensity refers to the results of the fire and the intensity of the effects on watershed conditions, especially in regard to released sediments and threat of erosion (USDI 1995).

number of samples. Most species established themselves early, while a few weren't noted until the second year. California man-root (*Marah fabaceus*) vines were seen very early after the fire, (probably spouting from tubers) noted in 1996, but no longer present in 1997. The tiny annual, varied leaf collomia (*Collomia heterophylla*) was likewise seen early and was not seen the second season. Other species were not seen until the second year, especially the tiny, rare Vision ceanothus (*Ceanothus gloriosus* var. *porrectus*). Pink everlasting (*Gnaphalium ramosissimum*), bush monkeyflower (*Mimulus aurantiacus*), and red flowering currant (*Ribes sanguineum*) all were present in samples only during the second year.

For the most part, the slight increase in species diversity was due to the presence of a single plant representing that species. *Solanum* sp. was found only once, as was *Carex* sp.. Other species were seen but not sampled on the line transects. Of these, the most common was madrone (*Arbutus menziesii*), a gooseberry (*Ribes* spp.), and tan oak (*Lithocarpus densiflorus*). Many other species were quite common along the trail but rarely seen off the trail. These included pink everlasting and bush monkeyflower.

Some species were present throughout all sample areas. Bishop pine and bracken fern were present in every transect. Ceanothus was present in all but one transect. Huckleberry was present in all but two transects.

Table 1 lists the 20 most common species for comparison and shows results for the overall landscape area as well as the statistical significance of the changes.

The first year composition shows a large portion of all the areas sampled (17.5%) as bare ground. This composition changed radically by the second year when only 2 percent of the total area sampled was bare ground. When examined by geographic area, more differences are highlighted. Vegetation quickly covered the bare ground on Inverness Ridge protected sites with 20 percent bare ground cover in the first year and only 2 percent in the second year. Exposed sites on Inverness Ridge remained without vegetation in 3 percent of the area after the second year, whereas 15.5 percent of the area was without vegetative cover the first year. On the Bayview sites all but nine percent of the ground was covered after the first year. Vegetation regeneration was obviously more vigorous in the Bayview area. This was expected due to the deeper soils and lower intensity burn. The photos in Figures 1a and 1b illustrate the rapid change in cover of one of the Inverness Ridge protected area sites. Note the tall seedlings in the foreground in the photo on the right and the clump of ubiquitous fireweed in the background.

One of the most striking changes in species composition was the proliferation of blue blossom ceanothus (*Ceanothus thyrsiflorus*),

Table 1. Comparison of Vegetation Change in Overall Landscape for 20 most common species. Percent cover calculated from total times species intercepted the transect (hits) divided by 1500 possible hits (50 points X 30 transects). Significance values were calculated using t-tests and are based on 95% confidence levels with 29 degrees of freedom.

Species Name		Total Hits 96	% Cover 96	Total Hits 97	% Cover 97	Overall % change
Erechtites minima	Australian fireweed	12	0.8	159	10.6	1225.0
Arctostaphylous sp.	manzanita	3	0.2	20	1.3	566.7
Chrysolepis chrysophylla	chinquapin	4	0.3	14	0.9	250.0
Rhamnus californica	coffeeberry	2	0.1	7	0.5	250.0
Lupinus arboreous	bush lupine	86	5.7	261	17.4	203.5
Ceanothus thyrsiflorus	blue blossom	248	16.5	728	48.5	193.5
Pinus muricata	bishop pine	165	11.0	443	29.5	168.5
Vaccinium ovatum	huckleberry	61	4.1	124	8.3	103.3
Helianthemum scoparium	rush-rose	16	1.1	28	1.9	75.0
Quercus agrifolia	Coast live oak	3	0.2	5	0.3	66.7
Gaultheria shallon	salal	7	0.5	9	0.6	28.6
Rubus ursinus	Cal. Blackberry	150	10.0	182	12.1	21.3
Polystichum munitum	w. sword fern	10	0.7	12	0.8	20.0
Lotus species	lotus	272	18.1	268	17.9	-1.5
Pteridium aquilinum	w. braken fern	151	10.1	125	8.3	-17.2
Stachys ajugoides	hedge nettle	31	2.1	20	1.3	-35.5
Rubus parviflorus	thimbleberry	8	0.5	3	0.2	-62.5
Calystegia purpurata	morning-glory	36	2.4	10	0.7	-72.2
- Litter -		297	19.8	76	5.1	-74.4
- Bare Ground -		262	17.5	32	2.1	-87.8
Total Cover			**121.6**		**168.4**	

Figure 1a (left)
1996 transect along Inverness Ridge with fireweed in the background and very small pine seedlings in the foreground.

Figure 1b (right)
The same transect in 1997, with bishop pine seedlings dominating the plot.

which increased significantly from 16.5 percent the first year to nearly 48.5 percent cover the second year. The greatest cover of this plant was found on Inverness Ridge protected sites (56%) as would be expected, although the exposed and Bayview sites also had extensive populations of this species (33% and 39.5% respectively). Ceanothus is known to frequently occur on arid sites prone to wildfire (Quick and Quick 1961). These plants are also known for their nitrogen-fixing properties, so the large-scale presence of ceanothus should have a positive effect on the soil along Inverness Ridge (Biswell 1989). The survival of these seeds in a high intensity burn points to the long-term storage of these seeds deep within the soil layer. Regrowth of this species has been rapid. Many plants were over six feet high by the second summer, and stands were very dense. Initial observations in year five found ceanothus continuing its codominance with bishop pine and reaching heights of over 3 meters (Figure 2).

Figure 2: Fall 2000. Bishop pine and ceanothus along Inverness Ridge Trail.

Another nitrogen-fixing species, the bush lupine (*Lupinus arboreus*), increased significantly in overall cover by more than 200 percent, from 6 percent during the first summer, to 17.5 percent during the second summer. Certain legumes will germinate only after intense fire due to seed storage deep within the soil layer (Whelan 1995). This may account for the total lack of lupine cover in the less intense burn area off Bayview Trail. Whether fire intensity is actually a factor is speculative however, since the actual life history of *Lupinus arboreus* off Bayview Trail is unknown. Excluding the Bayview transects, the mean percent cover of lupine in all Inverness Ridge sites was 20 percent. There was a fairly even distribution of *Lupinus arboreus* between the protected and exposed sites at the end of the second season (19% and 22% cover respectively). By year five lupine had virtually disappeared from the site.

Another legume, although relatively small as an individual plant, provided much of the low ground cover in areas where shrubs were not present. *Lotus* species constituted nearly 18 percent of the total cover by the end of the second year, changing relatively little from the first year. The most noticeable difference in distribution of this plant occurred in the Bayview sites where occurrence of *Lotus* dropped significantly from 18.5 percent in 1996 to 2.5 percent in 1997 (Table 1). Lumped together into the general category at the genus level, this group is most likely made up of two separate species, *Lotus humistratus E. Greene* and *Lotus purshianus* (Benth.) Clements and E.G. Clements var. *purshianus*. These plants formed dense mats throughout the study area on what might have otherwise been bare ground.

Erechtites minima, the invasive Australian fireweed, presented itself as a formidable challenge to management by its increased presence in the study area. First year sampling found only a dozen of these plants, although many more were seen along the trail, while 159 fireweed plants were found in year two. Fireweed spread dramatically after the first year's seed

Figure 3: Xeric site in 1996 (above) and 1997 (below). Within two years after the fire, most of the area is covered by vegetation.

dispersal, to 10.5 percent of cover by the second summer (Table 1). This overall average is misleading however, since a much larger increase in percent cover (20%) was seen in the xeric sites (Table1). Figure 3 shows a comparison of a xeric site in 1996 and 1997. Small fireweed plants can be seen covering the ground in the 1997 photo. Park service efforts to eradicate this plant along trails seemed only to increase its presence. Several of these plants were greater than seven feet tall, and its dandelion like reproduction strategy, light wind blown seeds, helps to ensure proliferation of the species. It grows in clumps, sometimes in dense thickets, and appears to have a detrimental effect upon soil stability.

California blackberry (*Rubus ursinus*), occurred in 12 percent of sites in 1997. In places this plant seemed to weave together the ceanothus, making penetration for sampling a thorny challenge. There was little change in overall cover of blackberry (Table 1). Bayview sites exhibited the largest increase (22% to 33%), although this change was not statistically significant.

Bracken fern (*Pteridium aquilinum*) was one of the earliest species to resprout from the charred ground. Its numbers dropped off slightly during the second year from 10 percent to just over 8 percent cover. The drop off was most significant on the exposed sites (15% to 10%).

Cover of huckleberry (*Vaccinium ovatum*) increased significantly in one year to over 8 percent. Judging by the large stumps of charred shrubs remaining, this resprouter will probably gain further dominance in the understory.

The rare Vision ceanothus (*Ceanothus gloriosus* var. *porrectus*) was sampled in only one spot in 1997. Several other sightings were made of this species in 1997. Although the presence of this species outside of the sampling sites was not part of this study, their occurrence was noted.

Bishop Pine

Bishop pine reproduction was phenomenal: 15,229 seedlings were counted in 1996 and 6,629 were counted in 1997. First year seedling density averaged 25 seedlings per square meter, and ranged from 71 to 33/m^2, while covering the landscape in 11 percent of the sampling sites. Second year density dropped to 11 seedlings/m^2 ranging from 32. to 2/m^2. Cover increased to 29.5 percent of the total area sampled.

Clumping of seedlings was also significant. ANOVA runs on data from 1996 and 1997 show a significant rate of variability among density values within each season. Figure 4 shows average seedling density by transect for 1996 and 1997. Figure 5 compares bishop pine seedling cover by transect for 1996 and 1997. These illustrate the inverse relationship between density and cover for the bishop pine seedlings. As density decreased cover increased. Highest densities were found in Inverness Ridge protected sites, with an average of 33 seedlings/m^2 in 1996 and 13 seedlings/m^2 in 1997.

Lower seedling densities were seen near the top of Inverness Ridge, as would be expected. Here, in the drier, exposed area changes were less significant. Seedling density decreased from an average of 6/m^2 to 4/m^2, while cover increased from four percent to 11.5 percent. Seedling density on Bayview sites decreased from 23.5 seedlings/ m^2 to 15 seedlings/m^2. Percent cover in the Bayview sites increased dramatically from 14.5 percent in 1996 to 40.5 percent in 1997, although the significance level is low due to the limited sites.

Associations

One of the more interesting aspects of this study is the representation of species associations. Several bivariate correlations were performed to see if relationships between the presence of certain species were related to the presence of bishop pine seedlings. Two of these tests stand out in particular. The first test was to determine if the thick regrowth of ceanothus was affecting the bishop pine seedlings. There was no apparent linear relationship between seedling establishment and ceanothus regeneration at this time. Although the high density of ceanothus is assumed to contribute to seedling mortality, it was not evident in year two. Other correlations were performed to examine if inverse relationships in seedling density between the sampling years with ceanothus cover existed. These were unexpectedly positively correlated. Future observations may show greater seedling mortality, but since seedlings established themselves well before the explosion of ceanothus, mortality may not have yet occurred. Additionally, there was a greater amount of moisture within the thick ceanothus, and this may have served as a type of nursery environment for the seedlings in the early stages.

In contrast, the effect of fireweed upon seedling establishment is pronounced (Figure 6). Fireweed seemed may have a desiccating effect

upon the soil, while ceanothus may provide soil moisture protection. However, it seems doubtful that fireweed will have a serious effect on the reestablishment of the bishop pine forest. Given the head start the seedlings possess, as well as their proliferation, quick growth rates, and ability to colonize poor soil conditions, a new bishop pine forest will develop relatively quickly. Casual observations at year four found little fireweed present.

Conclusions

Vegetation in Point Reyes National Seashore has demonstrated its resilience to fire. The rapid vegetative regrowth after the 1995 Vision fire characterizes this aspect of the ecosystem in the bishop pine forest. A large portion of vegetative cover after the second year (roughly 85 percent) was composed of three nitrogen-fixing plants. This is typical of early successional stages and indicates a healthy process. As all of these nitrogen-fixing species reproduce from seed, it would be interesting to compare their composition after this fire with their composition after controlled burns to see if any (or which) nitrogen-fixing species recover.

Nearly 30 percent of the area sampled was covered by bishop pine at the end of the second sampling season. The data suggest we will see another bishop pine forest in the near future, characterized by species that have long been documented as associations within this forest type. Although the invasive fireweed (*Erechtites minima*) increased initially it declined in later years. At this point its influence on the plant community seems minimal. There is a dearth of literature on Australian fireweed and more ecological studies need to be made of this species, especially with regard to its effects on the soil.

Fifth Year Update

Although detailed data on year five are not available at the time of this writing, initial observations were made. Bishop pine continues to co-dominate with blue blossom ceanothus. Both species reached heights of over three meters. Bishop pine also exhibited cone production five years earlier than previously reported (Krueger 1916). There was some seedling mortality under surviving bishop pines and ceanothus shrubs. Lupine and fireweed have virtually disappeared from the study area. The Bishop pine forest burned in the Mt Vision fire appears to be well on the way to becoming a healthy and vigorous plant community.

This ongoing research documenting the vegetation changes now at five year intervals, along with other research on the vertebrate, invertebrate and fungi population's response to the Vision fire, will help to provide a clearer picture of fire's effect on the unique natural community at Point Reyes.

Acknowledgements

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Literature Cited

Biswell, Harold H. 1989. Prescribed Burning in California Wildlands. Berkeley: UC Press.

Daubenmire, Rexford. 1968. Plant Communities: A Textbook of Plant Synecology. New York: Harper and Row Publishers, Inc.

Duffield, John Warren. 1951. Interrelationships of the California closed-cone pines: with special reference to *Pinus Muricata* D. Don. Ph. D. Dissertation. University of California, Berkeley.

Figure 4.

Comparison of average bishop pine seedling density per square meter by transect. Data shown compares 1996 to 1997. Bars represent +/- one standard deviation.

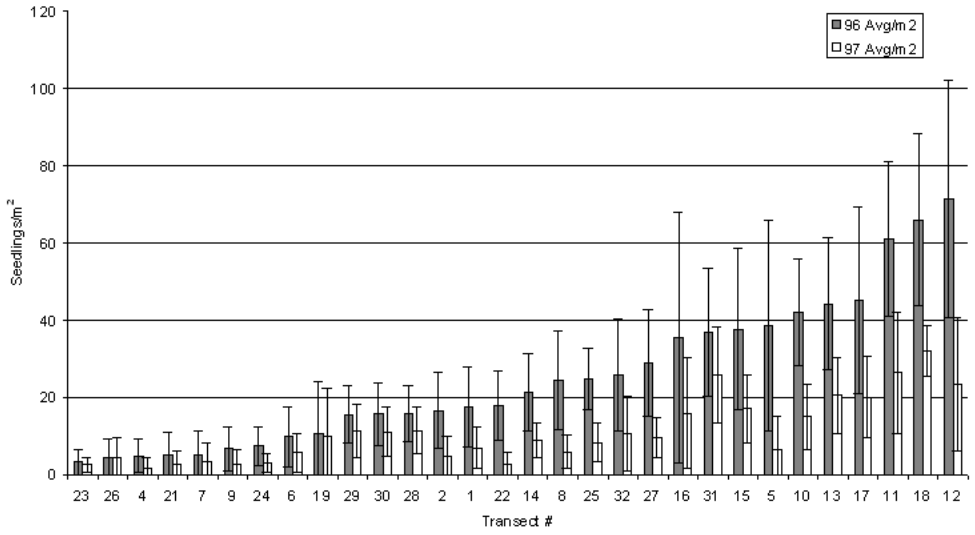


Figure 6. Presence of fireweed increased considerable from 1996 to 1997. Note the large patch with dried flower heads already to seed in August 1997.

Forde, Margot B. and Margaret M. Blight. 1964. "Geographical Variation in the Turpentine of Bishop Pine." New Zealand Journal of Botany 2: 44-52.

Galloway, Alan J. 1977. Geology of the Point Reyes Peninsula, Marin County California. Bulletin 202. Sacramento: California Division of Mines and Geology.

Hickman, James C., ed. 1993. The Jepson Manual: Higher Plants of California. Berkeley: UC Press.

Horton J. S. and C. J. Kraebel. 1955. "Development of Vegetation after Fire in the Chamise Chaparral of Southern California." Ecology 36(2) : 244-262.

Johnston, Verna R. 1994. "Closed-Cone Pines and Cypresses". In California Forests and Woodlands: A Natural History. Berkeley: UC Press, pp. 58-74.

Keeley, Jon E. 1995. "Future of California Floristics and Systematics: Wildfire Threats to the California Flora." Madrono 42(2):175-179.

Keeley, Jon E. and Paul H. Zedler. 1978. "Reproduction of Chaparral Shrubs After Fire: A Comparison of Sprouting and Seeding Strategies." The American Midland Naturalist 99(1):142-161.

Kilgore, Bruce M. 1976a. "From Fire Control to Fire Management: An Ecological Basis for Policies." In Transactions of the 41st North American Wildlife and Natural Resources Conference. Washington, DC: Wildlife Management Institute.

Krueger, Myron Edward. 1916. The Bishop Pine. M.S. Thesis. University of California, Berkeley.

Millar, Constance I. and William B. Critchfield 1988. "Crossability and Relationships of *Pinus Muricata* (Pinaceae)." Madrono 35(1): 39-53.

Norusis, Marija J., ed. 1997. SPSS 7.5: Guide to Data Analysis. Upper Saddle River: Prentice Hall.

Peattie, Donald Culross. 1981. A Natural History of Western Trees. Boston: Houghton Mifflin Company.

Point Reyes National Seashore. 1997. Fire: Burning Issues in Fire Management. Pamphlet. Point Reyes: Point Reyes National Seashore.

Quick, Clarence R. and Alice S. Quick. 1961. "Germination of Ceanothus Seeds." Madrono. 16: 23-30.

Schoenherr, Allan A. 1992. A Natural History of California. Berkeley: UC Press.

Sugnet, Paul Walsh. 1984. Fire History and Post-Fire Stand Dynamics of Inverness Bishop Pine Populations. M.S. Thesis. University of California, Berkeley.

Trimble Navigation. 1991. PFINDER Software Users Guide. Sunnyvale: Trimble Navigation.

US Department of the Interior. 1993. Fire Management Plan: An addendum to the Natural Resources Management Plan. Point Reyes National Seashore: National Park Service.

US Department of the Interior. 1995. Vision Fire Incident: Burned Area Emergency Rehabilitation (BAER) Plan. Point Reyes National Seashore: National Park Service.

Vogl, Richard J. 1980. "The Ecological Factors that Produce Fire-Dependent Ecosystems." In Cairns, John, Jr. (ed.) The Recovery Process in Damaged Ecosystems. Ann Arbor: Ann Arbor Science Publishers, Inc. pp. 63-94.

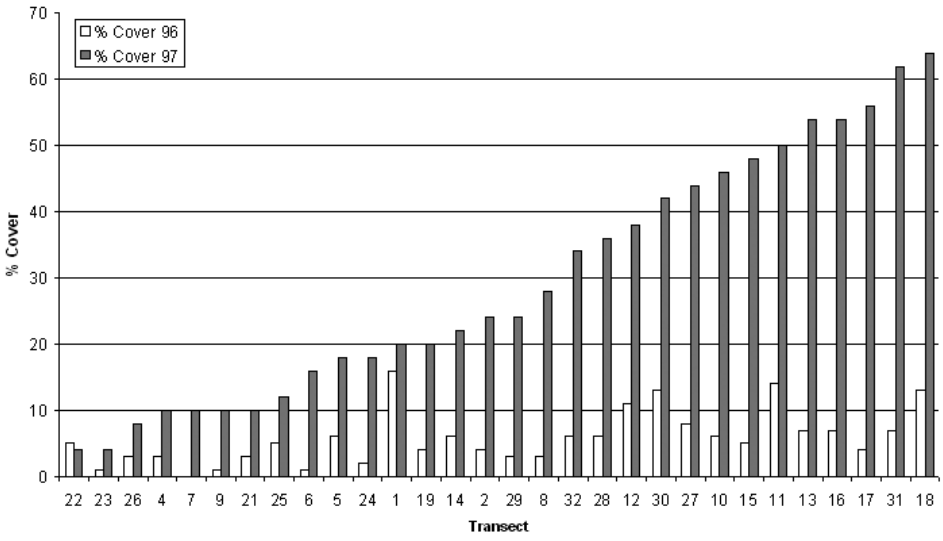
Vogl, Richard J., Wayne P. Armstrong, Keith L. White, and Kenneth L. Cole. 1988. "The closed-cone pines and cypresses." In Barbour, Michael G. and Jack Major (eds.) Terrestrial Vegetation of California. Sacramento: California Native Plant Society.

Wehtje, Walter. 1991. Response of Bishop Pine (*Pinus muricata*) to Cessation of Browsing of Feral Sheep on Santa Cruz Island, California. M.A. Thesis. University of California, Los Angeles.

Whelan, Robert J. 1995. The Ecology of Fire. Cambridge: Cambridge University Press.

Figure 5.

Comparison of bishop pine seedling percent cover per transect. Data shown compares 1996 to 1997. Percent cover was calculated by dividing the number of hits per transect by the total possible (50).



Fluvial Geomorphic Response of a Northern California Coastal Stream To Wildfire

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Abstract

The 1995 Vision Fire burned 100% of the 8.3 sq km (3.2 sq mi) Muddy Hollow Watershed at Point Reyes National Seashore. We monitored landscape response for two years in this watershed by measuring rainfall and stream flow, surveying channel change, and quantifying the amount of woody debris supplied to the channel and sediment deposited in a newly formed alluvial fan. Watershed response varied spatially and temporally. With a mean annual rainfall of about 1016 mm/yr (40 in/yr), recovery of fire-adapted vegetation was rapid. The upper and lower watershed sections had the earliest and most rapid geomorphic response. During the first post-fire year in the bishop pine forest of the upper watershed, the combined development of a soil surface crust and hydrophobicity caused excessive runoff.

Channels incised and eroded headward. Permanent drainage density, not including the ephemeral rill network, increased by at least 200%. Sediment was supplied from both hillside rills and channel bed incision. Dry ravelling was negligable. Yet, we expect that the upper section will be first to recover a stable geometry and return to usual rates of runoff and sediment supply. The middle watershed did not significantly change until the second post-fire year when excessive amounts of large woody debris fell into the stream from the riparian alder corridor. Subsequently, sediment supply increased from local bank erosion and sediment transport rates decreased as sediment became trapped and stored in the channel bed. Recovery through the entrenched middle reaches may take years as the large woody debris is redistributed during flood flows and as a mature riparian community develops. The lower watershed responded to high runoff and sediment supply by developing an alluvial fan with a braided distributary system across the valley floor. A minimum estimate of average post-fire sediment load during the first two years was 704 tonnes/sq km/yr (2,626 tons/sq mi/yr). Sediment supply to the alluvial fan was 2.7 times higher during the second post-fire year than the first, due to the changes that occurred in the middle reach. As sediment and water supply diminish, the lower channel will likely return to a single thread channel within a more extensive alder forest. Recovery may occur sooner in the lower watershed than in the middle, since the lower reach channel is not entrenched. High shear stresses during flood flows in the middle reaches will cause continued adjustments in hydraulic geometry. The post-fire channel response for this coastal Northern California stream did not follow the typical Fire/Flood Sequence that is often expected for coastal Southern California watersheds. We hypothesize that differences in watershed response in California's moist north coast environment as opposed to its semi-arid south coast are due to differences in apparent cohesion that is created by the dense root network from vegetation. Within the watershed, response varies spatially along the stream and elevational gradient.

Acknowledgements

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Introduction

The fluvial geomorphic response of a stream network in an 8.3 sq km (3.2 sq mi) watershed was monitored for two years to understand the initial influences of wildfire on a coastal Northern California stream. Virtually all of Muddy Hollow Creek Watershed was burned during the October 3rd, 1995 Vision Fire in the Point Reyes National Seashore (PRNS). This report, which was funded by PRNS, focuses primarily on data collected during the second year of monitoring post-fire effects. It draws upon results from first year monitoring (Collins, 1996), a project funded by the Marin Community Foundation for the West Marin Environmental Action Committee. A copy

of the first year report is on file in the PRNS Natural Resources Library.

Muddy Hollow Creek was chosen to learn about natural geomorphic and fluvial processes following intense wildfire for three principal reasons. One, most of the watershed was already in a relatively natural condition, except for previous land use impacts from grazing; it therefore allowed monitoring of natural processes without confounding influences of urban runoff. Two, fluvial and geomorphic processes were not influenced by-post fire disturbances such as aerial seeding, application of mulches, or construction of sediment basins because National Park Service policies did not require these post-fire erosion control techniques. Three, while much literature addresses landscape response for streams and watersheds for coastal southern California, and inland coniferous watersheds, yet scant information exists for north coastal watersheds, particularly bishop pine forests, which have very different floristic environments due to their moist coastal climate and greater rainfall regime.

Background Information about Landscape Response to Fire in Southern California

Until recently, much of the post-fire erosion control response to wildfire throughout California has been based upon landscape response of Southern California watersheds, even though climate, vegetation, and geology vary throughout the State. Post fire erosion and runoff monitoring following the 1991 Oakland Hills Tunnel Fire (Booker, 1998; Collins and Johnston, 1995; and Booker et al, 1993) has begun to document wildfire response in northern California that does not typically follow the often-anticipated Southern California model. If we first examine the typical landscape response for Southern California, differences and similarities in response for the MHC Watershed can then be recognized and understood. Such an understanding could lead to improved post-fire management response, cost–effective planning, and necessary future monitoring efforts for other north coast watersheds.

Current literature indicates that many coastal Southern California ecosystems have a fairly predictable response to fire, called the Fire Flood Sequence (Rice 1982; Wells 1981,1987). The scenario for these arid watersheds follows.

In the fire-flood model, the first significant winter storms produce unusually high runoff and sediment loading from burned watersheds. The high runoff is caused by a hydrophobic (water repellent) layer that forms in the shallow subsurface soils during fire. Vaporized organic compounds condense in the soil subsurface at a fairly uniform but relatively shallow depth to create a layer that impedes water infiltration (Debano, 1981). As a result, much of the rainfall that would normally percolate into the soil is shed as surface runoff. This causes a substantial increase in the amount of flow, as well as the height of the peak flood relative to the amount of runoff. Hence, the amount of runoff relative to the size of the drainage becomes uncharacteristically large for non-fire conditions.

Two very different geomorphic processes cause the high sediment load. One is associated with water, the other is not.

In the Fire-Flood scenario, the immediate process associated with sediment supply to the channel network is dry ravelling. It occurs either during or immediately after fire, whereby soil, rock, and debris ravel from steep hillsides directly into dry ephemeral or intermittent channels, filling them with large amounts of loose unconsolidated sediment. When the vegetation burns, its effect of buttressing the soil that has accumulated against the uphill side of the trunk is lost. On steep hillsides, substantial quantities of sediment can be rapidly delivered to the channel through this mechanism, especially if the soils lack cohesion.

Clays generally provide soil cohesion, and root networks provide apparent cohesion. Many of the soils in the Southern California Coast Ranges have granitic soils, which characteristically have very low clay content. For the Southern California arid climate, the fire-adapted vegetation is typically sparse, creating soils that tend to have a low density of root networks, low organic content, poorly developed litter layers, and abundant areas of exposed soil surface. In the absence of fire, the process of dry ravel has been reported to account for as much as 50% of the annual erosion in the steep mountains comprised of granite or coarse-grained sandstone bedrock (Anderson et al, 1959; Krammes, 1965; Rice, 1974). The sediment that accumulates in the stream beds from dry ravelling processes cannot be mobilized until there is sufficient winter runoff contained within the channel.

The second sediment supply process is associated with the hydrophobic layer and

requires rainfall and runoff to deliver sediment to the channel system. Overland flow on the hillsides concentrates into rills, causing soil loss and additional sediment and ash supply to the channel network from the hillsides. The coarse-grained characteristics and low clay content increase the probability that hydrophobic conditions will exist, even though pre-fire infiltration rates are usually high. Once a water repellent layer develops, the layer of soil and ash above it is easily removed by surface erosion processes, while the layer below remains dry. If the hydrophobic layer is pervasive, rill networks will develop over much of the landscape (Wells, 1986). If rainfall is intense or prolonged, and the soil deep, the rills will incise into gullies.

Sediment additions from both ravelling and rilling processes cause bulking of the first winter stream flows with loose unconsolidated sediment. These highly bulked flows, in association with increased runoff, can produce downstream mud flows that may have similar effects to debris torrents and debris flows. In particular, alluvial fans at the base of steep hillslopes may aggrade by deposition and spreading of bedload, while large volumes of fine sediments are transported farther downstream within the floodwaters.

The response of MHC was different than the post fire erosion response in the Northern California Oakland Hills Tunnel Fire that was studied by Booker et al (1993). In both cases, dry raveling was not an important process of sediment supply to the streams, yet development of a rill network and subsequent sediment supply was much greater in the MHC watershed than the Oakland Hills. In the Oakland Hills post fire erosion response was insignificant until after the soils were mechanically disturbed during post fire rebuilding efforts and by subsequent fuel management practices (Collins, unpublished data). We believe that the main difference in response of these two northern California environments was due to the different soil types that caused a surface crust and a greater degree of hydrophobicity to develop in the granitic soils of MHC as opposed to the more clay-rich sedimentary soils of the Oakland Hills.

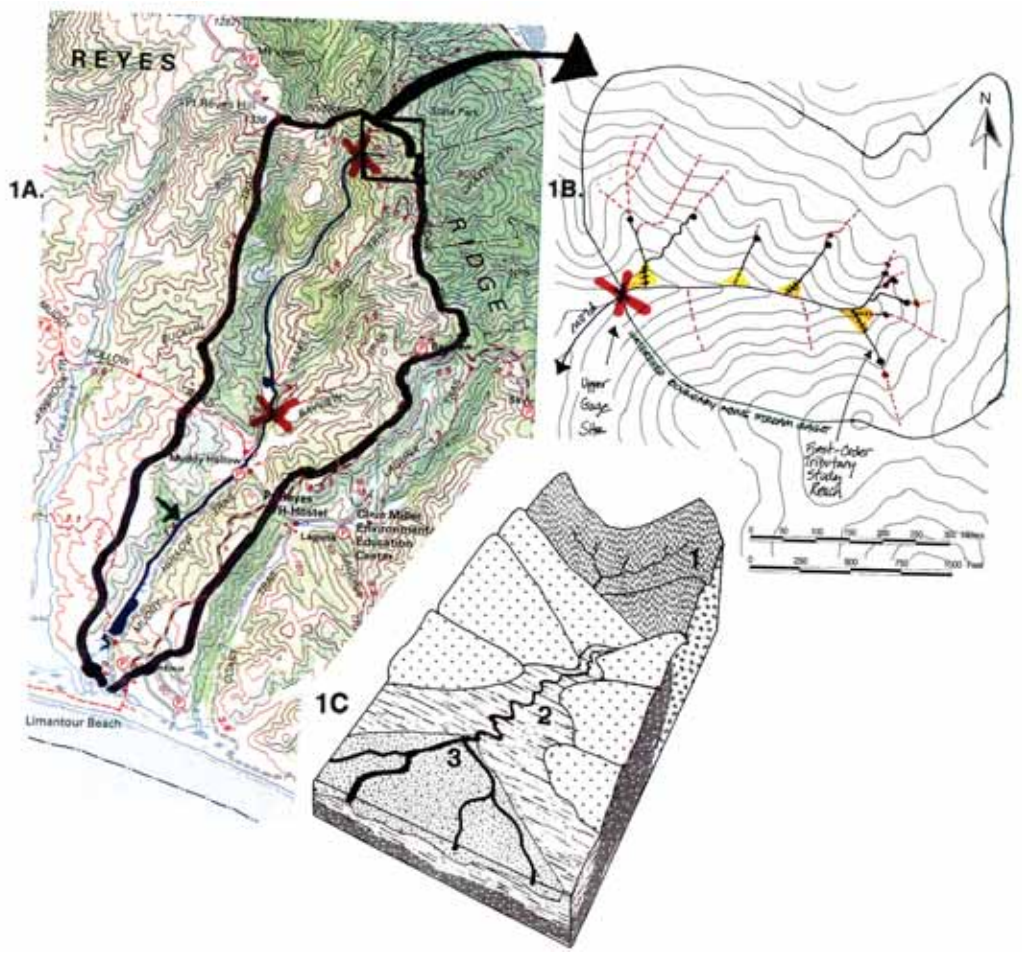


Figure 1

A) The boundary of Muddy Hollow Reservoir is outlined in black. The red X at the top of the watershed shows the Upper Gage Site location. The red X in the middle shows the Lower Gage Site location. The green arrow near the bottom of the watershed shows the location of the apex of the alluvial fan during the spring of 1998

B) A detail of the Upper Watershed Section is shown. Red dashed lines represent rills that developed after the fire along the main axis of zero-order basins. Blue dots show the original location of channel heads, while red dots show their post-fire location. Hatched blue lines show alluvial fans that became channelized after the fire, while dotted blue lines show alluvial fans that did not become channelized. Yellow areas show alluvial fans at the base of steep tributaries.

C) A conceptual diagram of the watershed is shown where point 1 represents the Upper Section which is primarily granitic rocks as shown at depth at point 1b. Point 2a and 2b represent the Middle Section which has both Quaternary sands and gravels (2a), and marine sedimentary rocks (2b) as shown at depth at point 2c. Point 3 represents the Lower Section as indicated by the alluvial fan which has very recently deposited sands and gravels.

Physical Setting of Muddy Hollow Watershed

Topography

The 8.3 sq km (3.2 sq mi) watershed of MHC flows 3.4 mi toward the Pacific Ocean on the southwest side of the PRNS in Marin County, California (Figure 1). Peak elevation is about 328 m (1,076 ft), which also corresponds to total topographic relief. The steepness of the hillsides approaches 60% in several upper headwater drainages, but as MHC flows southwestward, it eventually cuts through consecutively younger marine sedimentary bedrock that has increasingly gentle topography. After MHC reaches half its total distance, it flows into its alluvium-filled valley. At its downstream end, fresh water flows from Muddy Hollow Reservoir into an artificially truncated tidal salt marsh that is part of the southeastern edge of the Estero de Limantour. Limantour Spit blocks the Pacific Ocean from the estuary shoreline of Muddy Hollow.

Climate

The Mediterranean-type climate of the PRNS provides a mean annual rainfall ranging from about 1,000 mm/yr (40 in/yr) at Inverness Ridge to 600 mm/yr (24 in/yr) at the western shoreline (Evans, 1988). Snowfall is extremely rare. Most precipitation occurs between December and February. Figure 2 shows monthly precipitation at the Bear Valley Ranger Station for the 1995 Water Year(WY) preceding the fire and the three years following. Bear Valley Station is 3.2 km (2 mi) southeast of the Muddy Hollow headwaters.

1997 was the only year that did not have greater than average rainfall. Instead, it had nearly normal rainfall of 970 mm (38.2 in). 1998 was an El Nino winter. It had greater than 180% of normal rainfall totaling 1,840 mm (72.7 in). The first winter after the fire (1996) had 115% of normal rainfall. The year preceding the fire (1995) was also a wet year. Rainfall was 159% of normal.

Geology

The geology of Point Reyes has been recently re-described by Clark and Brabb (1997) following earlier mapping and interpretation by Galloway (1977). From the eastern headwaters to the western shoreline, the bedrock units in MHC drainage are as follows: in the Upper Section the mainstem MHC incises into mildly to strongly weathered, Cretaceous-aged granodiorite and granite for a distance of about 2,194 m (7,200 ft) before next flowing 122 m (400 ft) through a short segment of the Laird Sandstone. MHC then flows for about 488 m (1600 ft) through the Middle to Upper Miocene-aged Monterey Formation, which is a thin-bedded and laminated porcelanite and chert with several thin to medium interbeds of arkosic sand. Throughout the Laird sandstone and from this unit downstream, with a few minor exceptions, mainstem MHC flows through its own Quaternary Alluvial Sediments. The subsequent bedrock units from east to west of the watershed that are exposed in some of the adjacent low gradient tributaries include Miocene glauconitic Santa Margarita Sandstone, Upper Miocene Santa Cruz Mudstone, and Upper Miocene to Lower Pliocene–aged

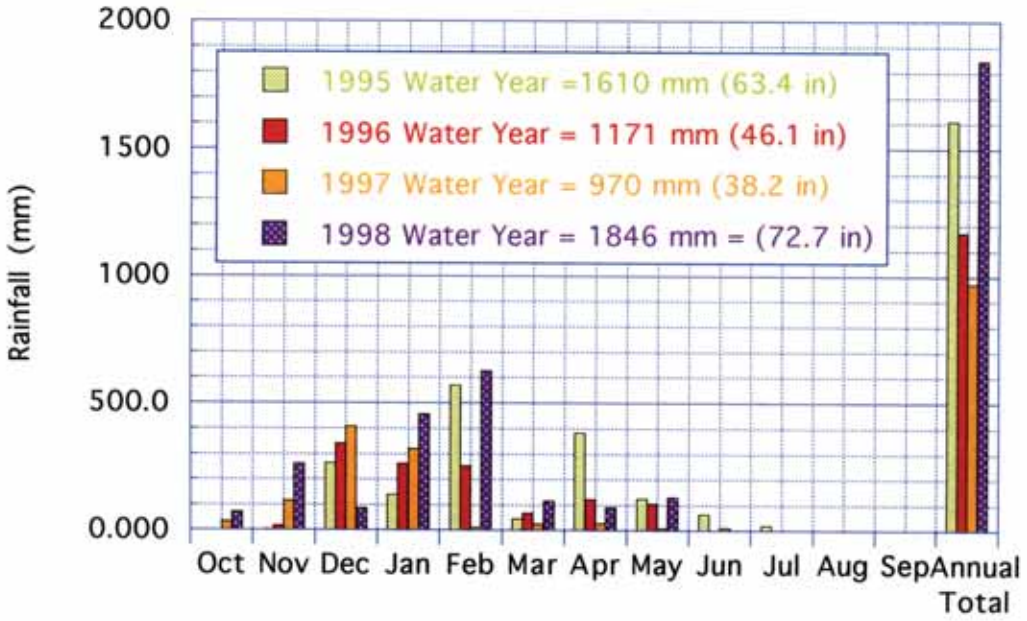


Figure 2

Monthly rainfall totals are shown for the Bear Valley Ranger Station. Water Year 1995 was before the Vision Fire. If the mean annual total for the area is 1,016 mm (40 in), then three of these four years had above average rainfall. The El Nino year of 1998 had 182% of normal rainfall.

diatomaceous mudstone and siltstone beds of the Purisima Formation. This latter unit at the western-most portion of the watershed was formerly referred to as the Drake Bay Formation by Galloway (1977). These geologic units comprise the Middle Section. The Lower Section is comprised entirely of Quaternary Alluvium.

Bedrock throughout MHC and the greater Point Reyes Peninsula has been mechanically weakened, fractured, and sheared by numerous faults that splay from the northwest trending San Andreas Fault. Tomales Bay demarcates the San Andreas Fault. It is located about 1.6 km (1 mi) east of the MHC watershed divide. A major earthquake of magnitude 8.3 on the San Andreas Fault affected this region in 1906. The long-term combined processes of landsliding and fluvial dissection have largely shaped the landscape of MHC. Both seismic shaking and high intensity rainfall have triggered landslides.

Vegetation

The vegetation in MHC watershed varies with geology and elevation. The granitic headwaters above an elevation of 171 m (560 ft) are dominated by bishop pine forest. The predominant understory includes a dense and often impenetrable thicket of huckleberry, manzanita, ceanothus, chinquapin, madrone, salal, coffeeberry, wax myrtle, and sword fern. Between the elevations of 98-171m (320-560 ft), the vegetation varies from bishop pine forest, to oak/bay woodland, or north coastal chaparral that is dominated by coyote brush. Below an elevation of 98 m (320 ft), the hillsides are predominantly north coast chaparral, and grassland to a lesser degree. Along the riparian corridor, downstream of the headwater pine forest, the channel is lined with red alders. They have a dominant understory of blackberry, thimbleberry, and sword ferns. Willow, elderberry, big leaf maple, hazelnut, and buckeye have minor presence. The riparian corridor changes to a wide alder forest at the lower end of the valley at the upstream end of the reservoir. The reservoir was constructed over former tidal marsh. Downstream of the reservoir, the tidal marsh is dominated by pickleweed.

Many of the plant species in Muddy Hollow are adapted to fire. In particular the bishop pines require fire to release seeds from their cones, while other species, such as bay, madrone, manzanita, coyote brush, huckleberry and chinquapin have adapted to fire by crown sprouting. Such fire-adapted vegetation has root networks that remain viable even when the surface biomass has been totally consumed by fire. This is important because the roots continue to add to the apparent cohesion in the soil.

Hydrology

MHC has perennial flow from its headwater first-order streams to its outlet. The lower MHC is a fifth-order channel. It has three major forks. The mainstem channel is the Middle Fork. Stream flow was not gaged prior to this post-fire study. Based upon published Regional Curves of discharge versus drainage area (Leopold, 1994), the predicted bankfull discharges for different segments of MHC are shown in Table 1.

Land Use

Historically, MHC flowed freely to the Estero de Limantour where it transitioned into a tidal slough that invigorated a tidal salt marsh. During the 20th Century, two reservoirs were constructed on the lower and middle sections of the valley. At the lower end of the valley, Muddy Hollow Reservoir was constructed by 1963 for recreational needs of a planned resi-

Table 1. Predicted MHC Bankfull Flow From Regional Curves

Drainage Area Sq KM (Sq MI)	Predicted Bankfull Discharge CMS (CFS)	Length of Channel Upstream KM (MI)	Study Site
0.28 (0.11)	0.18 (6.5)	0.5 (0.3)	Upper Gage Site at Downstream Boundary of Upper Watershed Section
5.07 (1.96)	2.55 (90)	3.2 (2.0)	Lower Gage Site in Middle Watershed Section
7.15 (2.76)	3.54 (125)	5.0 (3.1)	Downstream Boundary of Middle Watershed Section
8.29 (3.20)	4.11 (145)	5.5* (3.4)*	Downstream Boundary of Lower Watershed Section

*Before construction of Muddy Hollow Reservoir, MHC extended 5.9 km (3.7 mi) to the Estero.

dential subdivision. The dam was placed across the tidal marsh, truncating about 457 m (1,500 ft) of the upper salt marsh and its attendant slough that previously conveyed water and sediment to the Estero. Around 1952, prior to dam construction and perhaps in preparation of draining the valley for subdivision, about 701 m (2,300 ft) of the channel upstream of the Estero was ditched, straightened, and moved to the north side of the valley. 1942 photos indicate that much of the lower valley was covered by extensive, but young riparian forest at this time.

The upper reservoir, called Bonelli's Lake, was constructed in the late 1950's. It was located about 2.3 km (1.4 mi) upstream of the Muddy Hollow dam. It was developed for maintaining water supply to a commercial nursery that was located immediately downstream (verbal communication from Dewey Livingston, Historian Point Reyes National Seashore). The dam at Bonelli's Lake breached during a 1982 storm event that caused widespread flooding and landsliding throughout Marin County. The bedload upstream of the lake had been captured for about 32 years behind the dam. Presently, the channel has incised 7.6 m (25 ft) into its previously impounded sediments that now have a young alder forest.

Dairy farming began about 1857 at the Muddy Hollow Dairy. The farm was located in the main valley, 2.4 km (1.5 mi) upstream of the Muddy Hollow Reservoir dam. An 1859 topographic map from the US Coast and Geodetic Survey shows that the valley downstream of the dairy site had four small riparian groves (probably willow) through the lower, mostly grassland valley. According to Livingston, a circa 1900 photograph of the dairy shows that riparian vegetation and the former willow groves were entirely missing along MHC upstream of the dairy to beyond Bonelli's Lake. According to Livingston there was also another smaller, more short-lived dairy, called Sunnyside that was located along the ridge of MHC.

Dairy ranching dominated Muddy Hollow until the 1920's when sheep ranching and truck farming of pea crops took over the general vicinity of the Muddy Hollow Dairy. These latter activities only lasted until the 1940's when the land was sold for subdivision. From the 1950's through the early 1970's cattle herds from nearby beef ranches were allowed to graze the watershed. After the early 1970's the PRNS ceased all grazing activities in MHC watershed.

The influence of road construction in the watershed has been relatively minor. There are no paved roads within the watershed, yet there are a few maintenance trails and abandoned ranch roads. Two significant dirt roads exist. One crosses the creek near the location of the old Muddy Hollow Dairy. It was probably constructed during the truck farming era. When the land was sold for subdivision (around 1950's) its bridge was replaced with a 0.9 m (5 ft) diameter, corrugated metal culvert, which was removed later in 2001. The other dirt road which has been abandoned parallels the mainstem valley from the estuary to the headwaters. The section of road downsteam of Muddy Hollow Dairy was probably constructed between 1960-1963. Within a very small portion of MHC watershed near the top of the Inverness Ridge, there is a gravel road that provides access to several houses along the headwaters of the South Fork. Most of these homes were built during the 1970's and have little influence on MHC.

Fire

The incidence of naturally caused fires, such as those generated by lightening strikes, is probably very low for Point Reyes. Conversely, the incidence of anthropogenic-related fires was fairly high. The frequency of fires however, is probably less now than when the land was occupied by Native Americans that managed patches of their landscape by setting frequent fires. Barbara Moritch, PRNS Vegetation Management Specialist, suggests that fire frequency was about 8-14 years before non-native settlement. Only two fires may have affected the bishop pine forest in MHC during the last 150 years. The more recent fire may have occurred in the lower headwater forest prior to November 1942. This is indicated by examination of historical 1942 stereo photos. Based upon counts of annual growth rings of the oldest bishop pines in the upper northwestern Muddy Hollow ridge, we determined that the last fire in that area might have occurred around 1895.

The high intensity of the 1995 Vision Fire was most likely much higher than historical fires set by native people. In the bishop pine forest, the Vision Fire evolved into a high intensity crown fire, killing all pines in the upper watershed and removing all other vegetative surface cover. Nearly 100% of the watershed burned, except for some wetland areas at the upstream end of Muddy Hollow Reservoir. Fire intensity of the riparian alder corridor ranged from crown fire to understory burn.

Methods

Study Approach and Selection of Study Sites

This study was designed to provide an understanding of the overall landscape response to fire. We developed an approach to provide both extensive reconnaissance of different morphologic sections of the watershed, and detailed monitoring of representative stream reaches. Based upon stream gradient and valley conditions, the watershed along the mainstem MHC was conceptually divided into Upper, Middle, and Lower Sections (Figure 1C).

The Upper Watershed Section consisted of first- to third-order channels within granitic terrain. Average stream gradients are typically greater than 10%. The Upper Section has a drainage area of 0.26 sq km (0.10 sq mi or 68 ac). Following the fire, rainfall and stream flow were gaged at the downstream end of the Upper Section. The length of the upstream channel is about 0.48 km (0.3 mi). Repeated surveys of longitudinal profile and cross section were conducted on first- and third-order sections of the mainstem. Reconnaissance level mapping of the entire channel network was done with a fiberglass tape to measure channel distance. Observations of soil erosion and tests of hydrophobicity were also made over most of the Upper Section. Within it, the mainstem channel flows in a very narrow channel infall that is stepped, probably from former debris flow deposits. The valley fill frequently separates the channel banks from the base of steep hillsides. The pre-fire mainstem channel alternated from being barely incised in the valley fill (about 0.4 m or 1.5 ft), to occasionally deeply incised (up to 1.5 m or 5 ft) in bedrock. Montgomery and Buffington (1997) have described such headwater channels as "sediment source reaches".

The Middle Watershed Section includes the attendant tributary channels and the mainstem channel, which ranges from third- to fifth-order. It ends at the apex of a newly formed alluvial fan in the lower valley. Total drainage area is 7.15 sq km (2.76 sq mi 1,766 ac) and upstream channel length is 5.0 km (3.1 mi). The mainstem gradient ranges from 1% to 10%. Before the fire, few depositional bars existed within the mainstem channel. According to Montgomery and Buffington classification (1977), this might indicate that the Middle Section functioned as a "transport reach." Lack of bars might also indicate that sediment supply was low prior to the fire. Most of the channel was incised within its own, older alluvial fill in a narrow to moderate-sized valley that ranged in width from 15-107 m (50-350 ft). A continuous riparian alder corridor existed along the channel on the valley flat. Rainfall and stream flow were gaged at the lower third of the Middle Section. Drainage area above the gage is 5.07 sq km (1.96 sq mi or 254 ac) and channel length is about 3.2 km (2.0 mi). A detailed stream map of the channel reach at the lower gage site was made (Figure 6). A longitudinal profile was surveyed and several cross sections were repeatedly surveyed.

Before the fire, the Lower Watershed Section consisted of the fifth-order mainstem channel in the lower valley. This section now defines the extent of an alluvial fan that is presently defined by numerous distributaries. Its lower boundary is the open water of Muddy Hollow Reservoir. Valley width ranges from 60-150 m (200-500 ft). Stream gradient is generally less than 1%. The total drainage area upstream of the Lower Section is 8.29 sq km (3.20 sq mi or 2,048 ac) and channel length is 5.5 km (3.4 mi). Before the fire, the Lower Section would have been classified as a "response reach" according to Montgomery and Buffington (1997). Within the alluvial fan, soil pits were dug to quantify the amount of sediment deposition that followed the two post-fire winters. After each significant storm, the position of the apex of the alluvial fan in the lower valley was measured to determine changes in stream behavior.

During the first post-fire spring, the entire mainstem channel was walked. It was re-walked the following spring of 1997. Channel conditions were observed, and woody debris and landslides were quantified. Photographs were taken to document channel condition. By the second year, most of the channel was nearly impassable due to the numerous alder trees that fell into or across the channel.

Testing of hydrophobicity

The extent and significance of water repellency was determined by using a standard water drop test used by the Natural Resource Conservation Service. Small soil pits and trenches were dug to test the significance of hydrophobicity throughout the rainy seasons.

Stream Flow and Rainfall Gauging

The upper gage location was used to indicate runoff conditions from a third-order stream that was influenced by significant hydrophobic soils in the burned bishop pine forest. The lower gage location was used to indicate runoff conditions of the fifth-order stream system in a gently sloped alluvial valley that had low to moderate fire intensity in the riparian alder forest. Data on amount and intensity of

precipitation were collected at both sites with a continuously recording, tipping bucket gage. Rainfall was measured at both sites at intervals of two minutes at the upper site and 5 minutes at the lower site.

The range of stream discharge was determined by developing a rating curve for stream stage at different discharges. Stage was determined by using a continuously recording capacitive sensor at the Upper Site during the first year after the fire and at both sites during the second year. When it was not raining, stage was automatically recorded every half-hour at the upper site and every hour at the lower site. During precipitation, stage was recorded every 2 minutes and 5 minutes, respectively. For both sites, the gage was placed in a straight reach of channel where velocity measurements were taken by using a combination of orange peel floats and/or a Marsh McBurney current meter. Calibration between the two methods was conducted for low to moderate flow conditions. Velocity measurements from surface floats were multiplied by a correction factor of 0.8 to determine average velocity. Stream discharge was determined by surveying the cross sectional area of flow and multiplying by average velocity.

New rating curves had to be developed several times when the cross sectional area was altered by bank erosion and/or changes in bed elevation. At the lower site, maintenance of the gage site and calibration of a rating curve became increasingly difficult because of the effect of woody debris from alders falling into the channel during the second post-fire year. By the third year, the lower gage was moved downstream, but that site also had to be later abandoned due to large woody debris. At the upper site, the combination of re-growth of understory species and pine seedlings by the third year made access to the stream and the gage site nearly impossible.

Cross Section and Longitudinal Survey

Cross sections for discharge measurements were surveyed by using both standard surveying techniques with a self-leveling level and telescoping rod, and by level line measurement with a tape and rod. Stakes were placed to mark the end points of the survey to permit accurate re-measurement. Longitudinal profiles were surveyed with level and rod. Flagging was placed every 10 m (33 ft) to match the center line tape for re-survey. The long profile was surveyed twice at the Upper Gage Site and once at the Lower Gage Site. All elevations reported are relative. The longitudinal profiles were surveyed over at least 20 bankfull widths to be statistically representative of the channel conditions.

Soil Pits

Along the alluvial fan soil pits were dug at 10 m (32.8 ft) intervals along 6 transects that were spaced about 33 m (100 ft) apart. The pits were dug deep enough to intercept the ash layer that fell during the Vision Fire. It was usually no greater than 0.9 m (3 ft) deep. Stratigraphic changes were described and measured to develop a profile that represented the thickness of overlaying post-fire sediment deposits for both post-fire rainy seasons.

Geomorphic Map

A detailed geomorphic map was made of the stream reach at the Lower Gage Site within the Middle Watershed Section during the summer of 1997. Mapping was performed with a Brunton compass and fiberglass tapes placed along the centerline of the channel and at cross sections spaced about 10 m (33 ft) apart. Sorting patterns of the sediment on the channel bed were mapped as distinct size classes and all large woody debris and standing alders within the active channel and near bank zone were mapped.

Results of Monitoring and Discussion of Watershed Response

Upper Watershed Section

Hillside Soils

The combination of high intensity fire, soil texture, organic compounds in bishop pine

Photo 1 (right)
The Upper Watershed Section is shown for January 16, 1996. Light-colored trails mark areas of rilling where ash has been removed by overland flow.

Photo 2 (left)
The same general vicinity of the Upper Watershed Section is shown for February 11, 1996. An extensive network of rills has formed and much of the ash has been displaced.

litter, and possibly the root mycorrhizae associated with bishop pines caused the soils in the upper drainage of MHC to be intensely and pervasively hydrophobic during the first post-fire winter. These soils are granular and lack clays. A water repellent horizon extended from 5 to 20 cm (2 to 8 in) in depth. At the beginning of the first winter, ash covered the top 2 cm of soil. By mid December, after a cumulative rainfall of 260 mm (10.2 in), small rills defined the removal of the ash layer by surface flow (Photo 1). By mid February, after a cumulative rainfall of 600 mm (23.6 in), most of the ash had been displaced from the surface and an extensive network of rills carved the soil (Photo 2). The exposed soil quickly developed a crust of fine particles that sealed the surface from infiltration. This was also observed by Onda et al (1996). They determined that during the initial storm of the season, the runoff ratio was 10, and about 50% of the runoff was caused by the surface crust, not the hydrophobicity. We observed intense hydrophobicity in the presence of dense bishop pine root network where mychorizzae were present.

By the beginning of the second winter, hydrophobicity was patchy and had mostly disappeared by winter's end. Sprouting vegetation, including moss, had broken up much of the surface crust. Wetting of the subsurface was eventually accomplished by a variety of mechanisms including incision from rills, bioturbation by plants and animals of the surface crust and water repellent layer, flow along surface stems and roots of new plants, development of macro pores by plants, and decay of the organic oils. It may be possible that the condition of the mycorrhizae also contributed to the decline in hydrophobicity. It is worth noting that we determined by similar testing techniques that patchy hydrophobic conditions existed in an adjacent unburned bishop pine forest.

As the second and third winter proceeded, we observed that relatively few pines were uprooted. Instead, many seemed to break at their trunks; thus, soil disturbance was minimized. Prior to this time most of the pine trunks were still upright.

Linkages of Sediment Supply from Hillslopes to Channels

A reconnaissance was performed of the upper basin shortly after the fire to observe whether ravelling processes supplied dry sediment to the channel during the fire. None was observed. This was similar to the 1991 Oakland Hills Fire response (Booker et al, 1993). This is considered a significant difference in response from Southern California watersheds. The only place where sediment had been supplied to the stream, before rainfall, was in two locations where large trees, adjacent to the channel in narrow canyons, had fallen from steep slopes. Their upturned root boles caused soil to fan out into the channel.

Evidence of landslides active within the last 25 years was also sought within the Upper Section. A couple of very small bank slumps, caused by previous undermining by flow, were observed at sites where the channel flowed through narrow colluvial/alluvial deposits. During the second winter, there were some additional small bank slumps caused by bank erosion and bed incision. No new landslides were observed until winter February 1998 when El Nino conditions caused prolonged rainfall with periods of high intensity. At least 5 hillslope-generated landslides were observed from this event in the bishop pine headwaters. It is notable that slides were observed throughout various burned and unburned areas of Point Reyes after the February 1998 storms.

After the rains started sediment and ash from the hillsides were supplied to the channel by overland flow and rilling processes on the surface soils. By mid-winter, rills that became permanent channel extensions eroded as much as 0.3 m (1 ft) in depth along the axis of each small valley upstream of first-order channel heads. A dense network of smaller rills fed into each main axis rill. These previously unchannelized basins have been referred to as zero-order basins. The extent to which channels extended headward of zero-order basins is shown in Figure 1B. These new extensions are shown as red dashed lines. They function as ephemeral channels. Although they persisted during the second winter, they did not appear to change their width or depth after the decline in hydrophobicity.

Physical Changes in First-order Channels

The channel head locations of first-order streams throughout the Upper Section were mapped before the onset of winter rains. These channels were all spring fed by perennial flow. They are shown as blue dots in Figure 1B. Following the first post-fire year, the channel heads were remapped because increased runoff had caused most of them to extend headward. The new locations are shown as red dots. The dashes represent the headward channel extensions that went up the main axis of the hollow. These will most likely be ephemeral features that disappear after vegetation recovers and a litter layer develops.

The downstream extent of the first- and second-order channels was also mapped before the influence of winter rains. Most of these channels were not connected to the mainstem MHC because small unchannelized fans at the base of the slopes separated them. This means that upland sediment supply from these channel and hillside linkages was previously disconnected from mainstem MHC, and stored in deposits from fluvial and debris flow processes. Similarly, perennial flow from upstream channels was previously disconnected from the mainstem MHC because it went subsurface at the apex of the fans, thereby contributing to base flow. By mid January 1996, increased runoff carved permanent channels into three of the five fans (fans are showed with hachured lines in Figure 1B). By the end of the rainy season the depth of incision in the fans was as much as 1 m (3.2 ft). The fans shown with dotted lines did not incise, yet they contributed their post-fire runoff as overland flow to mainstem MHC.

As a result of post-fire headward extension and increased connection of first- and second-order tributaries to the main Muddy hollow channel, the drainage density (which is simply the length of channel divided by unit area) changed from 12 m/ha (17 ft/ac) to 25 m/ha (33 ft/ac). This 200% increase is a conservative estimate for the Upper Section for the first year, because it excludes the extensive shallow rill network that extended beyond the central axis of each zero-order basin. Such an increase may more accurately represent the second or third post-fire years, when the shallow rills began to diasapear because of the loss of the deterioration of the hydrophobic layer in the soil. If the spring heads stabilize at their present locations, the long-term change in drainage density may be about 118% in the Upper Section.

A first-order tributary was selected to monitor in detail. Its location (shown with black arrow in Figure 1B) was near the top the drainage. The channel was initially disconnected from the mainstem by an unchannelized alluvial fan. A portion of its middle reach was actually a short subterranean tunnel. Its changes in gradient are shown in Figure 3 for four different time periods: early November 1995 before rainfall; first post-fire mid winter (1/20/96); end of first post-fire year rainy season (2/26/96); and end of second post-fire year rainy season (10/1/97). Overall, the channel gradient of about 33% remained the same. The profile shows consecutive episodes of relatively parallel downcutting by as much as 2.1 m (7 ft), but averaging about 1 m (3.2 ft). The pre-fire average depth of the channel was about 0.6 m (2 ft) and a thin veneer of alluvium covered the bed. The post-fire channel became deeply entrenched by excessive amounts of runoff carving the slightly to moderately weathered and fractured granitic bedrock. Deposition was insignificant. The channel head has extended upstream about 7 m (22 ft), while the ephemeral, main axis rill extended another 76 m (250 ft). Photos 3 and 4 show the changes near the channel head between pre-runoff conditions and three months later.

The widening of the banks by erosion was not as significant during the first winter as it was during the second winter. However, it was more significant during the first summer than it was during the second summer and winter combined. Additionally, sediment supply from undermining of the banks at low to mid-level height exceeded the supply from widening at the top of the banks during both years. Slumping of the banks from undermining did not occur during the first year. It happened at a few

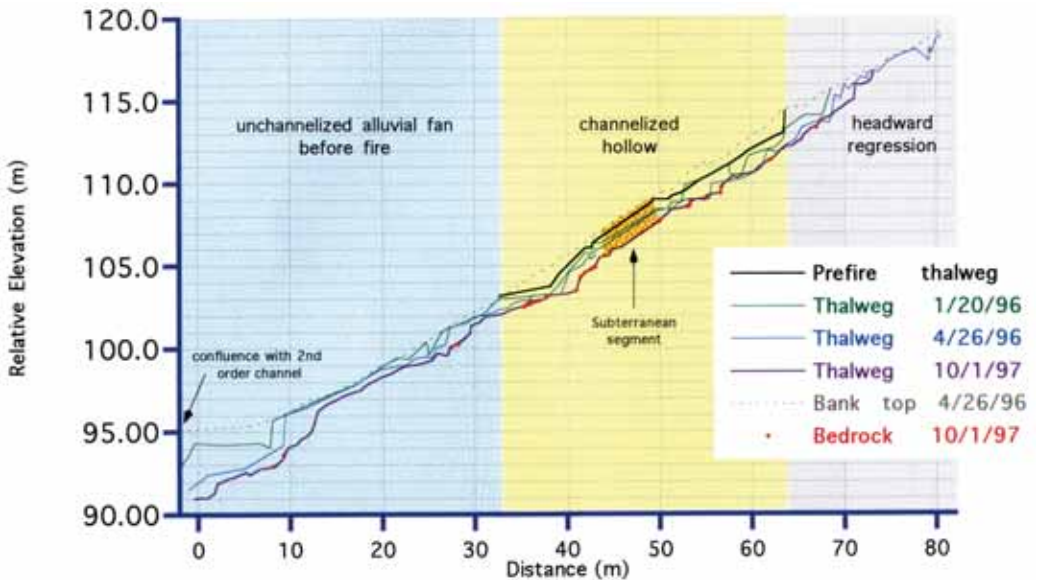


Figure 3 - Muddy Hollow First-Order Tributary Longitudinal Profile.

Repeated longitudinal surveys are shown for the First-order Tributary that is shown in Figure 1B. Before the fire, transport of water and sediment to the mainstem channel was inhibited by the alluvial fan.

sites during the summer of the second year, and happened pervasively during the third summer and winter. This may have been because of the draw-down effect of the water table, the loss of cohesion in the soil from drying, and the loss of root strength as the fine network of bishop pine roots decayed. The amount of channel widening exceeded 1.5 m (5 ft) in several areas and averaged about 0.7 m (2.3 ft).

As indicated by the mapping and surveying of first-order channels the increased drainage connectivity by incision of unchannelized fans accompanied by headward extension increased the rate and amount of water and sediment delivered downstream.

Physical Changes in Third-order Channel

The longitudinal survey of the third-order channel was conducted along 200 m (656 ft) of channel. The lower half was resurveyed four times, the upper half was surveyed twice. Most the sediment that was mobilized in the headwaters was transported through the entire Upper Section. Through the surveyed longitudinal profile reach, only a minor amount of sediment accumulated on 15% of the channel length. Trapping behind small woody debris was usually the cause. At one short site the maximum depositional thickness was 0.58 m (1.9 ft). The remaining 85% of the channel eroded, leaving 50% of its length as exposed bedrock. Average incision depth was about 0.17 m (0.6 ft), but ranged up to 0.46 m (1.5 ft). Most of the erosion took place during the first rainy season.

The small inset in Figure 4 shows the profile as surveyed after the second rainy season over a 200 m (656 ft) reach. The upper part of the reach has a slope of 15% and the lower part has a slope of 11 %. The position of the Upper Gage is shown. The larger graph in Figure 4 shows net erosion and deposition between 1/96 and 2/97 for a 50 m-long (164 ft) segment of the profile between distance stations 100 m and 150 m.

Hydrologic Conditions in Third-order Channel

The Upper Gage data indicate that the amount of rainfall that is required to produce a given flow decreased substantially for the second rainy season compared to the first. It is also highly probable that the amount of rainfall required to produce a given discharge for the post-fire years is substantially less than required for pre-fire conditions. If we assume that the pre-fire bankfull flow was about 0.18 CMS (6.5 cfs), bankfull flow was exceeded at least 7 times during the first post-fire winter and only once during the second winter. This discharge seemed consistent with geomorphic indicators in the channel before the first post-fire rainfall. During the first winter, only 8 mm (0.3 in) of rainfall was required to cause the bankfull flow to be exceeded.

During the second winter, it required 45 mm (1.7 in). This is at least 5.6 times the amount from the previous year and shows the influence of the loss in hydrophobicity and surface crust of the soil.

Photo 3 (left)
The head of the First-order Tributary that was surveyed in detail is shown for early November 1996, before post-fire rainfall influenced the site. The channel had a trickle of perennial flow. Average depth was about 0.6 m (2 ft) and average width about 0.3 m (1 ft).

Photo 4 (right)
The same site is shown as in Photo 3, but the channel has deepened, widened, and eroded headward by 4.5 m (15 ft). Note the exposed granite of the channel bottom that had been previously covered by a thin veneer of sediment before becoming incised by at least 1.3 m (4 ft).

Table 2 shows a number of characteristics for selected storms that occurred from January 1996 to November 1998 at the Upper Gage Site. The data are derived from the hydrographs that were developed from the site. Table 2 indicates the rainfall total used to determine the runoff coefficient and it shows the number of minutes of lag time from peak rainfall to peak discharge. It shows rainfall intensity for 2-, 20-, and 30-min intervals, runoff coefficients, and peak discharge.

Peak rainfall to peak hydrograph averaged about 9.5 min for WY 1996 and about 34 min for WY 1997. The lag time for the storm of Dec 29 for WY 1997 was considered an outlier. Based upon recurrence intervals for rainfall intensities at Point Reyes Station (BAER Team Report, 1995), none of the 30-min rainfall intensities exceeded a 2-year event, which require 10.4 mm of rainfall. MHC runoff coefficients are highly variable depending upon antecedent soil moisture and time of last rainfall.

The highest runoff coefficient, 42%, was for April 1,1996 where a significant amount of rain had fallen several hours earlier. The next highest runoff coefficient, 38%, was from December 29, 1996, which also had been preceded by intermittent rainfall for the previous two days. Interestingly, the April WY 1996 storm also had the highest peak discharge for the year, 0.41 CMS (14.5 cfs), but the Dec WY 1997 storm had a peak discharge of only 0.12 CMS (4.5 cfs), which was not the highest discharge of the year. The storm that had the highest peak discharge in WY 1997 was January 22. It had the highest 30 min rainfall intensity (9.4 mm or 3.7 in), a runoff coefficient of 28%, and a peak discharge of 0.18 CMS (6.5 cfs). The storm that had the highest 30 min intensity (8.8 mm or 3.5 in) in WY 1996 was three hours earlier than the previously reported April 1 storm. It had a runoff coefficient of 26% and a peak discharge of 0.26 CMS (12.8 cfs).

Autumnal base flow was also determined at the Upper Gage Site. During October 1996, base flow was 0.038 cfs, and for October 1997, it was 0.055 cfs. This difference may be significant because 1997 was a lower rainfall year that was just 96% of normal, while 1996 was 115% of normal rainfall. The greater baseflow during the second year may reflect the greater infiltration that occurred after the hydrophobicity and the surface crust diminished.

Middle Watershed Section

Physical Changes in the Third to Fifth-order Channel

In the Middle Watershed Section of MHC only patchy, non-uniform hydrophobicity was observed in the grassland, North Coastal scrub, and oak/bay woodland soils. Neither post-fire dry ravel processes nor rill development were observed along the mainstem channel or adjacent hillsides. This may be due to the more clay-rich soils developed from sedimentary rocks, more gentle topography, and perhaps, less natural patchy hydrophobic soil development within unburned north coastal scrub

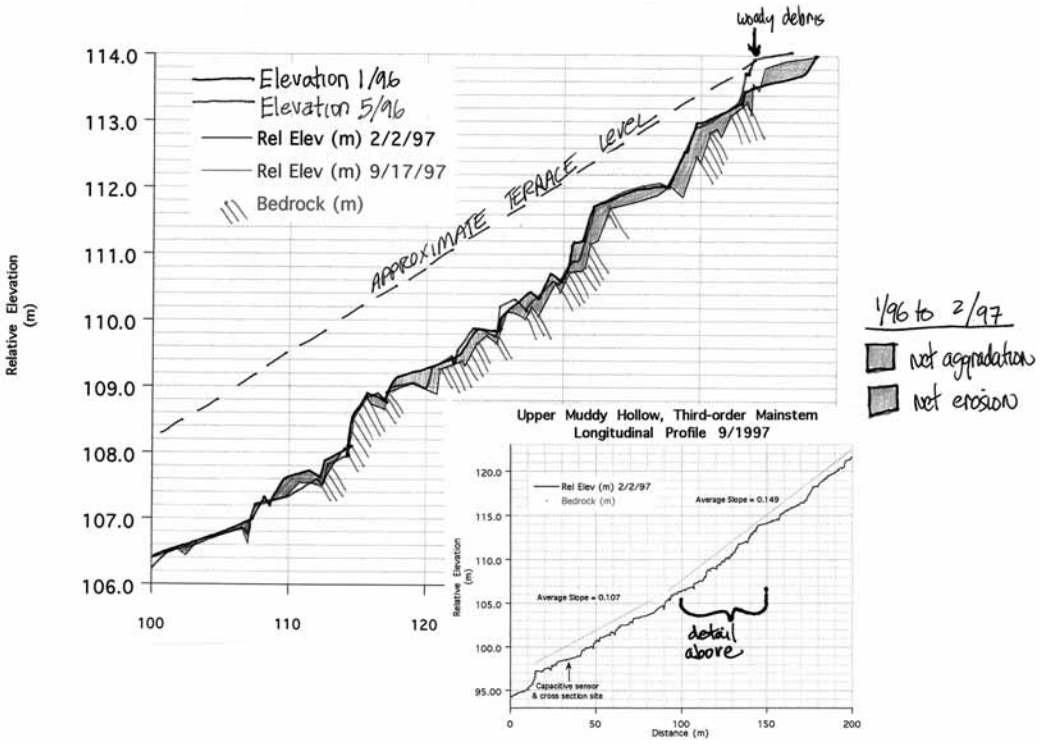


Figure 4
Upper Muddy Hollow Longitudinal Profile, 1996 to 1997.

A detail (large graph) of net erosion and deposition is shown for 50 m (164 ft) of the 200 m-long (656 ft) profile (small graph) of the third-order Muddy Hollow mainstem.

than within bishop pine forests because of different mycorrhizal associations. Rills and gullies were observed on some of the abandoned dirt roads but they had little influence on sediment delivery to the channel.

There were areas where the riparian alders had full crown fire, but more commonly, just the base of their trunks exhibited fire scars. Seared leaves were still visible on most of the alder crowns. Many alders remained viable, although clearly stressed, until the second spring season when they began to fall by either their root boles pulling from the soil or by their trunks snapping well above the ground surface. Subsequently, much more soil disturbance occurred on the valley flats with alder forest than in the steep slopes of the bishop pine forest. This was particularly pertinent along the stream banks where the falling trees literally ripped the banks apart, leaving loose unconsolidated soil in the channel and creating many woody flow obstructions that would initiate channel adjustments and more bank erosion.

During the first 1996 spring reconnaissance of MHC mainstem, we quantified the amount of active landslides and debris jams as we walked down the channel. The number of large woody debris (LWD) that was in the channel prior to the fire was possible to quantify because it did not have burn scars. During the second spring in 1997, landslides and debris jams were recounted. Additionally, the location of LWD was noted as to whether it was in the active channel bed or across the banks. This was important to note because the channel throughout the Middle Section was moderately to deeply entrenched, meaning that only high flood flows would intercept the wood across the banks or it would not be intercepted until it decayed or warped. The results of the reconnaissance are shown in Figure 5A.

The amount of LWD that was in the channel and/or across the banks increased from 260 elements from pre-fire 1995 conditions to 1,043 elements in 1997. The average 1995 wood spacing was 17.3 m (57 ft) over the 4.5 km (2.8 mi) Middle Section. Of the 1997 total of 1,043 wood elements, 86% were alders and most were whole tree trunks. About 10% were elderberries. The remainder included (in order of importance) bay, willow, oak, and juniper. About 56% (580 elements) of the total amount of 1997 LWD were in the active channel bed. This created an average 1997 LWD spacing of 7.7 m (26 ft). The remaining 44% of the total

Table 2. Characteristics of selected storms and flow at the upper gage site

	Rain-fall (mm)	Peak Rainfall to Peak Discharge (min)	RAINFALL INTENSITY TOTAL (mm)			Runoff Coefficient (%)	Peak Discharge (cms/cfs)
			Highest 2 min	Highest 10 min	Highest 30 min		
Water Year 1996							
Jan 20,1996	3.2	9.8	0.8	2.2	3.2	10	0.10 / 3.5
Jan 20,1996	8.0	7.6	1.2	4.2	7.2	19	0.25 / 8.9
Jan 26, 1996	5.6	8.9	1.2	4.2	5.8	17	0.25 / 8.5
Feb 3, 1996	11.0	12	0.6	3.6	3.6	28	0.26 / 9.1
Feb 3, 1996	8.4	7.1	0.6	2.4	3.6	28	0.26 / 9.1
Feb 4, 1996	4.0	11	1.2	2.8	8	22	0.14 / 5.1
Feb 19, 1996	8.8	11	1.0	3.2	3.2	22	0.18 / 6.2
Feb 21, 1996	8.8	13	1.2	3.4	4.4	12	0.25./ 8.8
Apr 1, 1996	3.6	12	0.6	2.2	4.8	10	0.12 / 4.3
Apr 1, 1996	3.6	10	1.6	3.4	4.2	11	0.13 / 4.5
Apr 1, 1996	8.4	11	1.2	5.2	8.8	26	0.26 / 12.8
Apr 1, 1996	8.0	8	2.0	5.6	6.2	42	0.41 / 14.5
Water Year 1997							
Oct , 29,1996	23.4	17	1.4	2.0	3.4	5	0.03 / 1.0
Nov 16, 1996	30.6	19	2.6	4.6	5.8	6	0.03 / 1.1
Dec 5, 1996	59.4	86	1.6	3.2	6.2	10	0.05 / 1.6
Dec 10, 1996	36.6	16	1.4	2.6	6.0	16	0.05 / 1.8
Dec 11, 1996	18.0	30	1.4	1.6	2.2	35	0.04 / 1.5
Dec 26, 1996	40.8	10	1.4	4.6	7.4	13	0.05 / 1.9
Dec 29 - Jan 1	182.2	109	1.6	3.6	6.8	38	0.12 / 4.4
Jan 22,1996	45.8	20	2.2	3.8	9.4	28	0.18 / 6.5
Apr 18, 1997	14	16	1.8	2.0	2.8	17	0.03 / 1.0
Water Year 1998							
Nov 10, 1997	5.0	86	0.6	1.8	3.8	8	0.05 / 1.7
Nov 26, 1997	21.8	28	1.4	3.8	7.6	32	0.04 / 1.4
Nov 29, 1997	20.4	16	1.4	4.4	6.4	19	0.04 / 1.4

Figure 5A
Pre- and Post-fire Woody Debris, Slides and Debris Jams along MHC Middle Watershed Section.

The amound of woody debris before and after the fire is shown. Of the 1,043 woody debris elements influencing the channel, 86% are alders. The amount of slumps and debris jams are also shown for the second two years.

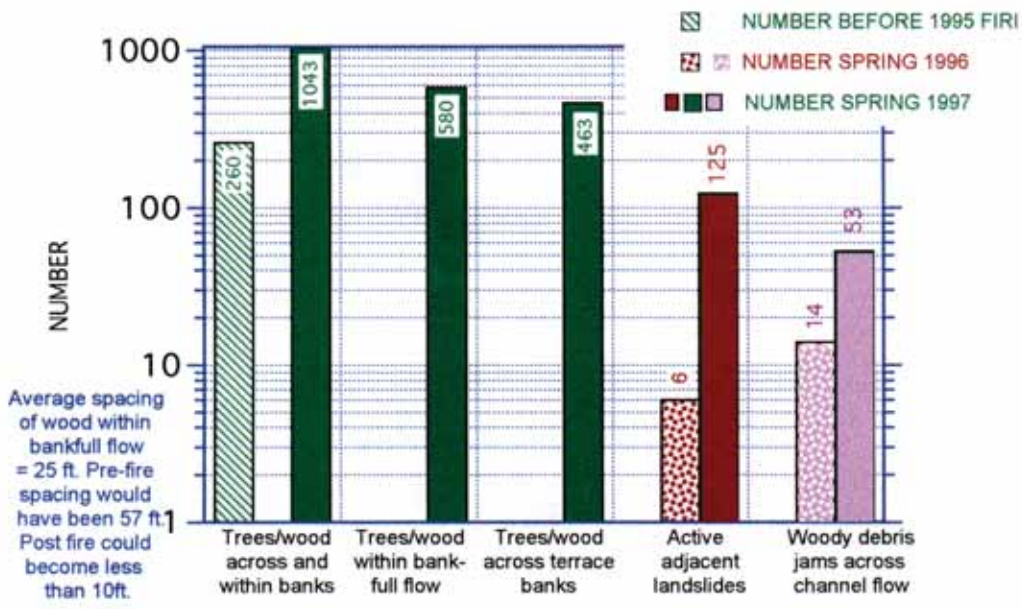
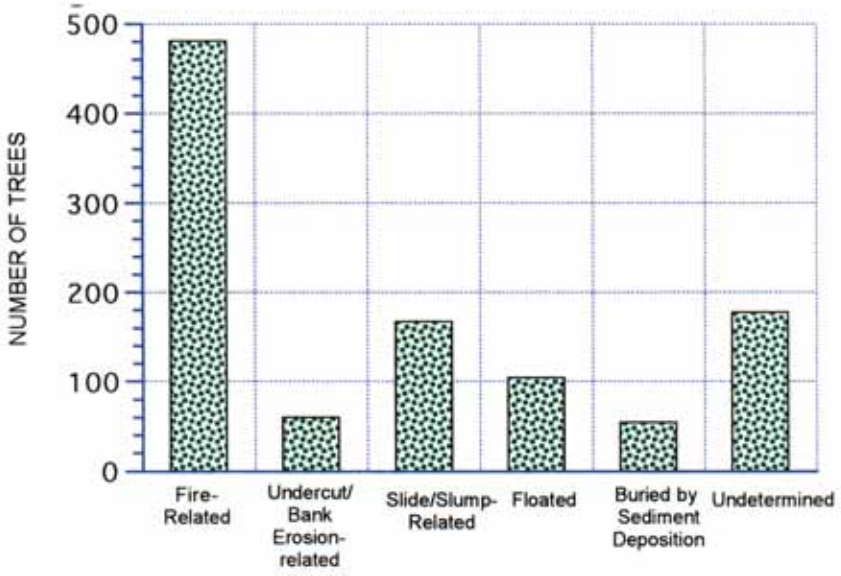


Figure 5B
Recruitment Processes of Woody Debris along the Middle Section of Muddy Hollow Creek, 1997.

The way that woody debris is recruited to the channel is shown. For wood listed as floated, its method of recruitment could not be determined.



number of LWD had fallen across the channel banks. By the third 1998 spring, many of the alders that were across the banks started to bend into the channel or broke apart and fell into the active bed. Thus, a conservative estimate of LWD spacing by 1998 or 1999 is about 3 m (10 ft). Between 1996 and 1997 the number of woody debris jams also increased from 14 to 53. There was no indication that there were existing debris jams before the fire. This number, in the short-term, will likely increase as large alders continue to break into transportable sizes. However, the number will eventually diminish as wood is redistributed during floods and as it continues to decay. Hence, some debris jams will increase in size and could cause channel avulsions.

The number of active slides and slumps along the channel increased from 6 in 1996 to 125 in 1997. This represented an increase in local sediment supply during the second year, not just an increase in sediment supplied from the headwaters. The channel, prior to adjustments caused by the fire, appeared to have become quite stable after it had adjusted to earlier land use impacts of grazing and reservoir construction, which likely contributed to its entrenchment.

The process of LWD recruitment to the channel was also quantified during the 1997 reconnaissance. Figure 5B shows that in 1997, 46% of the LWD elements were directly supplied by the effects of fire, 16% were supplied by landsliding, 10% had already floated downstream

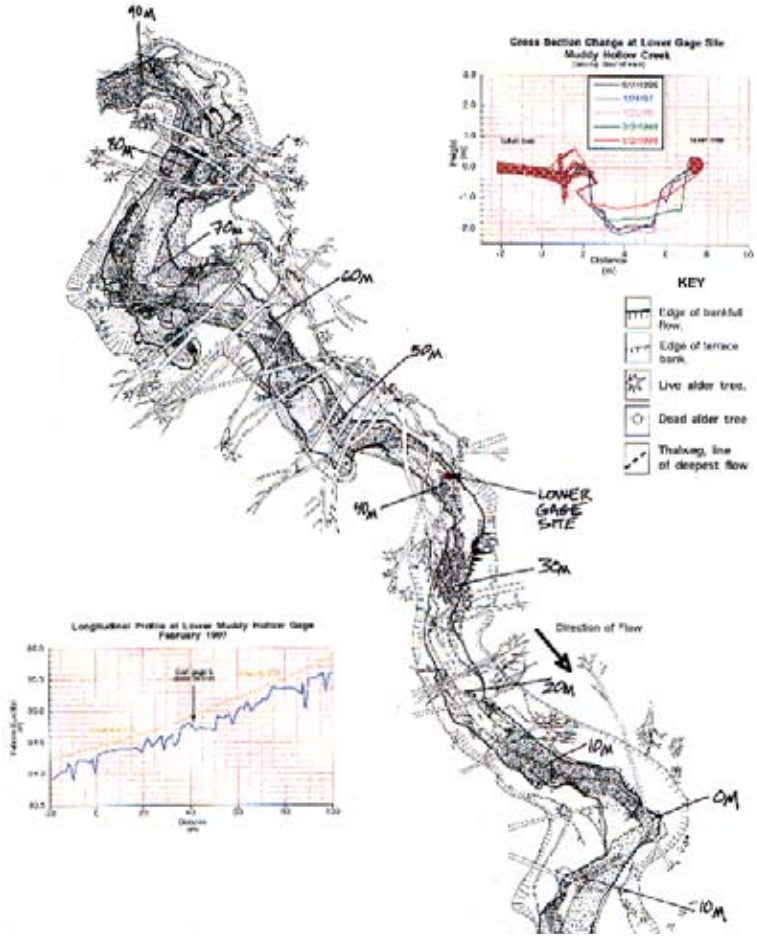


Figure 6
A map of the Lower Gage Site in the Middle Watershed Section is shown for summer conditions of 1996.

Distance stations on the map match the distances shown on the longitudinal profile.

(input mechanisms could therefore not be determined), 6% were supplied by stream flow eroding or undermining the banks, and 5% were supplied by sediment building up on the bed, thereby incorporating LWD by elevating the bed level. Recruitment processes for about 17% of the LWD was not determined.

Figure 6 shows a 1996 map of a 100 m-long (328 ft) reach at the gage location. It depicts the large quantity of LWD that accumulated in the channel bed and across its banks. It is representative of much of the channel throughout the Middle Section. The longitudinal profile in Figure 6 indicates that the stream gradient was about 1.4%. The deepest pool was 0.4 m (1.3 ft) and the other pools were less than 0.3 m (1 ft) deep.

Pre-fire bankfull width was about 2.7 m (9 ft). Based upon a general finding that there should be a pool for every 5-7 bankfull widths (Leopold, 1994), the spacing of pools deeper than 0.3 m at 13.3 bankfull widths fails to meet this criterion. We expect that before the fire, pool spacing may have been greater than the immediate post-fire condition because sediment supply was lower (thus pools would not have been filled). Pre-fire wood spacing was about 6.3 bankfull widths.

The cross section at the upper right corner of Figure 6 shows the changes in width and depth that occurred near the gage site between June 1996 and May 1998. The cross section existing before fire was not expected to be much different from that shown for 1996 because no significant erosion was observed at the site at the time of the survey. The amount of cross sectional change prior to the 1998 February flood flow of the El Nino year was about 0.2 m (0.7 ft) of incision followed by the same amount of deposition. During the February flood, when high flows started to intersect much of the LWD, the banks widened by nearly 1.5 m (5 ft). The bed elevation increased in height by another 0.3 m. At the end of the rainy season the bed averaged about 0.7 m (2.3 ft) higher than its 1996 elevation. These changes were due to wood slowing the water velocity, trapping the sediment, reducing the channel gradient, and creating obstructions to flow that increased local sediment supply.

Hydrologic Conditions in Third- to Fifith-order Channel

During WY 1996, the amount of precipitation at the Lower Gage was about 60% of the amount at the Upper Gage. This corresponds with the expected rainfall gradient from the Pa-

cific Ocean to the Inverness Ridge. Based upon the flow records and bankfull indicators at the site, we expect that bankfull flow at the lower gage might be slightly less than the 2.6 cms (90 cfs) predicted by statistical tendency from Regional Curves (Leopold, 1994). Tentatively, the data suggests 2.0 cms (70 cfs). This might be due to the much lower amount of rainfall in the Middle rather than the Upper Section. It might also be due to recharge of the alluvial valley.

During WY 1997, the Lower Gage may have exceeded bankfull flow only once during January 22 where peak flow was estimated as 2.0 cms (70 cfs). In 1998, it was exceeded at least once on February 9 when discharge was estimated at about 3.3 cms (116 cfs). To determine discharge for WY 1996, we used 29 storms during our period of record to develop a linear regression between discharge at the Upper Gage and the Lower Gage (R=0.91) The maximum discharge during WY 1996 may have been about 4.5 cms (160 cfs) during the first of January.

Lag times from peak rainfall to peak flow, and runoff coefficients were not determined for the Lower Gage because we were not confident that the data logger recorded the highest peaks of all storms. This is because measurements were set to be recorded every 5 minutes at the Lower Gage Site, but some of the flashier storms may have produced floods that peaked between the recording interval.

Base flow at the Lower Gage Site during 1 October 1996 was 0.014 cms (0.51 cfs). During 1 September 1997, it was 0.0102 cms (0.36 cfs), and 1 September 1998 it was 0.0136 cms (0.48 cfs). Unlike the Upper Gage Site, the lowest base flow coincided with lowest mean annual precipitation. The highest baseflow at the Lower Gage occurred during the first year after fire, although the third year after fire had 1.6 times more rainfall. We speculate that baseflow was highest the first year at the Lower Site because there was reduced evapotranspiration by plants during the growing season. Unlike the small percentage of area represented by hydrophobic soils in the bishop pine forest that did not have shallow surface saturation, the larger proportion of the watershed had soils that did saturate during winter. Therefore, soils throughout the Middle section maintained a slightly elevated base flow for the first post-fire year compared to the following years when vegetation was rapidly growing. Recall that the Upper Gage Site had less baseflow during the first year after fire due to hydrophobic soils that also developed a surface crust. The amount of change in base flow from pre-fire conditions is not known since stream gaging had not been previously conducted.

Lower Watershed Section

Changes in the Location of the Apex of the Alluvial Fan

Shortly after the first post-fire rainfall, an alluvial fan developed near the downstream end of the valley, about 960 m (3,150 ft) upstream of the Muddy Hollow dam. The lower channel that flowed within an alder forest that developed after the 1982 flood, filled with sediment. By the end of the rainy season, the apex of an alluvial fan was positioned at about 1,010 m (3,314 ft) upstream of the dam. The channel across the fan became braided and split into several distributaries. It had been a single thread channel prior to the fire. During the 1996 high flows, the full width of the lower valley was submerged. This caused sand to coarse-sized gravel to splay over the charred soils. In some areas the entire width of the valley, which was as much as 150 m (500 ft) wide, was covered with post-fire-related sediment. Most sediment was sand-sized and granitic in origin. Very little silt was observed. If it was present, it probably did not represent much of the total sediment load. The fine suspended load may have been deposited in Muddy Hollow Reservoir.

During the second winter, there were two additional upstream breaches of the mainstem channel upstream of the 1996 fan apex. The first breach caused the fan head to move upstream by 15 m (50 ft). The fan head was now out of the alder forest and into open meadowlands. The second channel breach, which occurred in later January moved the fan head another 16 m (52 ft) upstream. During the entire rainy season, the channel continued to braid and separate into distributaries. In total, the fan apex moved 1,041 m (3416 ft) upstream of the dam, representing complete burial of the previous channel (Figure 7).

During the early 1998 winter, the channel did not breach its banks. In fact, flow and sediment supply was diminished enough that the channel formed into a single thread channel across its entire alluvial fan. As the channel increased its depth, it transported sediment that had been stored in the fan to points downstream of the toe of the fan. In this way, an extension of the fan was built by deposition occurring downstream of the fan instead of upstream. This occurred until the February flooding, which was associated with El Nino conditions. During the flood event, the apex of the 1997 fan moved

upstream by an additional 69 m (227 ft) and the channel began to braid again across the fan. By the middle of the 1998 rainy season the fan head moved another 95 m (312 ft) upstream. By the end of the season it had moved another 168 m (533 ft) upstream. Thus, by the end of spring the fan head was 1,414 m (4,641 ft) upstream of the dam (Figure 7).

During the growing season, even on the fan, vegetation recovery was rapid, even though several feet or more of sediment had been deposited onto the valley floor. A new alder forest began to expand in aerial extent. The open meadow section had lush growth of annuals and perennials including grasses, sedges, bulbs, and herbaceous plants.

Volume of Sediment Deposited in the Fan

Stratigraphic relationships were developed between the sediment exposed in each of the soil pits that were dug across the alluvial fan. In this way, the volume of sediment deposited in the alluvial fan could be estimated. Conservative estimates of first and second year post-fire sediment yield could be made with the qualification that the amount reported does not represent the total annual sediment load, which would include both the bedload and suspended load. The amount of sediment deposited on the alluvial fan represents only a portion of the total post-fire volume. It does not include suspended load carried downstream of the fan, and it does not include the amount stored in the channel bed upstream of the fan.

To facilitate comparisons of sediment yield with other streams, we converted sediment volume into a weight by using a bulk density value of 1.63 for gravel and sand (recommended by William Dietrich, UC Berkeley Department of Geology and Geophysics) to convert cubic yards into tons (Table 3).

The total volume of sediment deposited on the alluvial fan during the first post-fire year was 1,833 cu m (2,398 cu yd). During the second, it was 4,970 cu m (6,500 cu ft), which represents about 73% of the total. For both years the combined total volume was 6,802 cu m (8,897 cu ft), meaning that the minimum post-fire sediment yield over two years was 1,407 tonnes/sq km (5,252 ton/sq mi). This is probably much higher than the average annual load. It is notable that rainfall during the second post-fire year was 0.8 times less than the first year, hydrophobicity was prevalent in the upper watershed during the first year not the second, and yet 2.7 times more sediment was supplied to the fan during the second year. This was because the channel through the middle section started becoming a sediment source as its banks responded to the disturbance caused by falling alder trees that ripped the banks apart and provided large woody debris that created flow obstructions throughout the Middle Section.

Synopsis of Temporal and Spatial Watershed Response

Before the fire, the Upper Section of MHC watershed functioned as a source and storage reach. It had part of its channel network disconnected by alluvial fans and historically it had punctuated sediment supply from mass wasting processes during landslide producing storms. Otherwise, sediment supply may have been fairly gradual from soil creep along the steep hillsides. After the fire it changed to a source and transport reach dominated by surface erosion processes that develop from particularly intense wildfire. Sediment storage was virtually eliminated and drainage density increased. During the first few storms following fire, the channels adjusted to the increased runoff by eroding their beds and banks and increasing the drainage density to accommodate the increased supply of water and sediment. As the drainage density increased the channel adjusted its geometry by eroding more to accommodate more flow, which then generated more sediment. Runoff from the hillsides diminished during the second year. It was caused by the reduction of hydrophobicity and breaking up of the surface crust on the soils by plants and bioturbation. During the third post-fire year the soils were saturated enough to respond again to landslide processes while the importance of surface erosion began to diminish. As recovery proceeds in the headwaters, sediment supply will again be punctuated by landslide producing storms. The importance of the headwaters, as a source reach from surface erosion processes should rapidly declined following the first few years. Potential for increased frequency of landslides may stay higher until a dense root network develops from mature bishop pines.

The Middle Section of MHC watershed has functioned, at least over the last few hundred years, as a transport reach. This changed for a short period during the latter half of the 20th Century following the construction of Bonelli's Lake. While the dam was in place, it stored sediment upstream of the dam and cut-off the sediment supply downstream. This probably initiated a period of incision and subsequent entrenchment of the channel below the dam that subsequently became a local, but short-

term sediment source as the bed incised. After the dam was breached during the 1982 storm, most of the sediment supplied to the system was again transported through it. The channel functioned this way through the first year after fire as indicated by minimal sediment storage in the channel bed, but by the second post-fire winter, it changed from a transport reach to a combined source and response reach. This was primarily due to the largest amount of LWD supplied to the channel, not due to hydrophobicity through the Middle Section, or necessarily from increased peak flows. We expect that the channel through the middle section will continue to function in both these ways for a number of years for several reasons including: 1) water and sediment transport rates will decrease because of reduced gradient caused by the higher base level from deposition of sediments on the downstream fan and from local entrapment of sediment behind LWD; 2) the local banks will generate more sediment as LWD continues to fall into the channel and is remobilized; and 3) during large floods high shear stresses will initiate erosion on the bed and banks because the channel is entrenched and does not develop a functional 'inner" floodplain.

The lower watershed above Muddy Hollow Reservoir has periodically functioned as a response reach ever since the reservoir was constructed. Prior to its construction and to land use activities, we expect that the entire channel and tidal slough system functioned to transport a fairly low sediment load out of the watershed and into the estuary. After cattle grazing affected the watershed, we expect that an alder forest began to invade the meadow of the lower valley floor because of increased sediment deposited on the floodplain and from draining of the valley by ditch construction. After construction of Muddy Hollow Reservoir, the base level of MHC was elevated. This caused deposition at the upstream portion of the reservoir and the valley upstream of its open water. A newer alder forest began to invade the valley floor. For a short time following the breach of Bonelli's Lake during the 1982 flood, the valley upstream of the Muddy Hollow Reservoir developed a new alluvial fan that supported yet another new alder forest. Following the fire, the lower reach functioned as a response reach by building another alluvial fan upstream and on top of the 1882 fan. For a short time during the early 1998 winter, when both sediment and water supply were relatively low, the channel on the fan also became a source reach. This was because sediment was supplied during the reformation of a single thread channel that incised into the fan and redistributed its bed sediments downstream of the toe of the post-fire fan. When flood flows occurred during the middle of the 1998 winter, the lower reach again functioned as a response reach. It began to develop distributaries and braided channels. In the future, as sediment supply shuts down, the channel in the Lower Reach may develop enough sinuosity to primarily function as a transport reach except during extreme storm events.

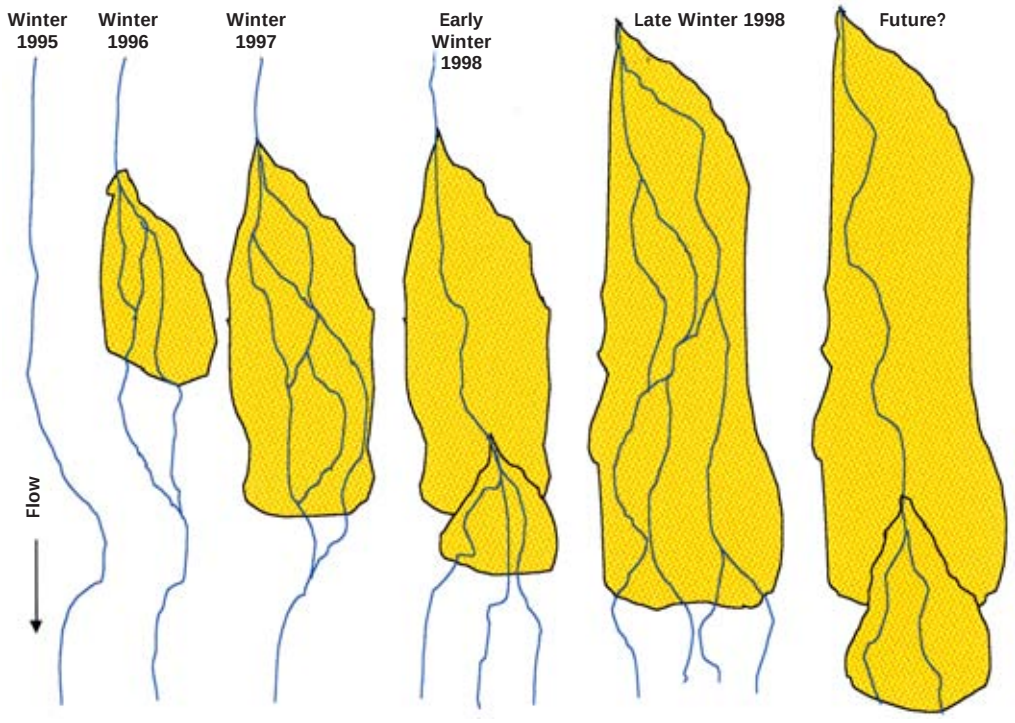


Figure 7
Conceptual sketch of the sequential growth of the post-fire alluvial fan upstream of Muddy Hollow Reservoir.

When the sediment supply is high, the alluvial fan grows as the channel braids and separates into distributaries. When the sediment supply from upstream fluvial transport decreases, a single thread channel incises and reworks sediment in the fan and deposits a new fan downstream.

The spatial and temporal effects of fire on baseflow appear to be dependent upon position in watershed and distribution of hydrophobic soils. If extensive hydrophobic conditions exist, baseflow may be diminished due to lack of infiltration. If these conditions do not exist and the watershed was previously densely vegetated, then baseflow may be increased due to the lack of evapotranspiration effects of vegetation.

Initial Hypotheses

A number of preliminary hypotheses were developed from this study. They are enumerated below.

1. Landscapes in coastal Northern California with similar geology to landscapes in coastal Southern California that both develop post fire hydrophobic conditions, have different geomorphic responses to erosion processes. These differences may be due to greater root density and their influence on apparent soil cohesion of plant assemblages that are adapted to Mediterranean-type climates compared to arid climates. As a result, post-fire sediment ravelling and bulking of flows was not an important process in Muddy Hollow watershed.
2. Large woody debris supplied from fire damaged riparian corridors can plays a critical role in altering transport reaches into source and response reaches for years after the event of fire.
3. Spatial and temporal effects of wildfire on peak and base flows depend upon position and distribution of hydrophobic soils in a watershed.
4. Amount of post-fire sediment supplied to a channel also depends upon position and distribution of hydrophobic soils, and upon the supply of LWD to the channel system.
5. Alder forests on the valley floor are a modern feature caused by increased sediment supply onto the valley floor. This hypothesis was developed by comparing historical circa 1860 maps to aerial photos dating back to 1942. The historical maps show minimal riparian vegetation on the lower valley floor. We suggest that as grazing and agricultural practices increased, sediment supply also increased, effectively elevating the valley floor enough to reduce seasonal saturation in meadowlands upstream of the tidal marsh. The water table began to fluctuate enough that alders could be supported in an environment that was not saturated for extended time and that was not stagnant.
6. In areas where surface crusts develop on hydrophobic soils, the first post-fire year base flow may be decreased below natural background rates because of the lack of saturation. In areas not dominated by hydrophobic soils base flow may be increased above natural background rates because of the loss of evapotranspiration.
7. Mycorrhizae in association with certain vegetation species may play an important role in development of hydrophobic soils.

Recommendations for Continued Monitoring by PRNS

The best way to determine change from catastrophic events is to have pre-existing data. The value of long-term background data far exceeds the value of short-term monitoring, such as this study that cannot definitively establish magnitude of response compared to background rates, and length of time required for recovery. Establishing baseline conditions to determine natural variability of a number of different but representative channels in the National Seashore is strongly recommended. Within a single channel system, measurements of hydraulic geometry at different positions along the watershed could be made to establish an appropriate picture of the spatial and temporal variability of the watershed. Their land use history and/or vegetation type could stratify watersheds. Monitoring could be accomplished by a variety of different efforts and associated costs. At a minimum, establish some permanent cross sections that can be reoccupied, even after the event of fire, by having their coordinates fixed by Global Positioning Stations. Fireproofing of permanent benchmarks that fix

Table 3. Sediment Volumes in Alluvial Fan

	Volume cu/m (cu yd)	Min. Yield cu m/sq km (cu yd/sq mi)	Min. Yield tonnes/sq km (tons/sq mi)
1996 First Winter after Fire	1,833 (2,398)	256 (869)	379 (1,416)
1997 Second Winter After Fire	4,970 (6,500)	695 (2,355)	1,027 (3,836)
Total for two years	6,802 (8,897)	951 (3,224)	1,407 (5,251)
Average per year	3,401 (4,449)	476 (1,612)	704 (2,626)

elevation and location for existing cross sections and profiles is also recommended. Cross sections and profiles could be surveyed once a year for a decade for example, and then less frequently based upon analysis of the data. The length of the long profiles should be 20 times the bankfull width. With this minimal amount of background data, the magnitude and type of change following a significant event such us fire or major flood could be better understood and post fire remediation effectively planned. It is important for land managers to understand the resilience of a system to recover, but also to recognize when thresholds have been crossed that initiate instability.

If a greater level of background information is desired, continuously recording stream and rainfall gages can be established on a number of channels to characterize storm and base flow. Still wells are a less expensive alternative that can also be used to establish peak flood heights.

The map of Lower Muddy Hollow Site can be used in the future to make comparisons of bankfull width, planform, distribution of woody debris, and particle size. The longitudinal profile could also be resurveyed, but patience will be needed to wait for the time when the channel is not impenetrable.

If stream gauging in MHC channel is desired, a permanent weir is highly recommended. Without a stable cross section, efforts to try to develop a rating curve for discharge and gage height will be thwarted by unstable conditions of the bed or banks.

To best determine impacts of the fire and of recent land use effects on sedimentation rates, we recommend that a program of coring and dating sediments deposited in Muddy Hollow Reservoir should be conducted. If this effort is done in concert with coring of the remaining tidal marsh at the foot of the Muddy Hollow Reservoir and deep coring of sediments in the upland valley, it could be possible to develop both historical sedimentation rates and fire frequency prior to non-native settlement.

Literature Cited

Anderson H.W., G. B. Coleman and P.J. Zinke, 1959. Summer slides and winter scour, dry-wet season in Southern California mountains: US Department of Agriculture, Forest Service, Pacific Southwest Forest and range Experiment Station, Berkeley, California, General technical report PSW-18, 12 p.

Booker, F.A., 1998. Landscape and management response to wildfires in California. Masters of Science thesis, Geology Department, University of California, Berkeley, 436 p.

Booker, F.A., W.E. Dietrich, and L.M. Collins, 1993. Runoff and Erosion after the Oakland Firestorm, expectations and observations, in California Geology, 1992, v.46, p. 157-173.

Clark, J.C., and E.E, Brabb, 1997. Geology of Point Reyes National Seashore and Vicinity, California: A digital database, an online US Geological Survey open-file resource, ftp: //wrgis.wr.usgs.gov/pub/open-file/of97-456/pr-geo.txt.

Collins, L.M., and Johnston, C.E., 1995. Effectiveness of straw bale dams for erosion control in the Oakland Hills following the fire of 1991, in Brushfires in California: Ecology and Resource Management, university of California, Irvine, International Association of Wildland Fire, p. 171-183.

Collins, L.M., T. Gaman, R. Moritz, and C. L. Rice, 1996. After the Vision Fire:

Restoration, Safety and Stewardship for the Inverness Ridge Communities, published by Environmental Action Committee of West Marin, 84 pp.

Debano, L.F., 1981. Water repellant soils, a state-of-the-art: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and range Experiment Station, Berkeley, California, General Technical Report PSW-46, 21 p.

Evans, J.G., 1988. The Natural History of the Point Reyes Peninsula, published by Point Reyes National Seashore Association, 226 p.

Galloway, A.J., 1977. Geology of the Point Reyes Peninsula, Marin County, California, Bulletin 202, California Division of Mines and Geology, 71 p.

Krammes, J.S., 1965. Seasonal debris movement from steep mountainside slopes in Southern California, in Proceedings of the Federal Inter-Agency Sedimentation conference, Jackson, Mississippi, 1963: US Dept. Agriculture Miscellaneous Publication, v. 970, p 85-88.

Leopold, L.B., 1994. View of a River, Harvard University Press, Cambridge, MA, 298 pp.

Montgomery, D.R., and J.M. Buffington, 1997. Channel-reach morphology in mountain drainage basins in Geological Society of America Bulletin, V. 109, p 596-611.

Onda, Y., W.E. Dietrich, and F. Booker, 1996. The overland flow generation mechanism in Mt. Vision fire area, Northern California, in Abstracts for the American Geophysical Union Fall Meeting December 15, pg. 209.

Rice, R.M., 1974. The hydrology of chaparral watersheds, in Rosenthal, M., ed., Symposium on living with the chaparral, proceedings: San Francisco, Sierra club, p.27-34.

Rice, R.M., 1982. Sedimentation in the chaparral: how do you handle the unusual events? in F.J. Swanson, R.J. Janda, T. Dunne, and D. N. Swanston, editors, Sediment budgets and routing in natural systems: US Dept. Agriculture, Pacific Northwest Forest and range Experiment Station, General Technical report PNW-141, p 39-49.

Wells, W.G., 1981. Some effects of brushfire on erosion processes in coastal Southern California, in Erosion and sediment transport in Pacific steeplands: International Association of Hydrological Sciences Publication no. 132, Christchurch, New Zealand, p. 305-343.

Wells, W.G., 1986. The Influence of fire on erosion rates in California chaparral, in Johannes DeVries, editor, Proceedings, Chaparral Ecosystems Conference, Santa Barbara, California, May 16-17, 1985: Water Resources Center Report 62, Davis, California, p 57-62.

Wells, W.G., 1987. The effects of fire on the generation of debris flows in Southern California, in J.E. Costa, and G.FD. Wiezorek, editors, Debris flows/avalanches: processes, recognition, and mitigation: Reviews in Engineering Geology, V. 7, p. 104-114.

Assessment of Northern Spotted Owls After The Vision Wildfire Point Reyes National Seashore

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Introduction

In October 1995, a wildfire burned over 12,000 acres in the Point Reyes National Seashore and adjacent lands located in Marin County, California. The Vision Fire encompassed 20% of the Seashore, and burned intensely in many areas, including portions of Bishop pine and douglas-fir forests with known or potential northern spotted owl (*Strix occidentalis caurina*) habitat. Fire suppression activities were in place throughout the area to control the burn, and combined with impacts from a severe fire, may have had adverse impacts on northern spotted owls in the area. As a federally threatened species (USDI 1990), the northern spotted owl is a species of special management concern in the park (50 CFR 402.1).

Northern spotted owls occupy relatively large home ranges and defend their territories from other northern spotted owls. Throughout much of their range, they remain on their territories year-round. Since 1988, limited surveys of northern spotted owls have been conducted within the Seashore and have documented their occurrence within what is now the burn perimeter. Daytime roosts, or nests, are at the center of activity for a territory and are used to estimate the location of each territory (Thomas et al. 1990). In Marin County, a territory is at least 0.5 mile diameter in size, based on the nearest distance between some roosts (Chow 1995).

We initiated this study following the Vision Fire to 1) determine how northern spotted owls may have been affected by the fire or fire-fighting efforts, 2) prescribe mitigation for negative, or potentially negative effects, to northern spotted owls, and 3) assess effects of mitigation in the burn area related to bulldozer-line placement, and the selection and removal of hazardous trees (USDI 1996). To assess the potential effects of the fire on northern spotted owl, we studied the post-fire occupancy of known sites, their productivity, habitat associations and diet.

Methods

The study area is comprised of both burned and unburned portions of the park with known or suspected northern spotted owl territories (Figure 1). The study area included 1) forested areas within the Vision wildfire of the Seashore, 2) a 0.5 mile area buffer around the fire perimeter, and 3) two areas of similar habitat within the park boundary but outside the burned area with known northern spotted owl sites (Tomales Bay State Park and Firtop peak areas). The study area encompassed approximately 5,500 acres (3,400 "burn" acres and 2,100 of non-burn acres) of forest dominated by either Bishop pine-hardwood (Bishop pine) or douglas fir forests (Figure 1).

Surveys were conducted immediately following the fire and during the spring and summer 1996, following the methods of Forsman (1983) and Franklin et al. (1990). Surveys were conducted to determine northern spotted owl occupancy and reproductive status. Within one

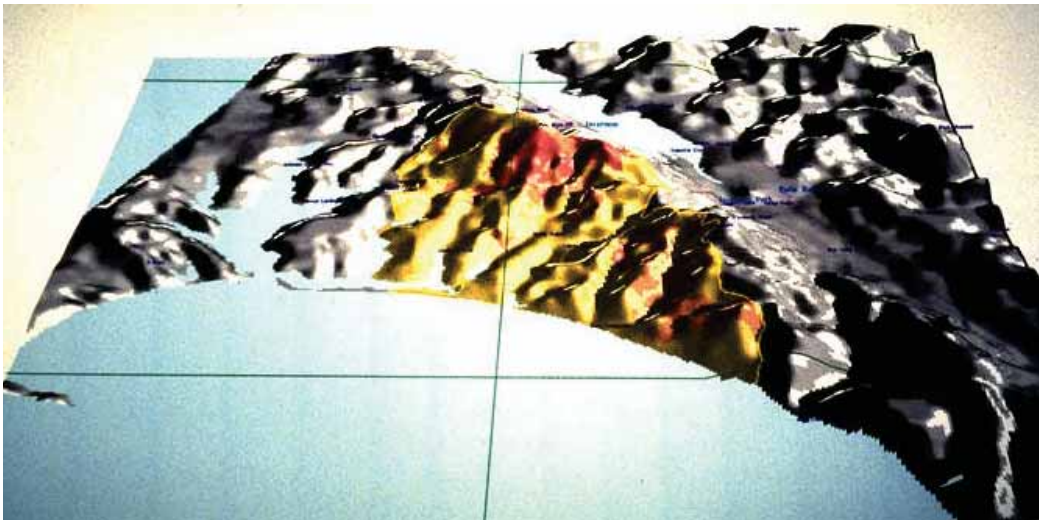

Figure 1
Extent of the 1995 Vision Fire.

Northern spotted owl (© Susan Van Der Wal)

month of the fire (October - November 1995), several informal surveys were conducted in some areas. All other data was collected during the breeding season (1 March - 31 August). Information on other current Marin County northern spotted owl sites was made available with the cooperation Golden Gate National Recreation Area and under contract with the Point Reyes Bird Observatory, Stinson Beach, California.

Initial post-fire calling surveys were conducted during the night and day in a limited area of the burn. In the spring of 1996, night surveys were conducted to detect owls, with at least three night surveys in areas of non-response to confirm non-occupancy. In areas where a response was detected, day visits were conducted to determine residency and reproductive status. Occupied sites were categorized as roost or nest sites. "Roosts sites" were daytime locations of a pair or at least one adult northern spotted owl, and "nest sites" were locations of nests. Evidence of fire and fire suppression activities within northern spotted owl territories were also noted.

Habitat data were obtained for occupied sites found in and outside of the burned area. The sites were mostly in Bishop pine and Douglas-fir forest. Three northern spotted owl sites outside of the study area were also included because of the small sample size of the study area: two nest sites in redwood habitat in south Marin and a roost site in a hardwood-dominated forest were included in the data analysis. To characterize northern spotted owl habitat, 26 variables were measured at each site or plot (Appendix A). For nest sites, an additional ten habitat variables were measured. We selected ten random sites were selected in Bishop pine and douglas fir habitat to determine if owls occurred in particular forested areas or were randomly distributed in the Seashore. Plots were centered on a roost or nest tree for all occupied sites, but trees were not necessarily plot centers at random sites.

Components of the forest habitat were measured following the methods of recent northern spotted owl habitat studies (Seamans 1994, LaHaye 1988, Call 1990). Plots were measured between 15 July and 1 August. The variable-radius plot method was used (Mueller-Dombois and Ellenberg 1974) with a 20 basal factor prism (Bell and Dilworth, 1988) to determine which trees were within each plot. To sample the understory types and estimate their percent cover, the line-intercept method was used (Mueller-Dombois and Ellenberg 1974) on a 22.9m line oriented in a random direction from the plot center. These data were collected to characterize and compare the forest structure of occupied sites (roost and nest sites) to the available Bishop pine and douglas fir forests. To determine if there were habitat variables significantly correlated with occupied sites, the data was analyzed using the Pearson correlation matrix, and a stepwise logistic regression with the (Wilkins, et al. 1992).

Egested northern spotted owl pellets were collected opportunistically in the 1996 breeding season from all occupied sites in the study area to obtain information on northern spotted owl diets among sites (n = 12) and to investigate important prey species and diet variations associated with habitat. Wherever possible, all pellets were cleared from sites. Throughout Marin County pellets were also collected by other land managers in cooperation with this project. A representative sample of pellets was analyzed for this report. Pellet age was estimated as "old," "this season," or "fresh," based on the condition of the pellet at the time of collection. Most prey items were identified to genus or species using mammal keys (Ingles, 1965, Jameson and Peeters 1988), but some birds and mammal bones could not be identified without comparison with museum specimens. Diet composition was estimated by frequency and percent biomass of prey items, based on mean weights of prey (estimated for unknown genera) (Jameson and Peeters 1988).

To obtain some indication of prey availability, sooted track plates were set in Bishop pine and douglas fir forest areas both inside and outside of the burn. Nine plots were established with two baited track plates each and run for 6-20 days (Barrett, 1983) between 5 June and 10 July. Plates placed within 200m of the burn perimeter were considered "burn" sites, and all others were considered "unburned" sites. All mammal tracks were identified using a track plate key (Taylor and Raphael 1988). The information collected is only considered anecdotal to the overall results since the sample (n = 9) was too small to give complete coverage of the study area.

Results

Occupancy

During the immediate post-fire surveys in November, only one female was heard in the evening just west of the Inverness Ridge area in a severely burned patch of Laguna Canyon, and during the day in Haggarty Gulch (outside burn; only positive results were considered

Adult owl with chick (© Susan Van Der Wal)

because they were conducted outside of the breeding season and owls are less likely to respond to calls). It is believed these were both responses from the female of Mt. Wittenberg (see Chow 1995). During the 1996 breeding season, a total of 14 sites were found occupied by northern spotted owls within the survey area (Table 1, Figure 1).

During the 1996 breeding season, all sites occupied in recent years were occupied within one mile of the burned area (seven) and one new pair was discovered (Meadow Trail). Of these eight sites closest to the burn, pair status and their roost location were confirmed at seven sites and a single male (unknown social status and roost) at one site. The male of unknown social status was from the Baldy Trail site where a pair has been heard in the area in the past (S. Bunnell, Cal. Academy of Sciences, pers. comm.). The precise day roost location was not found for the Baldy Trail site, which could have revealed whether the site was occupied by a pair.

Northern spotted owls with activity centers within the burn perimeter, Inverness, Mt. Wittenberg, and Drakes View were of special concern. Impacts on northern spotted owls from the fire and fire-fighting activities are not yet known but are evident near each of the three sites. Of these sites, only the activity center of Inverness site had been previously located. In 1996, the Inverness male was at the same roost location and the female was at a roost an estimated 200m away. The nest used in 1995 and the roosts were within approximately 400m of the burn. Thus firelines created by bulldozers along the burn perimeter were in their center of activity. In mid-August an adult (reported) northern spotted owl carcass was recovered from a highway less than 1 mile from both the Inverness and the Drakes View activity centers (J. Evens, Avocet Res., pers. comm.). Follow-up surveys have not yet been made for these sites, but it is possible the individual was from either of these sites.

Pre-fire and the initial post-fire survey detections of the Mt. Wittenberg pair indicated this

Table 1. Summary of spotted owl occupied sites

Site	Location	Owner	Occup	Reprod	#Young	Forest-type	Site-type
MR001	Tomales Bay -1	TBSP	Single	no		pine-oak	non-burn
MR002	Tomales Bay - 2	PVT	Pair	no		pine-oak	non-burn
MR003	Inverness	PVT	Pair	no		oak-bay-pine	burn
MR004	Bear Valley	PORE	Pair	no		fir-oak-bay	burn
MR005	Stewart Trail	PORE	Pair	active	1	fir-oak-bay	non-burn
MR018	Mt. Wittenberg	MIXED	Pair	no		fir-oak	burn
MR021	Drakes View Dr.	MIXED	Pair	no		fir-oak-bay	burn
MR022	Firtop	PORE	Pair	no		fir-oak-bay	non-burn
MR027	Pebble Beach	TBSP	Pair	no		oak-pine	non-burn
MR028	Invern Valley	PVT	male	unknown		pine	non-burn
MR029	Bear Valley Trail	PORE	Single	no		fir-oak-bay	burn
MR030	Baldy Trail	PORE	Single	unknown		fir-bay-oak	burn
—	Horse Trail	PORE	Pair	active	0	fir-oak-bay	burn
—	Tomales Bay 3	TBSP	Pair	no		oak-bay-pine	non-burn
—	Meadow Trail	PORE	Pair	no		oak-bay	burn
LC	Laguna Canyon	PORE	none	unknown		fir	burn
PP1	Predicted pr-Santa Maria *mMaria	PORE	none	unknown		fir	burn
PP2	Predicted pr-Bucklin	PORE	none	unknown		pine	burn

pair might have been using a portion of what burned forest habitat and may have shifted to a new roost site after the fire. During the 1996 breeding season, daytime locations were outside of the burned area, but within approximately 400 m of the burn perimeter on Inverness Ridge. Based on their daytime locations during the 1996 season, the portions of Sky Trail which were widened to create firelanes were within the territory of this pair. Habitat use patterns and activity center for the Mt. Wittenberg pair remains unclear, however, because evidence of regular occupancy was lacking at locations visited.

The Drakes View pair likely shifted to a new roost site after the fire, because the pair was roosting an estimated 30m from the burn perimeter, between two developed residential roads, and in a stand with more younger, and shorter subcanopy trees than other Douglas-fir roost sites. This pair was detected in several previous years higher in the drainage during nighttime surveys, in areas which were intensely burned (S. Bunnell, pers. comm.). The 1996 roost was located near residences, and efforts to save residences may have coincidentally preserved patches of northern spotted owl habitat.

In the non-burn portions of the study area, a total of six pairs were located. Daytime roosts had been located previously for five of the six pairs, and one new pair was located (at Shell Beach). One site was abandoned mid-season when the female disappeared and the male was found injured soon after and was eventually euthanized. No northern spotted owls were detected west of Inverness Ridge in 1996, both in the burned and unburned areas (Figure 1). A pair of northern spotted owls had been detected several times on the ridge in the area of the Fire Lane and Sky Trail junction, but was most likely the Mt. Wittenberg pair from Haggarty Gulch (east of the ridge). Outside the study area, an additional 21 sites were found occupied (twelve pairs and nine single owls).

Reproduction

Two pairs attempted breeding within the study area in 1996, Horse Trail and Stewart Trail. The Horse Trail pair was within 0.5 miles of the burn, and Stewart Trail pair was located outside the burn. Through nesting status surveys conducted in mid-June it was determined that the Horse Trail pair had failed to fledge young. We suspected that the young were predated upon because plucked fledgling feathers were found at the site. Another possible impact to the pair could have been noise disturbance caused by park staff when a tree was cut (with a chainsaw) within approximately 100m of the nest sometime between 15 May and 20 June. There were no other hazardous tree removals within 0.25 miles of an active nest in 1995 or 1996. The other pair which bred, MR005, had one fledgling.

The three sites closest to the burn did not breed in 1996 (Inverness, Mt. Wittenberg and Drakes View). Elsewhere in Marin County four pairs bred out of ten sites surveyed for reproductive status. These breeding pairs were all located in redwood forest habitat. Of these, three sites fledged one chick each, and one pair failed in their nesting attempt.

Habitat

Habitat data from fifteen occupied sites were analyzed and compared to ten randomly chosen plots within Bishop pine and Douglas-fir forests. In a preliminary analysis, thirteen of

Table 2. Habitat variables of spotted owl (n=15) and random (n=10) sites.

	Occupied Mean	Occupied % C.V.	Random Mean	Random % C.V.
Elevation (m)	101.4	47.7	155.8	60.6
Aspect (degrees)	169.3	30.8	125.3	93.2
Structural				
canopy (%)	80.3	17.5	57.1	36.7
strata (1-5)	3.4	36.7	3.3	20.5
shrub height(m)	1.8	93.2	2.2	96.9
Basal Area (sq m/ha)				
total	72.1	35.4	50.1	29.5
conifer	35.8	84.3	17.0	108.1
mature	18.8	81.9	13.3	88.2
large	16.7	131.6	6.9	122.7
young	9.3	117.7	7.4	98.6
Understory (%)				
shrub	16.1	144.6	26.6	110.3
small trees	7.6	134.4	7.5	270.1
duff	48.4	52.7	11.8	

the 26 variables collected were selected for this assessment (Table 2). The Pearson correlation matrix was run on nine variables and showed that the number of forest strata layers was the best predictor of occupied habitats ($p = .06$). In relation to other variables measured, strata tended to be correlated with aspect ($p = .07$) and significantly correlated with basal area of large trees ($p < .05$). Strata was negatively correlated with the elevation ($p = -.05$) and slightly negatively correlated with the basal area of conifers ($p = -.09$). In the stepwise logistic regression model, the amount of duff cover best defined occupied sites ($X^2 = 0.03$, *McFaddens's Rho-squared* = 0.4786, $p < .05$). When duff cover was eliminated from the regression, the variables which best defined northern spotted owl habitat were the percent of canopy cover ($X^2 = 0.072$, *McFadden's Rho-squared* = 0.4198, $p < .05$) and direction of slope aspect ($X^2 = 0.022$, *McFadden's Rho-squared* = 0.4198, $p = .07$). For occupied sites (both nesting and roosting), the averages for duff cover was 48.4%, for canopy cover 80.5%, and for aspect 169.3°.

Prey

Approximately 65 pellets from 12 sites were analyzed for diet. Both prey diversity and proportion of the total biomass for different prey types were compared between northern spotted owl sites in Douglas-fir and Bishop pine forest habitats (Table 3). A subsample of the pellets collected that were known or estimated to be egested in 1996 were targeted for analysis; however, when older pellets were included for all seasons, the results of analyses added a few more species (Table 4).

Dusky-footed woodrats (*Neotoma fuscipes*) were the most common prey type found at all sites. A variety of species was found in the pellets but species composition differed between the two forest types. At Bishop pine sites, prey types also included a jay-sized bird, and a shrew (*Sorex* sp). At douglas fir sites, other prey types included two mammal species similar in size to the woodrat (mammal "A" and "B"), and a California vole (*Microtus californicus*). In analysis of all-season pellets, other species in Bishop pine pellets included a microtine rodent, and at Douglas-fir sites, a small bird and a cricetine rodent.

Based on the biomass measured in weight of prey consumed, the most important prey was the dusky-footed woodrat for all sites, comprising at least 60% of the diet at douglas fir sites, and over 90% at Bishop pine sites (both spring-only and all-season analyses) (Figure 2,3). No skulls or other positive identifiable pieces were found of mammal "A" and "B," but given the large size range of dusky-footed woodrats (184-358g) and structural similarities to their bones, it is possible these two prey types were also woodrats, yielding over 90% prey biomass of woodrats at douglas fir sites as well.

Track plates were also used to identify potential prey near occupied sites. Tracks were detected on the sooted plates at all but one burned site. The non-burned plots ($n = 4$) generally had more species detected with average of 2.3 species compared to burned sites ($n = 5$) (average = 1.6 species). The most common species identified on the track plates in both site types was the gray fox (*Urocyon cinereoargenteus*) (six sites), followed by raccoon (*Procyon lotor*) (four sites). Equal samples were found in burn and non-burn sites for dusky-footed woodrat tracks (one each) and small mammals (two each).

Discussion

The results from 1996 surveys should be considered preliminary. The methods used to characterize habitat, analyze diet, and detect prey availability were exploratory, and will need to be refined in future years. Nevertheless, the methods used were useful in providing the first

Table 3. Spotted owl pellet analyses collected in spring 1996.

	mamm. A	Neotoma fusc.	mamm. B	Microtus calif.	bird	Peromyscus sp.	Mus sp.	Sorex sp.	Total Pellets
Mean wt. (g)*	300	271	270	53.5	50	24.5	18	10.5	
Individuals fr.									
Bishop pine:	0	10	0	0	1	7	0	1	12
Composition:	0%	92%	0%	0%	2%	6%	0%	0%	
Individuals fr.									
Douglas fir:	3	13	3	4	0	5	1	0	20
Composition:	16%	63%	14%	4%	0%	2%	0%	0%	

*Mean weight estimated for unknown genera

insights into needs of northern spotted owls in the Seashore and the potential effects of fire. Continued research will help to assess habitat preferences, determine how fire and fire-suppression activities affected northern spotted owl distribution and reproduction, and detect long-term effects on the northern spotted owls.

Occupancy and Reproduction

Historic occupancy information is primarily based on surveys conducted between 1988-1991 (S. Bunnell, pers. comm.). Prior to the 1995 fire, the National Park Service had conducted limited surveys in the study area (Chow 1995). In the first year- assessment (1996), all known sites from previous surveys were occupied during the breeding season. Thus occupancy rate of known sites did not seem to be significantly affected by the fire and fire suppression activities. No owls were detected coming from the west side of the Inverness Ridge, an area that had not had any known sites and had never been completely surveyed in the past. All known or suspected northern spotted owls pairs were pair-occupied except for one site, (Mt. Baldy), where a female was never seen or heard in 1996. Of the sites nearest to the fire, the roost habitat information from the Drakes View pair differs from other douglas fir roost sites measured and may be an indication that they shifted their roost site. The forest structure at their 1996 main roost had a larger component of younger trees compared to similar unburned roosts of other pairs.

Although site occupancy did not appear significantly affected by the fire, only two pairs in the study area attempted nesting, and both nests were at least 0.5 mi. from the burn perimeter. The number of sites (n = 13), though, was too few to draw conclusions. Some possible causes for low reproduction are 1) the fire and fire suppression activities were physiologically stressful to the owls; 2) the prey base was reduced, 3) the population is at carrying capacity, 4) the Seashore does not have good nesting habitat, or 5) severe winter storms early in the season (Franklin et al. 2000, Forsman et al. 2002). Trends in reproductive rates of northern spotted owls in Marin County is not known because past reproductive surveys have been limited.

Within the non-burned habitats surveyed countywide in 1996, four of six pairs in redwood forest type attempted breeding, while only two of eight pairs in Douglas-fir forest type bred. One pair in each of the two habitat types had nesting failure. One pair was suspected to have abandoned nesting after a late storm (a redwood site), and plucked fledgling feathers at another site indicated young were predated (a douglas fir site).

Habitat

Northern spotted owl roosts and nest sites tended to have higher amounts of duff cover and canopy cover, on slopes facing south by south east. These habitat variables appeared to be selected by northern spotted owls over what was generally available; however, a larger sample size is necessary to support these results. If additional data supports these findings, we may infer that the fire may not have affected the preferred northern spotted owl habitat because most of the burn was on the west-facing slope of Inverness Ridge.

Habitat studies of northern spotted owls in other areas (of all three subspecies; northern (*S.o. caurina*) , California (*S.o. occidentalis*), New Mexican (*S.o. lucida*) also found these variables to be good predictors of owl habitat (Seamans 1994, Call 1990, LaHaye 1988). Other researchers also determined that variables such as higher basal area of total trees, of conifers, of large trees, of mature trees and snags were good predictors; however, this study did not find similar correlations. Differences between habitat analyses may be due to different habitat, collection methods, or sample size.

Table 4. Spotted owl pellet analyses collected from all seasons.

	mamm. A	Neotoma fusc.	mamm. B	Microtine	Microtus calif.	bird	Cricetine	Peromyscus	Mus sp.	bird sp.	Sorex sp.	Total Pellets
Mean wt. (g)*	300	271	270	65	53.5	50	30	24.5	18	15	10.5	
Individuals fr.												
Bishop pine:	1	26	0	1	0	1	0	14	0	0	1	30
Composition:	4%	90%	0%	1%	0%	1%	0%	5%	0%	0%	0%	
Individuals fr.												
Douglas fir:	4	21	6	0	5	0	1	8	1	1	0	35
Composition:	13%	63%	18%	0%	3%	0%	0%	2%	0%	0%	0%	

*Mean weight estimated for unknown genera

Owl on Inverness Ridge
(© Susan Van Der Wal)

Prey

Pellet analysis results suggest that the northern spotted owl diet (by weight and frequency) in Douglas-fir and Bishop pine forests in the Seashore consists primarily of woodrats. A variety of other species are also eaten, but most of the biomass is composed of woodrats. Other studies of northern spotted owl diet yielded similar results, with northern spotted owls relying on one or two species of small mammals for the majority of their diet (Carey et al.1992, Laymon 1988, Thrailkill and Bias 1989). Two species of particular importance to northern spotted owls are the northern flying squirrel (*Glaucomys sabrinus*) and the woodrat (*Neotoma* sp.) (Zabel et al. 1995, Smith et al. 1999). Dusky-footed woodrats are fairly abundant in this region (Willy 1992), and may be supplying northern spotted owls with ample prey. Interestingly, pellets collected outside the study area from sites in redwood and hardwood forest types in Marin had a predominance of California pocket gophers (*Thomomys bottae*) and California meadow voles (*Microtus californicus*).

The track plate method used was insufficient to determine prey availability for northern spotted owls. The group of most interest were small mammals from which the availability of potential prey could be compared between sites; however, the results of the initial plots indicated this method was not detecting sufficient samples of the target group. Attracting non-target species may have also disrupted the experiment since species such as gray fox or raccoon usually consumed all the bait. Additionally, interpretation of the tracks was compromised by fog drip distorting the track impression.

Recommendations

The immediate effects of the fire and fire suppression activities are not clear, and more monitoring is required to gain a better understanding. For a relatively long-lived and reclusive species such as the northern spotted owl, it is inappropriate to draw conclusions from a single breeding season. Nevertheless, these data yield some insights into northern spotted owl habits that can guide future research.

Areas of current and future research that will guide management in understanding the potential effects of fire and fire suppression actions on northern spotted owl include 1) broaden the study area and increase sample size in order to strengthen power of analyses of habitat associations, 2) study primary prey of northern spotted owl, dusky-footed wood rats, to determine their distribution and abundance pre and post hazard fuel reduction, 3) study variability of northern spotted owl diet by season, by year and by habitat, and 4) continue long-term monitoring of known sites to enhance understanding of factors affecting reproduction, productivity, juvenile dispersal, and survivorship.

Finally, continue adaptive management by the Seashore for the protection of northern spotted owls in order to mitigate effects of the fire and fire suppression activities. Adaptive management includes coordination of park activities with park biologists to insure northern spotted owl sites are not disturbed during the breeding season by maintenance activities such as the removal of hazardous trees, trail maintenance, hazard fuel reduction, or exotic plants. Additional mitigation measures and management strategies may be needed if monitoring studies determine that annual recruitment rate is lower than annual mortality.

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Literature Cited

Barrett, R. H. 1983. Smoked aluminum track plots for determining furbearer distribution and relative abundance. Calif. Fish and Game. 69(3): 188-190.

Bell, J. F., and J.R. Dilworth. 1988. Log Scaling and Timber Cruising. Oregon State University, Corvallis. 396pp.

Call, D. 1990. Home range and habitat use by California spotted owls in the central Sierra Nevada. M.S. Thesis. Humboldt State Univ. Arcata, California 83pp.

Carey, A.B., S.P. Horton, et al. 1992. Northern

Juvenile northern spotted owl (© Susan Van Der Wal)

spotted owls: influence of prey base and landscape character. Ecol. Monog. 62(2):223-250.

Chow, N. 1995. Spotted owl annual report, 1994-95. Golden Gate National Recreation Area, San Francisco, California. unpubl. 10pp.

Forsman, E.D. 1983. Methods and materials for locating and studying spotted owls. U.S. Forest Service Gen. Tech. Rep. GTR-PNW-162. 8pp.

Foresman, E.D., S. DeStefano, M. Rafael, and R.J. Gutierrez (eds.). 1996. Demography of the northern spotted owl. Studies in Avian Biology. Cooper Ornithological Society. Vol. 17. 122 pp.

Franklin, A.B., J.P. Ward, R.J. Gutierrez, and G.I. Gould. 1990. Density of northern spotted owls in northwest California. J. Wild. Manag. 54:1-10.

Franklin, A.B., D.R. Anderson, R.J. Gutierrez, and K. Burnham. 2000. Climate, habitat quality and fitness in northern spotted owl populations in north eastern California. Ecol. Monog. 70: 539-590.

Ingles, L.G. 1965. Mammals of the Pacific State: California, Oregon, and Washington. Stanford University Press, Stanford. 506pp.

Jameson, E.W. and H.J. Peeters. 1988. California Mammals. University of California Press, Berkeley. 403pp.

LaHaye, W.S. 1988. Nest site selection and nesting habitat of the northern spotted owl in northwestern California. M.S. Thesis. Humboldt State Univ. Arcata, California. 111pp.

Laymon, S. 1988. Ecology of the spotted owl in the central Sierra Nevada, California. PhD. Dissertation, Univ. of California, Berkeley. 238pp.

Mueller-Dombois, D. and H. Ellenberg. 1974. Aims and methods of vegetation ecology. John Wiley and Sons, New York. 547pp.

Seamans, M.E. 1994. Breeding habitat ecology of the mexican spotted owl in the Tularosa Mountains, New Mexico. M.S. Thesis. Humboldt State Univ. Arcata, California. 63pp.

Smith, R., M. Peery, R.J. Gutierrez, and W. LaHaye. 1999. The relationship between spotted owl diet and reproductive success in the San Bernardino Mountains, California. Wilson Bull. 111:22-29.

Taylor, C.A., and M.G. Raphael. 1988. Identification of mammal tracks from sooted track stations in the Pacific Northwest Calif. Fish and Game. 74(1):4-15

Thomas, J.W., E.D. Forman, J.B. Line, E.C. Meslow, B.R. Noon, J. Verner. 1990. A conservation strategy for the northern spotted owl. Interagency scientific committee to address the conservation of the northern spotted owl. Portland, Oregon. 427pp.

Thrailkill, J. and M.A. Bias. 1989. Diets of breeding and non-breeding california spotted owls. J. Rapt. Res. 23(2):39-41

U.S. Department of Agriculture. (USDA), U.S. Forest Service. 1992. Protocol for surveying proposed management activities that may impact northern spotted owls. rev. March 17, 1992. Unpubl. 17pp.

U. S. Department of Interior (USDI), U. S. Fish and Wildlife Service. 1990. Endangered and threatened wildlife and plants; determination of threatened status for the northern spotted owl. Federal Register. 55, 26:114-194.

—, BAER Team, North Zone. 1996. Mount Vision Fire Incident Burned Area Emergency

Rehabilitation (BAER) Plan. Department of the Interior. unpubl. rev. March 1996. 116pp and appendices.

Wilkinorthern spotted owln, L., M. Hill, J.P. Welna, G.K. Birkenbeuel. 1992. SYSTAT for windows: Statistics. Version 5. Edition. SYSTAT, Inc. Evanston, Illinois. 750pp.

Willy, A.G. 1992. The habitat associations of the dusky-footed woodrat in Marin County, California. M.S. Thesis. Univ. of California, Berkeley. 66pp.

Zabel, C.J., K. McKelvey, and J.P. Ward. 1995. Influence of primary prey on home-range size and habitat-use patterns of northern spotted owls. Can. J. Zool. 73:433-439.

Interactive Application of GIS During the Vision Wildfire at Point Reyes National Seashore

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Abstract

A wildfire spread rapidly through the Point Reyes National Seashore in the fall of 1995, burning over 12,000 acres. At the height of the fire suppression campaign, 2,164 personnel, including 74 handcrews, 27 bulldozers, 7 air tankers, 7 helicopters and 196 fire engines, were involved. During and immediately following the fire, Geographic Information Systems (GIS) and Global Positioning Systems (GPS) were utilized to monitor the daily/hourly spread of the fire, measure fire suppression actions, assess damage to natural resources, and evaluate damage to adjacent homes in the wildland/urban interface. Examples of GIS/GPS data layers created included fire intensity, bulldozer lines, and fire perimeter over time. Within two weeks, with the aid of GIS, a team of specialists were able to make a comprehensive assessment of the fire impacts and recommend specific actions to the park service for short and long term restoration and rehabilitation.

Introduction

Many land management agencies such as the National Park Service (NPS), have some type of Geographical Information System (GIS) and Global Positioning System (GPS). In the fall of 1995, a wildfire occurred on the at Point Reyes National Seashore, California, and during this emergency, GIS/GPS was utilized to monitor the daily spread of the fire, measure fire suppression actions, and to assess damage to structures and to natural and cultural resources. Additionally, as part of the NPS fire recovery effort, several studies were initiated to evaluate fire and fire suppression effects on the ecological integrity of communities within and adjacent to the burn area. GIS/GPS will help to monitor treatments and effected resources to determine the efficacy of measures taken to mitigate suppression and rehabilitation actions. GIS was a significant tool for integrating the data layers in a form that enabled the park to measure, monitor, and map several data themes simultaneously, providing a more comprehensive understanding of the effects of the fire. GIS, for example, will and has aided in comparing vegetation pre- and post-fire, georeferencing noxious weeds for removal, and georeferencing permanent vegetation plots and photo points. With a more complete base of information, the park can conceive of and initiate more ecologically sound mitigation projects.

Objectives of this paper are to describe the application of GIS during a wildfire in a wildland/urban interface; to demonstrate the application of GIS to assess fire, fire effects and rehabilitation methods; and to recommend how land managers might better prepare for an emergency.

The Setting

Point Reyes National Seashore is located in Marin County, California, approximately 40 miles northwest of San Francisco. It was established by Congress on September 13, 1962 "to save and preserve, for the purpose of public recreation, benefit, and inspiration, a portion of the diminishing seashore of the United States that remains undeveloped" (PL 87-657). The Wilderness Act of 1976 (P.L. 95-544) established 25,370 acres of wilderness and 8,003 acres of potential wilderness in the Point Reyes National Seashore, thereby adding special protection. Today, nearly half of the Seashore is included in the National Wilderness Preservation System. Encompassing 71,046 acres of coastal dunes and prairies, marine terraces, coastal scrub and forests, this geologically unique peninsula has appropriately been called an "Island in Time".

Point Reyes National Seashore shares boundaries with many entities. Marine boundaries are shared with the Gulf of the Farallones National Marine Sanctuary; terrestrial boundaries are shared with Golden Gate National Recreation Area (another National Park), Tomales Bay State Park, The Nature Conservancy, and many private land owners. In 1988, UNESCO's

Morning PIO briefing with GIS-generated map.

Man in the Biosphere program designated the Central California Coast Biosphere Reserve (CCCBR) under the Internal Biosphere Program; CCCBR includes the entire Seashore, the Golden Gate National Recreation Area and other public lands in the region. Much of the eastern border of the park is shared with private landowners, many of which have large homes on multi-acre parcels. Within the park boundaries, there are several dairy and cattle ranches which have continued operation after park acquisition.

The Mediterranean climate produces heavy summer fog and moderate winter rains (average of 30 in/year). Fall tends to be hot and dry and is the period of highest fire danger when vegetation is desiccated. The year-round ambient temperatures are moderate, around 55°C; the difference in the monthly temperatures is only around 6.5°C.

The Vision Fire

The wildfire at Point Reyes was the most devastating wildfire in sixty years with more than 12,000 acres of state, federal and private lands burned. The wildfire was aptly named the Vision Fire after the site of ignition (Mt. Vision); however, the lessons learned from this fire also provided tremendous insights into fire management. The fire began in an illegal campground on State Park lands, and propelled by hot, dry 50 mph winds, spread rapidly through several decadent vegetation communities from the Bishop pine/Douglas fir along the Inverness Ridge to sand dunes along the Pacific Ocean. The rate of spread of the fire reached 3,100 acres per hour.

By the third day, the fire had consumed 12,040 acres from mountain ridges to the sea, including 45 privately owned homes in the wildland/urban interface. Although news agencies reported that 20% of the fire was contained on the second day, a discerning eye could easily distinguish from a map of the fire that the 25% containment represented the Pacific Ocean. The fire continued to flare up over the next 10 days, re-threatening homes along the park boundary. Not until October 16, 12 days after ignition, was the fire declared controlled. Over 95% of the fire burned within the Point Reyes National Seashore. Within the Seashore the Vision Fire burned a total of 11,410 acres, representing 25% of the park's designated wilderness.

At the height of the fire suppression campaign, 2,164 personnel including 74 hand crews, 27 bulldozers, 7 air tankers, 7 helicopters and 196 fire engines, were involved. Park Headquarters were converted into a self-contained city overnight with personnel from several agencies (NPS, California Department of Forestry, U.S. Forest Service, California Department of Corrections, California State Parks, U.S. Weather Service, Bureau of Indian Affairs, and fire departments extending from northern and central California) and organizations (Salvation Army, American Red Cross, etc.) forming an encampment on park lands.

Shortly after the fire began, the superintendent of Point Reyes National Seashore called in

Firefighters walking through the smoke-filled landscape.

a team of experts from the Department of Interior, the Burn Area Emergency Rehabilitation (BAER) team. This team is composed of multi-agency, multi-disciplined resource specialists that are assembled to assess fire damage, fire suppression effects and prepare mitigation measures. The BAER team was made up of resource specialists with expertise in plants, animals, soils, water resources, cultural resources, structures, roads and trails from agencies including the U.S. Forest Service, National Park Service, U.S. Fish and Wildlife Service, U.S. Bureau of Land Management, and Bureau of Indian Affairs. Although not specifically designated, a major component of the team was GIS related.

The primary task of the BAER team was to produce a report documenting the fire and fire suppression effects on the park and to make recommendations for mitigation and management; they were charged to complete the task within two weeks of fire containment. The report noted that there were extraordinary changes in the physical, chemical and biological status of park natural resources. Vegetation resources were impacted by varying degrees as burn intensities varied across the landscape, and these burn intensities in turn have the potential to influence the recovery of the ecosystems. Many areas within the fire perimeter were burned at high and severe burn intensities. Approximately 11% of the fire area was impacted by high intensity fire, 19% by moderately intense fire and 70% burned with low intensities. The plant communities within the fire area include marshland, coastal prairie, coastal grasslands, riparian, coastal dune, northern coastal scrub, bishop pine forest, and Douglas fir forest. Each of these communities has associated species that are unique to California and to the world. Within the burn perimeter, many species of plants (23), mammals (8), birds (24), insects (8), amphibians (4), reptiles (2) and fish (4) are sensitive or endemic to the park. Several species have special recognition under the U.S. Endangered Species Act and the California Endangered Species Act.

A number of plant communities and associations received burning at very high to severe intensities including Bishop pine forests, coastal scrub, northern coastal prairie and some Douglas fir forests. Many of these plant communities occur on steep slopes exceeding 60 percent. Soil associations within the burn are highly prone to erosion and are hydrophobic following exposure to moderate to intense fire. Erosion potential is very high in some region within the burn area due to this combination of factors and to locally high rainfall (14 in./mo.).

Fire suppression actions associated with containing and controlling the Vision Fire relied heavily upon direct and indirect mechanized bulldozer fireline construction. Bulldozer line construction totaling 23 miles of line occurred primarily within the wilderness; much of the dozer line construction occurred in the upper reaches of watersheds with impacts to existing

Superintendent Don Neubacher using GIS-produced maps for morning briefings.

Reviewing fire progression to develop strategy.

Figure 1 (opposite page)

An example of the maps given to firefighters every morning.

trails with direct line construction on extremely steep and unstable slopes. Mechanized dozer lines in some areas traversed locations of known noxious weeds and increased the potential for spread of these species. Downslope of the line construction activities are numerous watersheds, riparian areas, wetlands and estuaries.

Both the fire and the fire suppression efforts exposed many cultural resources; Native American midden sites were uncovered, as well as historical ranch dump sites dating back to the turn of the century. In addition to the 45 structures destroyed by the fire, tens of telephone poles were damaged and an estimated 2,000 hazardous trees posed a risk to park visitors along roads and trails.

GIS Support

During and immediately following the Vision fire, GIS was utilized to map and monitor the hourly/daily spread of the fire, measure fire suppression actions, assess damage to natural resources, and evaluate damage to adjacent homes in the wildland/urban interface. These tasks were possible, though, only through the efforts of many personnel and the generous support of state and federal agencies, private organizations and vendors. Point Reyes National Seashore, like many parks, had a fledgling GIS program with some equipment in place and was in the process of upgrading and moving to new quarters when the fire occurred. Fortunately, within only a 12 hours of fire ignition, the California Office of Emergency Services dispatched a strike team of GIS specialists to aid in fire analysis. This team was a self-contained unit including four specialists and hardware and software capable of assessing the spread of the fire. Upon this foundation, a fully operational GIS lab was in place within two days of fire ignition. The GIS lab extended through three offices and connecting hallways. Cables snaked through offices networking hardware between GIS platforms.

At the height of the operation, hardware consisted of two Sun Microsystems UNIX based workstations (with Arc/Info and ArcView software), two DOS based personal computers (one with PC ARC/INFO and the other with MapInfo), two laptop computers, two Hewlett Packard HP650C Designjet printers, a digitizer and various smaller printers. During the fire effort, the GIS team consumed five rolls of plotter paper, four color cartridges, several reams of paper and tens of diskettes.

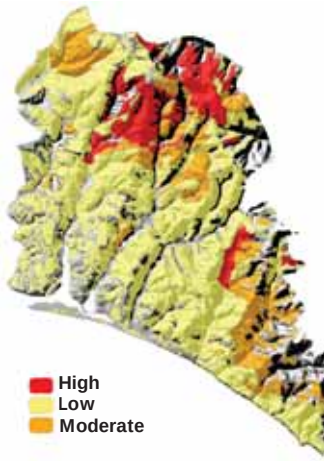
A collection of people with special skills in computer systems administration GPS, and GIS from NPS (Regional Field Office and Golden Gate National Recreation Area); the University of California, Berkeley; the California State Lands Commission; and a member the Department's BAER team rotated through the GIS lab and provided services to keep the operation running smoothly, 24 hours per day. Additionally, one person acted as a liaison between the GIS lab and the outside world, helping to interpret the needs of the 'customers' and what the lab could produce. The language of users and producers oftentimes necessitated translation because many disciplines (geology, hydrology, ecology, computer science, etc.) were combined into the GIS. At one point late at night, a fire

Mapping variation in fire intensity preserved valuable data.

Vision Fire
Point Reyes National Seasore
PACIFIC
OCEAN
DRAKES BAY
POINT REYES
PACIFIC OCEAN
BOLINAS RIDGE
TOMALES BAY
INVERNESS RIDGE
SAN ANDREAS FAULT ZONE
GOLDEN GATE NATIONAL RECREATION AREA
SAMUEL P. TAYLOR STATE PARK
BOLINAS BAY
PRIVATE PROPERTY
N
W
E
S
Vision Fire Perimeter
California
National Park Service
Point Reyes National Seashore
GIS Team
0 1 2 3 4 5 Miles
1:209,960 1 inch equals 3.31 miles

Figure 2

Burn Intensity Zones



fighter asked to find the 'gif illuminators'; after some mental gyrations and clarification, we determined that he was looking for the GIS lab and we came to call ourselves from thence forth the 'gif illuminators'.

GIS Products

Map 'users' ultimately defined the products generated; however, the demand for and the sophistication of products evolved over time as users perceived the value and capability of the GIS output. Users included decision-makers from all disciplines, fire fighters, public information officers, BAER team members, researchers from universities, and the general public.

Initially, the most critical information required from the GIS lab was the fire perimeter. Twice per day, a helicopter with a GPS unit on board flew the fire perimeter, and a map was promptly produced for the fire fighters (Figure 1). Another critical datalayer was the location and condition of structures destroyed by the fire. The California Department of Forestry, Marin County Fire and NPS personnel surveyed homes in the burn area with GPS units (Trimble Navigation, Lt. ProXL and Basic Plus) and collected data on the condition and location of structures. Within four days of the fire ignition, and while the fire was still burning, these data were converted to a GIS datalayer and overlayed with a county parcel map to identify the owners of the structures.

Data were also gathered using GPS on location of hand lines, bulldozer lines, roads, trails, fire suppression effects, noxious weeds, vegetation plots, photo points, and survey points. GIS was then used for mapping, measuring, and monitoring post-fire analysis of burn effects and rehabilitation prescriptions. Examples of preliminary products generated include generalized location and identification of high to moderate burn intensity zones (Figure 2), of fire suppression measures (Figure 3), of cultural resources in relation to bulldozer lines, and of threatened and endangered species in relation to fire suppression actions. As users perceived the ability of GIS to measure and calculate information, they requested reports on acreage's, linear distances, etc.

The park had several existing datalayers including soils, DLGs, DEMs and a few U.S. Geological Survey orthophoto quads; however, crucial, missing datalayer was a digital vegetation map. A recent Landsat TM image was available but it was not ground truthed and could only be used for general reference. Instead, a vegetation map was created using the U.S. National Resource Conservation Services (NRCS) digital soils map and cross-walking this information with the associated vegetation types. This proved surprisingly useful for some of the analyses with some adjustment based on spot checking with aerial photos.

More precise and inclusive information was added to the existing GIS databases. Examples include measuring more precisely the areal extent and location of potential soil erosion sites; monitoring the spread of noxious weeds, the recovery of vegetation communities; and more accurately locating roads and trails (the USGS DLGs were inaccurate). These data were crucial in assessing fire effects and guiding rehabilitation and mitigation prescriptions.

To speed up production of maps and to assure conformity in style, specially tailored, pre-existing programs (AMLs) were brought in and new ones developed. At one point, a California Department of Fish and Game Heritage Program species list of concern was plotted from the State Lands Commission in Sacramento via the internet on a plotter at Point Reyes.

How To Be Better Prepared

There were several challenging problems facing fire teams using GIS, and precious time was lost when problems arose. With just a little pre-planning, many of these issues could be eliminated. During informal discussions and during a debriefing meeting several months after the fire, the GIS team identified several problems and proposed solutions.

Problems:

1. GIS support was informally linked to the Incident Command structure; this led to the GIS team responding to many queries without prioritization.
2. GPS data dictionaries had to be created for field data collection.
3. AML's had to be converted to run on different platforms.
4. Problems with permission access to files on the UNIX workstations.
5. Inadequate disk workspace on the UNIX workstations.
6. Poor translation between platforms.
7. An experienced system administrator was always needed but not always present (night and day).
8. Different map projections and different scales of data.
9. Data was incomplete and sometimes out of date or of poor quality.

10. Lack of physical space for bulky computer systems, digitizers and plotters.
11. Lack of understanding of GIS capabilities by fire staff resulted in redundant work effort.
12. A computer virus brought in on a laptop computer plagued the systems for a couple of days before being identified and removed.
13. Non-standard file naming conventions and data categories.
14. Too many people and too few computers.
15. Lost opportunity for tracking fire history because hand drawn maps by fire fighters were lost early on as GIS lab geared up.
16. No tracking method for system administration of the workstations.

Solutions:

1. Place GIS support formally in the Incident Command System.
2. Prioritize products and place OPS first.
3. Prepackage GPS data dictionaries.
4. Prepackage AML programs for products with maps of several sizes. Ensure that the AML's can easily be transferred across platforms and are well documented.
5. Make sure that more than one person at a land management agency has GIS experience (training) and understands the capabilities and limitations of GIS.
6. Scan for viruses as new equipment is brought in.
7. Establish solid contacts with vendors so that problem solving can occur swiftly during a disaster.
8. Maintain a standard projection within the land management agency and have the AML's to convert.
9. Keep one complete set of hard copy maps in a secure place.
10. Maintain connections with GIS specialists with other agencies, organizations and universities through conferences, societies, etc. These contacts were the key to getting the GIS lab jump-started.

Figure 3

Map of fire suppression measures.

11. Maintain a list of contacts with names and home phone numbers. Disasters usually occur in the evening, over the weekend or on holidays.
12. Provide a bin for placement of early hand drawn maps so fire history data is not lost.
13. Provide a log book for each workstation tracking problems and serving as a reference for team members.
14. Develop a metadata form and maintain it during the fire.
15. Schedule a debriefing meeting of GIS team shortly after the event while ideas are still fresh.

The Vision fire at Point Reyes is a wake-up call for many private and public agencies. Although it is impossible to contemplate and identify every problem in providing GIS related services during fires or other emergency operations, we are convinced that GIS is, and will continue to be, a vital tool to emergency responders in the future. It is our hope that by documenting our experiences, identifying the problems we encountered and identifying pre-planning considerations, more public and federal agencies will be better prepared to handle emergency incidents more effectively.

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Literature Cited

BAER. 1995. Mount Vision Fire incident: Burned Area Emergency Rehabilitation Plan. U.S. Dept. of Interior.

Point Reyes National Seashore 1980. General Management Plan. U.S. Dept. of Interior, National Park Service, Point Reyes National Seashore.

Point Reyes National Seashore 1993. Statement for Management. U.S. Dept. of Interior, National Park Service.

Appendix A

Habitat variables and methods (following Seamans 1994, Call 1990, LaHaye 1988, Bias 1992).

Variable	Method
Elevation	elevation measured with an altimeter
Aspect	aspect measured with a compass
Slope	slope measured with a clinometer
Canopy closure	percent of canopy closure of plot, taken as average of visual estimates from each cardinal direction (4)
Number of	number of trees in tally with a) a cavity >48cm along its axis, b) a broken top
potential nest	>53cm diameter with live branches over it, or c) an existing platform nest trees>60cm diameter
Plot maturity	estimated visually for the plot (overmature, old growth, mature, maturing, young)
Strata	number and type of distinct vegetative layers (canopy, subcanopy, understory, shrub, herbaceous) within the plot
Max. shrub height	height of tallest shrub, measured or estimated in meters
Tree height	height of tallied trees, measured with a clinometer or estimated from a nearby measured tree, and estimated for trees less than 80'
Total basal area	basal area (sq m/ha) of all stems tallied with a 20 basal area factor prism, multi-stem trees were counted as separate trees (Mueller-Dombois 1974)
Conifer basal area	basal area of conifers (see above)
Hardwood basal area	basal area of hardwoods (see above)
Snag basal area	basal area of snags (see above)
Sapling basal area	basal area of any live tree 10.0-12.4cm dbh (see above)
Young basal area	basal area of any live tree 12.5-27.4cm dbh (see above)
Medium basal area	basal area of any live stems 27.5-52.4cm dbh (see above)
Mature basal area	basal area of any live stems 52.5-89.9cm dbh (see above)
Large basal area	basal area of live stems >90cm dbh (see above)
Trees <10.1cm dbh	percent of ground covered by dominant tree species <10.1cm dbh, estimated using the line- intercept method (Mueller-Dombois 1974)
Shrub cover	percent of ground covered by shrub species
Herbaceous cover	percent of ground covered by herbaceous plant species
Duff cover	percent of ground covered by duff
Inorganic litter cover	percent of ground covered by inorganic material, such as rocks
Small dead and down dbh, at largest end	percent of ground covered by dead and down woody material >2.5 - <30cm
Large dead and cover	percent of ground covered by dead and down woody material >30cm dbh, at largest end

Nest site variables and methods (adapted from Seamans 1994, Call 1990, LaHaye 1988).

Variable	Method
Nest type	type of nest - broken top, cavity, or platform. For platform nests, observer postulated primary nest occupant
Nest tree species	species of tree which nest is located in
dbh	diameter at breast height of nest tree measured with a dbh tape or measuring tape
height	height of nest tree measured with a clinometer
Nest height	height of nest measured with a clinometer
Orientation	for platform nests, the orientation of the nest from the bole, measured in degrees
Distance to bole	for platform nests, the distance from the bole, measured in meters
Canopy cover	percent canopy cover directly over the nest, estimated visually from the ground
Number of shelter trees	number of trees with live branches directly over nest
species	species of shelter trees
dbh	diameter of breast height of shelter tree (see above)
height	height of shelter tree (see above)

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