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BIOASSESSMENT OF STREAM CONDITIONS AND RESPONSE TO LAND USE IN LOS PADRES NATIONAL FOREST

A Group Project submitted in partial satisfaction of the requirements for the degree of Master of Environmental Science and Management for the Donald Bren School of Environmental Science and Management

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The Group Project is required of all students in the Master's of Environmental Science and Management (MESM) program. It is a three-quarter activity in which small groups of students conduct focused, interdisciplinary research on the scientific, management, and policy dimensions of a specific environmental issue. The final Group Project report is authored by MESM students and has been reviewed and approved by:

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Abstract

The United States Forest Service supervises Los Padres National Forest to protect wildlife and vegetation for multiple uses and ecosystem management. This project examines two bioassessment methods for use as management tools in the Forest: the River Invertebrate Prediction and Classification System (RIVPACS) and the Index of Biological Integrity (IBI). Unlike traditional physical and chemical measurements of water quality, bioassessment uses aquatic organisms as indicators of ecosystem health. The RIVPACS and IBI models analyze benthic macroinvertebrate data collected from Forest streams in 1999 and 2000 to assess anthropogenic effects on aquatic ecosystems. The results of the IBI analysis reflect that invertebrate populations respond to human activity in sample sites downstream of certain land uses, with cattle grazing and recreation showing the most obvious effects. Data limitations encountered in the construction of the RIVPACS model introduce significant uncertainties that reduce its utility as a management tool in Los Padres. Based on these results, we recommend the IBI model for monitoring aquatic impacts and site restoration through time. Examination of taxa abundances and error balancing can be used to strengthen conclusions drawn from raw IBI scores.

Executive Summary

Background

In the past decade, federal and state agencies have begun to develop and implement bioassessment methods to evaluate water quality and the condition of aquatic ecosystems (USEPA 1998b, USEPA 1996b). This project examines two bioassessment models, the River Invertebrate Prediction and Classification System (RIVPACS) and the Index of Biological Integrity (IBI), for use by the United States Forest Service (USFS) in managing Los Padres National Forest in California.

Bioassessment is a method used to evaluate water quality by measuring taxa occurrence and community characteristics of the aquatic habitat. Unlike chemical analyses, such as dissolved oxygen or nutrient loading, bioassessment uses organisms and communities as proxies for ecosystem status. The RIVPACS and IBI bioassessment models rely on benthic macroinvertebrate (BMI) communities to gauge the health of streams throughout the Forest. BMI assemblages are used because their analysis is inexpensive, data collection is relatively easy, and individual taxa typically exhibit known responses to different human-induced stressors.

High-quality, or undisturbed streams typically exhibit high biological diversity, and a suite of taxa (mayflies, stoneflies, caddisflies) that are indicative of good habitat conditions. Disturbed stream sites feature more soft-bodied organisms, worms, nematodes, and taxa that are tolerant of pollutants or sedimentation. Both models were developed by comparing undisturbed reference sites to test sites that may have been impaired by anthropogenic influences. Presence and absence of taxa were used in both models, and IBI also incorporates some abundance values.

The 1.75 million acre Los Padres National Forest is located in central California, stretching from Monterey County south to Los Angeles County. The Forest is managed for multiple uses, including wilderness protection, wildlife habitat, fire control, recreation, mining, and maintaining municipal water sources.

In 1999 and 2000, the USFS sampled BMIs as part of individual management projects in Los Padres National Forest. Some of these data were collected as part of a larger RIVPACS bioassessment project by Hawkins *et al.* (2000), while the remainder have not yet been used in any bioassessment evaluation prior to the current investigation.

Most of the stream reaches were sampled to analyze a certain land use or potential impact on the aquatic ecosystem. Reference sites were usually placed upstream from the suspected impact and test sites were located downstream. Suspected impacts included campgrounds and recreation, fire, roads and bridges, cattle grazing, landslides, oil and gas operations, and pick and shovel mining. Some sites were also chosen to assess recovery and mitigation efforts. Sample sites were located throughout the Forest to assess water quality at individual sites, streams, and entire watersheds.

Problem Statement

In this project, previously collected BMI data were assessed using the two bioassessment methods. The data were subjected to an existing IBI model that was developed for southern California (Ode *et al.* 2005), and a pilot RIVPACS model that was developed specifically for sites in Los Padres. Each model produced a score for all sites. Scores were compared with potential sources of impacts near the sample locations; scores were also compared between models and across test and reference sites.

The results of the comparisons were used to assess stream condition at sites throughout the Forest, identify land uses which may threaten aquatic ecosystems, and evaluate the differences in the two bioassessment methods. Our findings provide the basis for management recommendations for Los Padres and other interested agencies.

The project is organized around three repeating themes: IBI, RIVPACS, and land use analysis. Each theme is addressed in methodology, results, and discussion of the relevant findings. The discussion also includes a comparison of the models and management recommendations.

IBI

IBI is one of the most commonly used forms of bioassessment (Simon 1999) and is the foundation for U.S. federal programs for biological monitoring (Karr and Chu 2000). It is a multi-metric model that utilizes community attributes that vary in predictable ways to human disturbances. We used the SoCal IBI, developed by Ode *et al.* (2005), as the bioassessment protocol in our investigation. This model was developed for southern California, and 56 of the 275 sites used to construct the model were in Los Padres National Forest. The following seven metrics were used: percent collectors (gatherers and filter feeders), percent non-insect taxa, percent tolerant taxa, Coleoptera richness, predator richness, percent intolerant individuals, and EPT taxa richness (Ephemeroptera, Plecoptera, and Trichoptera orders). These seven metrics were found to respond to environmental stressors (Ode *et al.* 2005) as described below in section 1.6. Each metric was assigned a ranked point system by which a BMI sample can be scored. IBI scores range between 0 and 100 with lower scores indicating reduced site quality.

RIVPACS

The RIVPACS model is a predictive model that relies on the comparison of BMI assemblages between reference and test sites (Moss *et al.* 1987, Wright *et al.* 1984, Coysh *et al.* 2000). We constructed a pilot RIVPACS model based on models developed by Hawkins *et al.* (2000) and the Australian version of the RIVPACS model, AUSRIVAS. Construction of the RIVPACS model can be broken down into five simplified steps:

1) Reference sites are classified into clusters based on similarity of BMI taxa.
2) Habitat variables (e.g. annual precipitation, elevation) that can discriminate between the clusters are determined.
3) Habitat variables for the test sites are used to place them into the established clusters.
4) Based on the taxa composition of each cluster, the taxa that are expected (***E***) to occur at each site are determined.
5) Observed (***O***) taxa are compared to the corresponding expected taxa (***E***) calculate the RIVPACS score: ***O/E***.

RIVPACS scores vary about a score of one. RIVPACS scores that fall outside the 95% confidence interval around the mean reference score are considered impaired.

Land Use

To test the suspected impacts that the USFS considered to be relevant, both IBI and RIVPACS scores were analyzed for their correlation with available land use data. Reference and test site pairs were examined with respect to the impact zone between them. Additionally, multiple sources of potential impacts at all sites were examined using an analysis of variance (ANOVA). Potential impact variables used in the ANOVA were: grazing (within 75 meters of grazing allotment), fire history (size, distance, and age of fire), roads (nearest upstream road, road type), and recreation (nearest recreation area).

Results

Of the 50 Los Padres samples scored by the IBI, Prewitt Creek Test 2 was the only sample which ranked poor (20-39) with a score of 38, while 20 sites ranked fair (40-59), 26 ranked good (60-79), and three ranked very good (80-100). There was a significant difference in IBI scores from reference versus test sites (ANOVA $p = 0.006$).

The results from our RIVPACS analysis indicated that there was no significant difference between the test sites and reference sites ($p > 0.05$). Five test sites and one reference site were considered impaired:

- Matilija Creek Test 1
- Sisquoc Test 2
- Chorro Creek Test
- Tar Creek Test
- Sespe at Tule Test
- Santa Paula Reference

Further analysis showed a lack of correlation between IBI and RIVPACS scores ($R^2 < 0.05$) indicating that the models did not score the sites in the same way, and that each model is sensitive to different parameters that are not consistent among sites.

Only the IBI scores showed a significant change in relation to any of the land uses. The eleven sites within 75 meters of a grazing allotment scored significantly lower than sites further away from grazing (ANOVA, $p < 0.042$). The apparent decrease in aquatic BMI health may be due to erosion, defoliation, and nutrient loading associated with grazing. The scores were consistent with findings that sedimentation can affect BMI communities (Cordone and Kelly 1961, Murphy *et al.* 1981, Ohmart 1996).

Fire history also appeared to be relevant (ANOVA $p < 0.007$), with higher IBI scores in sites sampled within the first two years after a nearby fire compared to sites sampled two or more years after a fire. BMI response to fire is highly variable (Minshall 2003), thus it is unclear if our results are indicative of consistent forest-wide trends.

Recommendations

We presently recommend the SoCal IBI bioassessment model as a tool for managing Los Padres National Forest. Suggestions for improving the IBI model include refining the tolerance estimates of native taxa to improve the accuracy of some metrics, and constructing a more specific model tailored for Los Padres National Forest. We cannot presently recommend the RIVPACS model because a proper assessment of the model could not be made. Developing a RIVPACS model requires a large number of sites to satisfy guidelines for construction. The data available for our analysis were not sufficient for a suitable development and analysis; the results of the pilot RIVPACS model cannot be viewed with much confidence.

Management recommendations include expanding and standardizing the sampling protocol to include riparian habitat and percent shade cover, minimizing grazing allotments in aquatic endangered species habitat, and monitoring burned areas through time to further investigate the nature of this impact.

The results of our work can be strengthened by including replicate samples taken in 2004 in subsequent analyses. In the future, aquatic bioassessment in Los Padres may be relevant in habitat classification for the endangered southern and south-central coastal steelhead (*Oncorhynchus mykiss iredeus*), California red-legged frog (*Rana aurora draytonii*), and arroyo toad (*Bufo californicus*).

Commonly Used Acronyms

ANOVA	Analysis of Variance
BMI	Benthic Macroinvertebrate
Bug Lab	Utah State University National Aquatic Monitoring Center
CDFG	California Department of Fish and Game
EPT	Ephemeroptera, Plecoptera and Trichoptera Orders
HUC	Hydrologic Unit Code
IBI	Index of Biotic Integrity
RIVPACS	River Invertebrate Prediction and Classification System
SoCal IBI	Southern California Benthic Macroinvertebrate Index of Biological Integrity
TSO	Transformed Stream Order
USEPA	United States Environmental Protection Agency
USFS	United States Forest Service
USGS	United States Geological Survey

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1 Introduction

1.1 Background

With the passage of the Clean Water Act in 1972 and the Porter-Cologne Act of 1977 efforts were made to "maintain and restore the chemical, physical and biological integrity of the Nation's waters" (Federal Water Pollution Control Act Section 101 (a), Act 33 USC 1251 seq.). To track physical changes to a water body, various monitoring techniques have been developed to assess the health of a water body. Bioassessment methods measure water quality by analyzing the taxonomic composition of aquatic communities. This is accomplished by comparing undisturbed reference sites to sites with suspected impacts to differentiate between natural and human-induced changes to biotic assemblages (Karr and Chu 2000). The United States Environmental Protection Agency (USEPA) has implemented bioassessment criteria into water quality standards as stated in the Clean Water Action Plan (1998a) and in the Water Quality Criteria and Standards Plan (US 1998b). In 1993, the California Department of Fish and Game (CDFG) developed the California Stream Bioassessment Procedure, modeled after the USEPA's Rapid Bioassessment Protocols for Streams and Wadeable Rivers (USEPA 1996b). Since the development of the California procedure, individual Regional Water Quality Control Boards, the United States Forest Service (USFS), and the CDFG all have developed regionally specific bioassessment models.

Two commonly employed bioassessment techniques are the River Invertebrate Prediction and Classification System (RIVPACS) and the Index of Biological Integrity (IBI). Both IBI and RIVPACS models focus on a particular biological system. RIVPACS uses invertebrate fauna in the river benthos, while IBIs have been created using aquatic invertebrates, fish, algae, and plankton. Both models compare relatively clean and undisturbed 'reference' sites with 'test' sites that are potentially impaired.

In its effort to protect the resources in Los Padres National Forest, the USFS has sampled aquatic bottom-dwelling invertebrates (benthic macroinvertebrates, BMIs) for assessment of biotic status at individual sites, streams, and entire watersheds. BMI assemblages are used because they are easy to collect, inexpensive to process, and they exhibit a variety of sensitivities and tolerances to different stressors. Common categories of measurements that classify BMI assemblages are temperature preferences, functional feeding groups, sediment preferences, and riparian conditions. The presence, absence, and abundance of different taxa can thus indicate the biotic status of a stream reach.

BMI data in Forest streams were collected in 1999 and 2000 during USFS surveys. Reaches sampled during this time include river mainstems, tributaries, and coastal streams in Monterey, San Luis Obispo, Santa Barbara, and Ventura counties. Reaches that were potentially impaired by forest activities were identified and surveyed along with non-impacted areas of similar environmental characteristics, usually upstream of the impaired site. Raw BMI data, in addition to preliminary biotic indicators, such as the Shannon and Simpson biodiversity indices, are available for use in the RIVPACS and IBI models. Both models are dependent on the ability to identify conditions expected in the absence of anthropogenic influences. This is important because it can aid in distinguishing natural variation from human induced changes when comparing communities across sites.

1.2 Problem Statement

This project aims to provide the USFS and other interested agencies with a management tool to assess ecosystem condition and land use practices in Los Padres National Forest. Identifying impaired streams and watersheds, and the associated activities responsible, will help prioritize restoration, remediation, and conservation efforts for the multiple watersheds in the National Forest and provide a method for evaluation and monitoring of these efforts over time. Our analysis includes an evaluation of the previously collected BMI data and other relevant watershed information through two complimentary bioassessment models, IBI and RIVPACS, in an effort to:

- Assess stream conditions at each of 20 monitored sites in five watersheds in Los Padres National Forest.
- Correlate BMI assemblages with potential threats to water quality.
- Identify land use activities that may be responsible for any observed impairments of BMI populations.
- Evaluate conditions of the watersheds as a whole.

- Recommend and prioritize additional sampling, analysis, and restoration activities.
- Critically evaluate the two prominent methods for benthic macroinvertebrate bioassessment.

A BMI analysis can indicate the relative impact of specific land use practices on the health of Los Padres streams and watersheds, and provide a basis for investigating the relationship between impaired sites and specific Forest activities.

As one of the first projects to compare the two bioassessment techniques in California, the results of our project may be of use in constructing management plans in other National Forests. Data compiled for the RIVPACS and IBI models and the outcome of our investigation are also of interest to the USFS for analyzing habitat suitability for the endangered southern and south-central coastal steelhead (*Oncorhynchus mykiss iredeus*). The methods and results of habitat evaluation can be instrumental in monitoring the recovery of such species. Although our research will not examine these aspects, results from our analysis may be of importance for possible future research.

1.3 Setting

Los Padres National Forest encompasses approximately 1.75 million acres of central California coast and rangeland stretching between Monterey and Los Angeles Counties. It is separated into two divisions, a northern division within the Monterey and San Luis Obispo counties and a 'main' southern division in San Luis Obispo, Santa Barbara, Ventura, and Kern Counties (Figure 1).

Due to the expansive area that it covers, the climate in Los Padres National Forest varies from Mediterranean (mild winters and intense dry summers) in the coastal areas to semi-desert in the more inland eastern areas. Climate differences also occur when moving from north to south as well as west to east due to factors such as topography and air and water temperatures. The spatially and temporally varied climate causes large variations in water flows with many rivers and streams having surface flow through the fall and spring months and little or no surface flow in the summer months, especially in the more southern areas. This can be exemplified in the typically higher annual precipitation and consequent water flow in the northern division of the Forest that encompasses more coastal land and is subjected to different climate as compared to the southern division (USFS 2005a).

The Forest stretches over 220 miles from its northern to southern ranges and is comprised of varying vegetation, wildlife, and ecosystems. Within its boundaries chaparral, evergreen, oak, juniper, redwood, and conifer forests, as well as inland desert areas provide habitat for over 450 species of fish and wildlife. Currently twenty-six endangered or threatened plant and animal species can be found on Los Padres National Forest land, such as the California condor (*Gymnogyps californianus*), subpopupuations of steelhead (*Oncorhynchus mykiss iredeus)*, California red-legged frog (*Rana aurora draytonii*), and arroyo toad (*Bufo californicus*) (USFS 2005a).

Figure 1. Index map of Los Padres National Forest.

Multiple uses of Los Padres National Forest include rangeland, minerals, recreation, and a reliable water source for rural and metropolitan communities from Monterey to Los Angeles. Recreational uses of the Forest include fishing, hunting, camping, hiking, and off-road vehicle trails among others. No significant commercial or private logging is conducted in Los Padres. Rangeland for livestock is limited to special-use permit areas and 873,000 allotment acres of which 410,945 are capable for grazing. Most of this rangeland is on annual grassland habitats and is managed with the goal of sustainable grazing. Nearly all rangeland users are from families with a history of livestock use dating back to before the National Forest system was established. Minerals such as gold, uranium, sandstone, phosphates, and gypsum can also be found throughout the Forest and oil and gas are commercially mined, primarily from the Sespe Oil Field in Ventura. Rainfall in the Forest flows into various reservoirs and recharges groundwater supplies that are then used as a major source of drinking water for many downstream communities (USFS 2005a).

The arid climate, dry vegetation, and anthropogenic activities cause wildfires to affect habitat in Los Padres National Forest. From 1912-2002 a total of 2,366,970 acres were burned, for an average of 25,000 acres per year. The majority of fires are human-induced, while lightning accounts for the remainder. The frequency of natural fires has ecological consequences in forest vegetation. Fires help distribute chaparral of different age classes and many plants require fire for seeding or dispersal. Prescribed burnings are one method used by the USFS to decrease the dry and dead chaparral vegetation that is fuel for most fires (USFS 2005a), but some fire-dependent species have now declined. Los Padres National Forest is also subject to landslides that are often a natural occurrence, but can also be exacerbated by anthropogenic activities such as agricultural practices, grazing, fire suppression or fire effects, road cuts, and inadequate culverts.

Management of Los Padres National Forest is overseen by the USFS Supervisor's Office located in Santa Barbara County. This office coordinates the five other USFS district ranger offices into which the Forest is divided. The districts offices are: Monterey, Santa Lucia, Santa Barbara, Ojai, and the Mount Pinos Ranger Districts. The main management goals of the Forest are to safeguard its watersheds and water resources, to provide for outdoor recreational and wilderness activities, to protect the diversity of wildlife and vegetation that it supports, and to encourage scientific research and exploration (USFS 2005a).

1.4 Index of Biotic Integrity (IBI)

An Index of Biotic Integrity (IBI) measures ecosystem status by integrating multiple biological indicators into a numerical assessment score (Karr and Chu 2000). These individual biological indicators, or metrics, attempt to capture ecosystem impacts by showing changes in aquatic community composition along a gradient of human disturbance.

IBI has been in use for more than twenty years and has been applied and tested on every continent except Antarctica (Karr and Chu 2000). IBI is one of the most commonly used bioassessment tools and "arguably one of the most effective" (Simon 1999). Because of its reliability, IBI has become the foundation of U.S. Federal Programs for biological monitoring (Karr and Chu 2000). IBI was first developed for use with fish and has since been constructed for use with invertebrates, algae, plankton, and vascular plants (Karr and Chu 2000). IBI can be used in a variety of environments including wetlands, streams, coastal estuaries and terrestrial ecosystems (Karr and Chu 2000).

IBI is seen as an effective assessment tool because it captures human disturbances in all of the five water resources attributes identified by Karr and Chu (2000): water quality, habitat structure, flow regime, energy sources, and biotic interactions. Metrics are chosen because they vary in a predictable way to human disturbance, are easy to measure and are sensitive to a range of physical, chemical, and biological factors (Karr and Chu 1999). Metrics included in an IBI must be able to effectively distinguish between the effects of human disturbance from the 'noise' of natural variation (Ode *et al.* 2005).

1.5 RIVPACS

The River Invertebrate Prediction and Classification System (RIVPACS) is a family of predictive models that has only recently been applied to streams in the United States (Hawkins *et al.* 2000). RIVPACS has been used with success in Great Britain (Moss *et al.* 1984), and a similar model (AUSRIVAS) was developed in Australia (Coysh *et al.* 2000).

The RIVPACS model provides a means of comparing the biological community of reference and test sites (Coysh *et al.* 2000, Moss 1987, Wright *et al.* 1984). Reference sites are used to generate baseline macroinvertebrate communities to which test site communities are compared. By comparing test and reference sites, a list of expected taxa is generated for each test site. This list is the macroinvertebrate community that would be expected if the site were not impaired by human activity. The expected list (***E***) is then compared with the observed taxa in the site sample (***O***). The ratio ***O/E*** expresses the degree of biological impairment (Hawkins *et al.* 2000). A score of one indicates that the BMI community is equivalent to what would be expected at an undisturbed site. Scores that differ from one indicate that the community is different from expected, and that anthropogenic disturbance may presumably be the cause.

1.6 Land Use Impacts

The USFS identified seven different potential land use categories within our study area, both natural and anthropogenic, that may impact aquatic ecosystem health in Los Padres. These categories include: campground/recreation, roads/bridges, cattle, fire, landslide, oil and gas activity, and gravel mining. Each of these activities can impact the health of aquatic ecosystems. Our project uses the bioassessment models to evaluate the chosen streams and the corresponding forest activities which may produce negative impacts. This section reviews the potential impacts of each activity on streams and BMI communities.

1.6.1 Campground / Recreation

Recreation can have four types of impacts on forest streams: exploitation, disturbance, habitat modification, and pollution (Knight and Cole 1995). While some of these impacts are water-related, others can be results of terrestrial activity. Habitat modification, physical disturbance, and pollution can occur directly within the stream, such as washing in streams with soaps in a camping area. Activities that occur outside of the stream, such as trampling of bank vegetation, sediment erosion from roads, trails, and camp areas, trash, and human waste disposal have similarly destructive impacts on aquatic ecosystems. These impacts can alter water temperature, chemistry, and organic matter content, as well as increase sedimentation and turbidity through bank destabilization (Cole and Landres 1995). Effects on flow regimes and substrate characteristics are also commonly observed and can encourage or discourage the presence of different species through habitat alteration and food availability.

Recreational activities can alter the physical nature of the stream and affect the habitats and living spaces of aquatic animals (Cole and Landres 1995). A number of studies have demonstrated how changes in the physical characteristics of streams and flow regimes strongly influence the biotic assemblages found within. In 1983, Hawkins *et al.* showed how shade cover from vegetation and overhanging banks protect fish and salamanders due to an associated decrease in water temperature and solar radiance. Speaker *et al.* (1998) found that the efficiency of large woody debris in capturing leafy material and increasing organic matter, while Stanzer *et al.* (1988) demonstrated how mean water velocity and turbidity strongly control the presence and distribution of macroinvertebrates, microorganisms, and fish within a reach. In addition to organic matter and flow regimes, substrate characteristics such as size, surface area, texture, heterogeneity, and stability of particles are known to provide very specific aquatic habitats such as attachment and oviposition sites (Minshall 1984). Activities that alter physical characteristics of a stream's flow and substrate, such as changes in riparian vegetation, should have visible effects on BMI assemblages.

Changes due to terrestrial and water activities in recreation areas also alter the availability of food for BMIs and other aquatic animals. Organic input from vegetation is often a primary source of food and nutrients, and its disturbance is reflected in community structures and assemblages. Cummins *et al.* (1989) showed how peaks in the biomass of shredders feeding on vascular plants tissues is often correlated with the availability of litter. Increases in sediment loads can cover and kill periphyton, bacteria, and fungi on surface rocks that sustain arthropods, amphibians, and fish (Cordone and Kelly 1961, Murphy *et al.* 1981). Thus the effects of recreational activities that decrease organic input to streams by removing vegetative cover are reflected in changes in macroinvertebrate community structure.

Though the product of exploitation, disturbance, and habitat destruction can be seen in a relatively short period of time, consequences of some off-site activities may have a considerable lag-time before their influence is recognized. This occurs with polluting activities such as sewage effluent, gasoline and oil leaked from off-road vehicles, as well as destabilized banks that release excess sediment during heavy rain events. Such activities can alter the chemical, nutrient, and sediment loading of a stream, which in turn can affect BMI populations.

1.6.2 Roads / Bridges

Activity along transportation corridors has obvious impacts on terrestrial plants and animals, but it can also affect water quality and aquatic ecosystems. The specific impact of a road depends on a variety of factors (e.g. paved, un-paved, frequency of use, etc.), but commonly results in some amount of altered physical and chemical stream characteristics (Trombulak and Frissel 2000).

Altered physical characteristics include changes to patterns of runoff and sedimentation, and fluctuation in temperature, light levels, and dust. There are a number of reasons why road networks cause increased sedimentation. Road construction and maintenance requires vegetation removal, which reduces soil stability from root support and decreases canopy cover, increasing erosion during heavy rain events. Because roads have decreased friction and permeability, they are good conduits for runoff and efficiently route surface water toward stream crossings. The result is increased levels of fine and suspended sediment and altered flow regimes.

Other concerns with adjacent or intersecting roads include elevated levels of heavy metals, salts, organic molecules, ozone, and nutrients. These chemical concentrations are correlated with road traffic and are usually associated with gasoline additives and road deicing (Goldsmith *et al.* 1976, Dale and Freedman 1982, Leharne *et al.* 1992).

1.6.3 Cattle

There are three main concerns regarding cattle grazing and stream health: erosion/sedimentation, nutrient loading, and bacteria or parasite contamination. Cattle can affect stream sedimentation locally through trampling and vegetation removal which results in bank destabilization and erosion near the stream (Ohmart 1996). This can also impair the ability of the soil to retain water and support riparian vegetation. Remote impacts can include improperly managed grazing upland that increases overland flow and sediment transport into a stream. Nutrient loading due to excrement may increase the levels of nitrogen and/or phosphorous in streams, an effect which can be exacerbated by erosion. Exogenous nitrogen and phosphorous additions can lead to microbial blooms and turbidity, changes in pH, lowered dissolved oxygen content, and the buildup of organic matter in sediments (Schlesinger 1997). Any number of these effects may alter the ability of streams to support macroinvertebrate communities.

Cattle can also transport harmful bacteria and parasites, such as *Giardia*, *Cryptosporidium*, and *Escherichia coli* (E. coli) into surface waters. The factors influencing such contamination include the volume of manure, the concentration of bacteria and parasites in the manure, barriers to bacteria and parasite movement, and the extent to which previous deposits are resuspended (OAGBC 1998). While some BMI taxa are more sensitive to bacteria and parasites than others, the presence of bacteria and contamination is not generally captured by BMI communities. Sedimentation however, as discussed above, does impact macroinvertebrate communities.

1.6.4 Fire

The direct effects of fire on macroinvertebrate communities are generally minor or indiscernible. Comparison of reference and burned streams in prior studies show essentially identical communities, with the exception of intense heating experienced in small or shallow streams, extended smoke exposure, or excessive retardant drops (Minshall 2003, Figure 2).

Figure 2. Fire control near Lake Cachuma (USFS 2005b).

Indirect effects, resulting primarily from increased rates of runoff and channel alteration, have the greatest impacts on macroinvertebrate community metrics and foodweb responses. The relative impact of these effects depends on the nature of the fire and environmental characteristics of the habitat. Post-fire effects are variable in time and space, but are generally observed in smaller size streams (first to fourth order) in the first five to ten years following fire and are associated with the more intense burns (Roby 1995). Studies have shown that changes in macroinvertebrate community structure in response to burn intensity and extent are variable with regard to stream size and gradient, amount of precipitation, timing of runoff events, vegetation cover, geology, and topography (Minshall 2003, Roby 1995). The influence of each of these variables has been difficult to quantify as streams often show different responses based on a combination of influences. The most dramatic impacts, however, appear to be associated with physical alterations caused by flooding and mass movements. Increased erosion, runoff, and sediment transport and deposition resulting from fires impacts the habitat and food availability to aquatic ecosystems. Since these impacts depend on the timing, intensity, and amount of precipitation, effects can vary between regions and often appear some time after the actual fire event.

The intensity of precipitation during the first post-fire year is an important consideration with regard to the indirect effects of fire on macroinvertebrate communities. Flooding and mass movement accompanied by channel alteration and sediment transport and deposition can have dramatic impacts on the macroinvertebrate community (Rinne 1996, Minshall *et al.* 2001a). In the Rocky Mountains and intermontane areas, these events usually are associated with snowmelt runoff or intense mid-summer rainstorms following the July–early September fire season. In southwestern montane watersheds, flood events often occur during the July–August monsoon period immediately following the May–June fire season (Rinne 1996). A study which investigated the first-year post-fire sediment deposition in the Malibu Creek Watershed (Los Angeles County, California) found little evidence of degradation of habitat due to post-fire sediment deposition. However, the amount of soil erosion and sediment deposition that could have been produced during the first post-fire wet season may have been reduced by various factors, including below-average precipitation in that particular year (Spina *et al.* 2000) .

Results from studies of macroinvertebrate response to fire in Yellowstone National Park indicate that the size of the stream combined with the size and timing of the fire influence the magnitude of macroinvertebrate response. Minshall *et al.* (2001a, b) showed a positive correlation between the area of the catchment burned and macroinvertebrate response, and an inverse relationship between stream size and the degree of impact. Taxa richness, abundance and total biomass may return to pre-fire conditions within a year or two following the disturbance, but variations in the communities may persist for much longer. Studies have shown some communities can recover in five to ten years, while others exhibit differences from adjacent reference sites up to 45 years later (Albin 1979) The average recovery however, appears to be between ten and 15 years (Roby and Azuma 1995, Minshall *et al.* 2001a, c).

Though few studies have been conducted on food-web dynamics in post-fire streams, it is expected that changes in the quality and quantity of available food, such as leaf litter and organic matter, would affect the presence and abundance of different species. The most common response seen in BMI community composition is a shift toward increased dominance and relative abundance of disturbance-adapted strategists (Mihuc *et al.* 1996). Studies of functional feeding group composition of Cache Creek following fires in Yellowstone National Park showed that trophic generalists capable of using two or more resources for growth dominated post-fire communities (Mihuc and Minshall 1995). Evidence of macroinvertebrates switching food sources in response to altered resource availability was also found in Yellowstone.

Despite localized observations and general expectations, strict adherence to patterns of feeding group replacements is not generally observed in post-fire BMI communities. The reason for these deviations appears to be greater influence of altered physical features (e.g. turbidity, sedimentation, and scouring) observed in post-fire environments (Mihuc *et al.* 1996).

1.6.5 Landslides

Landslides affect streams by inputting large pulses of sediments and woody debris. These disturbances are part of the natural input to streams and many ecosystems depend on them to replenish substrates and provide habitat. A study of macroinvertebrate response to landslides in Washington's Hoh River indicated that there was little difference in community structure between reference and test sites (McHenry 1991). Though absolute abundance of BMIs differed between the reaches, the relative abundance of different trophic groups was not significantly different.

1.6.6 Oil

Many hydrocarbons and their biotransformation products released during oil and gas operations are toxic, and can damage DNA and produce mutations or cancer. Benthic macroinvertebrates that filter sediments or feed on suspended organic matter are capable of transferring these toxins to higher trophic levels. Bendell-Young *et al.* (2000) found decreases in diversity of BMI communities in wetlands receiving effluent from oil operations in northern Canada, with dominance by Chironomidae taxa, but no evidence of mentum deformities or mutagencity.

Macroinvertebrate recruitment may also be inhibited, especially as eggs and larval organisms are typically more sensitive to polyaromatic hydrocarbons (Hoffman *et al.* 2003). Ephemeroptera, Plecoptera, and Trichoptera taxa, as well as predatory invertebrates have shown a decreasing response to hydrocarbons (Crunkilton and Duchrow 1999). Physical effects of oils can include alteration of pH and dissolved oxygen content, as well as coating of substrate (Poulton *et al.* 1998).

Additional concerns of industrial oil operations include increased road density and traffic. The impacts of these are discussed in the previous section regarding roads and bridges.

1.6.7 *Pick and shovel mine*

There are few studies which investigate the effects of gravel mining on macroinvertebrate assemblages, yet there are some general impacts from mining operations. Most mining sites progress through the following four stages: site clearing through removal of vegetation and soil, mining (often pit mining), processing (e.g. crushing, washing, and stockpiling), and reclamation. These activities may have indirect effects on aquatic ecosystems through decreased vegetation, increased road density, and increased erosion and runoff. Other environmental impacts from aggregate mines include air quality degradation from stack emissions and direct site disturbance (Blodgett 2004).

1.7 BMI Characteristics

BMI bioassessment is based on observed responses of different taxa to different stressors. The major stressors captured by BMI population analysis can be characterized by five broad categorizations: feeding preferences, nutrient tolerance, sediment preferences, riparian conditions, and temperature preferences. In addition, life history strategies (e.g. complete, incomplete, or no metamorphosis) and life cycle length (a few months to a year) can also help determine taxa response to altered habitat conditions.

1.7.1 *Feeding preferences*

The five dominant feeding strategies include shredders, scrapers, collector-gatherers, filter feeders, and predators. Shredders subside on a diet of woody debris such as leaves and twigs, and the bacteria and fungi associated with this debris. Scrapers are so-called because they scrape or 'graze' algae and detritus off the surfaces of objects such as rocks or twigs. Collector-gatherers depend on algae, detritus and bacteria stored in sediments. Filter feeders filter bacteria, algae, detritus, and animal matter directly from flowing water. Each of these four classes consumes the same primary ingredients: detritus, bacteria, algae, and fungi, but employ different methods of collection. The mode of collection is important in that it requires certain habitat characteristics, and the prevalence of one group over the others generally reflects the availability of food in the preferred form. The fifth category, predators, feed primarily on other invertebrates (Appendix 1a).

1.7.2 Nutrient tolerance

Aquatic insects can be divided into two groups regarding nutrient availability: oligotrophic, or low nutrient preference, and eutrophic, or high nutrient preference. Different species of Diptera, Ephemeroptera, and Plecoptera can prefer either low or high concentrations (Appendix 1b gives an example of some of the more common species).

1.7.3 Sediment preference

Since BMIs are bottom dwelling, their presence and abundance depends on the stream substrate. Some taxa prefer coarse substrates, such as cobbles and gravels, while others prefer fine substrates like sands and silts. Others still prefer erosional or depositional substrates such as those found with abundant aquatic vegetation (Appendix 1c). Changes in sediment transport and deposition caused by natural events or anthropogenic activities can alter the substrate habitats for BMIs and result in shifts toward certain substrate preferences.

1.7.4 Riparian conditions

Riparian vegetation controls woody debris input, temperature, and flow velocity of a stream. Heavy tree cover is preferred by many taxa, such as shredders, that immediately utilize the fallen leaves and twigs. Taxa that depend on in-stream primary production, such as scrapers or gatherers, prefer open canopy streams (Appendix 1d).

1.7.5 Temperature preferences

BMI taxa exhibit different degrees of sensitivity to water temperature. Stenothermic taxa are more sensitive; they survive in limited temperature ranges, either warm or cold. Eurythermic taxa are adaptable to a wide range to temperatures. Some taxa, such as Dytiscidae, have a range of preferences and can survive in both warm and cold environments (Appendix 1e).

1.8 BMI Response

Stream health is often characterized by its level of biodiversity (Fore and Wildrick 1998). High biodiversity usually includes various types of mayflies, stoneflies, and caddisflies (Ephemeroptera, Plecoptera, and Trichoptera respectively), while moderate biodiversity is usually characterized by a decline in stoneflies (Plecoptera) followed by declines in mayflies (Ephemeroptera) and caddisflies (Tricoptera). Sites with low biodiversity are also often dominated by soft-bodied animals such as fly larvae, oligochaetes, nematodes, leeches, and planaria, as well as amphipods. Stoneflies are generally entirely absent, while the family Baetidae (Ephemeroptera) may be present. Certain tolerant types of caddisflies may also be present.

2 Methods

2.1 Site Selection

BMI samples were collected from sites both in the northern and southern divisions of the Forest by USFS crews during 1999 and 2000 (Appendix 2). Various stream habitats were sampled, including chaparral, intermontane mixed forest, coastal forest, redwood groves and semi-arid interior shrub land. Samples were drawn in main river stems, tributaries, and coastal streams. Sampled sites were identified as either 'reference' or 'test' sites by Forest Service personnel, indicating whether there was a management activity or potential stressor at the site.

Figure 3. Reference and Test reaches on the Sespe River near Lions Camp.

Reference sites were selected to reflect high biological quality and a minimal degree of human impact. In practice, reference sites were typically located upstream of campgrounds, road crossings, cleared forest areas, landslides, infrastructure, and other sources of human disturbance. Reference sites were not necessarily pristine, but were selected to represent the least impaired reaches within the geographic range or habitat type under study (Hawkins *et al.* 2000). Test sites were selected from habitat types similar to reference sites, although test sites were located throughout disturbed zones, in areas of suspected impact, or in areas of unknown biological quality.

In 1999, sampling locations in the Forest were determined to fulfill specific objectives relating to the seven different land use activities mentioned above: campgrounds/recreation, bridges/roads, cattle grazing, fire, landslides, oil and gas operations, and pick and shovel mining. Upstream reference sites were used for many of the reaches to provide a direct, undisturbed comparison to the downstream test site. Test sites were chosen in areas of concern where the USFS either suspected impact, or wanted to monitor recovery from previous mitigation actions. For some impacts (e.g. fire and landslide), pre-impact references are not available due to the instantaneous nature of the impact. Sampling dates vary from May to December, with the majority of samples taken in summer months.

In July and August of 2000, BMI samples were collected in Los Padres National Forest as part of a larger USFS bioassessment project which involved the entire state of California (Hawkins *et al.* 2000). Of this statewide data set, eight sites were located in Los Padres and used as part of the current project.

Across both sample years, data were collected from 21 reference sites and 29 test sites. Fourteen of the test sites are unpaired, while 15 test sites feature a paired upstream reference site. Test-reference pairs allowed the bioassessment models to analyze impacts arising from known disturbances located near the test reach that do not occur near the reference reach. While reference sites do not contain the particular disturbance of interest (e.g. camping, trails, etc.), they may still be affected by other land uses in the area. Likewise, a test site may assess the cumulative impacts of all upstream land-uses. To account for this possibility, our land use assessment includes both a pair-wise comparison between test and reference sites and an analysis of variance (ANOVA) that includes all potential stressors, even those not observed by the USFS survey crew.

2.1.1 Campground / Recreation

The majority of our sample sites were chosen to assess the impacts from recreational activities in Los Padres National Forest. These activities include family and group campgrounds and day-use areas, as well as more remote and rural campgrounds. The USFS crews surveyed a total of 29 streams related to recreation including Arroyo Seco, Piney, Plaskett, Little Sur, and Willow in Monterey County, as well as Cherry, Lion, Matilija, Upper Matilija, Manzana, Piru, Reyes, and Santa Paula Creeks and Sespe and Sisquoc Rivers (Appendices 3-5).

2.1.2 Roads / Bridges

Though many of the recreational sites included impacts from roads or trails near campgrounds and day-use areas, seven of our locations were chosen to evaluate the effect of a particular road. Three North Fork Matilija sites where chosen to assess the impact of a bridge crossing downstream of Wheeler Gorge Campground in Ventura County; two Santa Lucia Creek sites are situated above and below the Indian Springs road crossing; and two Mill Creek sites are located near Nacimiento Road in Monterey County (Appendices 3-5) .

2.1.3 Cattle

In addition to road and recreational activities in Los Padres, this study examines two streams in Monterey County where cattle grazing allotments are distributed: Plaskett Creek and Prewitt Creek. In 1999, one test reach at Plaskett Creek was sampled, along with one reference and two test reaches at Prewitt Creek. An additional reference sample was collected in 2000 at the Prewitt reference reach (Appendices 3-6).

2.1.4 Fire

Sites chosen to assess the impact of specific fires include the Monterey coast Willow Creek and the Ventura County Sisar Creek. The Kirk Complex fire occurred in September 1999 near Tassajara Creek and the Zen Mountain Center. Willow Creek connects with Tassajara Creek downstream of the fire area. Reference sites are located upstream from this confluence, while the test reach is located downstream from the confluence. Sisar Creek was sampled in January 2000 following a December 1999 fire. Two samples were taken, one within the fire boundary and one outside of the burn area (Appendices 3-6).

2.1.5 Landslide

Two Sespe River sites were chosen to evaluate the impact of a large landslide. The two reaches are located near Tule Creek in Ventura County, one upstream from the landslide and the other downstream (Appendix 5).

2.1.6 Oil

The Tar Creek test reach was chosen to monitor the effects of roads and oil operations on the creek (Appendix 5).

2.1.7 Pick and shovel mine

Chorro Creek was sampled to assess the impacts of a nearby pick and shovel mine in San Luis Obispo County. One test reach was sampled downstream from the mine (Appendix 4).

2.2 Field Sampling

USFS sampling procedures call for a series of subsamples to be collected from contiguous riffles within a given reach. Riffles are characterized by broken surface tension, turbulent water, and a slight gradient, and generally have higher levels of dissolved oxygen, factors which encourage macroinvertebrate biodiversity (Rabeni 2000, Metzeling and Miller 2001). In each of the adjacent riffles, BMI subsamples were collected from the substrate and sediments using a 0.5 mm mesh, 0.1 m^2 Surber sampler to a depth of approximately ten centimeters (Figure 4). Surber subsample locations were pseudorandom, but subject to the limitations of the Surber device, such as protruding rocks, fallen vegetation, and stream depth.

Figure 4. Ben Livsey collecting a macroinvertebrate sample with a 0.5mm mesh Surber sampler.

In 1999, the individual Surber subsamples from each site were captured and packaged separately. Most sites contained three or four riffles, each sampled twice, for a total of six to eight subsamples. Changes in field crews and lack of a defined protocol during the 1999 sampling sessions resulted in an inconsistent number of subsamples from each reach, with some subsamples being omitted or composited. For this reason, 1999 sites featured different numbers of subsamples, ranging from three to eight. By contrast, the site subsamples in 2000 were not preserved separately. Instead, nine subsamples were composited into a single container in the field (Figure 5). The inconsistencies in sampling method and intensity resulted from two different research projects and different field crew standards.

The preserved samples were processed by the Utah State University National Aquatic Monitoring Center (Bug Lab). Specimens were identified to a taxonomic level consistent with laboratory protocols. Some invertebrates are classified to species level, others to genus, others to family based on morphology and the practical limitations of laboratory techniques. The resulting classifications are referred to as Operational Taxonomic Units, or simply taxa. Each sample returned a count and density of each taxon, as well as percentages of non-insect taxa, predatory taxa, and other community metrics.

Figure 5. Corey Chan preparing a macroinvertebrate sample for preservation.

2.3 Environmental Data

Along with the BMI sample, a variety of habitat parameters were also collected (Figure 6). Geographic coordinates and elevation were recorded with handheld GPS units; alkalinity was tested using field kits during the 2000 sampling season. GPS verification and alkalinity samples for the 1999 sties were conducted again in 2004 during subsequent site visits.

Figure 6. Catherine Ravenscroft collecting environmental parameters.

Other habitat parameters required for the RIVPACS model were compiled from USFS records and public databases. Unless otherwise stated, all spatial data analysis was conducted in ArcGIS 9.0 (Appendix 7).

2.3.1 Annual precipitation

Annual precipitations for test and reference sites were calculated from the Spatial Climate Analysis Service at Oregon State University, PRISM website (Daly 2004). The annual precipitation was calculated by summing the monthly average precipitation for each sample location. To estimate the actual precipitation that occurred prior to any given sampling date, an average of the four-kilometer resolution Monthly Analysis (1895-Present) PRISM Explorer over the water year was calculated. The water year extends from October 1 of the previous calendar year to September 31 of the current calendar year (e.g. the 1999 water year extends from October 1, 1998 to September 31, 1999) (Appendix 7a).

2.3.2 Wet days

The number of wet days refers to the days in the water year when rainfall exceeded 0.01 inches. ArcGIS 9.0 was used to extract grid values from the 'United States Average Monthly or Annual Days with Measurable Precipitation, 1961-90' for each site location (Daly *et al.* 2000). The Wet Days grid was produced with the PRISM modeling system, published by the Spatial Climate Analysis Service at Oregon State University, and distributed by Climate Source. Unlike annual precipitation, which measures only the previous year, wet days is an average measurement which integrates precipitation history over 30 years (Appendix 7b).

2.3.3 Stream order

We calculated the Strahler definition of stream order (1-6) for each site based on the USFS Los Padres stream layer. Strahler (1957) identifies the smallest streams in the upper reaches of a watershed as 'first order.' Two first order streams join to become a second order stream; two second order streams converge to form a third order stream, and so on. Los Padres streams analyzed in this study range from first to sixth order.

2.3.4 Hydrologic Unit Code

Hydrologic Unit Codes (HUCs) (Steeves and Nebert 1994) differentiate between regions of different hydrologic characteristics. Each unique HUC identifies the region, subregion, accounting unit, and cataloging unit of the hydrologic basin. The data were originally collected by the Geographic Information Retrieval and Analysis System, and made digital by the USGS Office of Water and Data Coordination in 1994. We used the 1:250 kilometer scale HUC for the RIVPACS habitat parameter input and extracted polygon HUCs based on our site locations (Appendix 7c).

2.3.5 Ecoregion

All of our sites are located in the United States Geological Survey (USGS) Aquatic Ecoregion 8, Southern California Mountains (Appendix 7d). Omernik (1987b) identifies aquatic Ecoregions based on "perceived patterns of combination of causal and integrative factors including land use, land surface form, potential natural vegetation, and soils."

Ecoregions are used to characterize general geographic differences in aquatic habitats. We used the digital USGS Aquatic Ecoregions of the conterminous United States layer to identify the region containing our sample sites.

2.3.6 Distance to coast

Distance to coast was calculated in ArcGIS 9.0 with the Spatial Analyst Hydro Tools 'Near' function. This tool calculates the shortest distance from each sample site to the California Coastline polygon coverage provided by the USFS.

2.3.7 Gradient

Most gradients were provided by the USFS in previous datasets. The majority of these values were calculated from topographic maps. Where gradients were lacking, ArcGIS 9.0 was used to measure the differences in horizontal and vertical distances from beginning to end of each reach polygon. This was done by clipping the USFS Los Padres stream layer to digitized endpoints of reaches, calculating the length of each stream segment, and extracting the elevation of each endpoint from 30 meter digital elevation models (DEM). The stream gradient was the elevation difference over the length.

2.4 IBI Development and Method

We employed the Southern California benthic macroinvertebrate index of biological integrity (SoCal IBI), developed by the CDFG's Aquatic Bioassessment Laboratory (Ode *et al.* 2005). This IBI was chosen specifically because it was developed for coastal streams in Southern California and 56 of the 275 sites used to construct the model were on USFS land.

2.4.1 SoCal IBI construction

The SoCal IBI was developed specifically for the semi-arid, populous southern California coastal region (Ode *et al.* 2005). This region extends from Monterey County in the north to the Mexican border in the south and inland to the eastern extent of the Southern Coast Range. This region is comprised of two Level 3 Omernik Ecoregions: Chaparral-Oak Woodland and Southern California Mountains (Omernik 1987a). The region shares a common geology, dominated by recently uplifted and poorly consolidated marine sediments; and hydrology, with average precipitation of 10-20 inches per year in the lower elevations and 20-30 inches per year in upper elevations, and even reaching 30-40 inches per year in highest elevations of some coastal watersheds (Daly *et al.* 2000).

Ode *et al.* (2005) developed the SoCal IBI using data collected from 275 sites during base flow periods between April and October of 2000-2003. Of these 275 sites, 144 were from several regional Water Quality Control Board bioassessment programs that used the California Stream Bioassessment Protocol (Harrington 1999). The USFS contributed 56 sites from streams in the National Forest using the Hawkins *et al.* (2001) targeted riffle sampling protocol. The remaining 75 sites were drawn from the USEPA's Western Environmental Monitoring and Assessment Program (Ode *et al.* 2005).

Ode *et al.* (2005) selected reference sites using GIS land use analyses at several scales, and local condition assessments (in-stream and riparian) to quantify stressors within each study reach. They then calculated the proportion of different land use classes as well as other measures of human activity upstream of each site at four different spatial scales to assess potential stressors acting on each site. Frequency histograms of land use percentages were used subjectively to establish thresholds to eliminate sites from the potential reference site pool. Sites were further examined on the basis of reach scale conditions (obvious bank instability, evidence of recent fire, grazing, etc.) and eliminated accordingly. The analysis produced 88 reference sites, leaving 206 sites as test sites (Ode *et al.* 2005).

2.4.2 Metrics

A total of 61 possible metrics were examined for use in the SoCal IBI. Each metric was screened for the following traits: sufficient range to be used in scoring, responsiveness to watershed scale and reach scale disturbance variables, and lack of correlation with other responsive metrics (Ode *et al.* 2005). Twenty-three metrics passed the first two screens (range and dose-response) and were analyzed for redundancy with Pearson Product-Moment correlation. Seven minimally correlated metrics were selected for the SoCal IBI: Coleoptera richness, EPT taxa richness (Ephemeroptera, Plecoptera and Trichoptera), predator richness, percent collectors (gatherers and filterers), percent intolerant individuals, percent non-insect taxa, and percent tolerant taxa (Table 1).

Table 1. Scoring ranges for seven component metrics of the SoCal IBI (Ode *et al.* 2005).

Metric Score	Coleoptera Taxa	EPT Taxa		Predator Taxa	% Collector Individuals		% Intolerant Individuals		% Non-Insect Taxa	% Tolerant Taxa
	All Sites	6	8	All Sites	6	8	6	8	All Sites	All Sites
10	>5	>17	>18	>12	0-59	0-39	25-100	42-100	0-8	0-4
9		16-17	17-18	12	60-63	40-46	23-24	37-41	12-9	8-5
8	5	15	16	11	64-67	47-52	21-22	32-36	13-17	12-9
7	4	13-14	14-15	10	68-71	53-58	19-20	27-31	18-21	13-16
6		12-11	13	9	72-75	59-64	16-18	23-26	22-25	17-19
5	3	10-9	12-11	8	76-80	65-70	13-15	19-22	26-29	20-22
4	2	8-7	10	7	81-84	71-76	12-10	14-18	30-34	23-25
3		6-5	9-8	6	85-88	77-82	9-7	38638	35-38	26-29
2	1	4	7	5	89-92	83-88	6-4	38512	39-42	30-33
1		3-2	6-5	4	93-96	89-94	3-1	38388	43-46	34-37
0	0	0-1	0-4	0-3	97-100	95-100	0	0-1	47-100	38-100

2.4.3 *Taxa metrics: Coleoptera, EPT, Non-insect*

Coleoptera richness is the total number of distinct taxa within the Coleoptera order. Coleoptera richness generally decreases with impairment (Ode 2003). The number of EPT taxa is calculated by summing the Ephemeroptera, Plecoptera, and Trichoptera taxa richness. EPT taxa richness generally decreases with impairment (Ode 2003). Percent non-insect taxa is measured by dividing the non-insect taxa abundance by the total sample abundance. Non-insect taxa are phyla such as Annelida and Colenterata that often increase with impairment (Ode 2003).

2.4.4 *Feeding group metrics: Predators, Collectors*

Each feeding group metric was calculated using the Bug Lab's determination of macroinvertebrate taxa feeding group membership (Vinson pers. comm.). Feeding group designations were based on Merritt and Cummins (1995). The number of predator taxa is determined by the number of macroinvertebrate taxa that feed on other organisms. Predator taxa generally decrease with impairment. Percent collectors (gatherers and filterers) is a measure of impairment to this particular functional feeding group. Collector-gathers feed on the deposited fine particulate matter while collector-filters feed on suspended fine particulate organic matter; both of these functional feeding groups increase with impairment (Vinson 2003). This metric is calculated by dividing the total number of collector individuals by the total abundance for the sample, where abundance is number of aquatic macroinvertebrates per unit area.

2.4.5 *Tolerance level metrics: Tolerant, Intolerant*

Percent tolerant taxa is calculated using Robert Wisseman's measure of tolerant taxa divided by the sample's taxa richness. Wisseman measures are regionally specific tolerance values for different taxa based on sample studies in the Pacific Northwest (ABL 2003). Each taxon is assigned a numerical value reflecting its ability to persist in disturbed environment. Tolerant taxa are ranked from zero to ten with higher scores indicating more tolerance.

Taxa richness is the number of distinct taxa found within the sample. Tolerant taxa generally increase with impairment (Vinson 2004). Percent intolerant individuals is calculated using Wisseman's measure of intolerant taxa richness divided by the sample's taxa richness. The percent of intolerant individuals generally decreases with impairment.

2.4.6 Adjustments and scoring

In initial model construction, unadjusted IBI reference scores were consistently lower in the chaparral Ecoregion than in the mountain Ecoregions. Consequently, three metrics (EPT richness, percent collectors, and percent intolerant individuals) were adjusted by creating separate scoring scales for Ecoregions 6 and 8 (Table 1) (Ode *et al.* 2005).

Since there are only seven metrics in the SoCal IBI, with a total possible score of 70, IBI scores were multiplied by 1.43 to adjust the scoring to a 100 point scale. Ode *et al.* (2005) used the statistical criteria of two standard deviations below the mean reference site score to define the boundary between 'fair' and 'poor' conditions (Ode *et al.* 2005). They determined a score of 39 to be the impairment threshold. The scoring range below 39 was divided into two equal condition categories and the range above 39 was divided into three equal condition categories: 0-19 = 'very poor,' 20-39 = 'poor,' 40-59 = 'fair,' 60-79 = 'good,' and 80-100 = 'very good' (Ode *et al.* 2005).

2.4.7 Implementation

We used the SoCal IBI to score each metric for Los Padres sites collected in 1999 and 2000. The BMI data from the 1999 USFS survey was collected as separate subsamples for each site while the 2000 USFS BMI data were combine into one composite sample in the field. Consequently, we averaged the separate subsample IBI scores for each 1999 site into one useable score. This facilitated analysis and the best consistency between the 1999 and 2000 IBI site scores. All of the stream sites are located in Ecoregion 8, Southern California Mountains, and were thus scored according to Ecoregion 8 metric scoring.

The Bug Lab at Utah State University provides two measures of BMI tolerant/intolerant assemblages: Wisseman's Intolerant measure and Hilsenhoff's Biotic Index. The

Hilsenhoff Biotic Index is based on original field correlations in Wisconsin but is of limited scope, stressors, and geography (Herbst pers. comm.). We chose Wisseman's Intolerance measure because it is based on western taxa and thus better suited for use in the SoCal IBI (Rehn pers. comm.).

Each metric for a particular stream was scored individually according to the corresponding BMI data. For example, if Prewitt Creek had a total of four coleoptera taxa, the metric 'number of coleopteran taxa' received a corresponding score of seven (Table 1). Each of the seven metrics was scored this way based on the unique BMI site data. Finally, all seven metrics were summed and multiplied by 1.43 to give a total site score between zero (highly impaired) and 100 (not impaired).

2.5 RIVPACS Development and Method

The River Invertebrate Prediction and Classification System (RIVPACS) is a family of predictive models that have been used in Britain, Australia, and more recently in the United States.

The RIVPACS model provides a means of comparing the biological community of reference and test sites (Moss 1987, Wright *et al.* 1984, Coysh *et al.* 2000). Reference sites are used to generate baseline macroinvertebrate communities to which test site communities will be compared. By comparing test sites with reference sites, a list of expected taxa is generated for each test site. This list is the macroinvertebrate community that would be expected if the site were not impaired by human activity. The expected list (***E***) is then compared with the observed taxa in the site sample (***O***). The ratio ***O/E*** expresses the degree of biological impairment (Hawkins *et al.* 2000).

The process can be casually divided into five simplified steps:

1) Cluster reference sites based on similar taxa composition.

2) Determine habitat parameters that correspond with the cluster groups.

3) Use habitat parameters of test sites to predict to which cluster they belong.

4) Predict the taxa that are expected to occur at the test sites.

5) Compare the observed sample with the expected taxa.

2.5.1 *Cluster the sites*

The macroinvertebrate assemblages in the 21 reference sites were used to generate clusters of reference sites with similar community structure (Wright *et al.* 1984). Bray-Curtis similarity analysis was used to calculate the level of pair-wise similarity among all reference sites (Equation 1):

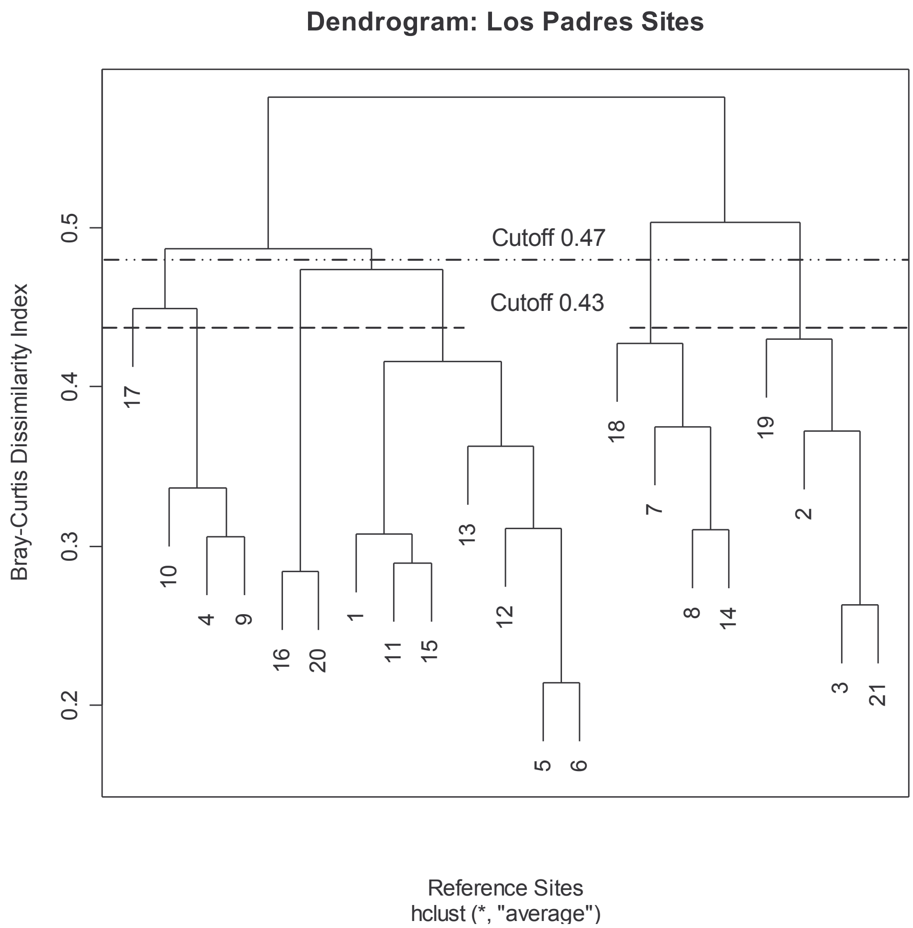
$$BC_{ij} = \sum \frac{\left| n_{ia} - n_{ja} \right|}{\left(n_{ia} + n_{ja} \right)} \qquad \text{(Eq. 1)}$$

where n_{ia} = the occurrence of taxa a in site i.

Hawkins *et al.* (2000) has used simple presence/absence information in calculating similarity indices. However, abundance and proportional abundance have also been used (Bailey 1998). Abundance values may preserve some of the nuances in the structure of communities and the differences between sites. We used both presence/absence and abundance methods for comparison and to determine which would provide the most useful clustering pattern given the inconsistencies in data collection.

Taxa that were observed in more than 95%, or less than 5%, of the reference sites were omitted from the cluster categorization. Taxa that have a high frequency of occurrence among all reference sites cause a loss in discriminatory power, and rare taxa may confound the cluster assignments by outweighing the effects of more significant differences in the BMI communities (Hawkins *et al.* 2000). Unweighted pair group arithmetic averaging (Marchant 1997, Hawkins *et al.* 2000, Coysh *et al.* 2000) was then applied to the Bray-Curtis indices to arrange the sites based on similarity. Unweighted pair group arithmetic averaging produces a classification in which the reference sites with similar taxa composition are grouped together. The distribution of the sites is represented as a dendrogram, plotting the degree of similarity along the *y*-axis (Figure 7).

Figure 7. Clustering example: site clusters determined by relative position in dendrogram.



The dendrogram was trimmed to separate reference sites clusters based on their relative similarities. The degree of dissimilarity at which the clusters are delineated can affect overall model output (Moss *et al.* 1999). We trimmed the dendrogram into clusters using two different levels of resolution for comparison. Under the first scheme, the sites were grouped into four clusters at a dissimilarity value of 0.47. A finer classification was made at 0.43, resulting in five clusters (Appendix 8a).

For each cluster of reference sites, the frequency of occurrence was calculated for each of 193 taxa. Frequency S_{aj} was calculated as the number of sites in cluster j in which taxon a was observed (Table 2). The taxa frequency is used in Step 4 to predict taxa in test sites.

Table 2. Cluster j taxa frequency.

	Presence/Absence			
	Taxon a_1	Taxon a_2	Taxon a_3	Taxon a_4
Site 1	1.00	0.00	1.00	0.00
Site 2	1.00	1.00	1.00	0.00
Site 3	1.00	1.00	1.00	0.00
Site 4	24.00	0.00	1.00	0.00
Frequency S_{aj}	0.75	0.50	1.00	0.00

2.5.2 Habitat parameters

After the reference sites were grouped into clusters, a suite of environmental variables was compared to the cluster assignments. The habitat parameters used were: annual precipitation, distance to coast, elevation, gradient, hydrologic unit code, latitude, longitude, stream order, and number of wet days (Appendix 7). Derivation of these parameters is described in Methods 2.3. Stepwise multiple discriminant analysis (MDA) was used to determine which, if any, of the habitat parameters can be used to predict or discriminate among clusters of reference sites.

Prior to running the MDA, all nine variables were checked for normality, and if the variable was not normally distributed the appropriate power transformation was found and the transformed values were used in the predictive model. Forward and backward stepwise MDA was performed on the 21 reference sites, for both the four and five cluster schemes, to select the set of environmental variables and cluster schemes which provided the most accurate prediction of cluster membership. We used two validation procedures which predicted group membership of each reference site from models constructed after removing habitat parameters one at a time from the data set. Results from the classification procedures were used to choose the discriminant analysis which had the highest percent classification. The output of MDA was a set of discriminant functions that estimates the probabilities of cluster membership of sites from environmental data.

2.5.3 Cluster test sites

The test site habitat parameters were then subjected to the discriminate function models to determine the cluster probabilities for the 29 test sites (Appendix 8b). Assigning the test sites to a single cluster may be problematic as this implies that the macroinvertebrate assemblages appear in discrete groups (Linke *et al.* 2005) and that all sites with a single cluster feature identical taxa composition. Fauna is more accurately estimated as weighted by proportional probabilities of cluster membership (Moss *et al.* 1987).

This calculation followed the procedures described by Clarke *et al.* (1996) in which the Mahalanobis squared distances (***Dj***) between a test site and each cluster are calculated (Equation 2):

$$D_j^{\,2} = \sum_{k=1}^{f} (X_k - M_{jk})^2 \qquad \text{(Eq. 2)}$$

where f is the number of discriminate functions (4), X_k is the value of discriminant function k for the site, and M_{jk} is the mean of function k for cluster j.

The probability of a site belonging to cluster j is equal to G_j (Equations 3-5):

$$G_j = M_j / M_T \qquad \text{(Eq.3)}$$

where:

$$M_j = e^{(-D_j^{\,2}/2)} \qquad \text{(Eq. 4)}$$

$$M_T = \sum_{j=1}^{f} M_j \qquad \text{(Eq. 5)}$$

2.5.4 Predict taxa

For any single site the probability (G_j) of belonging to each of the five clusters is combined with the probability of taxa frequency from Step 1 (Table 2) to determine the expected taxa that will occur at any one site. Following Step 1, if taxon a occurs in r_{aj} of the n_j reference sites in cluster j then the taxa occurrence is represented by S_{aj} (Equations 6 & 7):

$$S_{aj} = r_{aj} / n_j \qquad \text{(Eq. 6)}$$

$$Pc_a = \sum G_j S_{aj} \qquad \text{(Eq. 7)}$$

The probability of capture (Pc_a) of each taxon a at the test site is a function of the cluster probabilities and the taxa frequency (Table 3).

Table 3. Probability of capture of taxon a at Site 1.

Cluster	Cluster Probability G_j	Frequency S_{aj}	Proportional Probability G_jS_{aj}
j_1	0.700	0.750	0.525
j_2	0.100	1.000	0.100
j_3	0.100	0.500	0.050
j_4	0.100	0.000	0.000
Pc_a = Probability of Taxon a at site 1 = $\sum G_jS_{aj}$			0.675

The process is repeated for all taxa and all sites, generating a list of the probabilities of capture for each taxon at each test site. The probabilities for all taxa were summed to aggregate the total number of expected taxa (E) at the site (Table 4).

Table 4. Calculation of expected taxa (***E***) at test site.

	Pc_a
Taxon a_1	0.675
Taxon a_2	0.655
Taxon a_3	0.510
Taxon a_4	0.400
E = Total Expected Taxa = $\sum Pc_a$	2.230

The list of expected taxa may be trimmed based on a minimum probability of capture. Excluding taxa below a threshold value limits the number of taxa that are expected, reduces the value of E and weighs the metric towards more commonly occurring taxa. Hawkins *et al.* (2000) iterated the model in two different manners, first including all taxa with a probability of capture > 0, and again with a minimum probability of capture threshold of 0.5. We repeated these iterations, and used a third threshold of 0.75.

2.5.5 Compare expected with observed

The expected taxa from each iteration were then compared to the observed taxa to obtain the RIVPACS score, a ratio of observed:expected taxa. An expected taxon which is found in the sample contributes one point to the observed total (O). The observed total is the sum of all points earned by taxa in the sample (Table 5, Equation 8).

Table 5. Calculation of observed taxa (***O***).

	Pc_a	O_a
Taxon a_1	0.675	1.00
Taxon a_2	0.655	1.00
Taxon a_3	0.510	0.00
$E = \sum Pc_a$	2.230	
O = Total Observed Taxa = $\sum O_a$		2.00

$$\frac{O}{E} = \frac{2}{2.23} = 0.897 \qquad \text{(Eq. 8)}$$

Note that a taxon which is observed, but not expected, does contribute to the $\boldsymbol{O}$ summation. If a taxon is not expected, it does not appear in the table. The $\boldsymbol{O}$ calculation reflects only those taxa which are expected and observed at the site.

The reference site parameters were subjected to the same MDA and taxa classification. Model error and natural variation were estimated by comparing the number of taxa observed at reference sites to that expected from the model and generating a distribution of reference site $\boldsymbol{O/E}$ ratios. This population of $\boldsymbol{O/E}$ ratios describes the distribution of errors in predicting taxa richness and is used to determine if the $\boldsymbol{O/E}$ ratio of a test site is different from that expected from model error alone. We used the mean and standard deviation of the reference site $\boldsymbol{O/E}$ ratios to determine a 95% confidence interval for unimpaired $\boldsymbol{O/E}$ scores. Scores outside of the interval were classified as impaired. Analysis of variance (ANOVA) was used to determine whether $\boldsymbol{O/E}$ values from test sites differed from those of reference sites.

RIVPACS scores vary from zero to an upper bound determined by the $\boldsymbol{Pc_a}$ threshold used; a lower $\boldsymbol{Pc_a}$ threshold allows a higher maximum score by including more taxa. A score of one implies that the site is not impaired. A ratio that differs from one, either greater than or less than, indicates that the macroinvertebrate community differs from what would be expected in a high-quality site. In contrast to RIVPACS $\boldsymbol{O/E}$, the IBI scores follow a continuous scale. For comparative purposes, we normalized RIVPACS scores so that they fall between zero and one, and adjusted such that higher scores correspond with less impaired sites (Equation 9):

$$\mathbf{1} - \left|\mathbf{1} - \frac{\boldsymbol{O}}{\boldsymbol{E}}\right| \qquad \text{(Eq. 9)}$$

2.6 Land Use Method

We compared both the IBI and RIVPACS scores with USFS activities and land uses that were believed to affect the aquatic ecosystem. While many of our sites were strategically chosen to examine the potential impacts of a known disturbance between reference and test reaches, we also subjected all of the sites to the same land use tests to determine if any forest uses could be correlated with the scores across the entire sample space. We analyzed recreation, roads, cattle grazing, and fire history, but omitted landslides, oil and gas, and gravel mining because they were not within the direct vicinity of most reaches. Standard ANOVA tests were used to determine if any of the land use variables contributed to the variation in scores of either bioassessment model.

This section describes the processes used to prepare the multiple land-use layers for analysis. Decisions made about specific aspects of each layer were based on the current literature and data availability.

2.6.1 Fire

Fire layers were provided by the USFS dating back to 1985. We chose to restrict our analysis to fires to the past 20 years because recent studies indicate that most BMI communities recover in 10 to 15 years (Minshall 2003). We analyzed fires within three different five year periods (1985-1989, 1990-1994, and 1995-1999), and calculated the distance from each sample site to the three different fire layers. The fire year, area burned, and distance to sites from all time periods were all used in the land use ANOVA. In instances where sampling occurred prior to the 1999 fires, the second closest polygon for that layer was used (Appendix 6a).

2.6.2 Recreation

Recreation layers were provided by the USFS from three different origins. They include the following categories:

- Family Campground
- Family Picnic
- Group Campground
- Observation Site
- Trailhead
- Campground, USFS and Non-USFS
- Campsite, USFS and Non-USFS

Because some layers were of different geometries (points and polygons), we normalized them by calculating the centroid for each polygon and computed the distance from each site to the nearest upstream recreation point. The aerial extent, frequency of use, and capacity were not known for each recreation site and thus were omitted from the analysis. This information would be useful in refining the statistical analysis of land use layers.

2.6.3 Roads

Road layers for Los Padres were provided by the USFS and categorized into five road types; trail, unimproved dirt, gravel, improved light duty/paved, and secondary highway. For each site, the distance to the nearest road and road type were determined. Roads running either adjacent to, or upstream from (e.g. stream crossing) the stream were considered. Downstream roads were omitted.

2.6.4 Cattle

Sites were assessed depending on their relative location to USFS cattle allotments. GIS layers of designated allotments were provided by the USFS and contained the outer boundaries of the entire allotment area. Since particular grazing patterns within the allotments are unknown and are associated with some degree of computational error, we categorized sites into two groups: those within 75 meters of an allotment, and those not within 75 meters of an allotment. The area of the allotment was calculated using ArcGIS. Information about the distribution and intensity of use within the allotments would be very useful in refining the influence of particular cattle allotments on stream health (Appendix 6b).

2.6.5 Other

In addition to the potentially influential parameters listed above, significant effort was put toward identifying both the watershed, and the geologic setting for each sample location. Lack of complete data forced us to abandon these efforts. Future assessments may benefit by including these two parameters.

3 Results

3.1 IBI

The IBI scores were partitioned into qualitative categories based on Ode *et al.* (2005) condition ratings. Ratings ranged from poor to very good (Table 6).

Table 6. IBI condition ratings.

IBI Score	Quality Rating
0-19	Very poor
20-39	Poor
40-59	Fair
60-79	Good
80-100	Very Good

Los Padres test sites had an average IBI score of 60 with a 'good' condition rating. Reference sites had an average IBI score of 62 with a 'good' condition rating. The Forest had a total of zero sites ranked 'very poor,' one site ranked 'poor,' 20 sites ranked 'fair,' 26 sites ranked 'good,' and three sites ranked 'very good.' (Appendix 9a). The site scores were compared with the particular land use that was targeted in the assessment (Table 7) and across reference and test site pairs (see Land Use Analysis 3.3 for details). Our statistical analysis, consisting of a paired t-test and ANOVA, helped answer the question: Is land use affecting streams in Los Padres National Forest?

Table 7. Targeted land use and RIVPACS and IBI scores for each site.

Sample #	Sample Year	Stream and Site	Targeted Impact	IBI	Rating	RIVPACS	Rating
CAMPGROUND/RECREATION							
112337	1999	Santa Paula Reference	Dispersed recreation on Santa Paula Creek	52	fair	0.706	impaired
112433	1999	Santa Paula Test A	Dispersed recreation on Santa Paula Creek	52	fair	0.947	-
112439	1999	Santa Paula Test B	Dispersed recreation on Santa Paula Creek	60	good	1.029	-
112487	1999	Santa Paula Test C	Dispersed recreation on Santa Paula Creek	65	good	1.056	-
112349	1999	Manzana Reference	Nira Campground	56	fair	1.130	-
112343	1999	Manzana Test	Nira Campground	45	fair	0.936	-
112373	1999	Matilija Reference 1	Wheeler Gorge Campground	72	good	1.025	
112367	1999	Matilija Test 1	Wheeler Gorge Campground	69	good	1.356	impaired
112461	1999	Sespe at Oak Flat Ref.	Oak Flat Campground	52	fair	0.948	-
112455	1999	Sespe at Oak Flat Test	Oak Flat Campground	44	fair	0.947	-
112498	1999	Willow Reference	Willow Creek day use area	54	fair	1.016	-
112492	1999	Willow Test	Willow Creek day use area	44	fair	0.991	-
112516	1999	Arroyo Seco Reference	Arroyo Seco River recreation and Indian Road	63	good	1.040	-
112522	1999	Arroyo Seco Test	Arroyo Seco River recreation and Indian Road	49	fair	0.880	-
112325	1999	Cherry Test	Cherry Creek campground closure	50	fair	0.805	-
112391	1999	Piney Test 1	Lower Piney Campground and closed road of Piney Crk.	72	good	1.045	-
112399	1999	Piney Test 2	Lower Piney Campground and closed road of Piney Crk.	75	good	1.134	-
112403	1999	Piney Test 3	Upper Piney Campground	63	good	1.011	-
112534	1999	Plaskett Creek Test 1	Plaskett Creek Campground	64	good	1.209	-
112445	1999	Sespe at Lion Test 1	Lion Campground closure	52	fair	1.224	-
112546	1999	Sespe at Lion Test 2	Lion Campground closure	40	fair	1.033	-
112510	1999	Sisquoc Test 1	Dispersed recreation on Upper Sisquoc River	73	good	1.019	-
112504	1999	Sisquoc Test 2	Dispersed recreation on Upper Sisquoc River	51	fair	1.396	impaired
115340	2000	Lion Reference	Recreation along Lion Creek	74	good	1.115	-
115345	2000	Little Sur Reference	Pico Blanco Boy Scout Camp	77	good	1.077	-
115342	2000	Piru Reference	Recreation along Piru Creek	64	good	0.720	-
115341	2000	Reyes Reference	Recreation along Reyes Creek	61	good	0.846	-

Table 7. Continued.

Sample #	Sample Year	Stream and Site	Targeted Impact	IBI	Rating	RIVPACS	Rating
CAMPGROUND/RECREATION Continued.							
115347	2000	Sisquoc Reference	Dispersed recreation on Upper Sisquoc River	76	good	0.808	-
115343	2000	Upper Matilija Reference	Matilija Camp	77	good	1.000	-
ROADS/BRIDGES							
112361	1999	Matilija Reference 2	Bridge construction downstream of Wheeler Gorge	55	fair	1.083	-
112355	1999	Matilija Test 2	Bridge construction downstream of Wheeler Gorge	60	good	1.013	-
112385	1999	Mill Reference	Nacimiento Road sediment input	63	good	0.942	-
112379	1999	Mill Test	Nacimiento Road sediment input	58	fair	0.800	-
112427	1999	Santa Lucia Reference	Indian Road crossing	59	good	1.164	-
112421	1999	Santa Lucia Test	Indian Road crossing	64	good	1.177	-
115349	2000	Matilija Test 3	Bridge construction downstream of Wheeler Gorge	77	good	1.057	-
GRAZING							
112415	1999	Prewitt Reference 1	Cattle grazing	59	good	1.116	-
112409	1999	Prewitt Test 1	Cattle grazing	56	fair	1.103	-
112540	1999	Prewitt Test 2	Cattle grazing	38	poor	0.869	-
112528	1999	Plaskett Test 2	Cattle grazing	44	fair	0.940	-
115344	2000	Prewitt Reference 2	Cattle grazing	67	good	0.890	-
FIRE							
112552	1999	Willow Reference 1	September 1999 fire	80	very good	0.928	-
112555	1999	Willow Test 1	September 1999 fire	80	very good	0.751	-
112564	1999	Sisar Reference	December 1999 fire	71	good	0.905	-
112558	1999	Sisar Test	December 1999 fire	72	good	0.983	-
115346	2000	Willow Reference 2	September 1999 fire	84	very good	0.862	-

Table 7. Continued.

Sample #	Sample Year	Stream and Site	Targeted Impact	IBI	Rating	RIVPACS	Rating
PICK/SHOVEL MINE							
112331	1999	Chorro Test	Pick and shovel mine	64	good	1.269	impaired
OIL							
112480	1999	Tar Test	Road and oil operations	50	fair	0.622	impaired
LANDSLIDE							
112472	1999	Sespe at Tule Reference	Landslide at Tule Creek	50	fair	1.159	-
112466	1999	Sespe at Tule Test	Landslide at Tule Creek	56	fair	1.307	impaired

The first statistical test, a paired t-test, was run using the statistical program R 1.9.1. A paired t-test assesses whether the average difference between paired test and reference sites is different from zero. We could not determine with 95% confidence ($p = 0.224$) that the average difference between test and reference pairs was not zero. The paired t-test results suggest that there was likely no observable effect of land use activities on test sites within the Forest. However, this analysis was limited to those stream reaches with a test/reference pair.

The second statistical test, ANOVA, applied to all sites and analyzed whether a site score was affected by its designation as a test or reference site. The ANOVA used the dependent variable 'score' and tests whether the independent variables 'stream,' 'site,' and 'stream:site' affected the score. We termed 'stream' the individual stream reach name, 'site' 0 or 1 (coded for whether the site is test or reference), and 'stream:site' the interaction effect of stream and site. The interaction term 'stream:site' tested whether the main effect of 'stream' on 'score' is modified by its 'site' orientation. The ANOVA analyses showed that all three independent variables affected IBI scores. In this instance, the independent variables 'site' and 'stream:site' are of concern because these terms identify a site as test or reference. The p-values of 0.006 for 'site' and 0.0004 for 'stream:site' indicate that a site's test/reference status had a significant observable effect on its IBI score. We determined that forest activities at test sites (ANOVA $p = 0.006, 0.0004$) were affecting IBI scores and stream health.

That the stream:site variable appears more significant indicated that the particular stream being sampled affects the scores as well. Due to geographic differences, temperature, natural variability, or sampling uncertainty, the macroinvertebrate communities differ across streams independent of the test or reference status of the site, and presumably independent of any specific management activity.

3.2 RIVPACS

Our analysis found that the dissimilarity among BMI communities and clustering methodologies can strongly impact RIVPACS scores. For this reason we compared various techniques to determine the most applicable method for the Los Padres National Forest data.

We performed Bray-Curtis dissimilarity analysis using presence/absence and again using taxa abundance. When abundance values were used in assigning clusters, the sites sampled in 1999 were clustered separately from those sampled in 2000. This is presumably due to the different sampling methods and intensities in each year, resulting in different ranges of abundance values. The same occurs when relative abundance was used; reference site clusters reflect only the year of the sample.

When the clustering equations were calculated using binary presence/absence data, the 1999 sites and 2000 sites became intermingled to some extent. Since the cluster assignments reflected similarities in taxa occurrence rather than sampling intensity or sample year, we used the presence/absence calculations in subsequent analyses consistent with previous methodologies (Hawkins *et al.* 2000, Moss 1987).

We delineated the clusters based on two different levels of resolution, resulting in a scheme of four clusters and a scheme of five clusters (Tables 8 & 9, Figure 7). The scheme of five clusters breaks down Cluster B into two separate groups. Wright (1993) has found that a higher number of clusters yield more discriminatory power across macroinvertebrate communities, but also that clusters should contain at least five sites. Due to small sample size in the Los Padres National Forest study, we relaxed these requirements.

Table 8. Site groupings under the four cluster scheme.

Cluster A	Cluster B	Cluster C	Cluster D
Manzana	Arroyo Seco	Mill	Lion
Piru	Matilija 1	Prewitt 1	Little Sur
Sespe at Oak Flat	Matilija 2	Prewitt 2	Reyes
Sespe at Tule	Santa Lucia	Willow	Willow 2
	Santa Paula		
	Sisar		
	Sisquoc 2		
	Upper Matilija		
	Willow 1		

Table 9. Site groupings under the five cluster scheme.

Cluster A	Cluster B	Cluster C	Cluster D	Cluster E
Manzana	Sisquoc 2	Arroyo Seco	Mill	Lion
Piru	Upper Matilija 1b	Matilija 1	Prewitt 1	Little Sur
Sespe at Oak Flat		Matilija 2	Prewitt 2	Reyes
Sespe at Tule		Santa Lucia	Willow	Willow 2
		Santa Paula		
		Sisar		
		Willow 1		

3.2.1 Stepwise discriminant functions

Precipitation, transformed stream order (TSO), latitude, and longitude were all found to be significant predictors of cluster membership for both clustering schemes (Tables 10 & 11). In addition to these variables, we found elevation to be a significant predictor of cluster membership for the five cluster scheme (Table 10), and wet days to be a significant predictor of cluster membership for the four cluster classification scheme (Table 11).

Table 10. Discriminant functions for 5 clusters. Discriminant functions are used to predict cluster membership based on site specific environmental and geographic features for all 29 test sites. For each test site, four discriminant functions are used.

	Five clusters backward			
Variable	1	2	3	4
Precipitation	1.362	0.117	0.832	0.762
Stream Order	- 0.452	1.119	- 0.540	- 0.053
Latitude	- 2.059	- 1.060	6.946	- 1.688
Longitude	- 1.608	0.144	7.716	- 0.331
Elevation	1.091	0.079	- 0.603	- 0.646
	Five clusters forward			
	1	2	3	4
Precipitation	1.362	0.117	0.832	0.762
Stream Order	- 0.452	1.119	- 0.540	- 0.053
Latitude	- 2.059	- 1.060	6.946	- 1.688
Longitude	- 1.608	0.144	7.716	- 0.331
Elevation	1.091	0.079	- 0.603	- 0.646

Table 11. Discriminant functions for four clusters. Discriminant functions are used to predict cluster membership based on site specific environmental and geographic features for all 29 test sites. For each test site, three discriminant functions are used.

	Four clusters backward.		
Variable	1	2	3
Precipitation	1.721	1.204	0.052
Stream Order	- 1.132	0.507	0.473
Latitude	3.684	2.861	- 0.617
Longitude	3.336	3.828	- 1.692
Wet Days	- 1.032	- 0.876	- 1.091

	Four clusters forward.		
Variable	1	2	3
Stream Order	1.157	0.131	0.550
Longitude	1.289	0.539	- 0.608
Elevation	- 0.335	- 1.201	0.087

3.2.2 Discriminant function models classification errors

Results from the regular classification procedures indicate that cluster misclassification was 5 % for the five cluster models and 9-18% for the four cluster model. The jackknife classification indicated a misclassification percentage of 32% for the five cluster models and an error of 36% for the four cluster model. Based on this result, we chose the five cluster model for prediction of invertebrate fauna at the test sites.

3.2.3 Discriminant function models predictor variables

Variables determined to be important predictors of cluster membership were used in the discriminant models. A total of four discriminant models were derived (Table 12).

Table 12. Predictor variables identified by stepwise discriminate analysis. Variables are listed in order of their importance as measured by model F values.

Variable	F	Tolerance	DF 1	DF 2	DF 3	DF 4
Stream Order	4.84	0.495611	- 0.452	0.119	- 0.540	- 0.053
Elevation	4.47	0.482376	1.091	0.079	- 0.603	- 0.646
Precipitation	3.36	0.265309	1.362	0.117	0.832	0.762
Latitude	2.93	0.017706	- 2.059	- 1.060	6.946	- 1.688
Longitude	2.87	0.016048	- 1.608	0.144	7.716	- 0.331

The habitat parameters for all sites, both test and reference, were subjected to the discriminant models, producing the probability of cluster membership of each site for all five clusters. While all five cluster probabilities were used in the RIVPACS calculation, the sites were placed in the cluster of maximum probability ($\boldsymbol{Max\ P_{Clus}}$) for illustrative purposes (Table 13, Appendix 8).

Table 13. Cluster Assignments.

SAMPLE	SITE ID	NAME	MAX P_{Clus}	DATE	CLASS	STREAM	DISTRICT
Cluster 1							
	112349	Manzana Reference	0.87	7/15/1999	Reference	Manzana	SLRD
	112343	Manzana Test	0.87	7/14/1999	Test	Manzana	SLRD
	115342	Piru Reference	1.00	8/5/2000	Reference	Piru	MPRD
			0.98				
	112445	Sespe at Lion Test1		5/25/1999	Test	Sespe	ORD
	112546	Sespe at Lion Test2	0.96	12/2/1999	Test	Sespe	ORD
	112461	Sespe at Oak Flat Reference	0.97	8/4/1999	Reference	Sespe	ORD
	112455	Sespe at Oak Flat Test	0.99	8/5/1999	Test	Sespe	ORD
	112472	Sespe at Tule Reference	0.99	6/2/1999	Reference	Sespe	ORD
	112466	Sespe at Tule Test	0.99	6/3/1999	Test	Sespe	ORD
	112510	Sisquoc Test 1	0.84	9/16/1999	Test	Sisquoc	SLRD
	112504	Sisquoc Test 2	0.84	9/15/1999	Test	Sisquoc	SLRD
Cluster 2							
	115340	Lion Reference	0.77	8/1/2000	Reference	Lion	ORD
	115349	Matilija Test3	0.89	8/17/2000	Test	North Fork Matilija	ORD
	115343	Upper Matilija Reference	0.90	8/7/2000	Reference	North Fork Matilija	ORD
	115347	Sisquoc Reference	0.98	8/15/2000	Reference	Sisquoc	MRD
Cluster 3							
	112516	Arroyo Seco Reference	0.78	36522	Reference	Arroyo Seco	MRD
	112522	Arroyo Seco Test	0.93	36522	Test	Arroyo Seco	MRD
	112373	Matilija Reference1	0.95	7/29/1999	Reference	North Fork Matilija	ORD
	112361	Matilija Reference2	0.95	7/7/1999	Reference	North Fork Matilija	ORD
	112355	Matilija Test2	0.95	7/7/1999	Test	North Fork Matilija	ORD
	112379	Mill Test	0.56	6/17/1999	Test	Mill	MRD
	112391	Piney Test1	0.52	7/21/1999	Test	Piney	MRD
	112399	Piney Test2	0.52	7/21/1999	Test	Piney	MRD

Table 13. Continued

SAMPLE	NAME	MAX P_{CLUS}	DATE	CLASS	STREAM	DISTRICT
Cluster 3 Continued						
112403	Piney Test3	0.53	7/22/1999	Test	Piney	MRD
112427	Santa Lucia Reference	0.84	6/29/1999	Reference	Santa Lucia	MRD
112421	Santa Lucia Test	0.82	6/29/1999	Test	Santa Lucia	MRD
112337	Santa Paula Reference	0.99	9/3/1999	Reference	Santa Paula	ORD
112433	Santa Paula Test A	0.99	9/2/1999	Test	Santa Paula	ORD
112439	Santa Paula Test B	0.99	9/2/1999	Test	Santa Paula	ORD
112487	Santa Paula Test C	0.99	9/3/1999	Test	Santa Paula	ORD
112564	Sisar Reference	0.99	1/12/2000	Reference	Sisar	ORD
112558	Sisar Test	0.96	1/12/2000	Test	Sisar	ORD
112480	Tar Test	0.97	6/9/1999	Test	Tar	ORD
112552	Willow Reference1	0.86	11/5/1999	Reference	Willow	MRD
112555	Willow Test1	0.86	11/5/1999	Test	Willow	MRD
Cluster 4						
112385	Mill Reference	0.82	6/15/1999	Reference	Mill	MRD
112534	Plaskett Test1	1.00	12/14/1999	Test	Plaskett	MRD
112528	Plaskett Test2	1.00	12/14/1999	Test	Plaskett	MRD
112415	Prewitt Reference1	0.97	5/12/1999	Reference	Prewitt	MRD
115344	Prewitt Reference2	0.80	8/9/2000	Reference	Prewitt	MRD
112409	Prewitt Test1	0.96	5/11/1999	Test	Prewitt	MRD
112540	Prewitt Test2	0.98	12/16/1999	Test	Prewitt	MRD
112498	Willow Reference	0.99	5/15/1999	Reference	Willow	MRD
112492	Willow Test	0.92	5/14/1999	Test	Willow	MRD
Cluster 5						
112325	Cherry Test	0.91	7/8/1999	Test	Cherry	ORD
112331	Chorro Test	0.94	7/14/1999	Test	Chorro	SLRD
115345	Little Sur Reference	0.76	8/11/2000	Reference	Little Sur	MRD
112367	Matilija Test1	0.62	7/28/1999	Test	North Fork Matilija	ORD
115341	Reyes Reference	1.00	8/3/2000	Reference	Reyes	MPRD
115346	Willow Reference	0.49	8/13/2000	Reference	Willow	SLRD

3.2.4 Prediction of the fauna at test sites: probability of capture

Fauna prediction at test sites was used to list macroinvertebrates taxa in order of decreasing probability. The probabilities of capture ($\boldsymbol{Pc_a}$) of 193 taxa are listed for each site. Prediction of fauna at test sites, and subsequent comparison of predicted (or expected) fauna with observed fauna, was performed for three taxa classifications based on probability of capture thresholds:

1. Taxa whose predicted probability of capture at test sites was at least 50% ($\boldsymbol{Pc_a} > 0.5$).
2. Taxa whose predicted probability of capture at test sites was at least 75% ($\boldsymbol{Pc_a} > 0.75$).
3. All taxa ($\boldsymbol{Pc_a} > 0$).

The number of taxa expected to occur within the samples was predicted for all three $\boldsymbol{Pc_a}$ thresholds. Within each category, the number of taxa expected to be found was calculated as the sum of the individual probabilities of capture for each taxon at the site. We compared the expected value with the observed taxa in the field samples for all taxa whose predicted probability of occurrence was above the threshold.

Mean $\boldsymbol{O/E}$ values for each probability of capture threshold were very similar, indicating that each classification provides similar estimates of the relative number of taxa expected and observed at the site. Standard deviations of $\boldsymbol{O/E}$ were similar for each of the three classification schemes, indicating that the prediction error is similar for both rare and more common taxa (Table 14).

Table 14. Comparison of distribution of $\boldsymbol{O/E}$ using four methods.

	Reference Sites			
	$\boldsymbol{Pc_a} > 0.75$	$\boldsymbol{Pc_a} > 0.5$	$\boldsymbol{Pc_a} > 0$	Weighted
N	21	21	21	21
Min.	0.70	0.71	0.64	0.50
Max.	1.21	1.16	1.22	0.79
Mean	0.98	0.98	0.95	0.68
95% CI Upper	1.23	1.24	1.21	0.84
95% CI Lower	0.72	0.71	0.68	0.52
Standard Dev	0.13	0.14	0.14	0.08
	Test Sites			
	$\boldsymbol{Pc_a} > 0.75$	$\boldsymbol{Pc_a} > 0.5$	$\boldsymbol{Pc_a} > 0$	Weighted
N	29	29	29	29
Min.	0.61	0.662	0.76	0.41
Max.	1.24	1.39	1.61	0.73
Mean	0.92	1.03	1.06	0.61
95% CI Upper	1.22	1.39	1.48	0.79
95% CI Lower	0.62	0.68	0.65	0.43
Standard Dev	0.21	0.18	0.21	0.09

3.2.5 Prediction of fauna at test sites: weighted observed probability

In the previous cases, any observed taxa whose $\boldsymbol{Pc_a}$ was above the threshold assigns one 'point' toward the $\boldsymbol{O}$ score. In addition to the three different $\boldsymbol{Pc_a}$ thresholds, we iterated the score using weighted observed scores. Under the weighted scoring system, all taxa $\boldsymbol{Pc_a} > 0$ are included. If observed, any taxon contributes only a weighted fraction of a 'point' towards the $\boldsymbol{O_w}$ score (Table 15). The point fraction was weighted by the $\boldsymbol{Pc_a}$ of each taxon. A common taxon $\boldsymbol{Pc_a} = 0.9$ would earn 0.9 toward the $\boldsymbol{O}_w$ score if it were observed. A more rare taxon $\boldsymbol{Pc_a} = 0.1$ would earn 0.1 points (Equation 10).

Table 15. Observed taxa weighted by probability of capture.

	$\boldsymbol{Pc_a}$	O_a	O_a $\boldsymbol{Pc_a}$
Taxon a1	0.675	1.000	0.675
Taxon a2	0.655	1.000	0.655
Taxon a3	0.500	0.000	0.000
Taxon a4	0.100	1.000	0.100
$E = \sum \boldsymbol{Pc_a}$	2.230		
O_w = Total Observed Taxa = $\sum O_a \boldsymbol{Pc_a}$			1.430

$$\boldsymbol{O_w} = \frac{\boldsymbol{O}}{\boldsymbol{E}} = \frac{\mathbf{1.43}}{\mathbf{2.23}} = \mathbf{0.641} \qquad \text{(Eq. 10)}$$

The weighted scenario constrains final scores between zero and one. Weighted scores also change the influence of rare and common taxa. A rare taxon that is not found has a minor influence compared to a common taxon that is not found. While a weighted system seems to preserve some of the biological information, previous RIVPACS models have not followed a weighted scoring system (Hawkins *et al.* 2000, Coysh *et al.* 2000, Moss 1987). Intermediate thresholds ($\boldsymbol{Pc_a}$ ~ 0.5) have produced the most robust results in some cases (Hawkins *et al.* 2000). The response of rare and common taxa to disturbance may drive this phenomenon. Rare taxa may be inherently rare, independent of any response to anthropogenic impact, and their inclusion in the RIVPACS score may occlude significant changes in biotic assembly. In contrast, common taxa may exhibit more fine-scaled responses. The influence of disturbance may not be whether or not they are observed in any single case, but whether or not their $\boldsymbol{Pc_a}$ is sufficient to meet the threshold (Hawkins *et al.* 2000) and be included in the ratio at all.

3.2.6 Error in predicting the expected fauna: the distribution of reference and test site ***O/E*** *values*

Results from the ANOVA indicated that the mean ***O/E*** values for the test sites were not significantly different than mean values for the reference sites for all three probability of capture classifications ($p > 0.11$). When observed taxa were weighted by probability of capture, mean ***O/E*** values for test sites were significantly lower ($p < 0.008$) than the mean values for reference sites (Table 16). The results indicated that without weighting the observed taxa based on their probability of capture, the difference in ***O/E*** values for test and reference sites could be interpreted as an indication that test sites are not impaired, or that the impairment is not outside the range of model error.

Table 16. ANOVA results for differences in ***O/E*** for impact and reference sites.

Source	F-ratio	p-value
Pc_a > 0.75	0.527	0.471
Pc_a > 0.5	1.428	0.238
Pc_a > 0	2.613	0.113
Weighted	7.666	0.008*

* Statistically significant.

The weighted observed probability of capture scheme indicated likely impairment at the Cherry Creek Test, Mill Creek Test, Tar Creek Test, and Sisquoc Reference 2.

Under the Pc_a > 0.5 scheme, RIVPACS scores indicated likely impairment at Chorro Creek Test, Tar Creek Test, Matilija Test 2, Sespe at Tule Test, Siqsuoc Test 1, and Santa Paula Reference.

Based on the differences in weighted and unweighted scoring systems, we considered the weighted scoring system experimental only, and the Pc_a > 0.5 iteration of the model was used for subsequent analyses (Table 7). A threshold Pc_a > 0.5 has been used with success in British RIVPACS models and models developed for the California Region (Hawkins *et al.* 2000, Moss *et al.* 1987).

3.2.7 Misclassification of taxa

The calculation of RIVPACS allows determination of the degree of impairment based on taxa presence or absence. It does not, however, provide a means of assessing differences in taxa diversity among test and reference sites which can be equally instructive in illustrating the degree of impairment. There are two types of misclassification error in RIVPACS:

1. Taxon observed but not expected
2. Taxon expected but not observed

A taxon is said to be expected to occur if the probability of capture at the particular site is above 50%. Expected occurrences were compared to observed taxa occurrence for each site. We determined the degree of misclassification error by summing the number of times a taxon is expected but not observed, or observed but not expected. Taxa that were expected but not observed decrease the ***O/E*** value for the site, while taxa that were expected with low probability (50%) and were subsequently observed inflate the RIVPACS score.

On average, taxa misclassification was higher for the test sites. Chironomidae (Diptera) had the highest number of overall misclassifications (17) for the test sites while Ceratopogoniadae (Diptera) had the highest number of misclassifications for the reference sites (9). The taxa with the most Type 1 misclassifications were Empididae (Diptera), Malenka (Plecoptera), and Mariuna (Diptera), each with 16 instances of a Type 1 misclassification error, and no instances of Type 2 errors.

Type 2 errors were similar, both in terms of taxa with the highest error, and number of errors, for both test and reference sites. The highest Type 2 error was for the Ceratopogoniadae (Diptera), which had 13 misclassifications for the test sites, and seven Type 2 misclassifications for the reference sites. The finding that taxa associated with, and the frequency of Type 2 errors, were similar for both test and reference sites implies that the taxa were equally underrepresented in the model, and result in decreased ***O/E*** values. The frequency of Type 2 error occurrence was similar for both reference and test sites for all taxa, indicating that these types of errors do not invalidate comparison of the RIVPACS score between test or reference sites.

The RIVPACS bioassessment model found six sites out of the 50 total sites to be impaired based on a 95% confidence interval around the mean reference score. Sites scoring above 1.237 or below 0.713 fall outside of the range and are considered impaired (Table 7). All of the impaired sites were test sites except for the Santa Paula reference site. Four of these sites scored above the (***O/E***) two standard deviations range and two of these sites scored lower than the two standard deviations range. Among the five test sites which returned impaired scores, we identified taxa with misclassification error rates that are higher than error rates in reference sites. These taxa were classified accurately in reference sites, signaling a strong predictive capacity of the model, but were frequently misclassified in impaired sites.

Chloroperlidae and Calineuria stonefly taxa, Chironomidae midge taxa, Ampumixis and Psephenidae beetle taxa, Baetidae, Eporeus, and Ephemerellidae mayfly taxa, Cheumatopsyche, Hydroptila, Lepidostoma, and Glossosomatidae caddisfly taxa, and Antocha and Tipulidae cranefly had the greatest difference in misclassification rates. The stonefly, mayfly, caddisfly, and beetle taxa are typically known to decrease in numbers as a response to impairment, while the cranefly taxa typically increase and the midge taxa show no trend (ABL 2004). Except for the cranefly and midge taxa, the results are logical because impaired sites would typically have fewer of these taxa. That these taxa were predicted, but not observed in the impaired sites implies that these taxa decrease with human disturbance, and that their absence partially drives the RIVPACS score.

3.2.8 Compositional differences between reference and test sites

Several taxa were substantially less abundant at test sites and thus less likely to be captured at test sites than predicted. These taxa would therefore have had a strong influence on the ***O/E*** values of test sites because taxa that are expected but not observed decrease the RIVPACS score. Densities of 43 taxa were 3-500 times less abundant at test sites than reference sites. Cleptelmis (Coleoptera), Baetidae (mayflies), Chironomidae (Diptera), Brachydentridae (Trichoptera), and Rhagovelia (Heteroptera) appeared to be the most indicative of reference conditions, as these taxa are more than 30 times more abundant at reference sites. Coleoptera, mayfies, and trichoptera typically exhibit a decreasing response to impairment (ABL 2004).

Densities of an additional 17 taxa were 3-45 times more abundant at test sites. Only one taxon, Maruina (Diptera), was found to be indicative of test sites conditions and was 45 times more abundant at test sites. Increasing occurrence of this taxon would cause an increase in the RIVPACS score. This could be interpreted that Maruina presence/absence may be indicative of land use impacts, such as higher sediment load.

3.3 Targeted Land Use Analysis

To assess the impacts associated with land use activities within the Forest we analyzed the IBI and RIVPACS scores for each stream site. Each stream site was analyzed according to the single land use targeted for assessment and included the IBI score(s) and corresponding condition rating (Ode *et al.* 2005), the RIVPACS score and condition, and any conclusions that could be drawn from the bioassessment scores (Table 7).

For the 32 sites which part of a test/reference site pair, the degree of difference in test/reference scores was used to determine the likelihood of impact. A rating scheme was derived for each model by dividing the spread of the absolute difference in scores into three different impact categories.

For the IBI model, a score difference of 0-5.49 indicated that the impact is 'uncertain.' When a reference site and downstream test site differed by a margin of 5.5-11, the land use was categorized as 'possibly' causing an impact, and a difference of 11-22 was categorized as 'likely' causing an impact (Table 17).

Table 17. Impact rating based on absolute differences in scores between reference and test pair.

Impairment rating	IBI	RIVPACS
Uncertain	0.0-5.5	0.0000-0.0874
Possible	5.6-11	0.0875-0.1750
Likely	11-26	0.1760-0.3500

Only streams with a test/reference pair could provide information on the degree of impact associated with the targeted land use. When only a test site was sampled for a given stream, the absence of a paired reference site limited the land use analysis. Lone test sites could be analyzed only in terms of their condition rating and not whether the targeted land use was possibly causing an impact. The IBI impairment threshold of 39, determined by Ode *et al.* (2005), was used in this analysis.

As with the IBI scores, the RIVPACS scores for each test/reference pair were analyzed. Impact was categorized as 'likely' for sites pairs with a score difference of 0.176 to 0.35, as 'possible' for score differences from 0.0875 to 0.175, and 'uncertain' for score differences from 0 to 0.0874 (Table 17). In addition to the change between reference and test location, a 95% confidence interval around the mean ***O/E*** score was used as the impairment threshold for individual sites. Under the $\boldsymbol{Pc_a} > 0.5$ scheme, a test site was categorized as impaired if the score was above 1.24 or below 0.71 (Table 7).

The land use analysis suggested some forest activities are possibly affecting stream health, as indicated by lower IBI scores at test sites, and/or RIVPACS scores that deviate significantly from the mean (Table 18). Although this analysis was based on a limited sample size, conclusions can be drawn about forest activities and their potential impacts on aquatic ecosystems. The land uses assessed were campground/recreation, roads/bridges, landslides, cattle grazing, fires, oil operations, and pick and shovel mines.

Table 18. Likelihood of impairment based on the difference in means between test and reference site pairs.

	IBI		RIVPACS		
Sites with pairs	Absolute Difference	Impairment	Absolute Difference	Impairment	Model Agreement
Arroyo Seco	14.78	likely	0.159	possible	No
Manzana	11.20	likely	0.167	possible	No
Matilija 2	5.00	uncertain*	0.070	uncertain	Yes
Matilija 1	3.81	uncertain	0.331	likely	No
Mill	4.91	uncertain	0.142	possible	No
Prewit A	3.34	uncertain	0.013	uncertain	Yes
Prewit B	21.69	likely	0.247	likely	Yes
Santa Lucia	4.29	uncertain*	0.013	uncertain	Yes
Santa Paula A	0.71	uncertain	0.242	likely	No
Santa Paula B	7.15	possible*	0.323	likely	No
Santa Paula C	12.78	likely*	0.351	likely	Yes
Sespe at Oak Flat	8.25	possible	0.001	uncertain	No
Sespe at Tule	6.14	possible*	0.147	possible	Yes
Sisar	2.88	uncertain*	0.078	uncertain	Yes
Willow	9.53	possible	0.026	uncertain	No
Willow 1	0.00	uncertain	0.178	likely	No

* Indicates that IBI scores were higher at test site than reference site.

3.3.1 Campground / Recreation - 29

Streams chosen to assess human recreation include Arroyo Seco, Piney, Little Sur and Willow Creeks in Monterey County; Cherry, Lion, Matilija, Upper Matilija, Piru, Reyes, Sespe and Santa Paula Creeks in Ventura County; and Manzana and Sisquoc Rivers in Santa Barbara County. None of the sites assessed for recreation impacts had IBI scores below the impairment threshold of 39 established by Ode *et al.* (2005).

IBI score evaluation showed that in Monterey County, Willow Creek was possibly and Arroyo Seco likely experiencing impacts associated with recreation. RIVPACS scores suggested possible impacts at Arroyo Seco and uncertain impacts at Willow Creek Test 2. Both models suggested no impact at Little Sur River.

Score evaluation was uncertain in regards to Matilija Creek; IBI scores indicated impact was uncertain at both sites, whereas RIVPACS scores indicated impairment at Matilija Test

1. The change in RIVPACS score from upstream to downstream location reflected a possible influence of the campground.

IBI scores indicated that impact at Manzana Creek due to recreation is likely; however, RIVPACS scores indicated impact with less certainty.

The models showed anomalous results at Santa Paula Creek. For both models, the reference site scored lower than all three of the test sites. According to RIVPACS, the reference site exceeded the impairment threshold, while none of the test sites fell outside this range. All three test sites were categorized at likely different from the reference site using RIVPACS scores.

IBI scores showed the difference in scores at Test A as uncertain, Test B as possible, and Test C as likely. These likelihoods were based on the absolute difference between reference and test scores, and may indicate an unforeseen disturbance at the Santa Paula reference location.

The models were not in agreement in Santa Barbara County at Sespe Creek at Oak Flat. The IBI scores showed a possible impact; however, impact was uncertain for RIVPACS.

Several unpaired test sites were also sampled in recreation areas. In general, these sites yielded good IBI ratings. RIVPACS scores indicated impairment at Sisquoc Test 2, but not at Test 1. The IBI scores also signaled a decrease in rating at Sisquoc Test 2, rated fair, compared with Sisquoc Test 1, rated good.

3.3.2 Roads / Bridges - 7

Though many of the recreational sites include impacts from roads or trails near campgrounds and day-use areas, three of our locations were chosen to evaluate the effect of a particular road (Figure 7).

The North Fork Matilija sites were chosen to assess the impact of a bridge crossing downstream of Wheeler Gorge Campground in Ventura County; the Santa Lucia Creek

sites were situated above and below the Indian Springs road crossing in Monterey County; and the Mill Creek sites were located downslope from Nacimiento Road in Monterey County. In all three streams, our analysis was uncertain as to possible effects resulting from roads/bridges.

Figure 7. Road over North Fork Matilija near Wheeler Gorge Campground.

3.3.3 Grazing - 5

This study examined two streams in Monterey County where cattle grazing allotments are distributed: Plaskett Creek and Prewitt Creek. Both models were in agreement that cattle grazing allotments were possibly affecting stream health at certain locations. The IBI scores suggested impacts at Prewitt Test A were uncertain, while Test B was likely impacted. Prewitt Test B was the only impaired site in the Forest according to IBI scores. The metrics driving the low IBI score at Prewitt Test B were predator taxa, percent collectors and percent intolerant individuals. The Plaskett test site was not paired with a reference site, and scored a fair IBI rating. The score for the grazed Plaskett site was lower than the score for the Plaskett creek test sample taken in the campground area.

3.3.4 Fire - 5

Sites chosen specifically to assess the impact of specific fires included the Monterey County Willow Creek site and the Ventura County Sisar Creek site. The Willow Creek site was located downstream from Tassajara Creek and the Zen Mountain Center, where the Kirk Complex fire occurred in late September of 1999. Test and reference sites were taken in November, shortly after the fire. The site was re-sampled in 2000 to assess the recovery of Willow Creek. Sisar Creek was sampled in January 2000 following a fire in December of 1999. The models were not in agreement as to the effects of the 1999 fire on Willow Creek. RIVPACS scores suggested a likely impact while IBI scores indicated an uncertain effect. Both scoring systems were uncertain as to the effect of the 1999 fire on Sisar Creek sites.

3.3.5 Pick and shovel mine – 1 Chorro Creek

Chorro Creek was chosen to assess the effects of a nearby pick and shovel mine in San Luis Obispo County. The results from the IBI and RIVPACS models differed with regards to BMI assemblage health at Chorro Creek. The IBI score suggested a good condition rating, while the RIVPACS score indicated impairment.

3.3.6 Oil – 1 Tar Creek

Tar Creek was chosen to assess the effects of roads and oil operations in the Tar Creek watershed. The IBI score indicated a fair condition rating, while the RIVPACS model indicated impairment.

3.3.7 Landslide – 2 Sespe Creek at Tule Creek

Sespe Creek at Tule Creek was chosen to assess the effects of a landslide that occurred in the area. The models differed in regard to the impact of the landslide on conditions at the Tule Creek test site. The IBI score at the test site produced a fair rating, while the RIVPACS score indicated impairment. Using both models, the difference in scores from the reference to the test site indicated a possible effect from the landslide.

Of the six sites identified as impaired by the RIVPACS scores, three were in areas of suspected high-level disturbance: Tar Creek (oil operations), Sespe at Tule (landslide), and Chorro Creek (gravel mine).

3.4 Land Use ANOVA

In addition to the targeted land use results, we used an analysis of variance (ANOVA) to determine whether the RIVPACS and IBI scores were significantly affected by different land uses for all sites. The effects of number of years between sample collection and the nearest fire, grazing within 75 meters of the reach, the type and distance of the nearest road, and the distance to the nearest recreational area were considered for their effects on the IBI and RIVPACS scores (Table 19).

Table 19. ANOVA: IBI and RIVPACS scores with land use.

Land Use	IBI (p)	RIVPACS (p)
Years Between Sample and Fire	0.007*	0.650
Grazed within 75 meters	0.042*	0.785
Road Type	0.071	0.579
Distance to Road		
25 meters	0.091	0.420
50 meters	0.244	0.090
100 meters	0.570	0.935
200 meters	0.551	0.140
Distance to Recreation		
50 meters	0.524	0.866
100 meters	0.370	0.963
200 meters	0.578	0.572
300 meters	0.845	0.922
400 meters	0.898	0.657
500 meters	0.730	0.991

* Statistically significant to $p < 0.05$. IBI scores were significantly different in response to a grazing allotment within 75 meters, and for sites sampled within two years after a fire. There were no significant differences in RIVPACS scores in response to land use.

Results from ANOVA of RIVPACS and IBI scores in response to different land use factors indicated that none of the land uses explain a significant degree of variation in RIVPACS scores. IBI scores were found to be significantly affected by years between sample and fire ($p = 0.007$) and grazing allotments ($p = 0.042$). IBI scores were higher in sites which were sampled 0-1 years after a fire as compared to sites that were sampled two or more years after the fire. IBI scores were lower for sites that had a grazing allotment within 75 meters. While the relative distance to roads was not a significant influence on IBI scores ($p > 0.05$), the general trend is that roads become less significant factors with distance. There was no observed correlation between the distance to recreational areas and IBI and RIVPACS scores.

4 Discussion

4.1 IBI

The SoCal IBI has proven useful for bioassessment in Los Padres National Forest. The SoCal IBI is an applicable bioassessment model for use within the Forest because 56 of the 275 sites used for model construction were from the Forest (Ode *et al.* 2005). Results derived from the SoCal IBI need to be taken in the context of statistical limitations and sample size. General criticisms of the IBI bioassessment model are that it does not utilize all the available biological information and that it is based on circular reasoning, criticisms which are addressed below. The SoCal IBI model could be improved through advancements in reference site selection and the inclusion of regionally specific tolerance values. Constructing an IBI based on the SoCal IBI would create a more powerful bioassessment tool for Los Padres Managers.

4.1.1 Criticisms

One criticism of IBIs is that they fail to utilize all the biological information that is collected (Norris and Hawkins 2000). Most of the BMI data that is processed and provided by the Utah State University Bug Lab is not included in the IBI and ends up being an inefficient resource and monetary waste. The current contract between the United States Forest Service and the Bug Lab requires a high taxonomic resolution that hinders the possibility of potentially saving money on a simpler sample analysis. A second related criticism is that because the outputs of the IBI model are reported as single index scores, there is a loss in rich biological data (Norris and Hawkins 2000). However, IBI scores are meant to be used in concert with analysis of variation in taxa abundance and diversity among and between reference and test site pairs. In addition, the high taxonomic resolution is used implicitly in the creation of the IBI model itself.

Another criticism directed at IBIs and other multi-metric indices is that the underlying reasoning is circular (Norris and Hawkins 2000), i.e., that biologists investigate a site to determine whether it is pristine or degraded and then afterward choose metrics that suggest a sites condition as first observed (Karr and Chu 1999). However, this argument is flawed for two reasons. Firstly, test sites are compared through metrics to a regionally defined reference condition and assemblage, and not according to the biologists' own observations (Karr and Chu 1999). Secondly, scientifically sound dose-response curves depicting the biological response to human activities are utilized in model construction (Karr and Chu 1999).

4.1.2 Improvements

Reference sites were chosen by Forest Service personnel familiar with the streams used in the analysis. The subjectivity of choosing reference site locations produces a certain degree of inconsistency and irregularity across the analysis. Reference sites may not be out of range of a particular impact targeted for assessment or could have other land uses present but simply not taken into account with site selection. Consequently, some test sites may score higher than their reference counterparts. This type of discrepancy could be due to sampling error, natural variation, a particular land use causing an impact at the reference site but not at the test site, or from reference sites being affected by stressors that were not realized during site selection and are located far enough upstream that there is no associated impact at the downstream test site. Another possibility is that, if the reference and test sites are distant from each other, there could be substantial difference in hydrology and geology that account for differences. As a result of the possible reference site problems discussed here, sites which are identified as impaired and the potentially responsible land uses should be used as a guide to focus future restoration efforts. However, we advise that additional information be obtained about conditions at the site before restoration actions are undertaken.

The SoCal IBI model relies on Robert Wisseman's tolerance values. These values were developed for streams in the Pacific Northwest and although there is a large degree of overlap there are still many California taxa without tolerance values (Ode 2003). Furthermore, California has many benthic macroinvertebrate species that are either endemic to California more similar to invertebrates in the southwestern states and Mexico (Ode 2003). For these reasons, Wisseman's numbers, while they are currently the best available, are inadequate for use in California (Rehn pers. comm.). For the IBI metrics of 'percent intolerant individuals' and 'percent tolerant taxa' to become more robust, advancements in California BMI physiological data need to occur.

4.2 RIVPACS

The results of the RIVPACS bioassessment model show that this particular model is not currently a useful management tool for Los Padres National Forest. Problems with the RIVPACS model developed in this study are likely due to the small pool of reference sites or the exclusion of habitat variables (e.g. basin area, stream slope). More robust models developed from larger datasets, such as the Utah State University Bug Lab and USFS Region 5 model, may be of more use in assessing impacts in the Forest.

One criticism of the RIVPACS bioassessment technique in general is that it does not account for quantitative changes in BMI assemblages. Other problems with model construction and interpretation of results include the classification or clustering procedure, the treatment of rare taxa, accounting for spatial and temporal variation, assessing model error, and taxa misclassifications. Increasing the number of reference sites and collecting additional habitat variables will likely improve the accuracy of the RIVPACS model.

4.2.1 Quantitative changes in community assemblages

The RIVPACS model does not account for quantitative changes in community assemblages. Construction of the model does not take into account taxa abundance, instead it uses simple taxa presence and absence measurements. Changes in abundances with little or no changes in presence and absence may reflect disturbances that may not be apparent in RIVPACS scores. A method to help curtail this problem is to examine both RIVPACS scores and changes in taxa abundances in any analyses.

4.2.2 Classification (clustering)

The role of the classification procedure, in which reference sites are clustered according to the degree of similarity in their biota, is often questioned. Simple factors such as the order of data input (Podani 1997) and the number of groups being determined by visual inspection (Simpson and Norris 2000), have been shown to influence classification. Moss *et al.* (1999) examined seven different clustering strategies for the classification step and found marked differences in group sizes and predictions between the methods.

4.2.3 Spatial and temporal variation in taxa

There is a great deal of debate regarding how to treat rare taxa in the RIVPACS model. Rare taxa may reflect natural conditions at the site, indicate an impact, or may be a chance occurrence. Many studies delete rare taxa from their data sets, arguing that rare taxa contribute little to community analysis but add noise to statistical solutions (Clarke 1996, Moss 1987, Marchant 1999, Hawkins 2000). Others criticize this strategy because valuable information can be lost (Karr and Chu 1997, Cao *et al.* 1998). Cao *et al.* (1998) showed that if rare taxa are eliminated from freshwater macroinvertebrate data, then the differences in richness between sites will change to a varying extent, dependant on abundance patterns at each site. Diverse sites with many rare taxa will lose a greater percentage of their fauna compared to sites with few rare taxa. Of the 193 taxa identified in the reference sites, 68 were found only once and are considered rare. Given the small sample size, and the restricted geographic range of the study area, problems associated with rare taxa are valid for our model.

It is argued that the interpretation of an ***O/E*** score is limited, because invertebrate communities are not evaluated in the context of the site specific habitat, which may show large spatial and temporal variation (Linke *et al.* 2005). The biological potential of a site may be limited by the quality of its habitat (Barbour 1991 in Linke 2005). By classifying or grouping sites based on similarities in their biota, these models assume that macroinvertebrate communities occur in discrete groups. Yet, species assemblages are commonly described as changing continuously (Vannote *et al.* 1980, Gauch 1982, Linke *et al.* 2005). This type of problem was minimized in the construction of this model, by calculating the probability of each taxon belonging to any one site as a sum of the individual probabilities of cluster membership (Clarke *et al.* 1996). Despite this effort it is expected that some uncertainty is still present in the model as a result of this assumption.

Although the RIVPACS model is designed to integrate changes in taxa occurrence through space and time, there still may be some degree of error due to temporal variation of macroinvertebrate communities and its consequence on model output. Natural stochastic events such as droughts and floods can result in marked changes in the value of ***O/E***. To determine the variation in taxa abundance and frequency, it is necessary to investigate the changes in taxa with regard to space and time, and how this type of variation influences the calculation of expected taxa. Studies indicate that the sampling season is of particular importance because macroinvertebrate phenology, distribution, and abundances vary widely between seasons (Furse *et al.* 1984); however, sampling date did not emerge as a relevant predictor variable in our analysis.

4.2.4 Model error assessment

Model error is generally evaluated by calculating the percentage of sites that are assigned to the correct cluster groups. It has become standard to evaluate the model using the reference sites that were employed to build the model (Moss *et al.* 1987, Clarke 1996). This method of model assessment is argued to be valid, as studies which tested model accuracy using evaluation reference sites not included in model construction found little difference in the distribution of ***O/E*** values between evaluation reference sites and modeled reference sites. The fact that this assumption is validated by results from previous studies does not preclude the need for independent evaluation reference sites in others. Ideally, such models should be tested with independent data (Fielding *et al.* 1997).

Regardless of which sites are used for model evaluation, the method of model assessment is also problematic. There are indications that the prediction accuracy might be affected by the frequency or prevalence of the test organism(s) being modeled (Fielding *et al.* 1997, Manel *et al.* 1999). The prevalence of a species is the ratio of observed occurrence over the number of streams in the survey. Prediction success may be misleading as it does not take into account prevalence effects. Manel *et al.* (2001) found that the overall success at predicting presence and absence was higher with more prevalent species and lower with rare species.

4.2.5 *Taxa misclassifications*

Taxa misclassification errors occur when a taxon at a site is observed but not expected, or when it is expected but not observed. These are respectively known as Type 1 and Type 2 errors. Examination of the misclassified taxa does not show any noticeable trends. At both test and reference sites, Type 1 and Type 2 errors occur. No trends were observed because the misclassified taxa are typically known to exhibit both increasing (Ceratopogoniadae, Tipulidae, Empididae, Maruina) and decreasing (Epeorus, Perlidae, Malenka, Perlodidae) occurrence as a response to impairment in both test and reference sites(ABL 2004).

4.2.6 *Improvements*

RIVPACS is a data intensive model that requires a large number of sites to construct. Increasing the number of sites is the most important improvement that can be made to our pilot model because the number of sites factors into the early steps in model construction and affects all subsequent results. The number of sites available for our analysis presented a problem that violated some general guidelines of RIVPACS construction. In this project, this problem was addressed in a number of ways. For example, it is suggested that there be at least five reference sites in each cluster. Based on this guideline, our reference site dendrogram could have been divided into only two clusters, which is insufficient for showing distinguishable characteristics for different groups of sites. We decided to use five clusters for the sake of the exercise of RIVPACS construction although there were less than five reference sites in each cluster in this scenario. Misclassification errors have been shown to be higher for smaller cluster groupings. Although violation of the cluster size rule was necessary to build the model, it negatively affected the accuracy of classification of test sites, and reduced our confidence in the results that the model generated.

Inclusion of more habitat variables in our analysis would involve refining the data collection process. More habitat variables in RIVPACS construction could strengthen the model because there are possibly other habitat factors that affect the community BMI assemblages such as riparian habitat area, percent stream shade cover, and sediment classifications. Hawkins *et al.* (2000) found basin area and stream length to be significant predictor variables that were not included in our pilot model due to data limitations. Effort should be made to collect a larger number of samples, and to collect the same list of habitat variables at all sites.

4.3 Comparison of Models

There are two main factors which reduce the confidence that we place on the IBI and RIVPACS scores, and therefore limit our ability to assess the degree of agreement between the two models. First, both models are based on data collected using inconsistent sampling methodology. Second, the lack of confidence in the accuracy of RIVPACS scores as a result of the inadequate number of reference sites makes the validity of comparison questionable.

An underlying problem in using both models is the sampling methods that were employed in 1999 and 2000 used inconsistent methods. For instance, different numbers of subsamples were taken in certain areas, resulting in higher intensity data collection at some sites. In our analysis we assumed that the data was collected using the same techniques because there was no way to standardize the varying methods. Using standard sampling protocol ensures that any sampling bias related to a particular survey method will be the same for both reference and test sites.

As discussed above in the RIVPACS discussion section (4.2), adjustments and exceptions to RIVPACS construction guidelines were made because of the limited amount of data available. This decreases the confidence that we can place in use of the RIVPACS model. In contrast, the IBI bioassessment model that we used was based on the previously established SoCal IBI developed by Ode *et al.* (2005), which is not dependent on the number of sites or amount of data available and we can thus view the results of the IBI model with more confidence.

Our results showed that reference and test site scores were not significantly distinguishable for the RIVPACS model, but for the IBI model reference site scores were significantly higher than test site scores. The difference in results between the IBI and RIVPACS scores, however, is not necessarily indicative of a better model; alternate explanations are possible. One possibility is that the IBI model is not working correctly but RIVPACS is actually more accurate, and there are no significant impacts at the test sites. Another possibility is that the IBI model is more sensitive than the RIVPACS model to the impacts at the test sites. This could indicate that RIVPACS is simply insensitive to particular impact types, or that the impacts are within the range of model error. The first hypothesis may be true, but data limitations inherent with the RIVPACS model make the second hypothesis more plausible.

Comparisons of RIVPACS and IBI scores showed that sites listed as impaired under the RIVPACS model did not match well with the lower IBI scores. The IBI model had 23 sites classified in each of the fair (40-59) and good (60-79) ranges and one site in the poor range (20-39). Of the six sites which were impaired based on RIVPACS scores, four fell under the fair range and two fell under the good range of IBI scores. Prewitt Test 2, the site in the poor range and listed by the IBI as the most degraded at 37, was not identified as an impaired site by the RIVPACS model.

Calculation of covariance did not reveal any significant trend between the IBI and adjusted RIVPACS scores ($R^2 = 0.005$). The lack of correlation reinforces that the two different scoring systems rely on different metrics, and that the RIVPACS pilot scores are founded on incomplete and suspect data. The IBI is based on a series of defined metrics, which incorporate a limited number of significant taxa. RIVPACS, by contrast, can be sensitive to any number of taxa. IBI uses a ranked system of scores based on percent composition, relative abundance, and trophic features, while RIVPACS used only presence/absence information. Each scoring system is not sensitive to the same metrics, and the correlation test shows that the metrics are not sensitive to the same impacts. Additionally, the small sample size and variable sampling protocol used in generating RIVPACS weakens statistical inferences that can be drawn.

Comparison of the two scores also reveals another facet that favors the IBI over the RIVPACS model. IBI scores are inherently easier to prioritize than RIVPACS scores because of their 0-100 ranking system, while once a RIVPACS score is identified as impaired, the actual level of impact is not as easily assessed.

As mentioned in the RIVPACS discussion section (4.2), managers should examine abundance trends when implementing either the IBI or RIVPACS models. IBI scores utilize taxa abundances in four of its seven metrics while the RIVPACS model does not account for abundance at all. If changes in abundance are indicative of impairment, we would expect taxa negatively affected by the impairment to decrease in abundance and taxa positively affected by the impairment to increase (at test sites relative to reference sites). Evaluation of abundance data did not show any trends. Of the five taxa listed as much more abundant in reference sites compared to test sites, some typically exhibit increases (Chironomidae), some decrease (Brachydentridae, Baetidae) in numbers, while some exhibit variable or unknown changes (Cleptelmis, Rhagovelia) in response to impairment (ABL 2004). The Maruina taxon was 45 times more abundant in test sites than reference sites, and it typically decreases in numbers in response to impairment, according to entomological literature (ABL 2004). This goes against what we would expect for this taxon, because it should typically be more abundant in less impaired sites.

4.4 Land Use

Results from the ANOVA indicated that grazing within 75 meters of the site and years between sample and fire were the only two land uses that showed a significant influence on IBI scores. No land use type caused a significant difference in RIVPACS scores. IBI scores were significantly lower at sites within 75 meters of grazing allotments, suggesting that grazing plays a large role in riparian and stream habitat and the resultant BMI assemblages. IBI scores were significantly higher in sites which were sampled one year or less following a fire compared to sites that were sampled two or more years after a fire. This is interesting to note because responses of BMI assemblages to fire are known to be highly variable (Spencer and Kingsley 1991). Depending on the particular fire, the rates of sedimentation, water temperature changes, and amount of leaf litter or detritus produced will vary and correspond with differing effects on BMI assemblages. That the IBI scores are lower after two years suggests that compositional changes due to fire effects still take place after two to three years, and that fire effects may not be unidirectional or consistent.

The conclusions drawn from this land use analysis should be taken in the context of sample size, site location and bioassessment score. Improvements in land use analyses would involve collection of more in-depth data such as frequency of use and visitation to recreational areas and campgrounds and intensity of grazing in cattle allotments as discussed in the land use methods section. These data would allow for more accurate calculations.

4.5 Management Implications

Forest managers can use bioassessment models to assess impacts and to track the progress of restoration efforts over time. We cannot presently recommend the use of a Forest specific RIVPACS bioassessment model for Los Padres National Forest because of data limitations that restrict model development. On the other hand, we do recommend the use of the SoCal IBI model for streams within the Forest.

With the required biological and habitat data, the IBI bioassessment model can be used to formulate scores for any stream site within Los Padres National Forest. These scores can then be used to examine and compare impact levels between individual sites, the same sites over time, or to evaluate differences between groups of sites. The IBI scores can provide indications of improvement, degradation, or no change in site quality. The scores do not inherently identify the impact or source of impact. By using additional knowledge of the site and surrounding areas, as well as the BMI response, the source of the can be identified with a reasonably high confidence level. In this study, we identified grazing and variable response to fire through time as two influential factors on aquatic ecosystem quality. Managing these impacts should help prioritize sites on which to focus improvements.

Managers need to maintain an adequate balance of Type 1 and Type 2 management errors when interpreting IBI scores and setting restoration priorities. A Type 1 management error occurs when a test site scores significantly lower than its reference counterpart when in fact no impairment exists. This represents a case in which forest managers may take restoration action when none is warranted (*Overaction* error). In this case, time, money, and effort may be devoted to restoring a stream where no impact has occurred.

A Type 2 management occurs when one believes the null hypothesis to be true when in fact there is a negative effect. No action is taken, although the site is threatened (*Inaction* error). Type 2 errors can cause impaired streams to be neglected and become more impaired over time. In statistical analyses, ecologists generally fix the probability of Type 1 error (α) at 5% (Downes *et al.* 2002). The probability of a Type 2 error (β) is inversely related to the probability of a Type 1 error (α), thus any reduction in α will only increase β and vice versa (Downes *et al.* 2002).

Managers must realize and assess the balance between *Overaction* and *Inaction* errors. One way to decrease the probability of both types of errors is to increase the sample size. Because bioassessment is a relatively inexpensive way to assess impacts, managers should consider using multiple test sites on a stream reach to minimize the probability of both types of errors from occurring.

An adaptive approach should be taken when determining α. Here, two scenarios are presented in which different values for α would be applicable. First, a manager is deciding whether to close a popular campground near a stream. If the campground closes, social, political, and economic repercussions may arise. In this situation, setting a low α will imply that *Overaction* errors are more important than *Inaction* errors. The manager is unlikely to falsely declare an impact but may miss an impact because of a higher β error rate. The manager determined that the social, political and economic cost of an *Overaction* error (closing the campground when no impact existed), is too high. In the second scenario, a manager is in charge of determining if a cattle grazing allotment is impacting an andromous stream containing southern steelhead. In this type of scenario, it is more important to minimize the probability of an *Inaction* error. An erroneous conclusion that the site is unimpaired carries a high cost because the stream contains an endangered species.

Managers must also be made aware of the sampling problems that are present in the data we used in our analysis, as discussed above. Regardless of the issues surrounding the use of non-standardized collection techniques, the use of the same sampling methodologies can still cause problems. For example, Mackey *et al.* (1984) found differences in taxon yield and community composition between operators sampling at the same site. Their findings may reflect variation in the physical effort exerted by the operators, although individual variation in technique and success of sampling in all available habitats may also be relevant. A study by Clarke (2002), which investigated variation in sample values among biologists following a standard operating procedure, found that inter-operator influence on sample values were negligible. This finding emphasizes the importance of having standard procedures and adequate training for long term bio-monitoring. Sampling variation may be a cause of errors in observed fauna. However, it is time and cost intensive to take and identify replicate samples at each site and time. Therefore, we recommend that the Forest Service have an estimate of the expected size of sampling variance obtained for a separate extensive replicated sampling study.

4.6 Data Recommendations

To fine tune the data collection process, we make the following recommendations:

Composite field sampling

Although some data are lost when subsamples are composited into the one field sample, the benefits of this technique may outweigh the losses. First, both the SoCal IBI and RIVPACS have been developed in the past using composite subsamples. The sampling method used in implementing the models should be consistent with that used in developing the model. Composite sampling also yields a cost savings as only one sample must be identified by the lab rather than eight smaller subsamples. The 2000 and 2004 data were collected using composite field samples, while the separate 1999 subsamples were composited *post hoc* via computer for use in this project. To maintain consistency in future sampling periods, field composited sampling is favorable.

Reference site locations

Some reference sites were located upstream of a designated impact, but were still within range of another impact. For example, a reference site may be placed upstream of a road crossing, but it remains within the larger campground impact area. While this does not affect the implementation of IBI, reference sites such as this may cloud RIVPACS development as the model does not account for different degrees of impairment from different forest activities. If reference sites are to be used in model development, it should be ensured that sites are located distant enough from all impacts as to be undisturbed.

Furthermore, the Santa Paula reference site scores lower than three test sites according to both models. Investigation of the reference site location should seek to identify any unexpected disturbances, or any activities which may be improving conditions at the test site locations.

4.7 Management Recommendations

Our evaluation of the scores and land use data have contributed to the following recommendations for management activity:

IBI over RIVPACS

Based on current model development and data availability, we have found the existing SoCal IBI to be advantageous over the pilot RIVPACS model. The existing IBI model used a larger sample size in development, and appears to be useful in determining some impacts in Los Padres. In the future, a RIVPACS model for Los Padres may yield additional fruitful information, but until a critical mass of reference sites is reached, the model is error-ridden. Existing RIVPACS models for Region 5 of the USFS have recently been developed using a larger sample size (WCMAFE 2004), and output from these may be more meaningful. However, the model is not tailored to central and southern California.

Grazing allotments

Based on our site analyses and ANOVA, we found that grazing may negatively affect the aquatic ecosystem, but it appears the effect is small and isolated. Consequently, we recommend that grazing allotments be kept away from threatened or endangered species habitat or sensitive areas. Otherwise, grazing patterns should be monitored along with other impact zones.

Fire-recovery monitoring

Fire is one of the largest impacts in Los Padres. Fire history has shown a statistically significant influence on variance in IBI scores ($p = 0.007$), but macroinvertebrate response to fire is inconsistent through both space and time (Minshall 2003). Our results show a higher quality rating shortly after a fire (0-1 years) compared with longer time intervals. Due to inconsistencies in BMI response, bioassessment appears to fall short in evaluating rehabilitation of burned areas. A longitudinal data set is needed, documenting the same sites at different periods after fires, to determine if any trends in fire-recovery become apparent. Samples acquired in 2004 can aid in this effort.

Lion Campground closure at Sespe River

The Sespe River at Lion Campground shows a fair quality level and minimal impact. The campground has been closed in an effort to protect the endangered arroyo toad (*Bufo californicus*) found at the site. The scores suggest that the aquatic ecosystem is at an intermediate condition. We recommend further monitoring and calculation of 2004 scores to determine if the campground closure corresponds with higher scores through time.

4.8 Future Work

The IBI bioassessment model is an applicable tool for managing Los Padres National Forest. Future work should be concentrated on strengthening the IBI model. We also believe that the RIVPACS model could be a valuable addition to the IBI model. Since the models incorporate different aspects of the community structure, use of both models could ensure that impairments which fall outside the range of detection of one model will still be identified. Future work should include the collection of more environmental data at all sites and adding additional references sites for RIVPACS model construction and evaluation.

The construction of an IBI model that is more narrowly confined to Los Padres National Forest rather than the central and southern California coast would strengthen the accuracy and interpretation of the IBI results. The Los Padres IBI could draw on work by Ode *et al*. (2005) and use existing and new BMI data gathered from Los Padres water quality surveys in model construction. The resulting model would be unique to Los Padres National Forest and would more adequately address the wide-range of environments present in the Forest. Similar to the SoCal IBI, which has metric scores for each of the two Ecoregions in southern California, the Los Padres IBI could have separate metric scores for each of its ranger districts. IBI construction is data intensive, and requires a large number of reference sites. Until a large body of sites is sampled and evaluated, the existing SoCal IBI appears to be an adequate set of metrics for assessing certain impacts in Los Padres.

Although we cannot presently recommend the RIVPACS model, it has the potential to be an appropriate and instructive model for use in the Los Padres Forest. Without incorporating additional sites into construction of the RIVPACS model, a proper assessment of the model cannot be done. New data collected in the summer of 2004 is currently being processed and could be incorporated into the current data set in an effort to improve the RIVPACS model as well as allow for comparisons of temporal variation in site scores over a five year period. Future sampling will be necessary for continued monitoring of stream conditions using the IBI. Only a few more measurements would be necessary to improve the RIVPACS model and the additional information gained from the use of RIVPACS model would be very beneficial to monitoring stream conditions in Los Padres.

The current contract between Utah State University National Aquatic Monitoring Center and USFS is to process the field samples that are collected and to provide a RIVPACS model. Unfortunately, we were not able gain access to the results or formulas used in the construction of this model due to disclosure and proprietary issues. However, an automated version of the model has recently become available for use (March 2005). The model requires additional habitat data that is not available for all Los Padres sites, but the information could potentially be captured without further site visits. Comparison and analysis of the Monitoring Center's RIVPACS model to the one we constructed, or future iterations of this model, would be beneficial and could provide further insight into the usefulness of the RIVPACS model in management of Los Padres National Forest.

The USFS has expressed a strong interest in a future project that builds upon work that we have done and examines streams for habitat suitability for the endangered southern steelhead (*Oncorhynchus mykiss iredeus*). This may be applicable to a future Bren school group project or work within the USFS.

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Appendices

1 Macroinvertebrate Habitat Preferences

Appendix 1a. Feeding preferences for macroinvertebrate taxa.

Shredders: Invertebrates that eat leaves, twigs and the bacteria and fungi that grow on this woody debris.	*Diptera*: Tipulidae *Ephemeroptera*: Ephemerellidae *Plecoptera*: Capnidae, Leuctridae, Nemouridae, Peltoperlidae, Pteronarcyidae *Trichoptera*: Lepidostomatidae, Limnephilidae, Sericostomatidae
Scrapers: Invertebrates that scrape or 'graze' attached algae and detritus from the surface of rocks, twigs, and other objects.	*Coleoptera*: Psephenidae *Diptera*: Blephariceridae *Ephemeroptera*: Heptageniidae *Plecoptera*: Perlodidae, Chloroperlidae, Trichoptera: Calamoceratidae, Glossosomatidae, Leptoceridae, Uenoidae
Collector-gatherers: Invertebrates that feed primarily on algae, detritus, and bacteria deposited on sediments in slow water areas.	*Ephemeroptera:* Baetidae, Caenidae, Ephemerellidae, Ephemeridae, Tricorythidae *Plecoptera:* Nemouridae, *Trichoptera:* Brachycentridae, Leptoceridae *Diptera:* Chironomidae, Psychodidae, Empididae *Coleoptera:* Elmidae
Filter feeders: Invertebrates that filter bacteria, algae, detritus, and animal matter from the water column.	*Trichoptera:* Philopotamidae, Polycentropodidae, Hydropsychidae *Diptera:* Simuliidae
Predators: Invertebrates that feed primarily on other invertebrates.	*Coleoptera:* Dytiscidae *Diptera:* Ceratopogonidae, Dolichopodidae, Muscidae, Tabanidae *Hemiptera:* all *Megaloptera:* all *Odonata:* Anisoptera, Zygoptera *Plecoptera:* Chloroperlidae, Perlidae, Perlodidae *Trichoptera:* Rhyacophilidae

Appendix 1b. Nutrient tolerances for macroinvertebrate taxa.

Oligotrophic: Waters characterized by low nutrient concentrations.	*Diptera:* Blephariceridae *Ephemeroptera:* Ephemerellidae, Heptageniidae *Plecoptera:* Chloroperlidae, Nemouridae, Peltoperlidae, Perlidae, Perlodidae, Pteronarcyidae *Trichoptera:* Rhyacophilidae, Glossosomatidae
Eutrophic: Waters characterized by higher nutrient concentrations.	*Diptera:* Muscidae, Simuliidae, Syrphidae *Ephemeroptera:* Baetidae, Tricorythidae *Trichoptera:* Hydropsychidae, Brachycentridae,

Appendix 1c. Sediment preferences for macroinvertebrate taxa.

Coarse: Substrates like gravel and cobbles	*Diptera:* Blephariceridae, Chironomidae, Deuterophlebiidae, Simuliidae *Ephemeroptera:* Ephemerellidae, Heptageniidae *Plecoptera:* Chloroperlidae, Perlidae, Perlodidae *Trichoptera:* Glossosomatidae, Hydropsychidae, Philopotamidae, Rhyacophilidae
Fine: Substrates like sands and silts	*Oligochaeta* *Coleoptera:* Notonectidae *Diptera:* Chironomidae *Ephemeroptera:* Caenidae, Ephemeridae, Tricorythidae *Odonata:* Gomphidae *Trichoptera:* Lepidostomatidae
Erosional or depositional: Substrates with abundant aquatic vegetation	*Coleoptera:* Dytiscidae, Gyrinidae, Haliplidae, *Hemiptera:* Belostomatidae, Corixidae *Odonata:* Coenagrionidae, Libellulidae *Trichoptera:* Leptoceridae

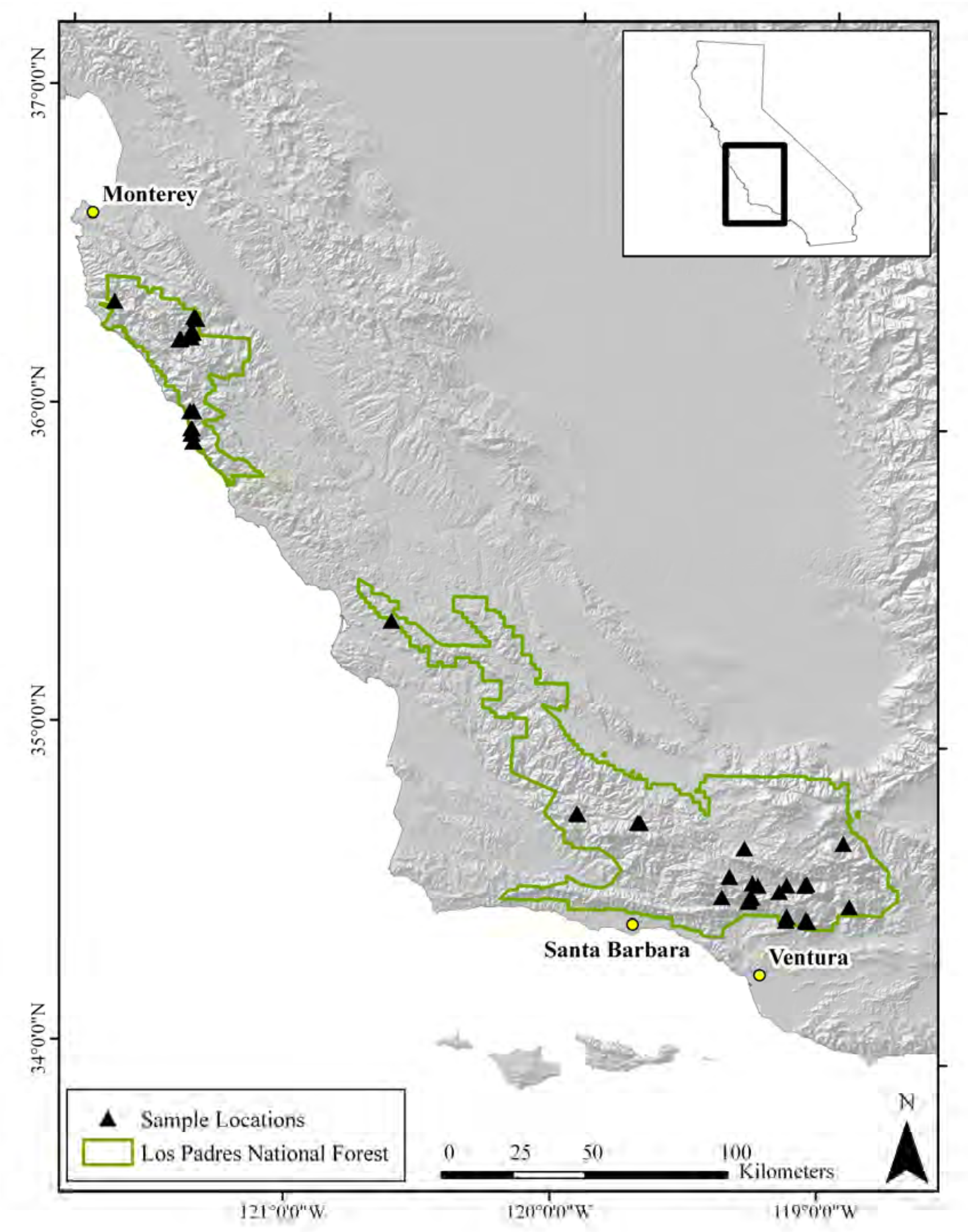
Appendix 1d. Riparian conditions for macroinvertebrate taxa.

Shaded streams: Streams shaded by dense riparian vegetation provide a lot of food in the form of leaves and wood (allochthonous material) that can be used by shredder invertebrates.	*Coleoptera:* Elmidae *Plecoptera:* Peltoperlidae, Pteronaricyidae *Trichoptera:* Calcamoceratidae, Lepodostomidae, Limnephilidae
Open canopy streams: Streams without heavy shading generally have more in-stream primary production than shaded streams and are dominated by bugs that prefer to eat algae or filter fine organic matter (autochthonous matter) from the water column.	*Ephemeroptera:* Baetidae *Trichoptera:* Hydropsychidae

Appendix 1e. Temperature preferences for macroinvertebrate taxa.

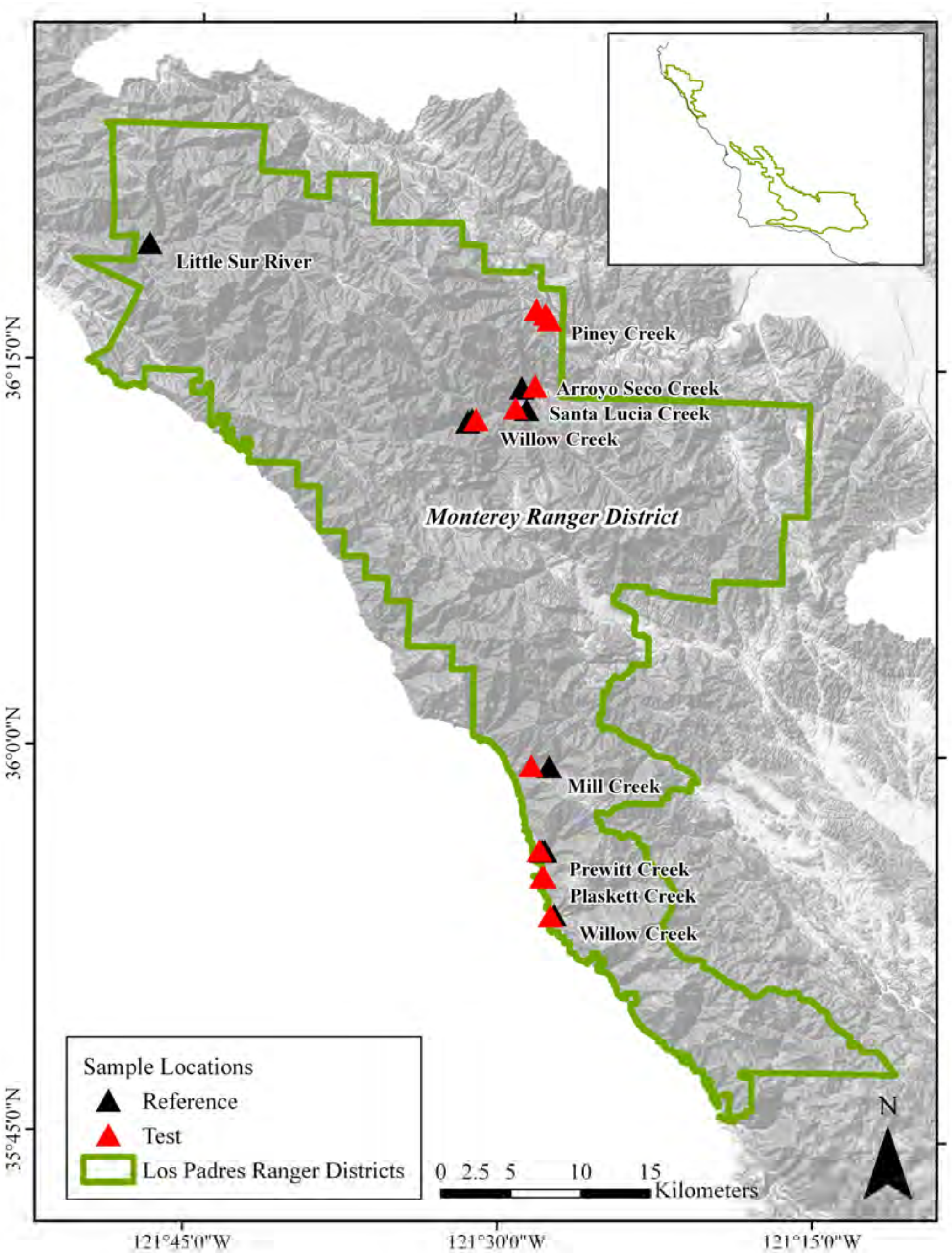
Warm stenothermic: Waters generally always warm. May include warm springs.	*Coleoptera:* Dytiscidae, Hydrophilidae *Diptera:* Ephydridae, Stratiomyidae *Hemipterans:* Saldidae, Corixidae *Odonata:* Coenagrionidae, Libellulidae
Cold stenothermic: Waters generally always cold. May include cold springs and high elevation streams and ponds.	*Coleoptera:* Dytiscidae *Diptera:* Chironomidae *Ephemeroptera:* Baetidae, Ephemerellidae *Plecoptera:* Leuctridae, Nemouridae, Perlidae *Trichoptera:* Lepostomatidae, Limnephilidae, Rhyacophilidae
Eurythermal: Waters that vary from cold to warm throughout the year. The majority of streams and ponds fit this category.	Many

2 Sample Locations



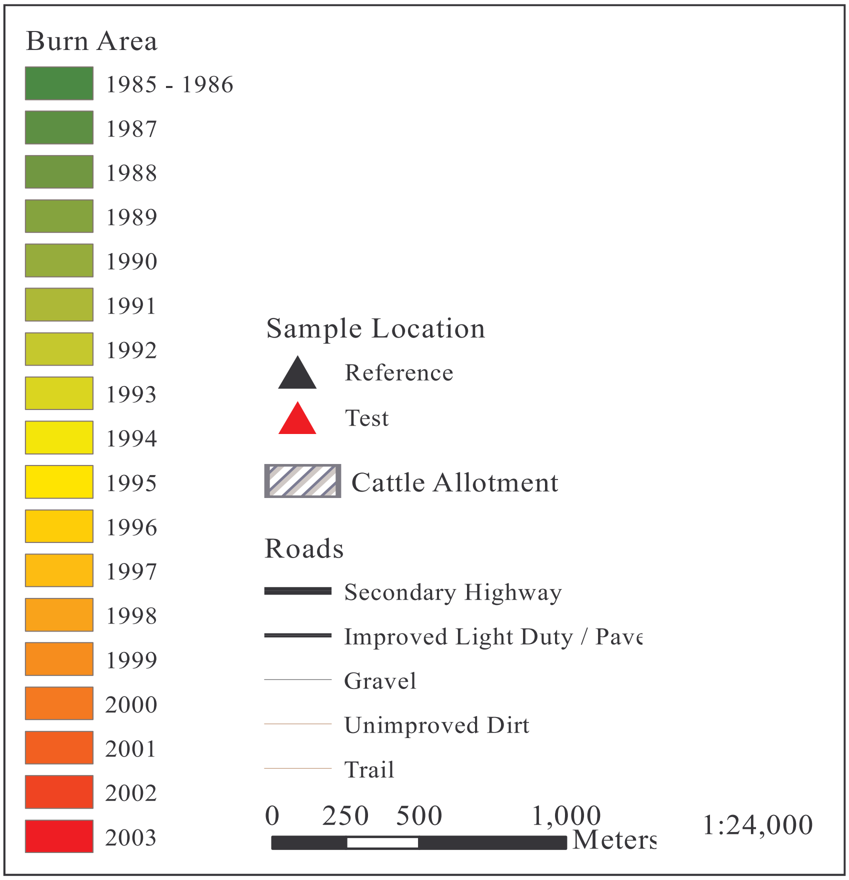
Appendix 2. Los Padres National Forest in Southern California. Locations of benthic macroinvertebrate samples are indicated by the black triangles. Total sample count is 50.

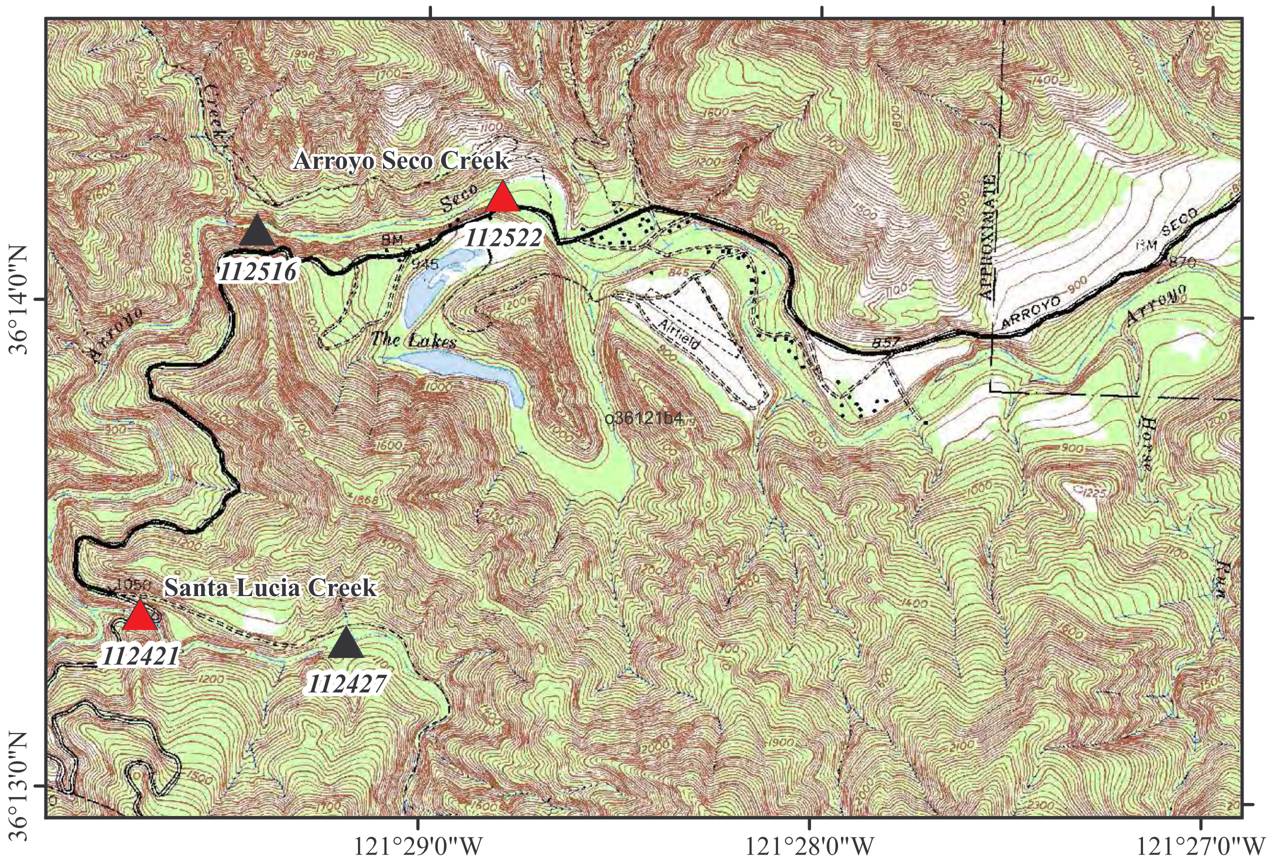
3 Monterey Ranger District Samples



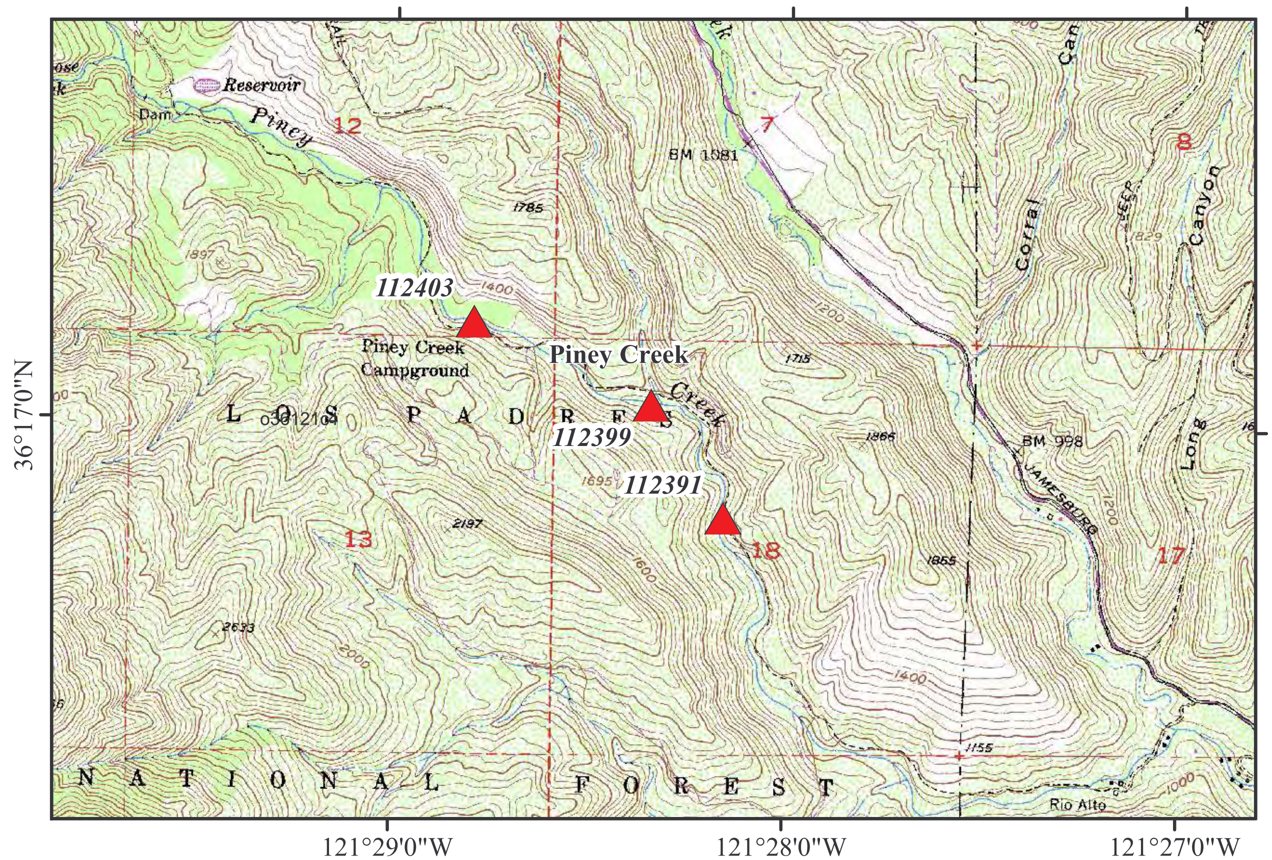
Appendix 3a. The Monterey Ranger District is the northern most ranger district in the Los Padres National Forest. There are a total of twenty-one sample sites in this district, nine are reference, and twelve are test.

Map legend for topographic maps showing sample locates and landuses (Appendices 3-5).

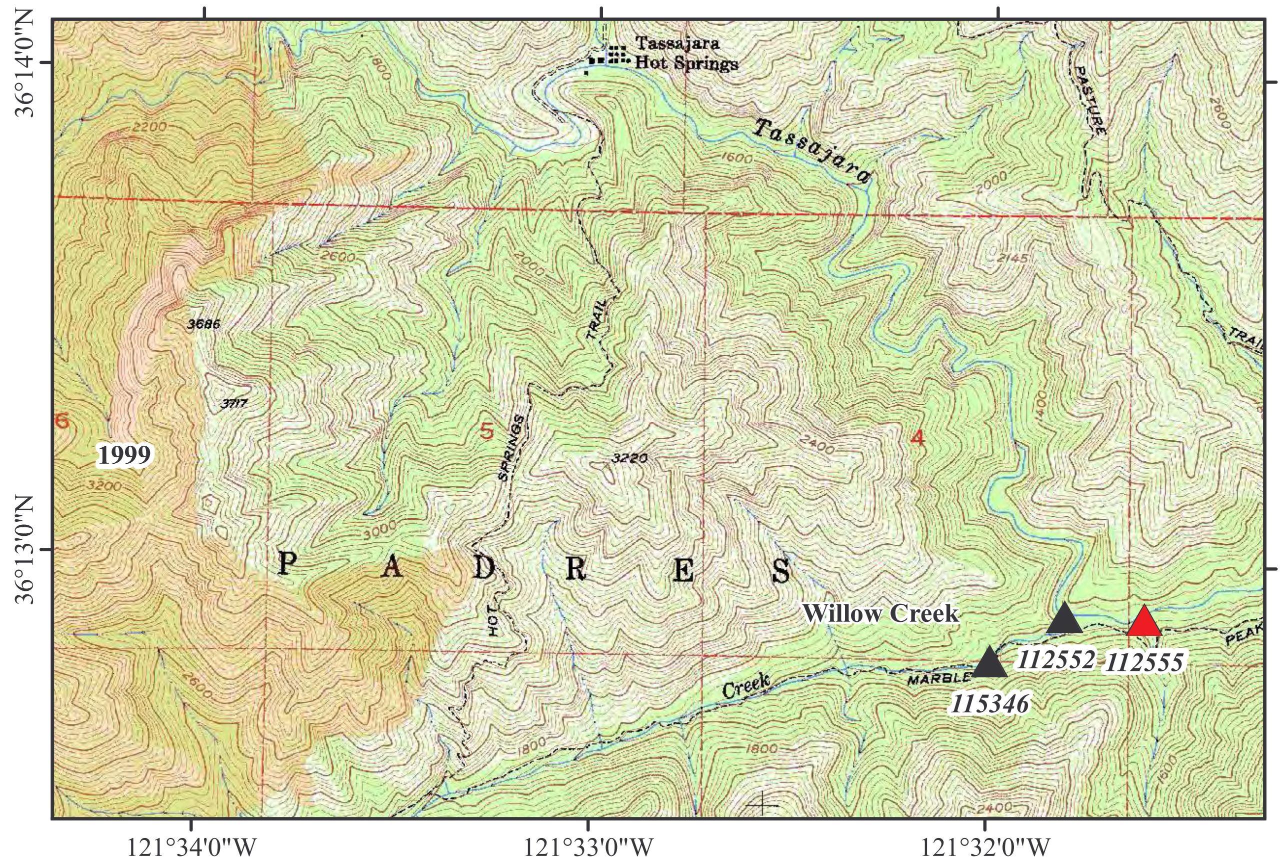




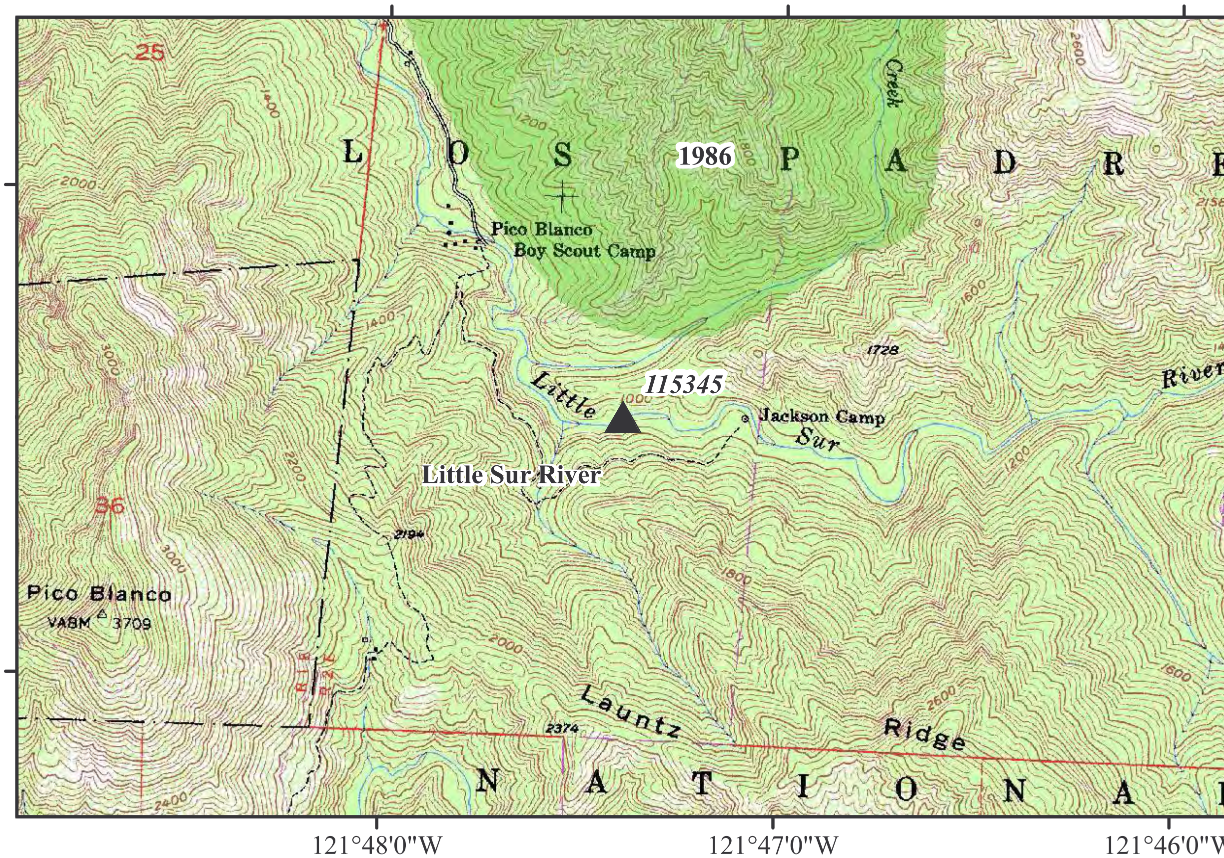
Appendix 3b. Arroyo Seco and Santa Lucia creeks, reference sites 112516 and 112427 (black triangles), and test sites 112522 and 112421 (red triangles) respectively. The objective of Arroyo Seco sampling is to evaluate impacts from Arroyo Seco Campground. The objective of the Santa Lucia Creek sampling is to evaluate impacts from the Indian Road crossing



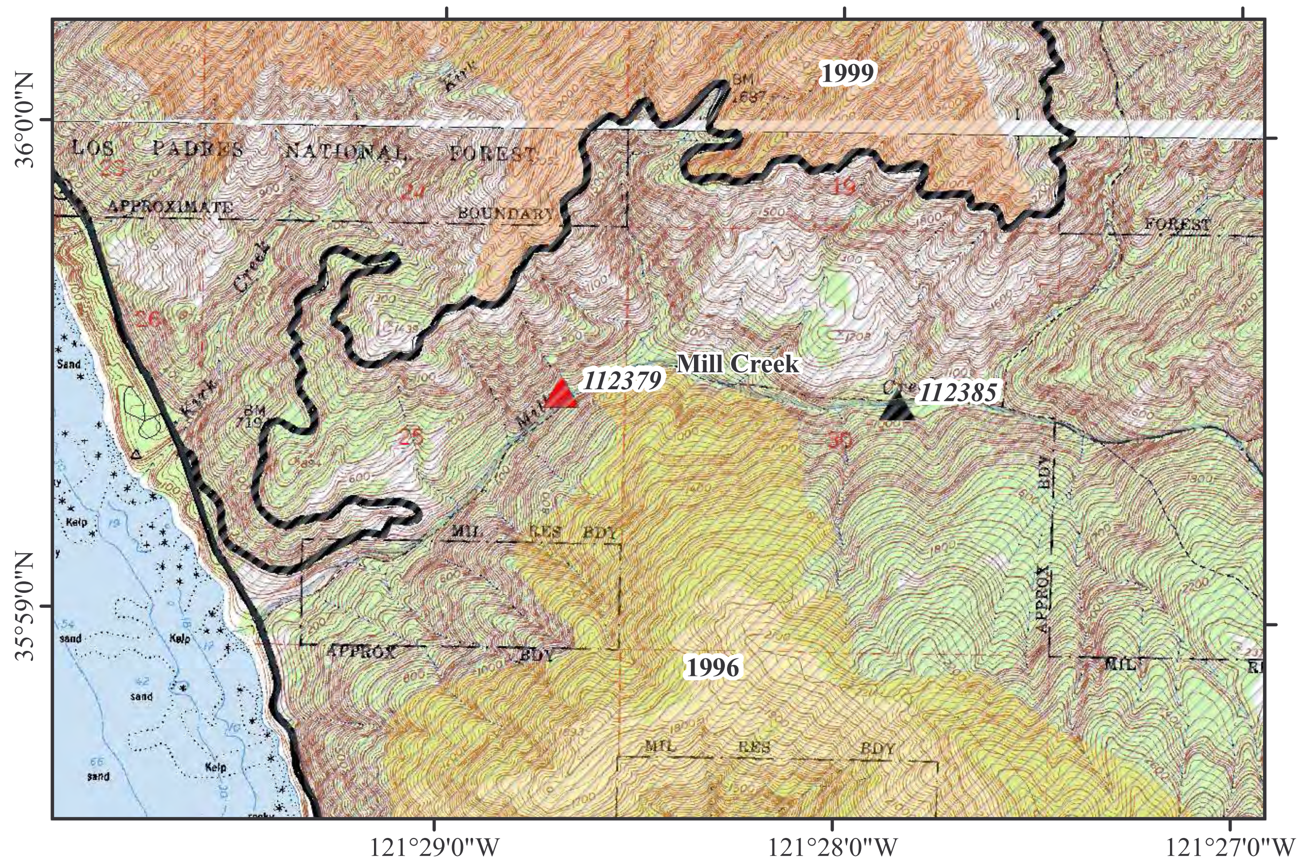
Appendix 3c. The three test sites on Piney Creek are to monitor the impact of Piney Campground.



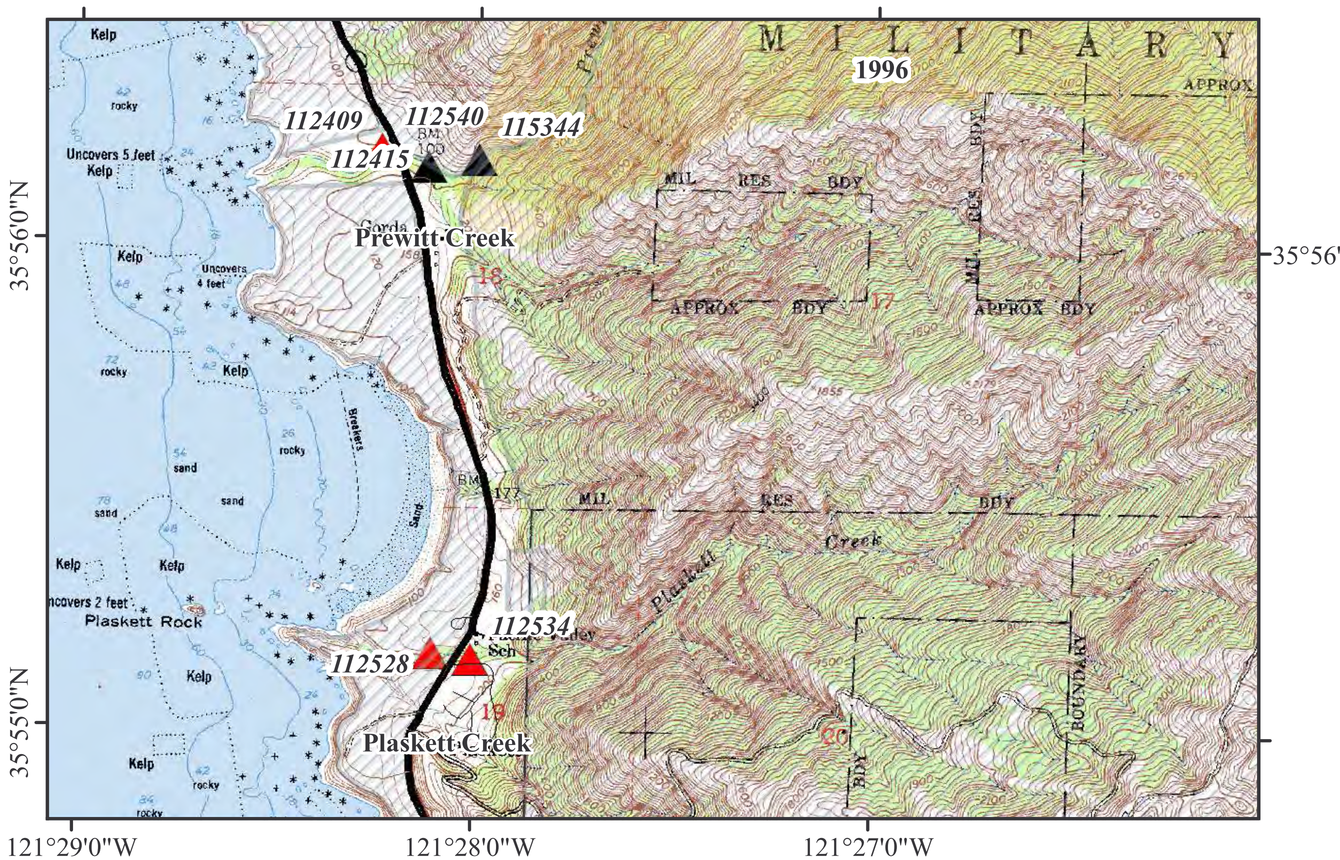
Appendix 3d. The inland MRD Willow Creek sites were collected to evaluate the impact of the September 1999 Kirk fire. Willow Creek's test site (112555) is located downstream from the confluence with Tassajara Creek, while reference sites (1999: 112552 and 2000: 115346) are upstream of the confluence. All samples were taken after September 1999.



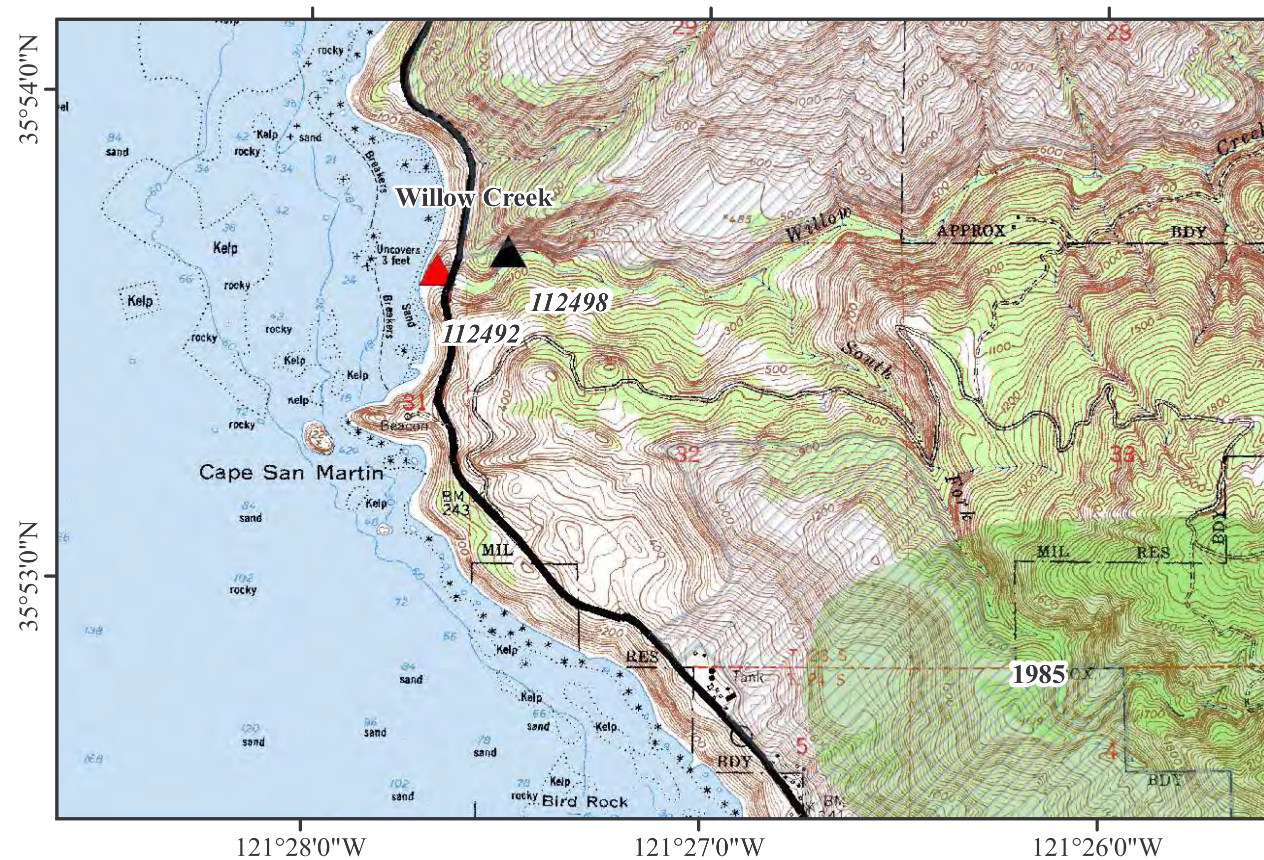
Appendix 3e. Monterey Ranger District Coastal Little Sur River was sampled in 2000 as a reference site for monitoring Pico Blanco Boyscout Camp.



Appendix 3f. Mill Creek was sampled in June 1999 to evaluate the impact of sediment from Nacimiento Road. It also appears to be near two burn areas, 1996 to the south, and September 1999 to the north. The entire Mill Creek region is within a USFS Cattle allotment, though it is not the objective of the sampling.

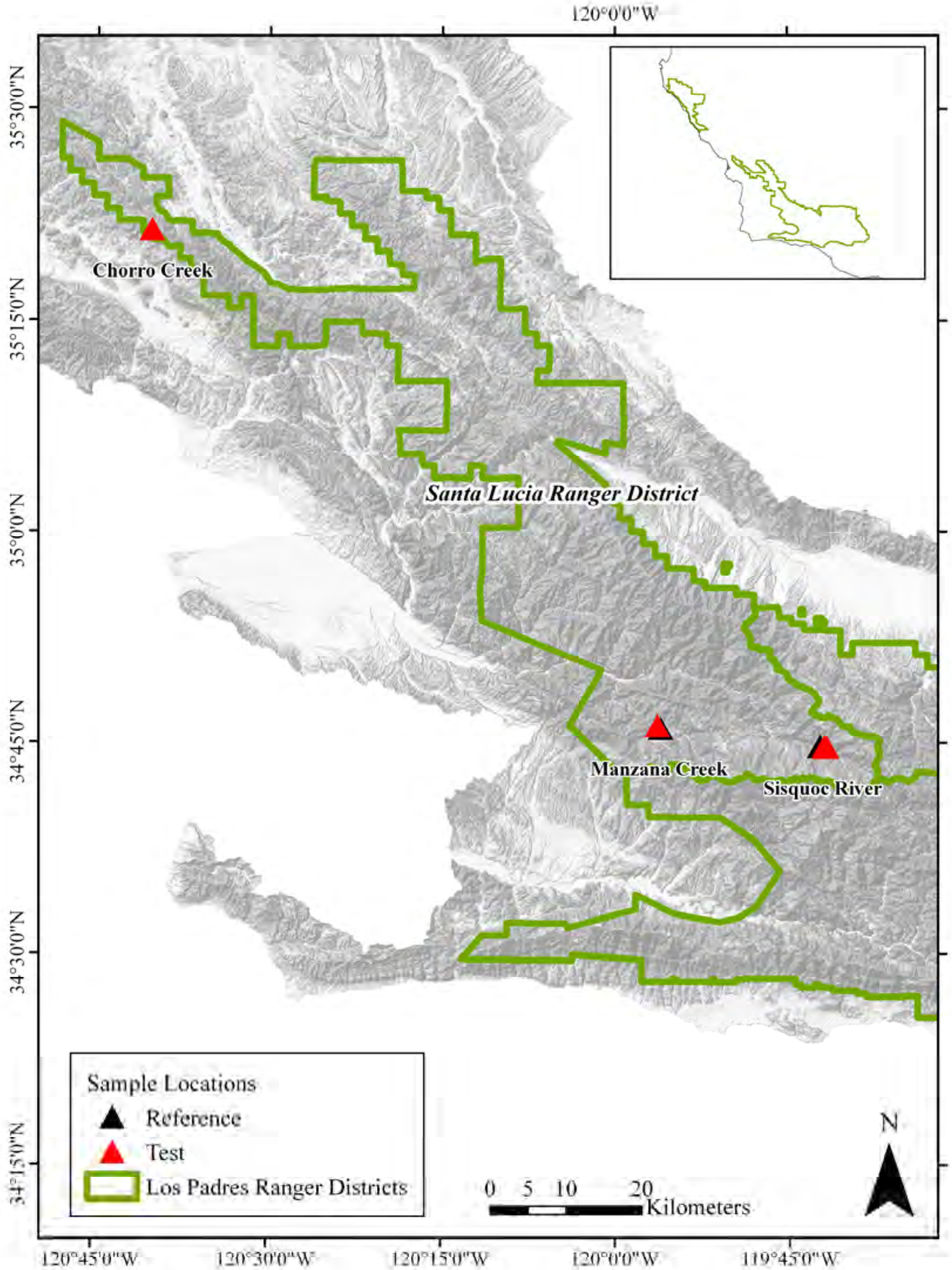


Appendix 3g. Plaskett Creek contains two December 1999 test sites, one to evaluate the effects of Plaskett Creek Campground (112534), and the other to evaluate cattle activity (112528). Prewitt Creek, just north of Plaskett Creek, has two reference site and two test sites sampled during 1999 (May: 112415, May: 112409, December:112540 respectively), and one reference site from 2000 (August: 115344). A 1996 burn area is also noted to the northeast of the map.

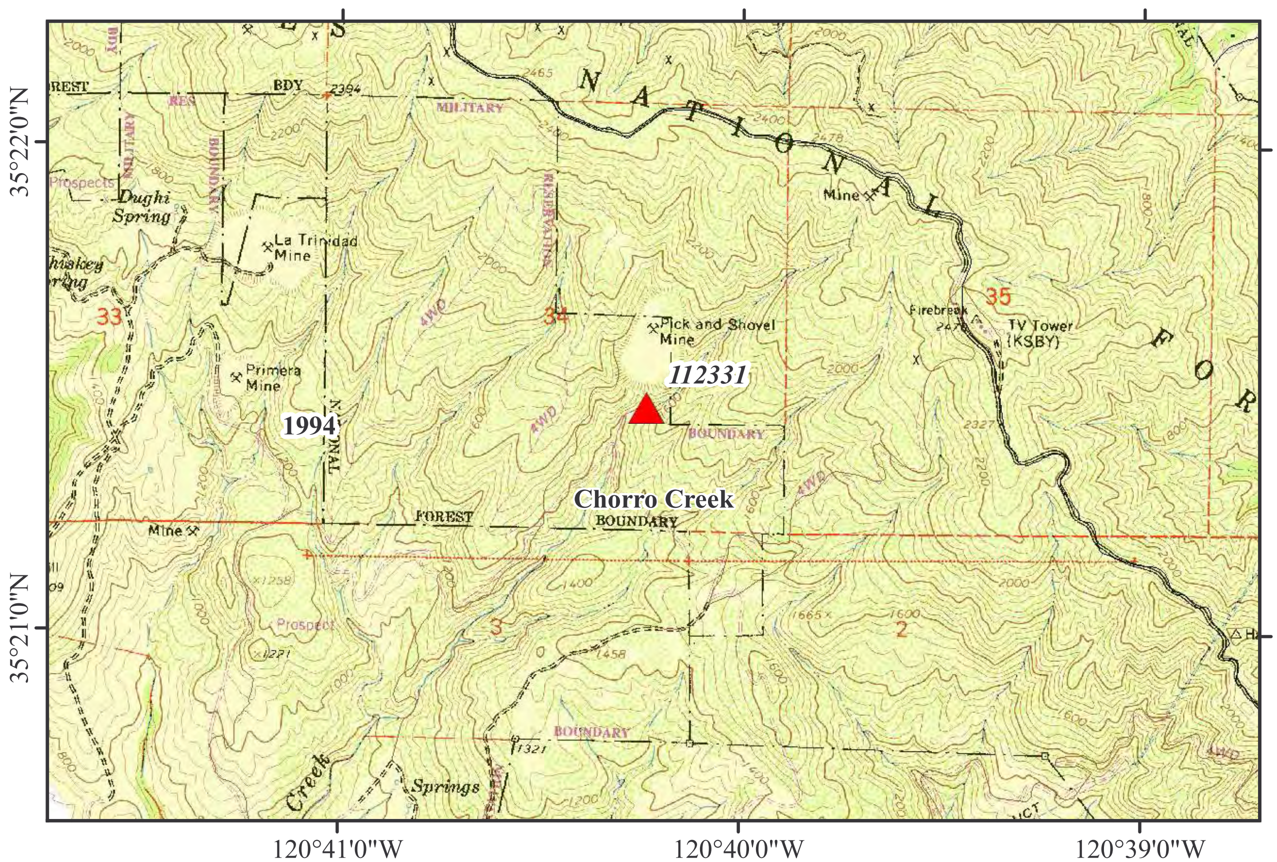


Appendix 3h. One test and one reference sample were taken from Monterey Coastal Willow Creek during May of 1999 to monitor the impact of Willow Creek day-use area (112492 and 112498 respectively).Cattle allotments are noted in the north and south area of the map, and a 1985 fire burn area in the southeast.

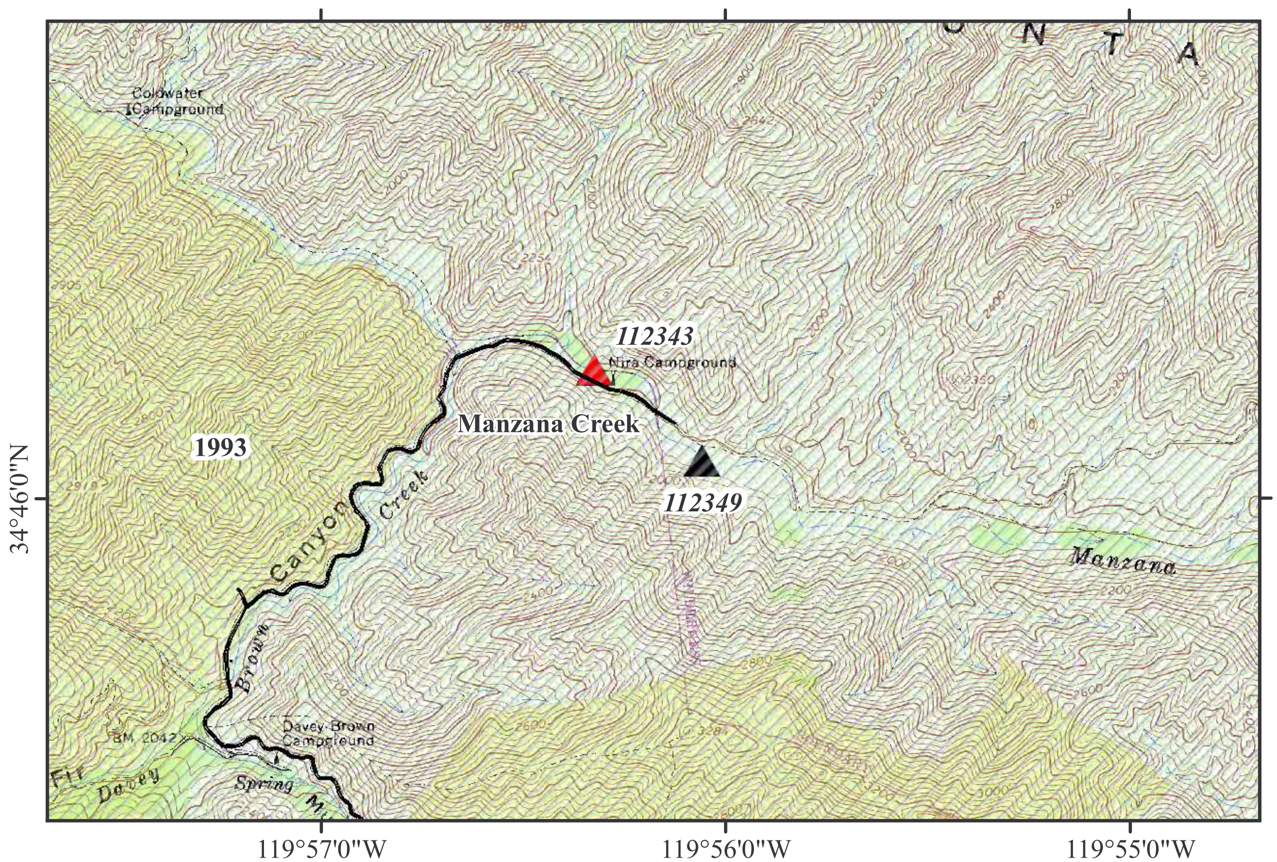
4 Santa Lucia Ranger District Sites



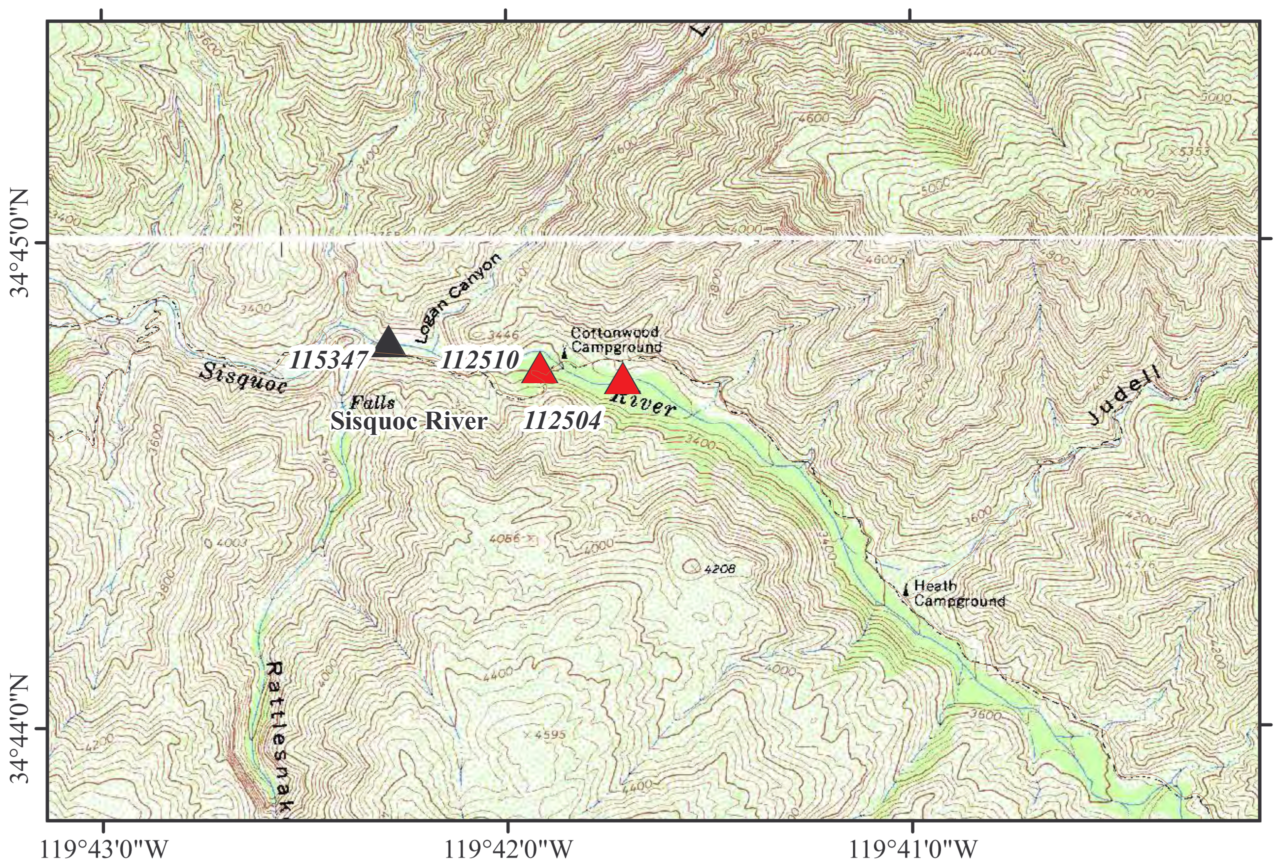
Appendix 4a. The Santa Lucia Ranger District is the central district in the LPNF. There are three streams in this district. There is one test site on both Chorro Creek and Manzana Creek, and two test sites on the Sisquoc River. Manzana Creek and Sisquoc River each have a single reference reach.



Appendix 4b. Santa Lucia Ranger District: Chorro Creek was sampled in July of 1999 to evaluate potential impacts from a nearby Pick and Shovel Mine. The area also overlaps with a 1994 burn area documented in the USFS fire layer.

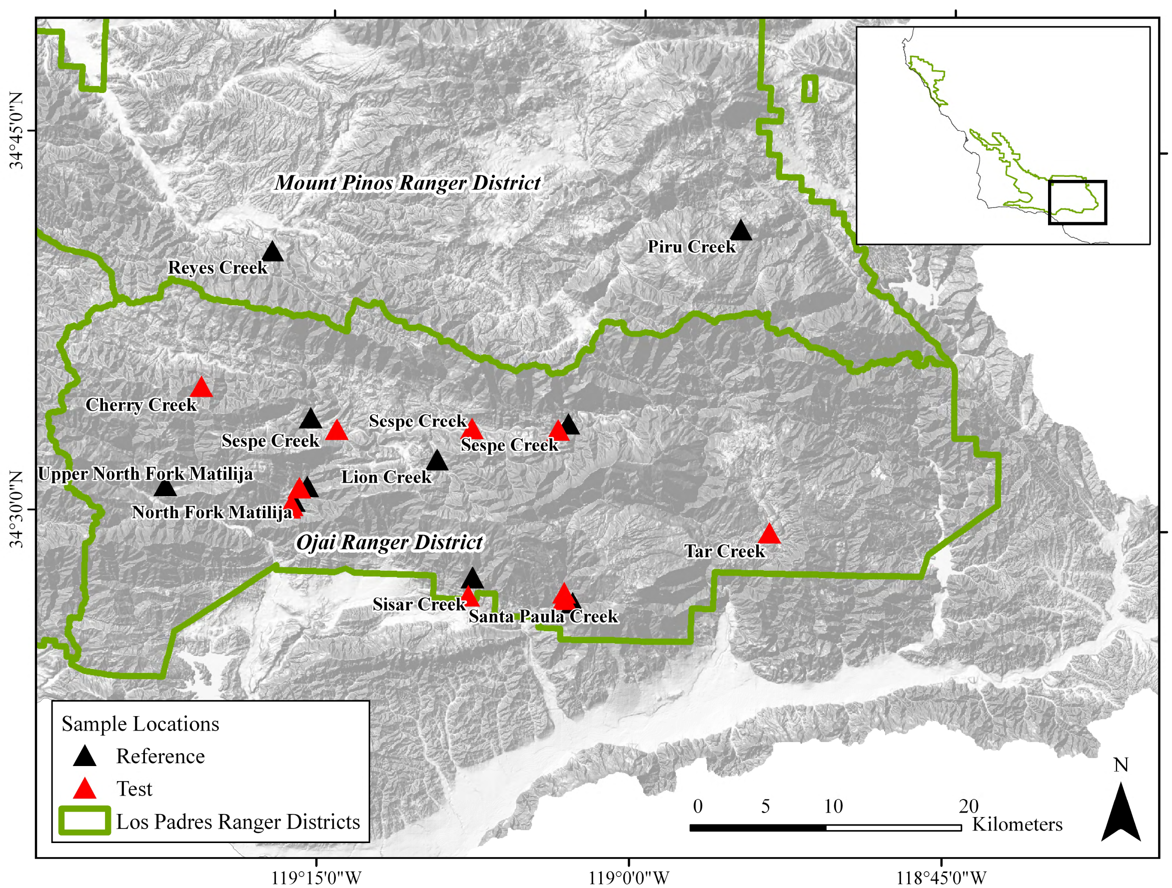


Appendix 4c. Individual reference and test samples were taken during July of 1999 along Manzana Creek to monitor impacts from Nira Campground. USFS fire layers show 1993 burn areas to the west and south of Manzana Creek. Both reaches appear to be located within a USFS cattle allotment.

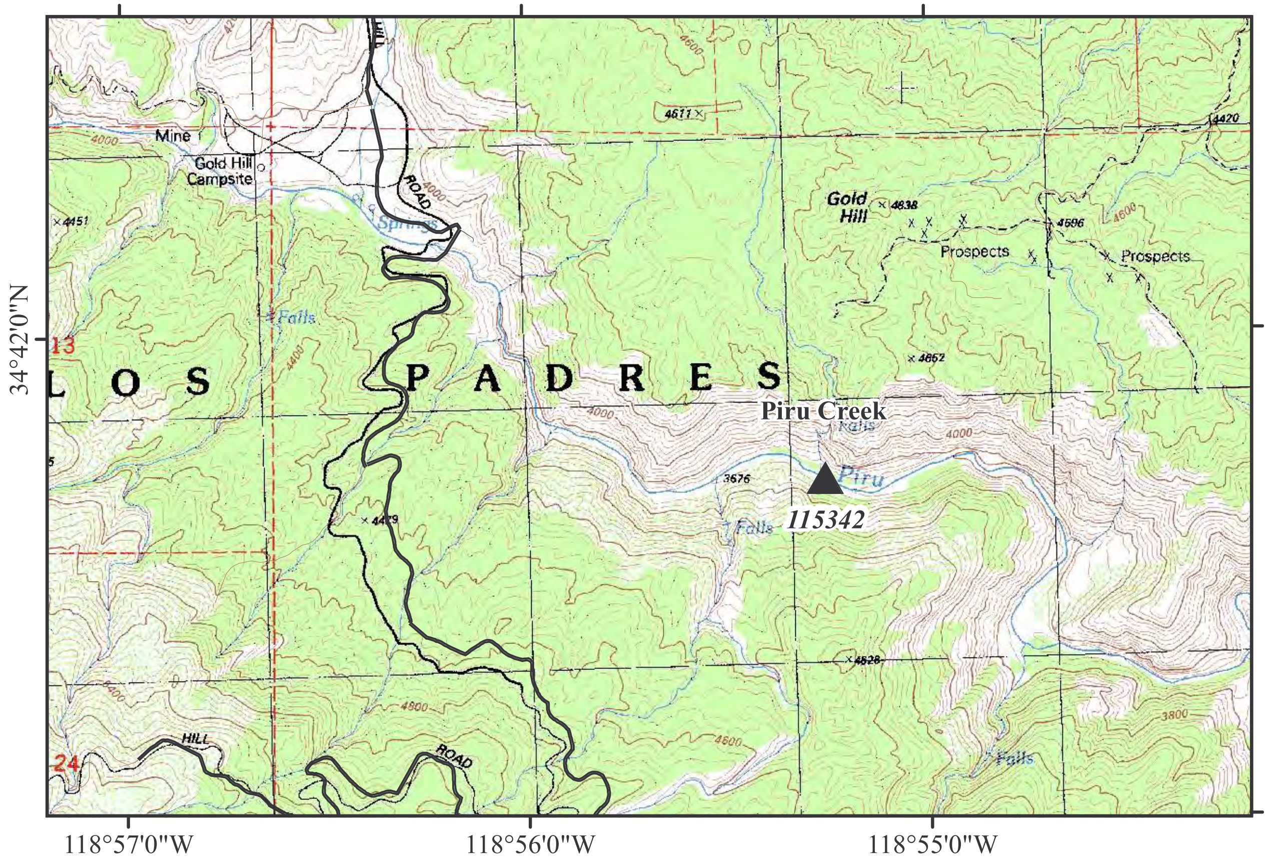


Appendix 4d. One reference and two test reaches are located on the Sisquoc River (Reference: 115347, Test 112510, and 112504). These sites are to evaluate impacts from Cottonwood Campground.

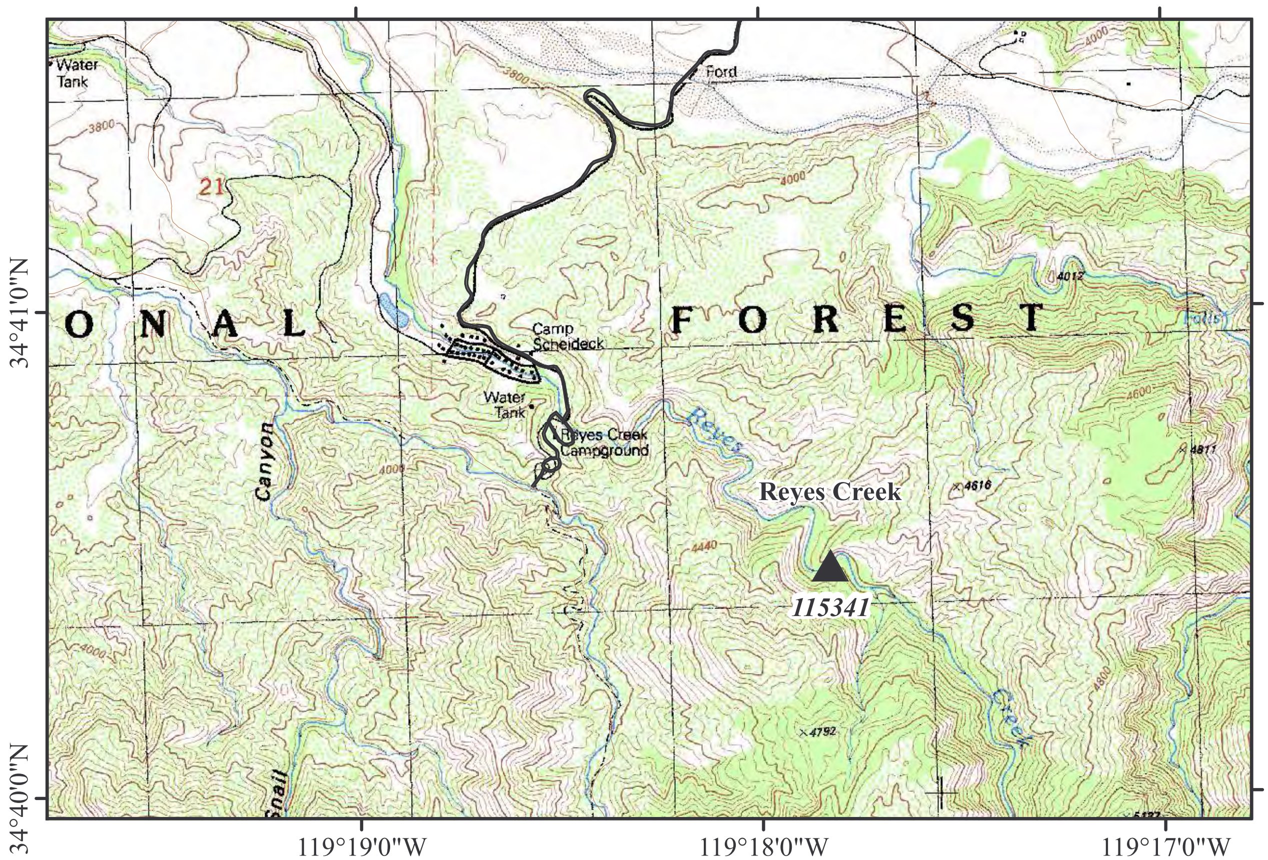
5 Mount Pinos and Ojai Ranger District Sites



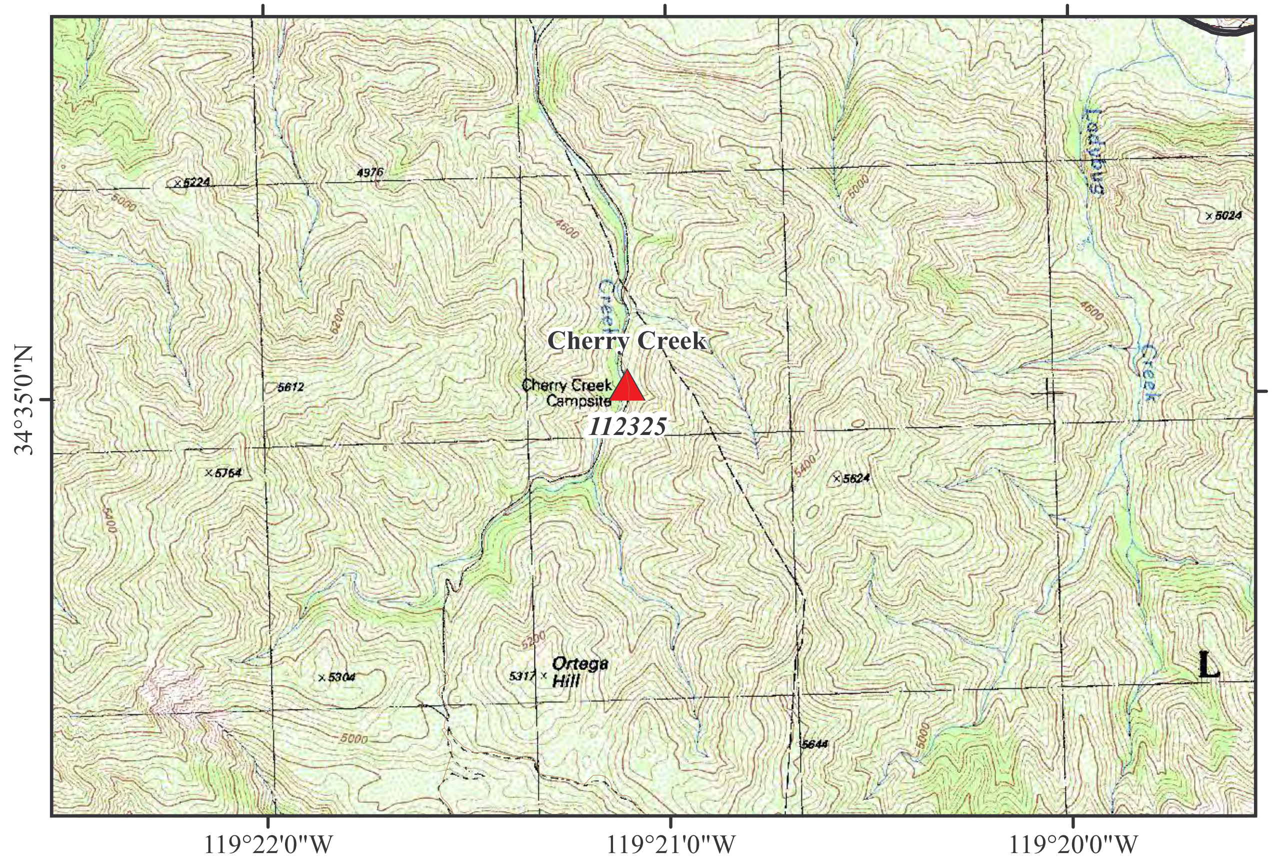
Appendix 5a. The Mount Pinos and Ojai ranger districts are the southern most districts in the LPNF. There are two creeks in Mount Pinos, Reyes and Piru, with one reference sample per reach. The Ojai Ranger District is the most heavily sampled district in the forest. There are a total of 21 sites in eight streams.



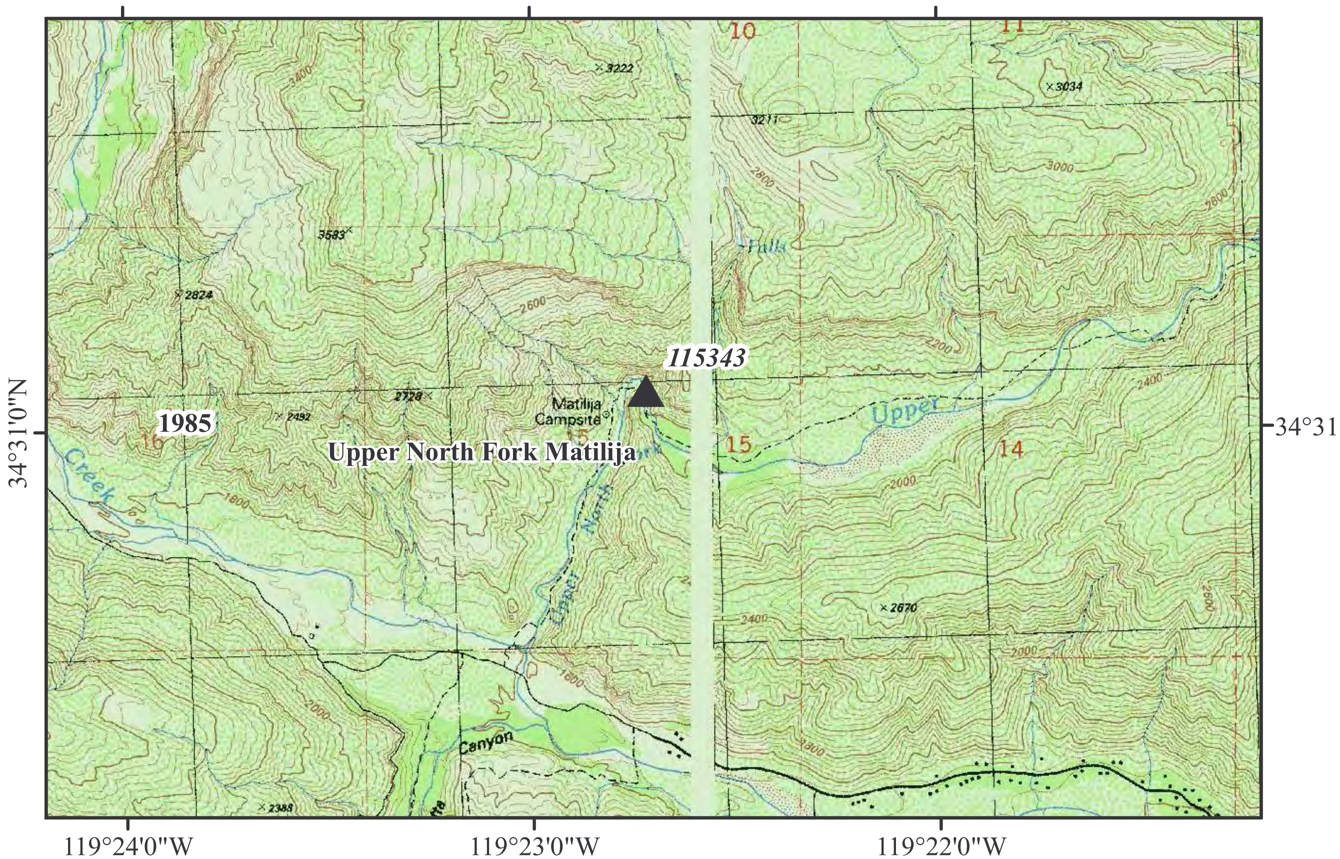
Appendix 5b. Mount Pinos Ranger District: The Piru Creek reference reach is located downstream from Gold Hill Campsite and off-road vehicle park. A single sample was taken in August of 2000 to evaluate recreation along the creek.



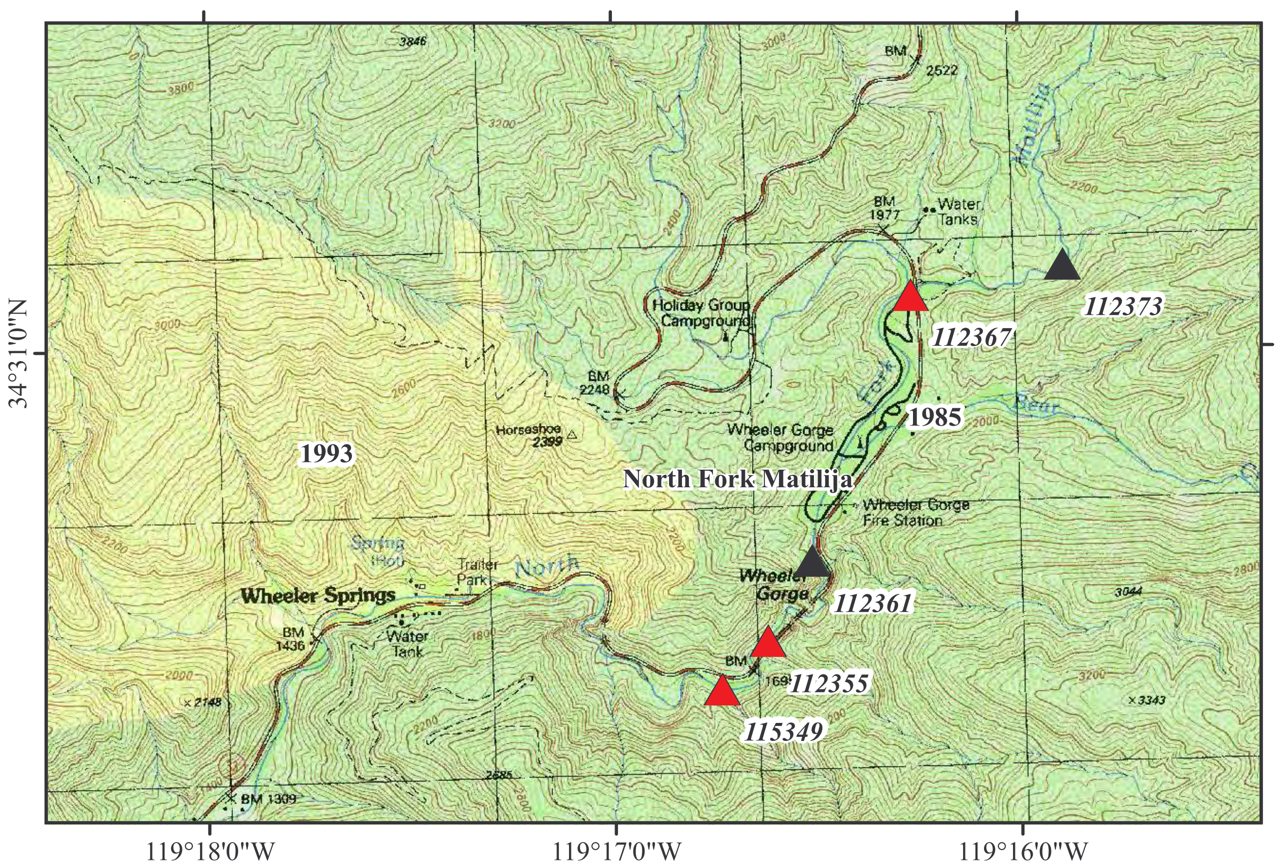
Appendix 5c. In August of 2000 the USFS collected a reference sample along Reyes Creek to evaluate recreation along the creek (sample 115341).



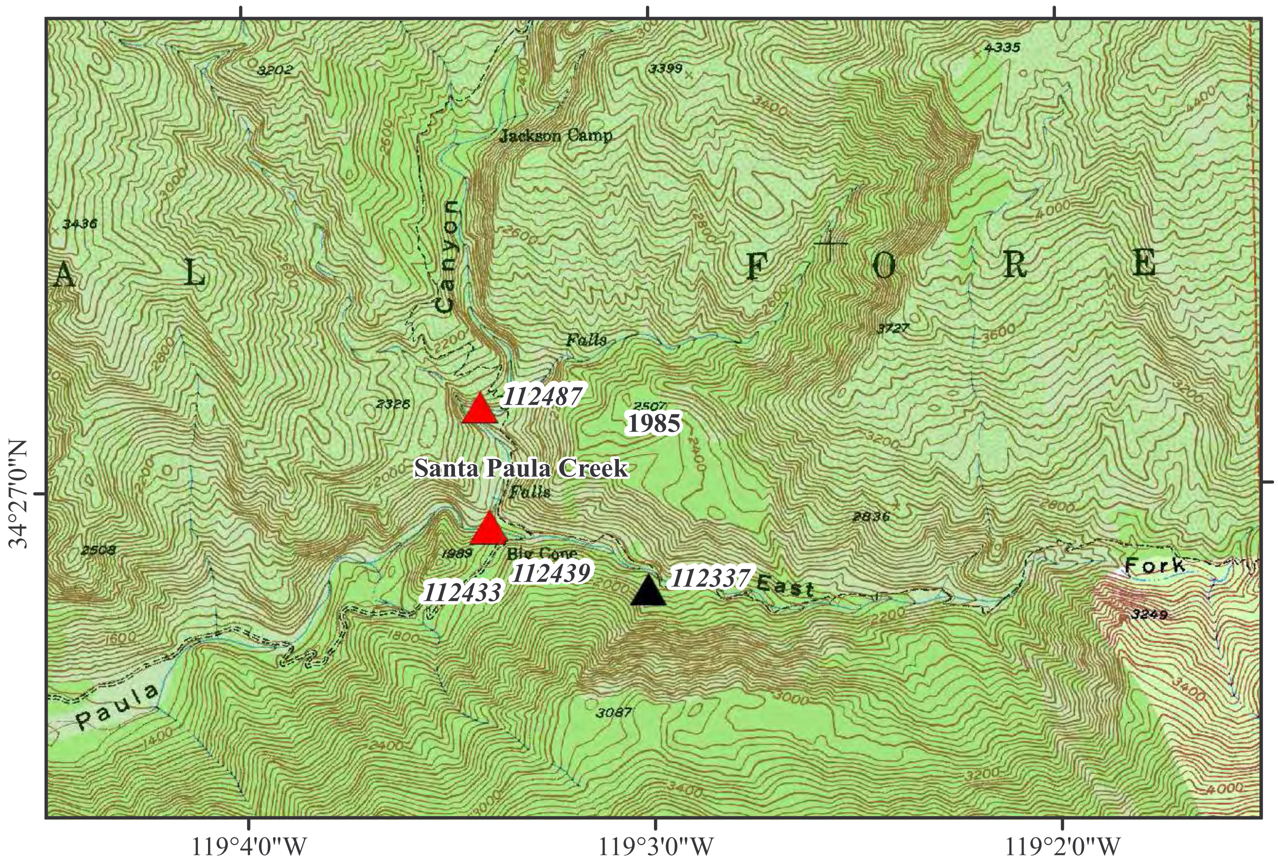
Appendix 5d. Ojai Ranger District: Cherry Creek was sampled in July 1999 to evaluate the effectiveness of Cherry Creek campground closure (112325).



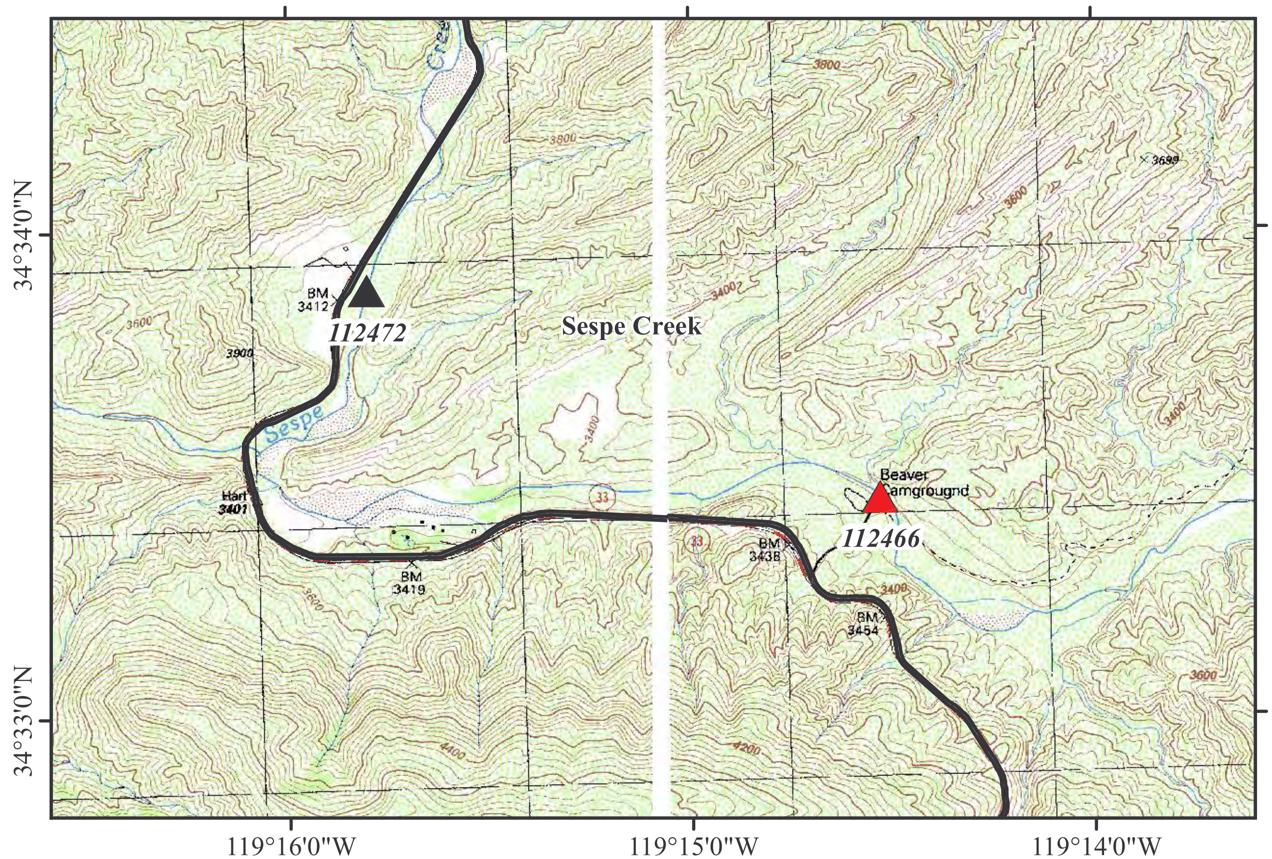
Appendix 5e. USFS reach layers did not identify an objective for the August 2000 Upper North Fork Matilija Creek reach, but based on its proximity upstream from the Matilija Campsite, we expect that sample 115343 is to evaluate recreational impact along the creek.



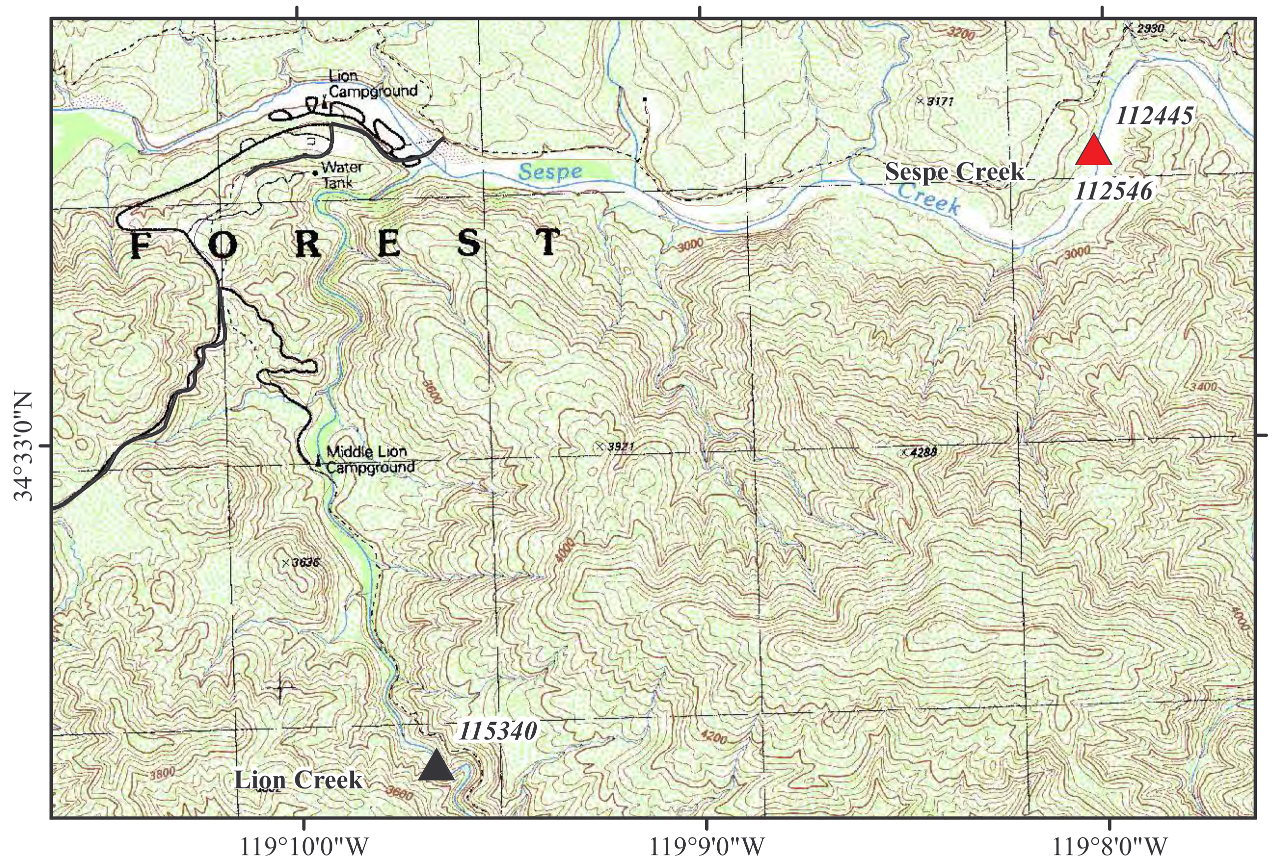
Appendix 5f. Five reaches were sampled along the North Fork Matilija near Wheeler Springs. The southern-most reaches (Test: 115349 and 112355, Reference: 112361) were sampled to evaluate the impact from bridge construction along highway 33. The two northern sites (reference: 112373, and test: 112367) were sampled to evaluate recreational activity near Wheeler Gorge Campground. USFS fire layers shows that much of the area was burned during a fire in 1985, with the western part of the map shows burn areas from 1993).



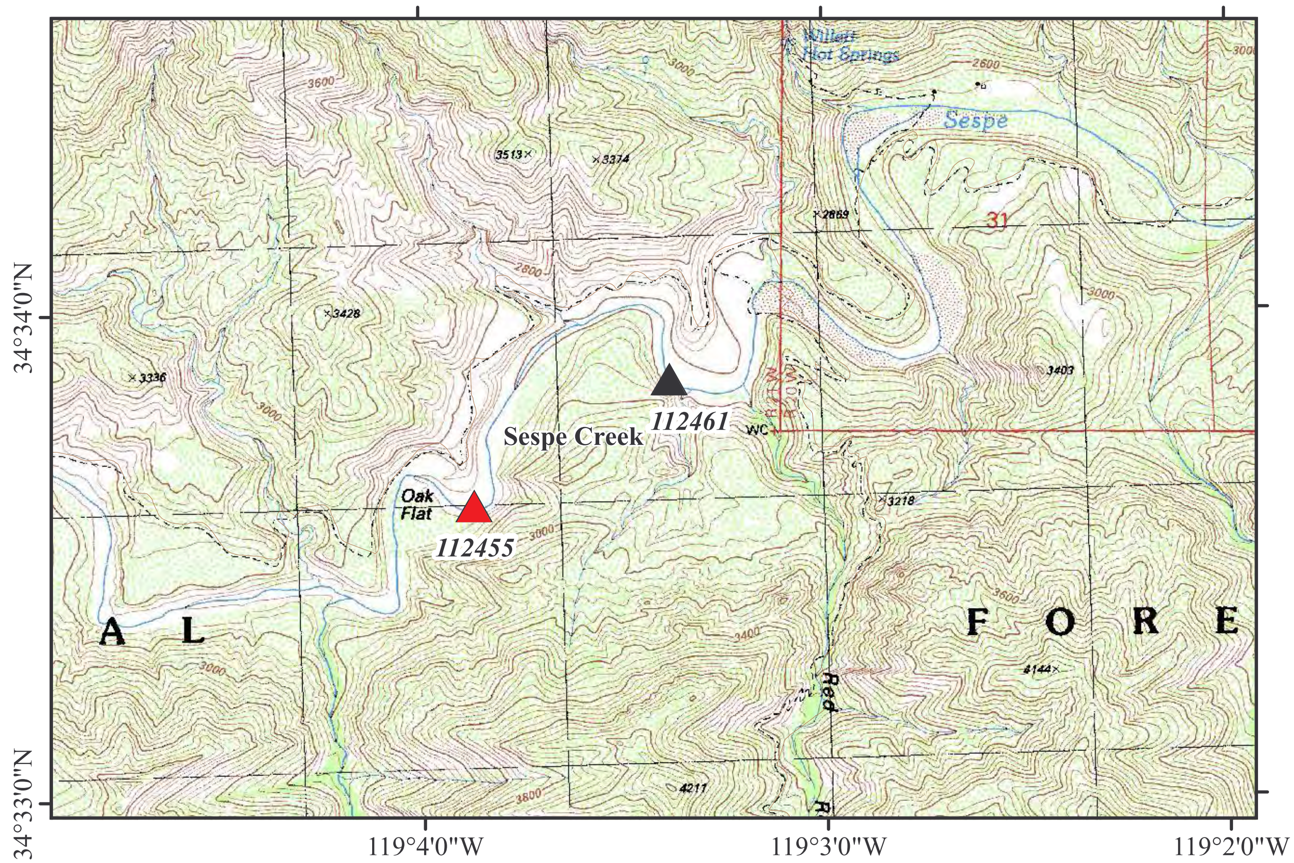
Appendix 5g. Three test reaches and one reference reach were sampled to evaluate recreational impacts along Santa Paula Creek (Test: 112487, 112433, and 112439, Reference: 112337). Aside from the campgrounds Big Cone and Jackson, there are a number of swimming holes and illegal camps on the creek. The entire area is located within the boundaries of a 1985 burn area.



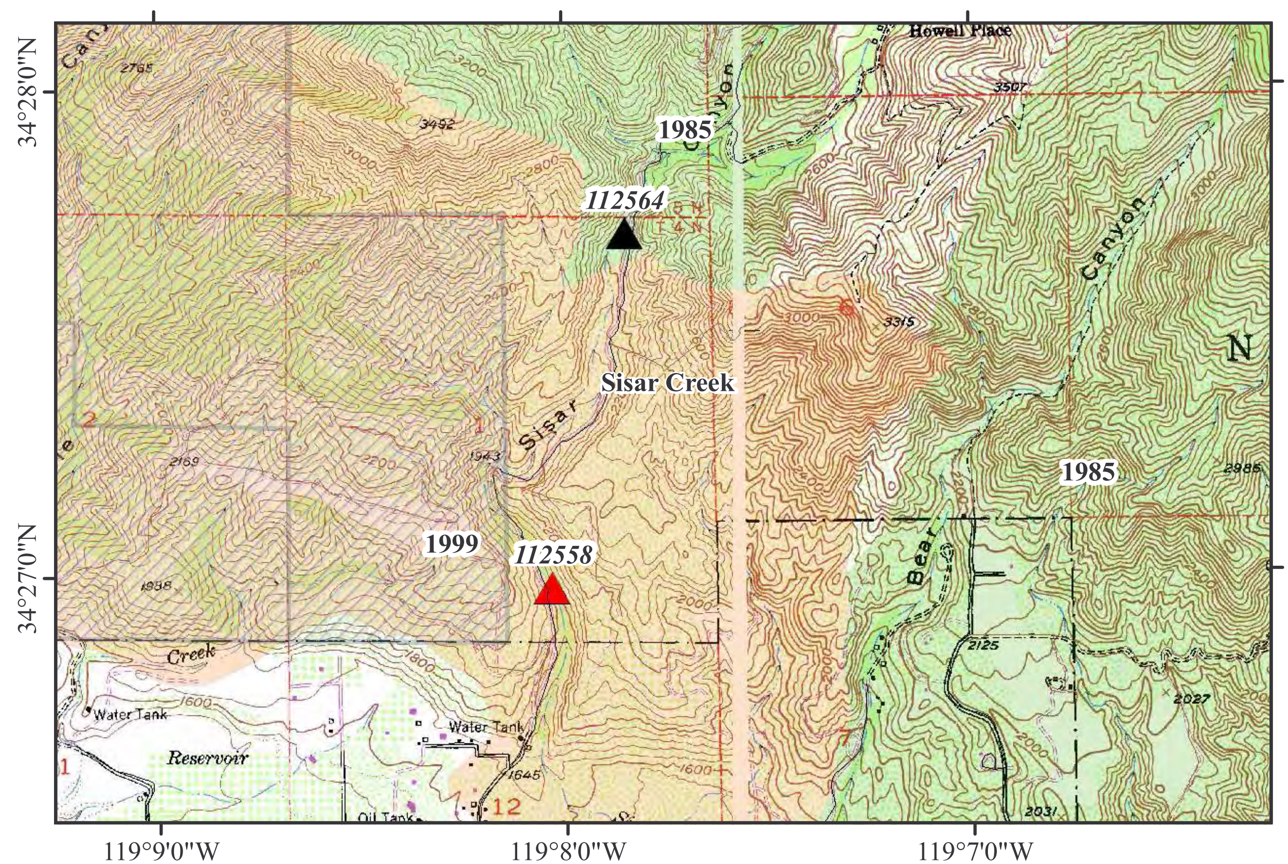
Appendix 5h. The two westernmost reaches along Sespe Creek were sampled in June 1999 to evaluate the impacts of a landslide. These reaches are located near the confluence with Tule Creek.



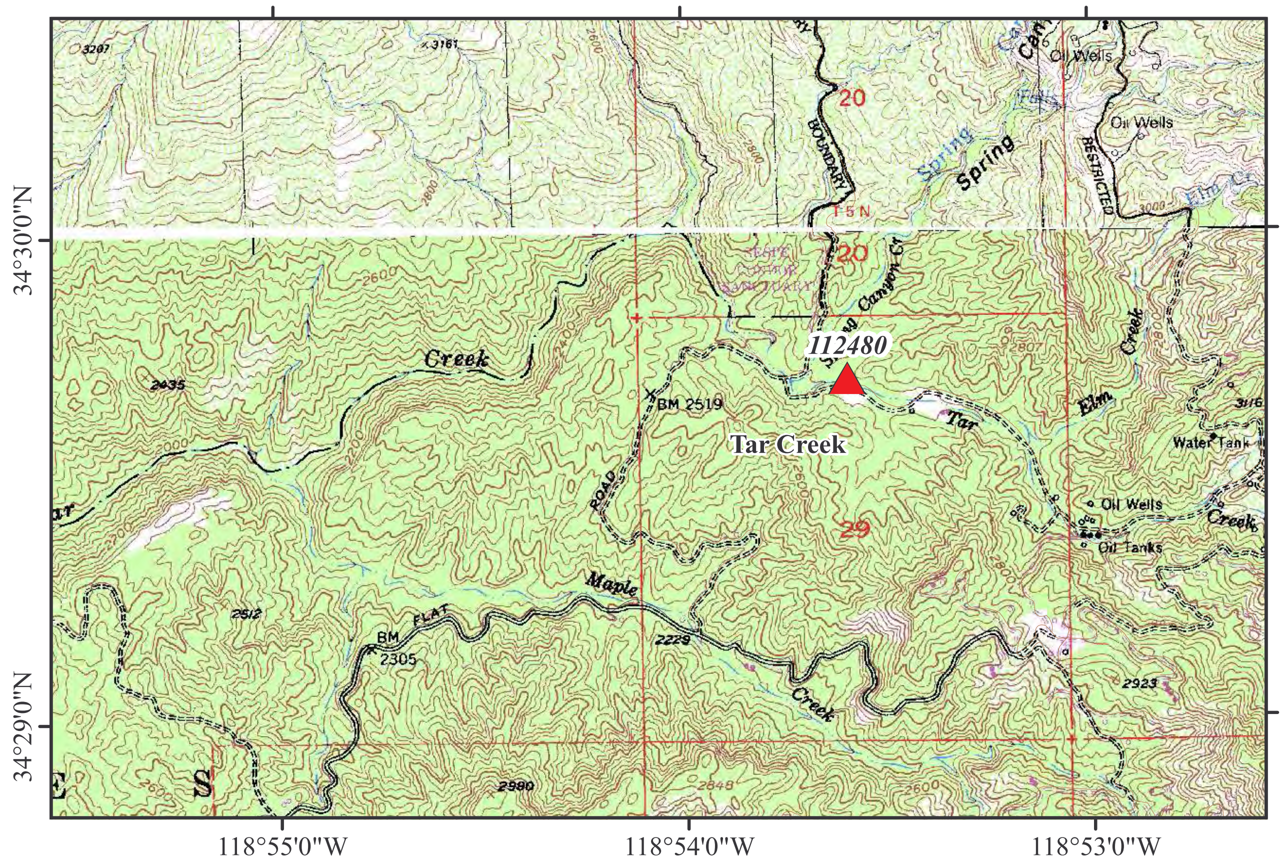
Appendix 5i. Two samples of a test reach near Lion Campground of Sespe Creek were taken to evaluate the effectiveness of the campground closure. The sites were sampled in May 1999 and December 1999 (112445 and 112546 respectively). South of Sespe Creek the USFS sampled one reference reach on Lion Creek to evaluate impacts from recreation along the creek.



Appendix 5j. One reference and one test reach (112461 and 112455 respectively) were sampled on Sespe Creek near Oak Flat in August 1999 to evaluate impacts from Oak Flat Campground.

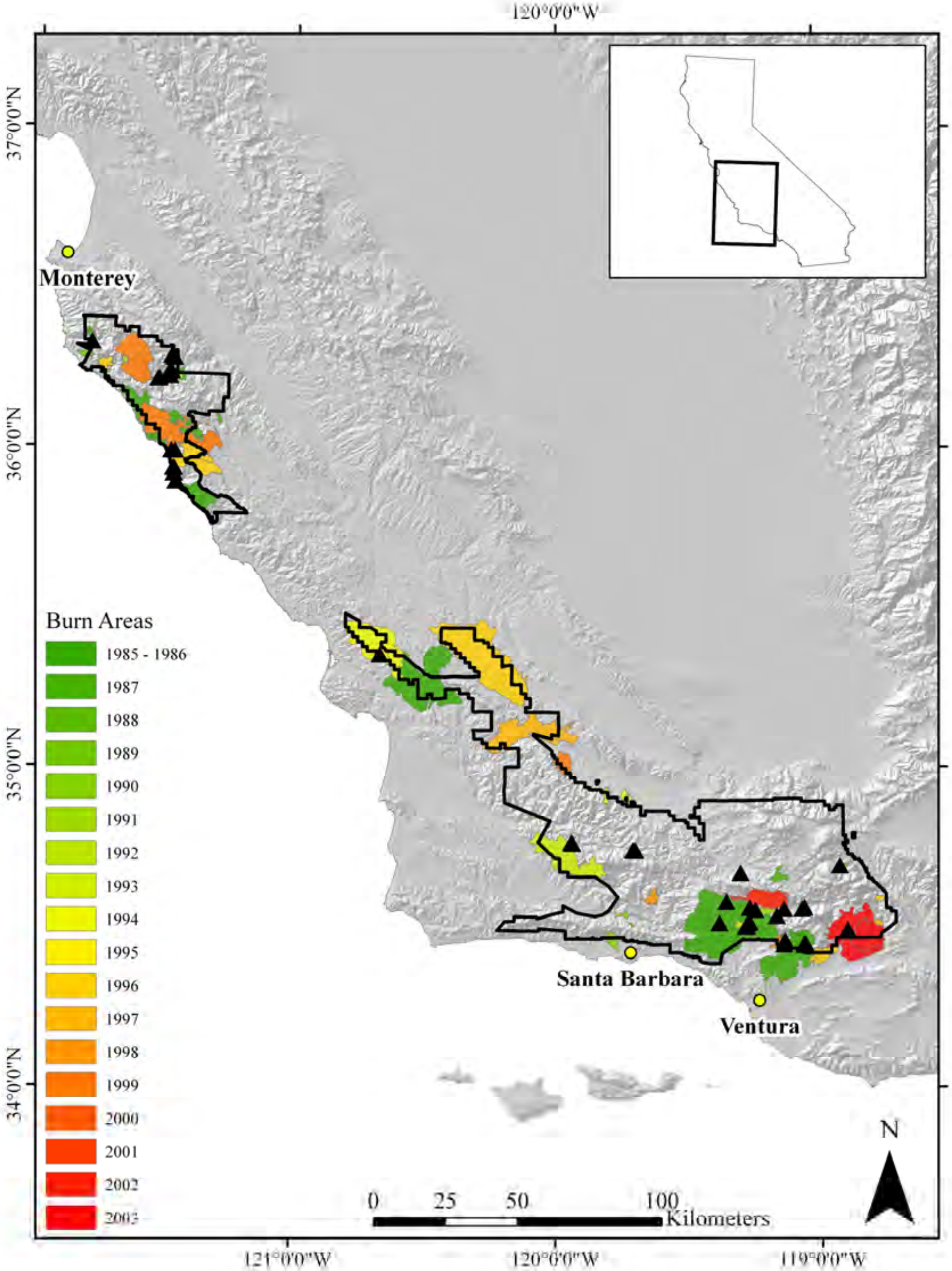


Appendix 5k. One reference and one test reach (112564 and 112558 respectively) we sampled along Sisar Creek in January 2000 to evaluate the impact of a fire in December of 1999. The reference reach (black) is just upstream of the burn area. USFS cattle allotment layers show a cattle allotment to the west of both reaches.

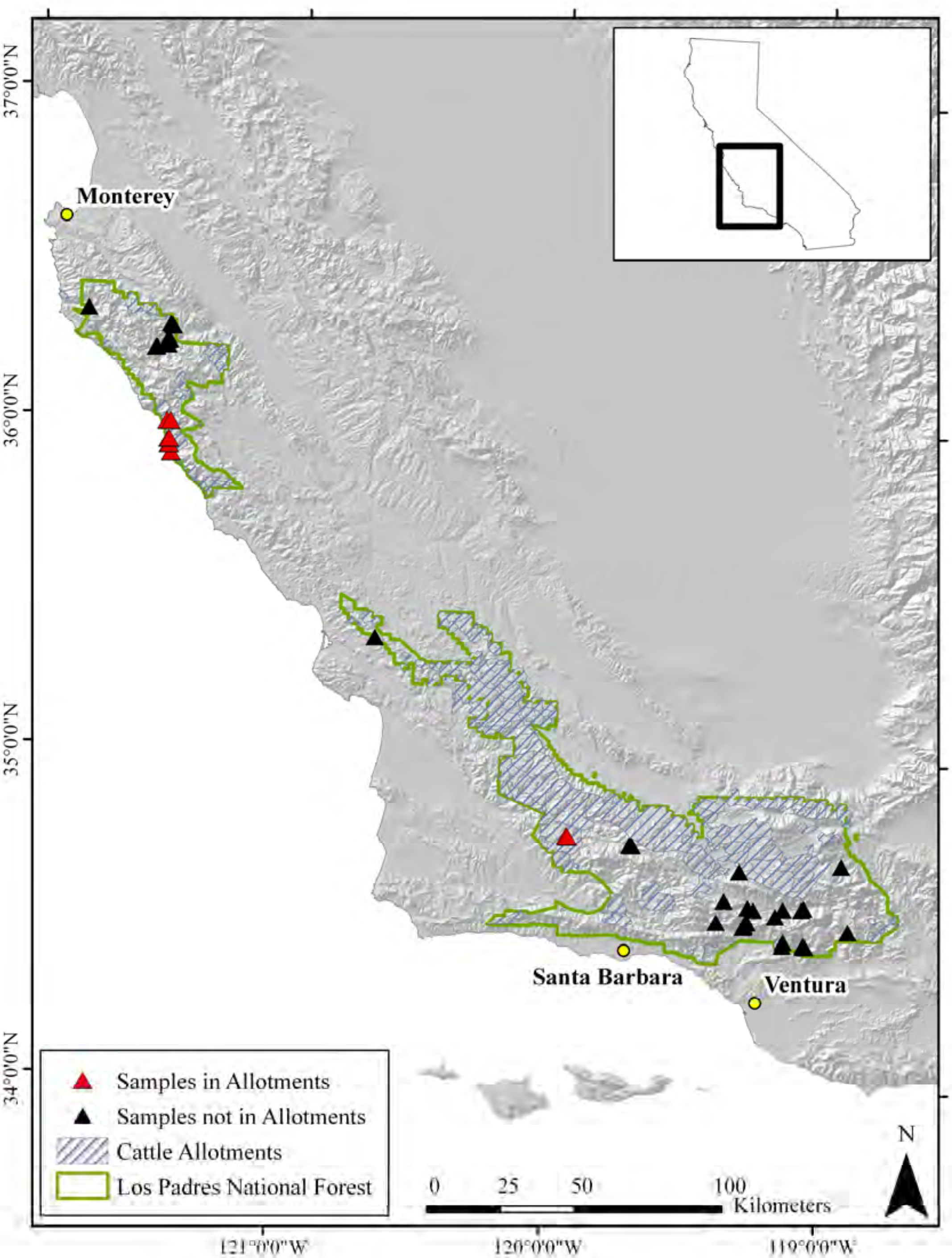


Appendix 5l. The test reach on Tar Creek was sampled in June 1999 (112480) to evaluate impacts from road and oil operations.

6 Land Use Variable Maps

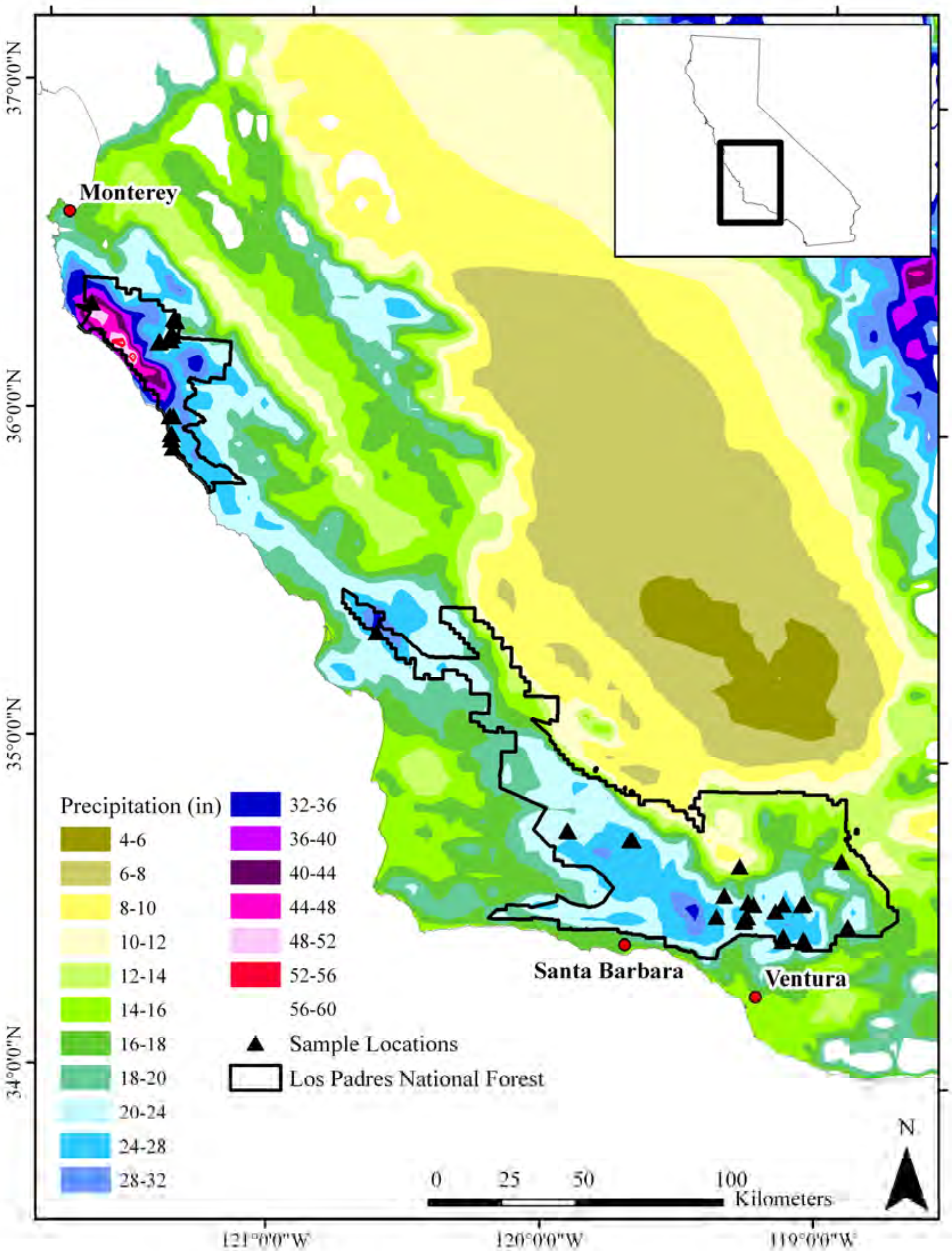


Appendix 6a. Fires from 1985 to 2003 that occurred within the Los Padres National Forest. Sample locations are in black.

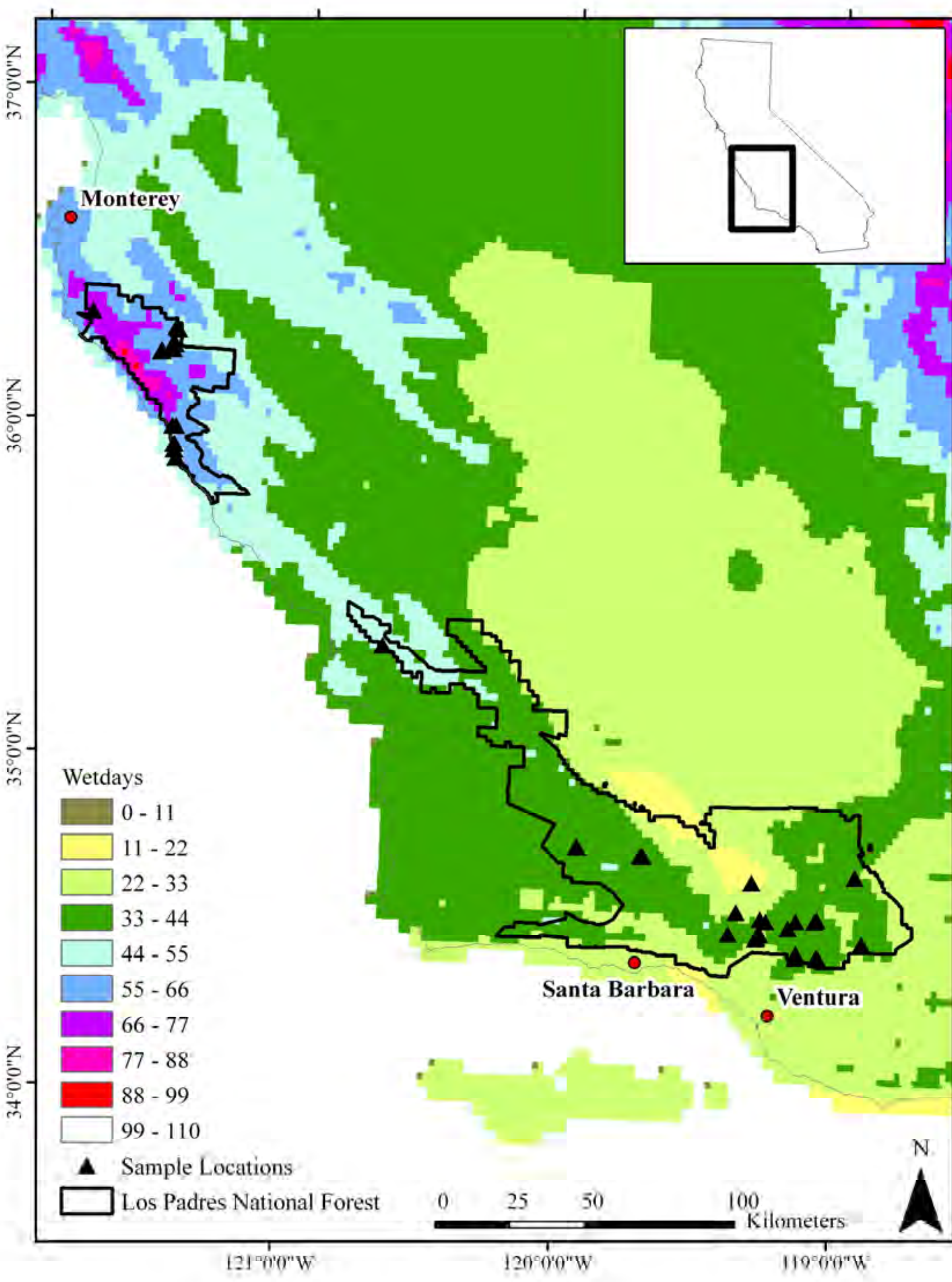


Appendix 6b. Cattle allotments in the study area, hashed blue lines. Red triangles represent samples located within 75 meters of an allotment. Black triangles are sample not within 75 meters of an allotment.

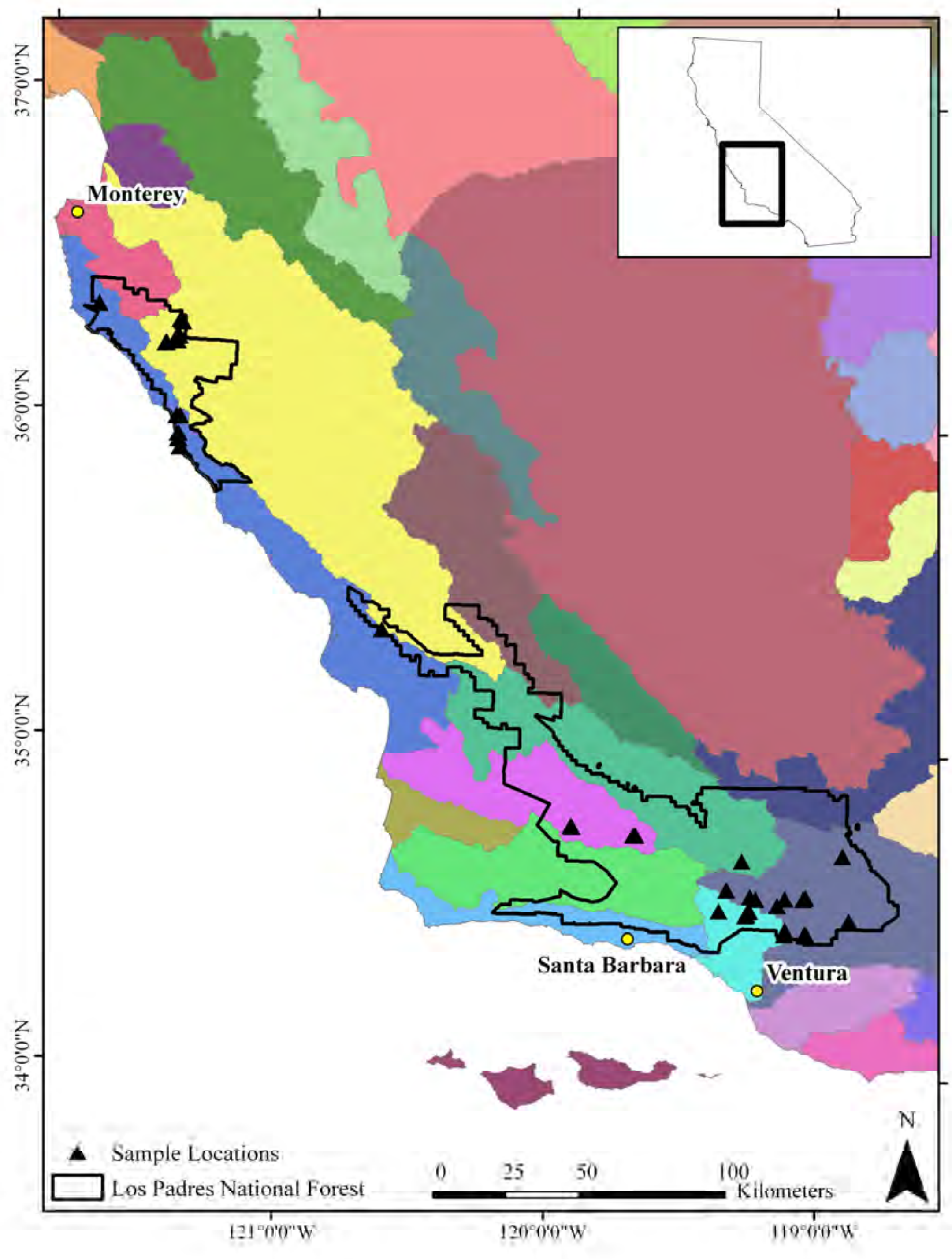
7 Habitat Variable Maps



Appendix 7a. Precipitation data were collected from the PRISM database. Wetter areas, in blue to red, are located south of Monterey and north of Santa Barbara. Drier areas, brown to yellow, increase further inland. The majority of the study reaches, shown in black, are in areas of moderate precipitation.



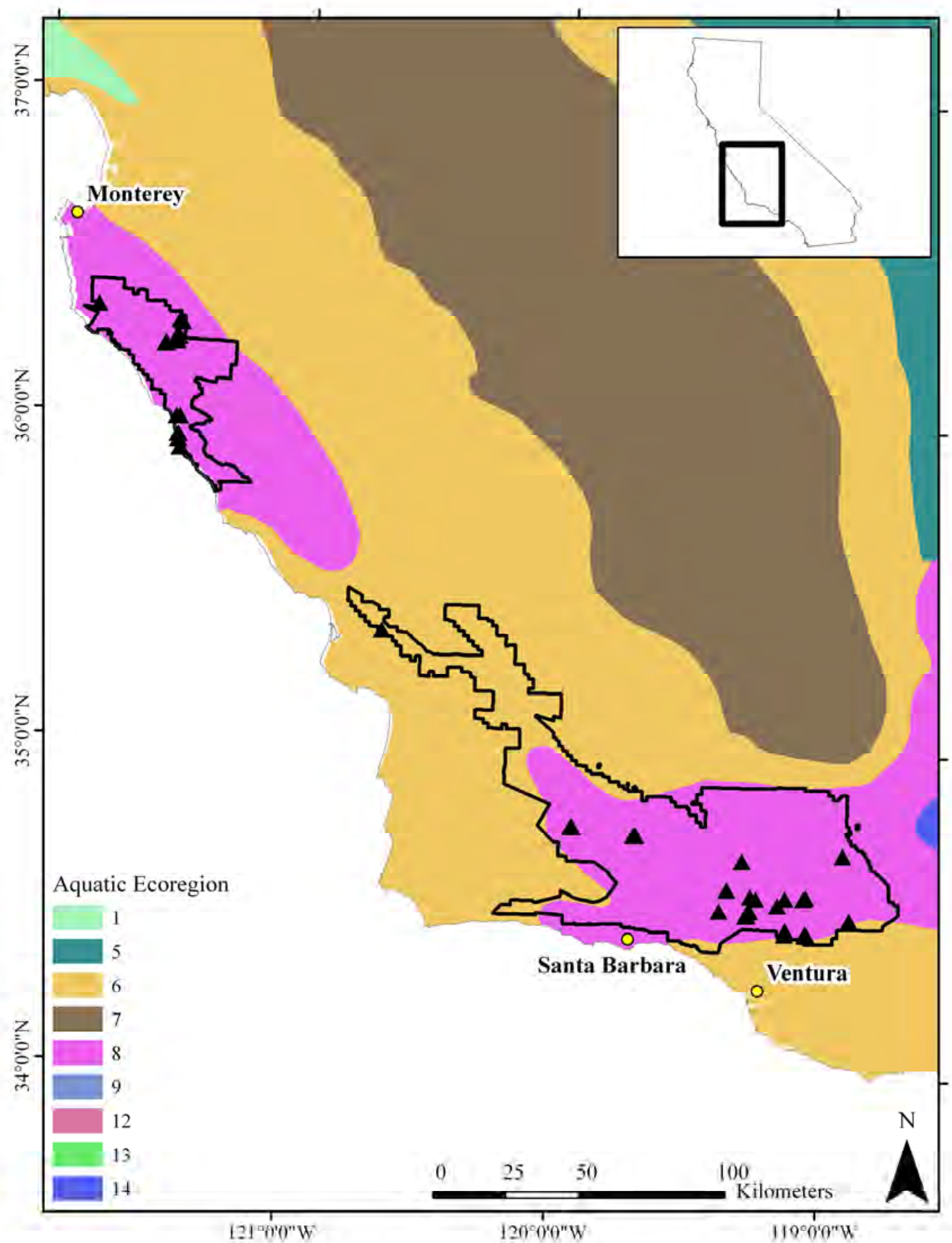
Appendix 7b. The number of days in the water year (October to September) where precipitation exceeds 0.1 inch. Most of the study area is between 33 and 66 days of precipitation.



Appendix 7c. Sample locations are within six different hydrologic unit codes (HUC). Each code represents an area of distinct hydrologic characteristics. From north to south these codes are the Central Coast, Salinas, Santa Maria, Ventura, and Santa Clara cataloging unit HUCs.

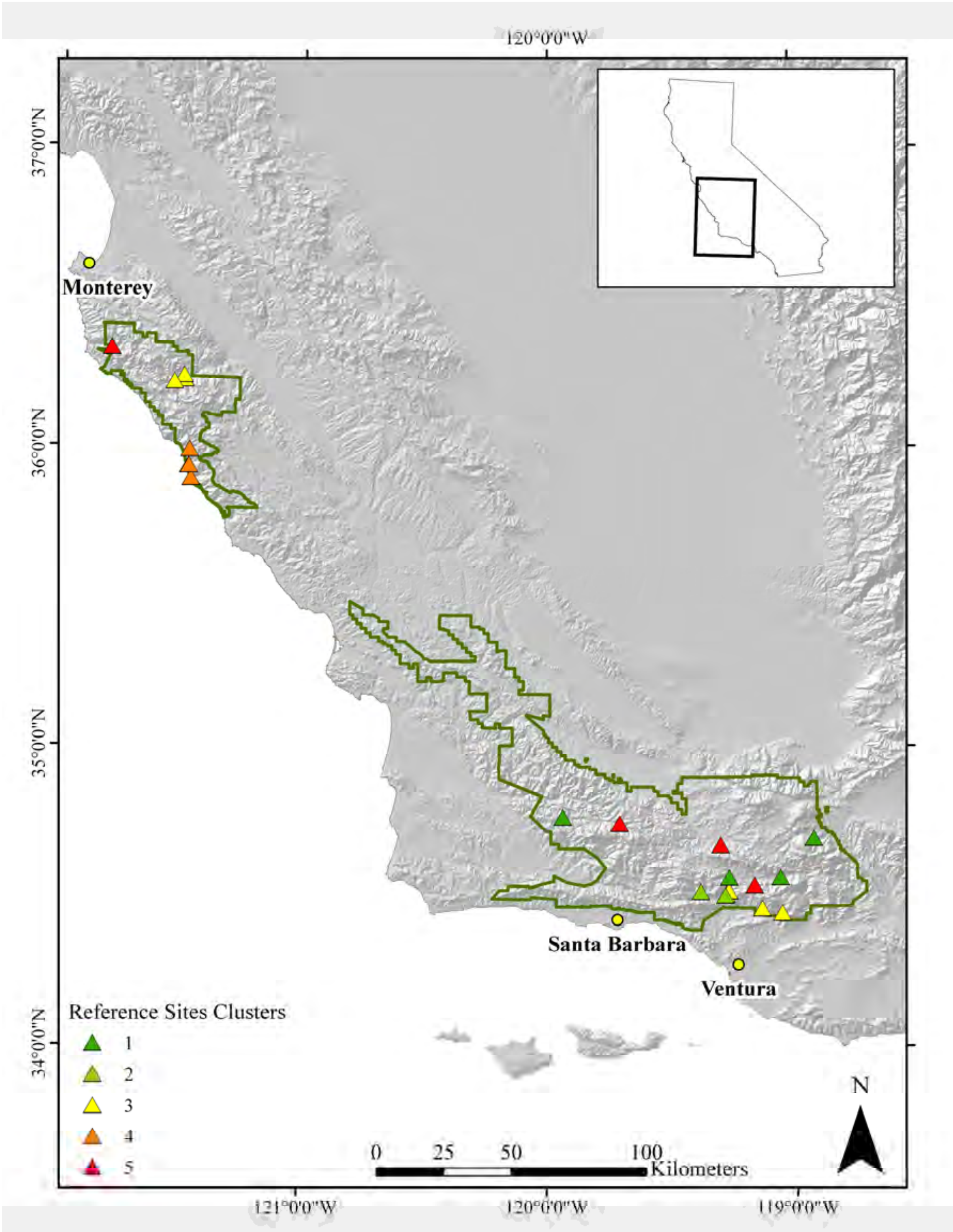
Hydrologic Unit Code

- ALISAL-ELKHORN SLOUGHS
- ANTELOPE-FREMONT VALLEYS
- CALLEGUAS
- CARMEL
- CARRIZO PLAIN
- CENTRAL COASTAL
- COYOTE
- CROWLEY LAKE
- CUYAMA
- ESTRELLA
- LOS ANGELES
- MIDDLE KERN-UPPER TEHACHAPI-GRAPEVINE
- MIDDLE SAN JOAQUIN-LOWER CHOWCHILLA
- MILL
- PAJARO
- PANOCHE-SAN LUIS RESERVOIR
- SALINAS
- SAN ANTONIO
- SAN LORENZO-SOQUEL
- SANTA BARBARA CHANNEL ISLANDS
- SANTA BARBARA COASTAL
- SANTA CLARA
- SANTA MARIA
- SANTA MONICA BAY
- SANTA YNEZ
- TULARE-BUENA VISTA LAKES
- UPPER CHOWCHILLA-UPPER FRESNO
- UPPER DEER-UPPER WHITE
- UPPER DRY
- UPPER KAWEAH
- UPPER KERN
- UPPER KING
- UPPER LOS GATOS-AVENAL
- UPPER POSO
- UPPER SAN JOAQUIN
- UPPER TULE
- VENTURA

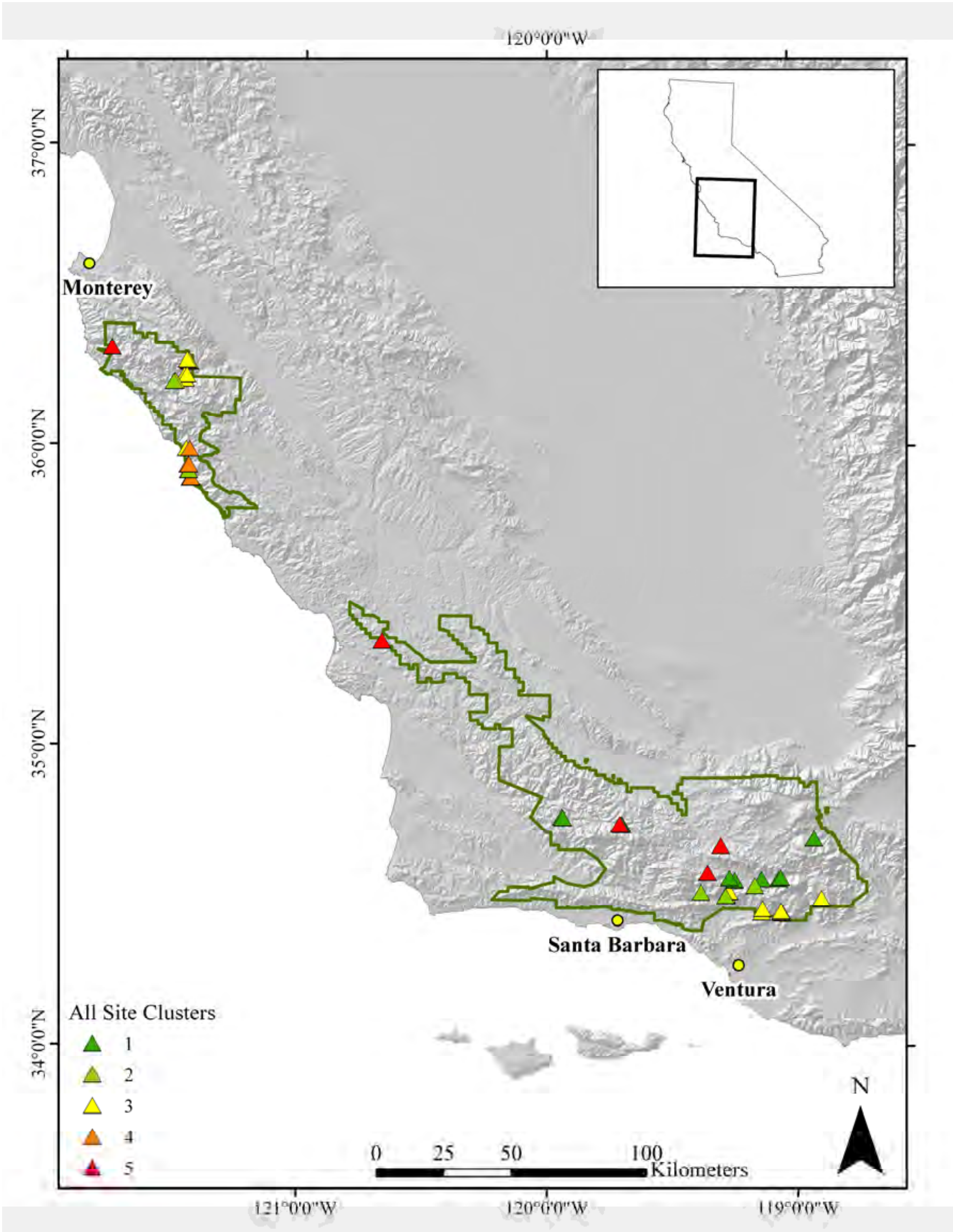


Appendix 7d. Aquatic ecoregions are categorized by differences in landuse, landsurface form, vegetation, and soil. All of the study reaches are located within ecoregion 8, Southern California Mountains.

8 Cluster Maps

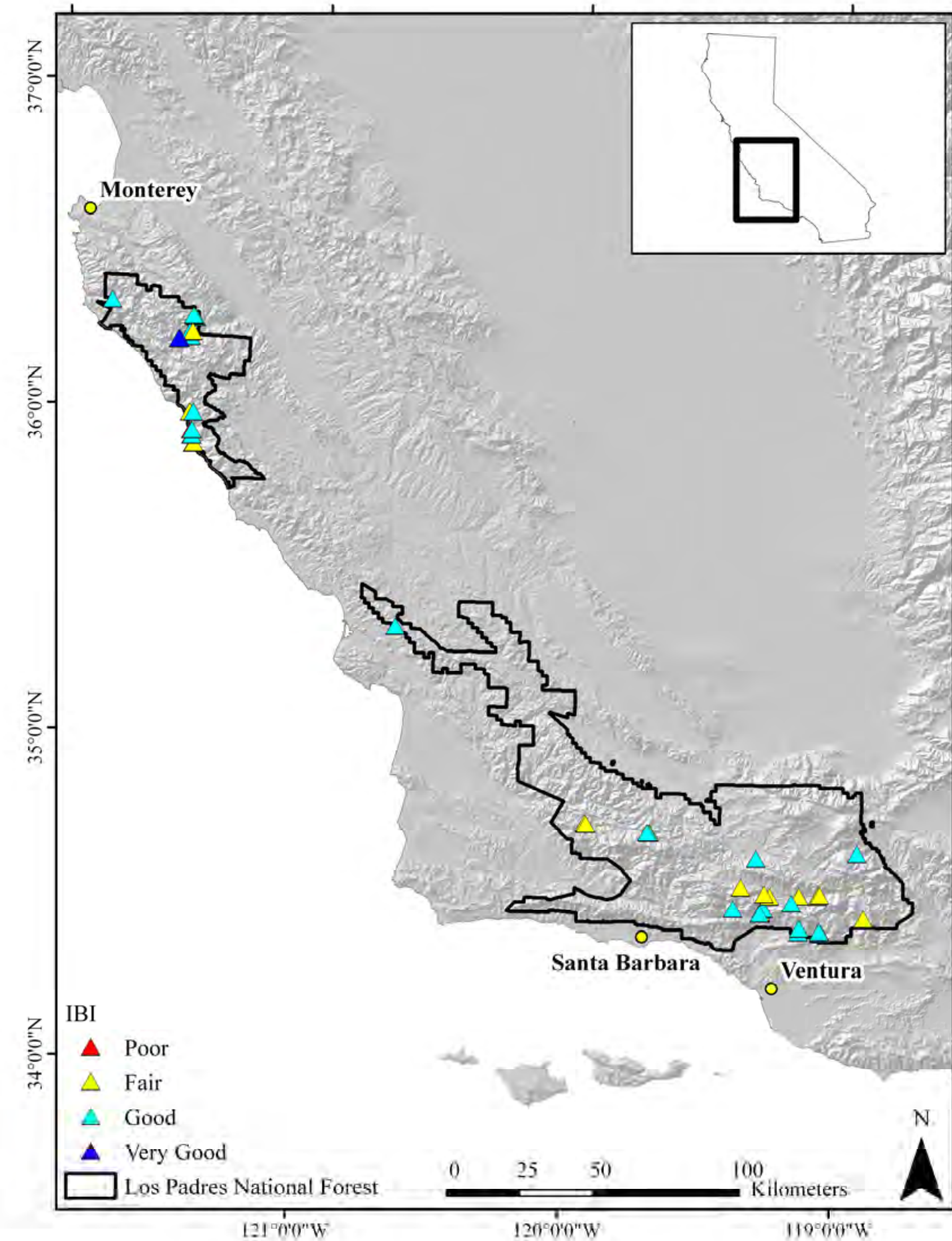


Appendix 8a. Refernce site clusters generated by the RIVPACS development.

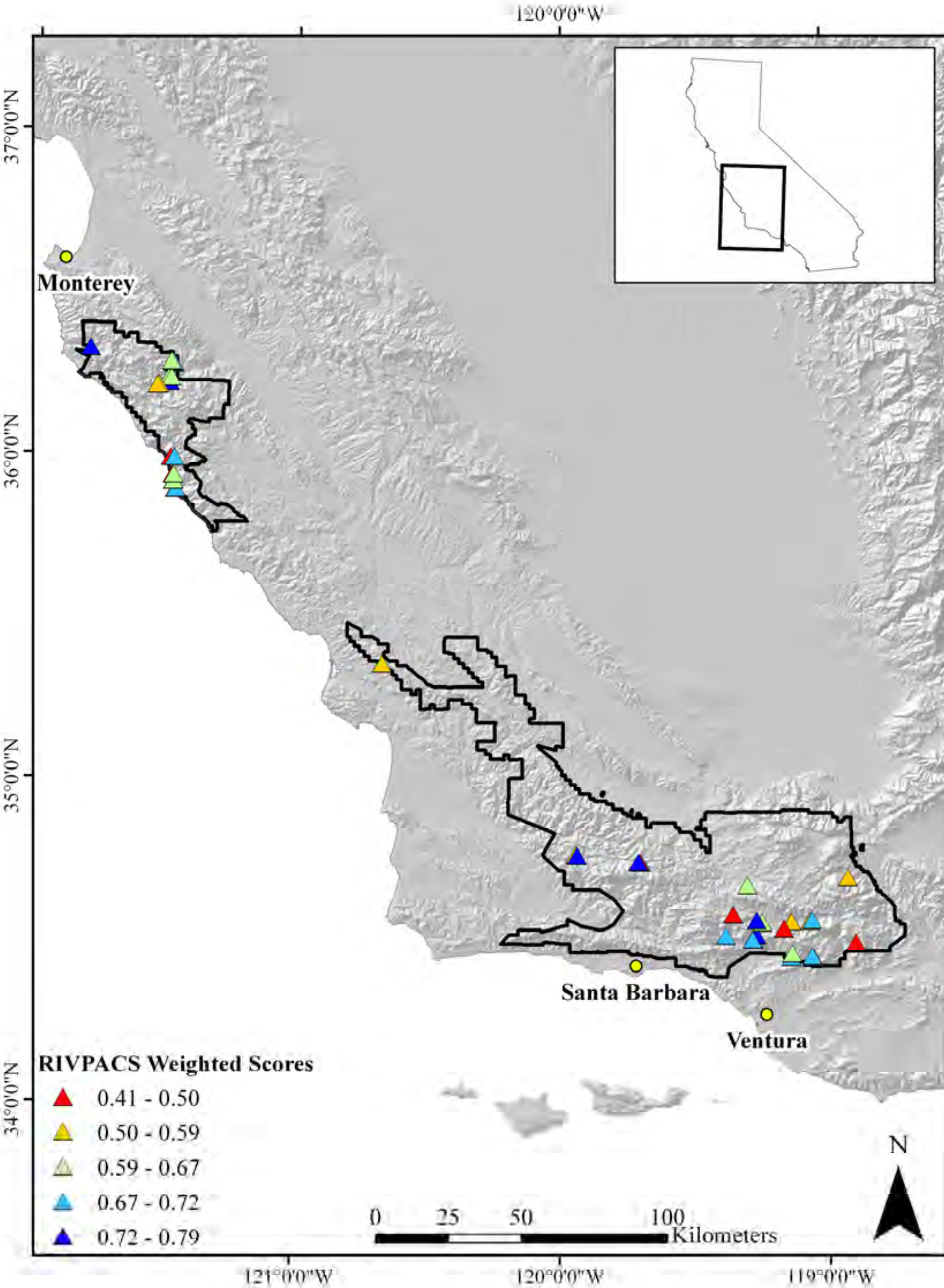


Appendix 8b. Cluster classification of BMI samples. Sites are colored according to their majority cluster (e.g. if Sample X is 70% cluster 1 and 30% cluster 4 it is colored as cluster 1).

9 Score Maps



Appendix 9a. IBI scores for each samples from the SoCAL IBI.



Appendix 9b. Weighted RIVPACS scores for each sample location. Higher scores represent better BMI health, blue, while lower scores show more impacted BMI communities, red.