

Final Report for “Creating and Index of Biological Integrity for Coastal Sage Scrub: A tool for habitat quality assessment and monitoring.”

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EXECUTIVE SUMMARY	4
INTRODUCTION	6
Monitoring in NCCP Reserves.	7
<i>Methods of assessing ecological condition</i>	7
<i>Existing approaches for measuring habitat quality</i>	7
<i>Geographic Information Systems (GIS) based approaches to measuring habitat quality</i>	9
Index-based indicators of ecological condition.....	10
<i>Step 3. Metric selection and statistical analyses.</i>	14
<i>Step 4. Validation and verification.</i>	16
<i>How disturbance is measured in IBIs</i>	17
IBI methods in Southern California	19
Spatial Scale of the IBI	19
Measuring disturbance	20
Methods.....	21
<i>Study areas.</i>	21
<i>Overall sampling effort.</i>	22
<i>Grid design and timing.</i>	27
<i>Taxa specific sampling</i>	28
<i>Data Analysis and IBI development techniques</i>	31
RESULTS	34
Section 1: Overall diversity trends and vegetation relationships	34
Patterns in species richness across taxa: Do taxa respond similarly to disturbance?	34
Patterns between native shrub cover and exotic invasion.....	35
Overall taxa response to levels of exotic invasion.	36
Section 2. Exploratory Data Analyses for responses to increased exotic cover, metric selection and scoring.....	38
Methodological overview:	38
<i>Ants.</i>	39
<i>Arthropods.</i>	40
<i>Birds.</i>	41
<i>Herpetofauna.</i>	43
<i>Mammals.</i>	45
<i>Vegetation.</i>	47
An IBI for CSS:.....	49
<i>Metric Clustering Methods and results</i>	52
<i>Year 1</i>	52
<i>Year 2</i>	55
<i>Both years</i>	57
<i>ANOVA results on Clusters.</i>	58
<i>Year 1.</i>	59
<i>Year 2.</i>	60
<i>Both years.</i>	60
<i>Summary.</i>	61
CONCLUSIONS AND FUTURE WORK.	61
Is an IBI for CSS possible?	61

Is an IBI for CSS ready for use?63
 CSS-IBI Validation.....63
 IBI Refinement.....64
 Is an IBI for CSS necessary?.....66
 Additional items worth mentioning.....69

EXECUTIVE SUMMARY

This report is the final deliverable for a project between Dr. Jay Diffendorfer and the California Department of Fish and Game (CDF&G) “*Creating and Index of Biological Integrity for Coastal Sage Scrub: A tool for habitat quality assessment and monitoring* “. The project aimed to develop an Index of Biological Integrity (IBI) for Coastal Sage Scrub (CSS) useful in NCCP monitoring programs.

The report contains three main sections and 3 appendices. We first review current methodology for assessing habitat quality, describe how IBI’s are developed, and describe the methods we used to develop an IBI for CSS in the introduction. In the results, we describe the development and testing of the IBI. In the conclusion we speak frankly about the implications of our research, the limitations of the IBI and the role it can play in CSS monitoring programs. The appendices support the main document.

Introduction. Currently, habitat quality assessment employs single species approaches such as indicator or umbrella species, GIS-based methods, or multi-metric approaches such as IBIs. Assessing habitat quality over large spatial scales for use in reserve design is typically done by a variety of GIS-based approaches. However, at smaller spatial scales, the reliability of the GIS-based approaches declines. At these smaller spatial scales, multi-metric indices, such as the Index of Biological Integrity have been successfully developed.

IBI’s are developed using a four-step process: establishing biological dose-response curves, developing scoring systems, selecting metrics and statistically analyzing the IBI, and IBI validation or verification. In the report, we explain each of these steps in detail giving examples from successful IBIs. Our study focused on steps 1 through 3 because it was not appropriate to include validation in the original contract until we had evidence an IBI was feasible.

Measuring disturbance independently of the biological data is a critical aspect of IBI development. Measurements of disturbance are often specific to the site of interest and to the type of disturbance. Professional judgment often categorizes sites between low, moderate and severely impacted. In our case, we performed a literature review (Appendix A) and concluded historic land use such as agriculture, grazing, and mechanical disturbance, as well as short fire return intervals, result in a loss of shrub cover and an increase in exotic plant invasion in CSS. Based on this review we used the levels of exotic plant cover to select our study plots and create a disturbance gradient.

Our original sampling strategy of 38 sites, each sampled four times a year was impacted by fires at some of the reserves so new plots were added as old plots burned. The timing of the 4 sessions was optimized to gain the maximum amount of information across all taxa sampled. At each site, a 50 x 50 m grid sampled vegetation, small mammals, birds, arthropods, while most plots were adjacent to a herpetofauna sampling array (trapped by United States Geological Service personnel). A variety of statistical methods, from exploratory data analysis, to ANOVA’s, to clustering algorithms were used to identify IBI metrics and test the final IBI.

Results. The fire-induced changes to the study forced us to create three separate datasets: plots sampled in year 1, plots sampled in year 2, and plots sampled in both years. Of the original 38 plots, 26 remained unscathed through the 2 years. We removed species, genera, or orders with low numbers of observations or species whose ranges did not overlap all our sites. We screened the remaining 370 species, using all 3 datasets and developed a 16-metric IBI. The IBI included metrics from Ants, Arthropods, Birds, Herpetofauna, Mammals, and Vegetation. Our level of confidence regarding the species included in each metric varied depending on the level of pre-existing information, our ability to discuss our findings with taxa specialists, and the strength of the empirical relationships across the disturbance gradient.

A combination of statistical approaches indicated the IBI for CSS effectively distinguished disturbance levels and was robust. A cluster analyses indicated up to 5 categories of disturbance could be identified by the IBI and the ANOVA's indicated average IBI scores across these categories often differed. We conclude the IBI can discriminate 4 or 5 levels of disturbance in CSS. The 16 metric IBI also showed a similar response to the Absolute Exotic Cover gradient across all three datasets, indicating robustness across drought and non-drought years and similar performance with 1 or 2 years of data.

Conclusions. We centered the conclusions around three main questions. We answered "Yes" to "Is an IBI for CSS feasible?" as our results indicate the IBI performs well even without additional refinement. We also asked "Is an IBI for CSS ready?" and answered "No" because the IBI must be validated (the critical final step in IBI creation). We also discuss issues related to refining the IBI by developing additional certainty and confidence regarding the species we included in the metrics and by considering better or more efficient sampling strategies. Finally, we pose the open-ended question "Is an IBI for CSS necessary?" as a method for highlighting the limitations and strengths of the IBI. We note the IBI, as developed now, measures disturbance caused by historic land use at scales of $\sim 2500\text{m}^2$, and may not be sensitive to other forms of disturbance occurring at larger spatial scales such as habitat fragmentation, isolation and patch size. We discuss ways in which multiple IBI sampling locations can be combined to measure status and trend of ecological condition at larger scales. We also discuss how the IBI can be very effective at measuring post-fire recovery and the impacts of management activities such as weeding or restoring CSS.

INTRODUCTION

Human settlement and population growth during the last century have increased displacement and fragmentation of southern California's (S.CA) natural systems. Coastal sage scrub's (CSS) regular occurrence on coastal lowlands makes it particularly vulnerable to both agricultural and urban displacement. Estimates of the extent of coastal sage scrub loss range from 36% to 85% (O'Leary 1990, 1995), with most experts agreeing with higher levels of destruction.

Efforts to conserve remaining CSS include complex, large-scale efforts at reserve design and management. These efforts, driven almost exclusively by Habitat Conservation Plans and the Natural Community Conservation Plan, are precedent setting and have received recognition at the highest levels of government (Babbitt 1999).

The success of the NCCP program will rest almost solely on the ability of local reserve managers to maintain the internal biological integrity of each reserve. Three main issues make this task difficult. First, the heterogeneous mix of land use patterns surrounding most of the NCCP reserves influences their internal dynamics by causing edge effects, altered fire regimes, and mechanical disturbance. Second, historical disturbance caused by grazing, altered fire regimes, agriculture and invasive species exist, to different degrees, in all reserves. Third, strong public pressure exists to use these lands as parks instead of reserves for endangered and threatened species. Managing these reserves, despite historic and continuing forms of disturbance, is possible if managers have the high-quality data necessary to make science-based decisions. Managers need to understand, for example, the responses of CSS food webs to both disturbance and feasible management actions.

The overarching goal of the "IBI project" (i.e. the California Department of Fish and Game (CDFG) grant to Dr. Diffendorfer "Creating an Index of Biological Integrity for Coastal Sage Scrub: A tool for habitat quality assessment and monitoring") was to investigate a framework for collecting, analyzing, and interpreting multi-taxa monitoring data in CSS habitats. Our goal was to create a method for comparing the biological integrity of different CSS sites or the same site before and after management actions. Developing a successful IBI is one approach towards creating and implementing a comprehensive, economically, and biologically relevant monitoring program in CSS throughout the NCCP reserve system.

We particularly point out our role in this process: We were not advocates for the implementation of an IBI in CSS monitoring when the project began, but we knew of examples in aquatic and terrestrial systems where IBIs had become effective tools in large-scale monitoring programs. We thought it was worth investigating whether a cost-effective CSS-IBI could be developed for use in NCCP monitoring. We view ourselves as scientists studying the possibility of implementing an IBI in CSS, rather than scientists insisting on its implementation. In this report, we primarily describe our methods and results, but in the discussion section, we do make recommendations, based on both our findings and on our opinions, about large-scale monitoring in CSS.

As will become painfully obvious to the readers of Appendix B, we wanted to honestly portray the data and the process we used while creating the IBI metrics. As such, we present large numbers of the figures we used while doing exploratory data analyses in Appendix B. These represent a little more than one of 3 datasets we compared while working on the metrics. Any detailed reader will recognize 'borderline' cases where a species might have been, but was not included in a metric or vice-versa. Tough calls and borderline cases are inherent in the data we collected and we welcome any thoughtful comments that may improve the metric selection process.

Monitoring in NCCP Reserves.

NCCP reserves, although legally driven by state and federal Endangered Species Acts, are considered multiple-habitat, ecosystem-based reserves. Monitoring is a fundamental aspect of management in the NCCP and required by law. Although monitoring and studying individual endangered species is necessary, given the ecosystem-based focus of the NCCP, so too is a method for measuring the ecological condition of the entire system. We were funded to investigate the potential of such a method for CSS habitat.

We note our focus was not to describe an overarching framework for an NCCP monitoring program. Other authors have suggested conceptual models for high quality monitoring and adaptive management programs and the United States Fish and Wildlife Service (FWS) described monitoring guidelines in their HCP handbook. Instead, we investigated the possibility of developing just one, of many possible tools, managers can use while implementing a reserve-wide monitoring program.

Methods of assessing ecological condition

Historically, most land stewards managed one or a few species at a time and typically targeted those of particular concern (e.g. sport fish, game animals, livestock, endangered species). Land stewards now attempt to manage multiple-species, entire ecological communities of organisms, or simply put, focal habitats (such as Coastal Sage Scrub).

No standard protocol exists for selecting and developing reliable methods of representing ecological condition for a diverse array of species or ecosystems. This lack of uniformity should not be surprising given the diversity of ecosystems, complexity of species interactions, and multiplicity of goals among land management organizations. Indeed, given the nuances associated with each ecosystem, monitoring protocols and approaches will likely differ considerably across systems.

However, without an accepted protocol for characterizing ecological condition, land managers face significant disadvantages when legislatures, courts, or the general public ask them to justify their management decisions. Public sentiment, as well as most federal and state environmental statutes, demands land managers base their actions on the best-available science. Objective and accurate methods for measuring ecological condition would help land managers communicate the logic and evidence behind their decisions.

We were tasked with describing “1) a method for estimating human disturbance in CSS habitat including a 2) review of the IBI literature.” In addition, the report was to “summarize how estimates of disturbance are used in the development of IBIs and how disturbance has been quantified in other systems.”

Here, we briefly review the approaches land stewards have commonly used to assess both ecological condition and the degree to which changing land-use may affect ecological condition. The literature related to this topic is enormous and system specific. Therefore, we limited our focus to assessment methods currently being used in management while realizing the development of ecological condition (often called “habitat quality”) as a concept is not yet synthesized by science historians. We end this section by describing why we chose to use an index-based approach for measuring ecological condition. In particular, we discuss the index of biological integrity, or IBI, an empirically tested approach for measuring habitat quality (Karr 1987, 1991a, 1992, Karr and Chu 1999).

Existing approaches for measuring habitat quality

In conservation planning, managers sometimes seek effective shortcuts to conserve biodiversity. Commonly, this involves managing for a few species and assuming protecting these surrogates will confer protection on other species in a region. In a review of surrogate species in conservation management,

Andelman and Fagan (2000) described the three most prevalent categories as: flagships (i.e. charismatic species that attract public support), umbrellas (i.e. species requiring large areas of habitat, whose protection serves to protect many co-occurring species), and biodiversity indicators (i.e. sets of species or taxa whose presence may indicate areas of high species richness). Other categories of surrogate species commonly used include: big carnivore, habitat generalists, habitat specialists, high age at first reproduction, long-lived, health indicators, population indicators, population turnover, residency status, etc. (Landres et al. 1988, Caro 1999, Andelman and Fagan 2000).

One of the simpler approaches to conservation management is to focus on a single keystone species, as a proxy for habitat quality. The rationale behind keystone-based measures of habitat quality is straightforward: habitats lacking keystone species must be significantly altered and of lower quality. Keystone species are typically considered those species which, when removed from an ecosystem, cause a cascade of changes in abundance of other species (Paine 1966, 1969). A classic example of a keystone species is the predaceous sea star, *Pisaster ochraceus*, whose presence enhances species diversity by allowing other invertebrates to colonize rocky substrates that would otherwise be dominated by barnacles and mussels.

Given the binary nature of the metric (i.e. keystone present or absent), this concept has limited utility in management. For example, many management units might lack historic keystone species, yet still support large amounts of diversity. Furthermore, the concept of keystone species has been thoroughly criticized by Hurlbert (1997), who considers it operationally impossible to define (i.e. pine trees might be considered a keystone species in pine forests as much as Grizzly bears) and thus meaningless. Mills et al. (1993) critiqued the use of keystone species in part because although it has applications for conservation and food web theory, it remains largely undemonstrated in nature. In fact, Mills et al. (1993) argue the formalization of the term in laws and policy guidelines would do more harm than good, and the use of this broadly applied and poorly defined term does not allow for the practical use of keystone species in management.

Another common approach is to use a single species as an indicator of biological diversity, species richness or composition. An indicator species is "an organism whose characteristics (e.g. presence or absence, population density, dispersion, reproductive success) are used as an index of attributes too difficult, inconvenient, or expensive to measure for other species or environmental conditions of interest" (Landres et al. 1988). This approach to habitat conservation has had mixed success. The probability a single species could serve as a surrogate measure of habitat quality given the complexity of natural systems is small. In CSS, indicator species of conservation concern could not be assumed to be indicators of hotspots for either bird or small-mammal richness (Chase et al. 2000). Furthermore, Chase et al. (2000), found a poor correlation between the presence of bird and mammal species, suggesting that maximizing diversity of one taxa, would not maximize diversity of another.

A related single-species approach, the umbrella species, attempts to manage for multiple species by conserving a species "with large area requirements, which if given sufficient protected habitat area, will bring many other species under their protection" (Noss 1990). Examples of umbrella species proposed to protect other species within their ecosystems include: spotted owls (Franklin 1994), desert tortoises (Tracy et al. 1995), black-tailed deer (Hanley 1993) and butterflies (Launer and Murphy 1994). However, landscapes managed for a single species may fail to meet the needs of other species in a complex ecosystem (Franklin 1994). For example, management plans for large-scale forest reserves to protect the umbrella species, Northern Spotted Owl, did not protect aquatic ecosystems, Marbled Murrelets, and failed to include a large portion of late-successional forests (Franklin 1994).

Conservation based on surrogate species is a common approach to habitat and ecosystem management because managing for a single species is easier and more practical than managing for

complex ecosystems, which require monitoring several biotic and abiotic factors. However, the ability of surrogate species to protect other species in the region is considered inadequate and cost-ineffective by some (e.g., (Franklin 1993, Lambeck 1997, Andelman and Fagan 2000). Debates on how to manage habitats and ecosystems have led to the continued use of both single and multiple species approaches.

In CSS, no evidence suggests either umbrella, or indicator species approaches are effective management tools. Given the fine scale variation in habitat types, the limited distributions of many species, and the small sizes of many reserves, it is unlikely either approach will adequately allow effective management. Furthermore, historic disturbances in most reserves and continuing levels of exotic invasion and disturbance concerns us because management based on maintaining populations of a few focal species may not guarantee other species will persist.

Geographic Information Systems (GIS) based approaches to measuring habitat quality

A number of GIS-based methods exist for modeling or predicting habitat quality or ecological condition. All of these methods rely on either known, or assumed relationships between focal species (typically an animal) and habitat (typically vegetation, but may also include other features such as snags, rock outcroppings, urban edge, etc). These methods model species distributions using the habitat relationships and maps of existing land-types, and are typically used to predict both the range of a single species or group of species as well as a map of habitat quality for the species of interest. These types of models are typically called Habitat Suitability Indices (HSI) (Terrell and Carpenter 1997). However, once the predicted species maps are in place, they are often over-laid to predict areas of high species richness and guide reserve design (Habitat Evaluation Procedures – HEP), or discover gaps in the extent of reserve systems (GAP analyses). More computationally elaborate methods allow the weighting of various map layers and rules for inclusion in a potential reserve. Mathematical algorithms then predict a potential reserve design across a landscape that optimizes the reserve selection given the constraints originally set (Chikumbo et al. 2001, McDonnell et al. In Press).

HSI models are best used as hypotheses of species-habitat relationships as opposed to causal functions (Morrison et al. 1998). HSI is defined as a linear index representing the capacity of a particular habitat to support a focal species (US Fish and Wildlife Service 1981). HSI's combine a suite of variables thought to correlate with the population size of a species or group of species. Variables might include the number of downed logs, old snags, percent cover of a particular vegetation type, etc. The index typically scales from 0-1 and is the ratio of actual habitat conditions compared to optimal habitat conditions for the species in a specified unit of measure (a km² for example). Optimal habitat is defined as that combination of variables resulting in the maximum carrying capacity. The HSI model produces an index assuming a linear relationship between HSI value and carrying capacity (i.e. units of biomass/unit area or units of biomass production/unit area; (US Fish and Wildlife Service 1981). The Bureau of Land Management, US Forest Service, and US Fish and Wildlife Service among others have developed several HSI models for different species (e.g. salmon, red tailed hawk), as technical notes to serve as the basis of management decisions. The California Department of Fish and Game (CDF&G) maintains the California Wildlife Habitat Relationship System a system of HSI models for 675 vertebrate species (CDF&G 1999).

The Habitat Evaluation Procedure (HEP) is another hybrid single-species / multiple-species approach. It combines the Habitat Suitability Index (quality of the habitat) with the total area of available habitat (quantity). The HEP is a collection of procedural and habitat suitability index models for fish and wildlife species (US Fish and Wildlife Service 1980). The models predict changes to carrying capacity of habitats supporting the particular species of concern. Some examples of applications of HEP include: assessment of timber-sale impacts on wildlife habitat in the Sierra Nevada, California (Doering and Armijo 1986), as well as developing a model for rocky mountain bighorn sheep (Smith et al. 1991). This

latter model "combined (1) a quantitative assessment of bighorn range to determine if there are adequate quantities of resources to support a minimum viable population of bighorn sheep, and (2) a qualitative assessment of a range to predict the probable density of bighorns the range can support." HEP guidelines suggest selection of indicators "can be arbitrary or according to some ranking scheme," where the "availability of habitat data" is used as a component of the ranking scheme (US Fish and Wildlife Service 1980).

HEP and HSI rely on evaluation species as indicators of habitat quality and can be used to predict future impacts to habitat quality. Morrison et al. (1998) argue the strength of HSI and HEP "lies in documenting a repeatable assessment procedure and providing an index to particular environmental characteristics that can be compared with alternative management plans." However, there are several critiques on the use of these models in habitat and ecosystem management. (Landres et al., 1988), for example, suggest the arbitrary nature of selecting indicator species based on availability of habitat data compromises the ability of the indicator species to reflect habitat quality. In addition, several reviews of HSI and other habitat-relationship models of birds and mammals have shown large deviations from species habitat requirements and model assumptions (Dedon et al., 1986; Raphael and Marcot, 1986; Stauffer and Best, 1986). Malanson and Westman (1985) argue HSI models developed from single-species experimental data assume optimum habitat for a species in isolation (i.e. absence of competition) is equal to optimum in the field (i.e. competition possibly present).

GIS based methods of assessing habitat quality can effectively aid decision-making in situations covering large spatial scales at relatively low resolutions. Implementation of these approaches is common in reserve design, or landscape-scale habitat conservation planning. For example, HEP's are used in a number of the NCCP plans to predict total loss of habitat for target species. Even more recently, optimization-based GIS approaches were used in the North County MSCHP to model particular species distributions and assess how well the reserve protected a species.

GIS based methods have limitations as tools in reserve management and assessing ecological condition. First, these techniques are not adequate to assist in management decisions requiring fine spatial resolution because the estimates of habitat quality they produce are limited by the resolution of the data. Second, GIS methods require tremendous amounts of technical expertise, GIS software/hardware, and labor to enter, error-check, and manage data. Thus, using such methods to assess short-term, or small-scale management actions is highly problematic. Furthermore, because GIS-based models of habitat quality use wildlife-habitat relationships, some of which are based on assumptions and best professional estimates, they can give little indication of the detailed processes causing changes in habitat quality. Indeed, the habitat quality maps derived from GIS-based approaches are, in reality, spatially-explicit working hypotheses requiring additional testing and study to verify both the patterns predicted and the processes generating those patterns (Beutel et al., 1999).

Index-based indicators of ecological condition

At smaller spatial scales, detailed sampling allows the accumulation of fine-scale data and has led to numerous efforts to develop indices or scoring systems containing information about ecological condition. An increasingly popular multi-species approach is the Index of Biological Integrity (IBI), which evaluates ecological condition by combining a series of empirically derived and tested curves representing species responses to disturbance. In this section, we familiarize the reader with the Index of Biological Integrity (IBI), the function it plays, and how it is developed. This general background is necessary to fully understand the goals of the project.

What is biological integrity? A concept frequently mentioned in this report and in large-scale, ecosystem management (as is occurring in the NCCP) is ecological (Norton 1992), or biological integrity (Frey 1975, Woodley et al. 1993, Angermeier and Karr 1994). Biological integrity, as defined by Frey (1975), is “*the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity and functional organization comparable to that of natural habitats in the region.*” Biological integrity incorporates not only components of systems (i.e. species), but also the processes sustaining those components (i.e. predator-prey dynamics, succession, population regulation, etc). In addition, biological integrity is a relative measure. A site’s ecological condition is based on how far away it is from a typical, intact system.

How is biological integrity measured? Indices of biological integrity are used to measure biological integrity. These indices incorporate many attributes of the biological community, which can encompass multiple functional groups (i.e. ground nesting birds, terrestrial insects), trophic levels (producers, herbivores, meso and top predators), or unique species to evaluate human disturbance effects on habitats (Karr 1991b). The IBI was first developed for fish communities in mid-western streams of the US (Karr 1981, Fausch et al. 1984), and is predominately used to evaluate aquatic ecosystems¹.

Scientists develop IBI’s by sampling across a gradient in anthropogenic disturbance and quantifying how the system responds. Thus, IBI’s are based on empirical relationships showing the response of multiple species in a system to varying levels of disturbance. Scientists who developed the IBI realized ecosystems vary naturally across space and through time yet managers and decision makers were more interested in the added variation caused by human disturbance than they were in natural levels of variation. By specifically sampling across the gradient of disturbance when developing the IBI, we attempt to extract and measure this additional, human-caused variation.

IBIs for different systems often vary in details, such as the taxa used, or the metrics included, but the general approach used and the overall goal is similar. IBIs are developed using 4 general steps. We detail these four steps below, using Figure 1 as a reference.

Step 1) Empirical studies develop species or guild response curves by sampling across gradients of disturbance. These empirical relationships are similar to dose-response curves used in toxicological studies and show how specific metrics respond to disturbance.

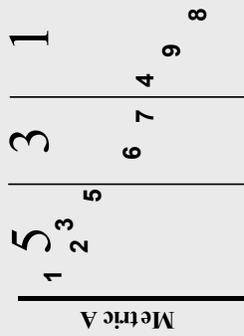
Step 2) A scoring system is developed for combining the various response curves into an easily understandable methodology. A standard approach used in most IBIs is to categorize each response curve into 3 sections, giving them a score of 1, 3 and 5, where a 1 indicates high levels of disturbance.

Step 3) A subset of metrics is chosen for the final IBI. In this step, metrics are screened to avoid including repetitive metrics measuring similar responses to disturbance. In addition, statistical analyses are performed to determine the best set of metrics to include in the IBI and the number of disturbance levels the IBI can distinguish.

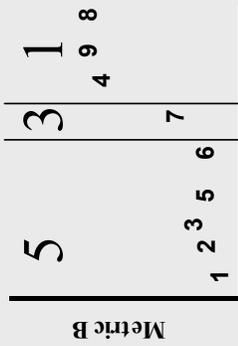
Step 4) Once the IBI is developed, field verification of the proposed scoring system must occur. Classification of new sites based on their level of disturbance is used to predict an IBI, then field studies generate an observed IBI, and the two IBI values are compared.

¹ Over 91% of biological assessments using IBI are in aquatic systems, according to a Biosis search spanning 1985-2001.

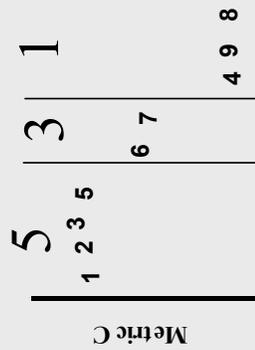
Step 1. 3 response curves. Data for each metric are collected at 9 sites specifically placed so they span a gradient of disturbance.



Metric A shows a linear response to disturbance.



Metric B is useful for separating low from high disturbance.



Metric C shows a sharp, threshold between disturbed and undisturbed sites, with little change within a disturbance category

Step 2. 5, 3, 1 A standard scoring system gives metrics a 5 at low disturbance and a 1 at high, with 3 the intermediate value.

Step 3. Scores for each metric are combined across a site to develop an overall IBI score. Statistical analyses determine the ability of the final IBI to distinguish levels of disturbance.

Site	Metric A	Metric B	Metric C	IBI Score
1	5	5	5	15
2	5	5	5	15
3	5	5	5	15
4	1	1	1	3
5	5	5	5	15
6	3	5	3	11
7	3	3	3	9
8	1	1	1	3
9	1	1	1	3

Step 4. Field validation. After field validation, unknown sites can be sampled and their overall integrity measured relative to the known sites.

Figure 1. The four general steps used in IBI creation and example metrics.

Perhaps the most critical point to emphasize, IBIs begin with *empirical relationships* derived from monitoring data showing how a group of species in a system respond to varying levels of disturbance. An IBI simply combines a number of these empirical relationships into a standardized, repeatable, and biologically interpretable scoring system. The strength of IBIs is their ability to incorporate many attributes of the biological community into a measurable response to disturbance. We detail the 4 steps in IBI development below, and illustrate these in Figure 1. However, for additional information refer to (Karr et al. 1986, Kerans and Karr 1994, Fore et al. 1996, Brooks et al. 1998, O'Connell et al. 2000).

Step 1. Establishing biological responses across disturbance gradients. The first step in developing an IBI is to establish empirical relationships between biological metrics and human caused disturbance. This step consists of two parts. First, a gradient of human disturbance must be developed and reference sites selected. The sites can either be ranked along the disturbance gradient based on a combination of abiotic factors, or disturbance can be measured directly using specific variables. In streams, abiotic factors include water quality variables, levels of urbanization or agriculture along a stream, and distance downstream from a point source of pollution. The ranking is often qualitative and can include a three-category scale of low, moderate, and severely disturbed sites. We discuss our methods for choosing a disturbance gradient and measuring disturbance in more detail below. Having established a gradient of human disturbance, the next step is to sample the biota. When combined across sites, a scatterplot showing the value of the metric relative to the disturbance gradient is created and indicates if and how particular species or group of species respond to disturbance. Figure 1, illustrates three metrics each having a different response across a disturbance gradient.

Step 2. Scoring systems. Once empirical relationships between disturbance and biological metrics of the system are established, a scoring system is developed to allow ranking of the sites and a method of comparison. In general, each site gets a score for each metric and then the sum of the scores across all metrics is used to rank sites.

Scoring systems are arbitrary and vary across investigators, the ecosystems where IBI's are developed, and the taxa used in the IBI. For example, in many aquatic IBI's, and in the methods described by Karr and Chu (1999), scoring entails trisecting the range of values of the metric and assigning a 1, 3, or 5 to each section, with 5 representing the least impacted site. However, O'Connell et. al (2000) used a ranking scheme of 1, 2, and 3 in their bird-based IBI. They ranked sites with highest occurrence of specialist guilds, reflecting highest biological integrity, with a 3, next highest a 2, and the lowest a 1. Despite the scoring system used, if it is reported, highly transparent, and the scale equal across disturbance categories, then it should not diminish the value of the methodology. Figure 1 illustrates a '1,3,5 scoring system' on each of the 3 metrics.

As a site's ranking is dependent on undisturbed sites, selecting reference sites plays an important role in developing an IBI. Biological communities at a disturbed site are compared to communities at a relatively undisturbed "reference site". However, because unimpaired ecosystems may no longer exist, an estimate of expected biological integrity in ecosystems is often based on "least-impacted" conditions (Davis and Simon 1995). It is these least-impacted sites that represent one end in the gradient of

disturbance. Suggested criteria for reference sites include: 1) easy accessibility for monitoring over multiple years (e.g. usually public lands), 2) land types and landscape settings representative of those commonly impacted during the permitting process, and 3) random selection (Brooks et al. 1996).

Step 3. Metric selection and statistical analyses.

A study conducted across sites varying in the degree of disturbance will generate massive amounts of data. A successful IBI does not need to include all possible metrics collected in a study and so exploratory data analysis is used to 'pare down' the data into a suite of useful metrics. Karr and Chu (1999) recommend an IBI include three categories of metrics: species richness and composition, trophic composition, and taxa abundance and condition.

The justification for the categories is not arbitrary as each of the categories makes up a unique aspect of the biological system. Species richness metrics include information about the make-up or composition of a system. Typically, metrics include overall species richness of tolerant taxa, or the number of invasive species. Trophic composition includes information about food web complexity, which can often decline in degraded systems because species interactions are altered. Thus, trophic composition metrics indirectly measure the integrity of consumer-resource interactions and include metrics such as the number of herbivores and/or carnivores, and the presence/absence of top-predators. Finally, taxa abundance and condition metrics reflect information about individual and population level processes. Metrics may include the proportion of individuals with deformities, average body condition of key species, or the relative abundance of particular taxa.

In selecting metrics for an IBI model, several criteria must be met. First, each candidate metric's sensitivity to human disturbance should be evaluated. Explicit hypotheses on how each metric responds to a particular disturbance should be tested. For example, a confirmed hypothesis that intensive cattle grazing in coastal sage scrub would decrease species richness of plants would demonstrate that plant species richness is a suitable biological metric for evaluating grazing disturbance effects on coastal sage scrub systems. This relationship between biological response (e.g. species richness) to different levels of disturbance (e.g. light, moderate, heavy grazing intensity) is an underlying principle in generating a "dose-response curve," where the biological attribute is plotted against a gradient of disturbance (Karr et al. 1986). For a successful biological metric, a clear relationship between the IBI and disturbance gradient should be detectable or obvious (Figure 1 and Figure 2).

Second, the metric should adequately distinguish sites with different levels of disturbance (Kerans and Karr 1994). Not only should plant species richness decrease in grazed areas, but species richness should have distinguishable responses to low, moderate, and heavy grazing intensity.

Third, the successful metrics should not be redundant. Because the IBI model is a multi-metric index, redundant metrics should be avoided. For example, declines in overall species richness may be caused just by declines in a suite of intolerant taxa. Including both overall species richness and the richness of the intolerant taxa would as 2 metrics effectively count the same biological response twice.

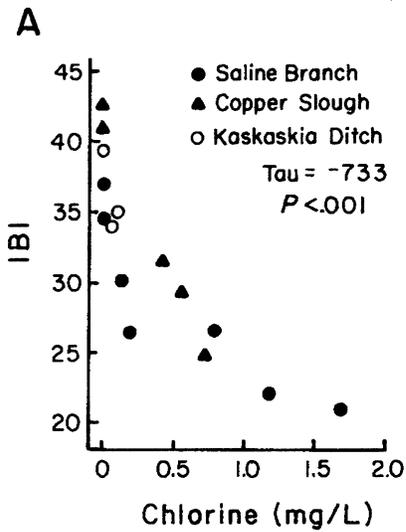
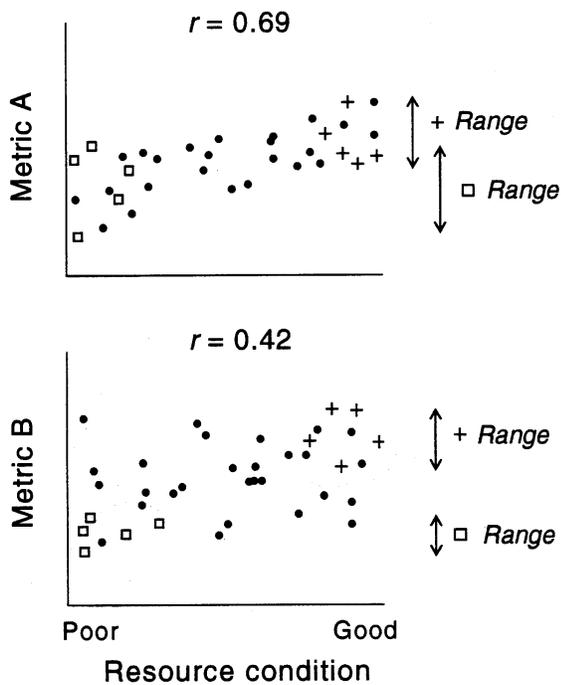


Figure 2. Index of biotic integrity (IBI) as a function of total residual chlorine content in three streams in east-central Illinois with wastewater inflow from standard secondary treatment with chlorination (from Karr, JR., R.C. Heidinger, and E.H. Helmer. 1985. Sensitivity of the index of biotic integrity to changes in chlorine and ammonia levels from wastewater treatment facilities. *Journal of the Water Pollution Control Federation* 57:912-915).

Fourth, a robust metric should correlate with more than one measure of disturbance (e.g. land use, soil condition, etc.) An IBI's ability to assess biological responses to human disturbance is strengthened by including in the model several measures of disturbance that can be independently quantified. If plant species richness were a robust metric, the metric would respond along the gradients of independent measures of livestock grazing, such as grazing intensity, soil compaction, and time since grazing.

Lastly, the metric should respond to disturbance despite natural temporal or spatial variability. One challenge to biological assessments is the inherent natural variations of populations, which may undergo statistically significant fluctuations even in undisturbed systems (McBride et al. 1993). The difference between statistically



significant and ecologically significant results is often clarified by graphing the biological attribute against the established gradient of disturbance to detect potential trends (Figure 3). Fore et al. (1996) argued it is more biologically useful to select metrics from plots of metric vs. a specific or cumulative measure of disturbance, than to rely on statistical tests that typically focus on organisms' abundance rather than their biology.

Figure 3. Hypothetical relationships between resource condition and candidate biological metrics. Metric A is more strongly correlated with resource condition (or r^2 is higher if using regression) than metric B, suggesting it is a

better metric. A more reliable test compares the metric's ability to distinguish between minimally disturbed (+) and severely degraded (square) sites (ranges noted by arrows). Note that moderately degraded sites (shaded circle) span the range between the extremes. Metric B is the more effective metric in spite of its smaller statistical correlation. (From Fore et al. 1996)

The process of selecting metrics and testing IBIs. A general methodology exists for selecting metrics given the framework described above. First, exploratory data analysis is performed with the goal of identifying potential metrics, then measuring their response to disturbance and their correlation with other potential metrics. Second, as metrics are chosen and a scoring system devised, statistical methods are used to determine the ability of the IBI to discriminate between levels of disturbance. These two steps may occur iteratively as initial metrics are placed in a scoring system, tested, and then metrics dropped or added to improve the IBI. The specific statistical tests or procedures used to select metrics and test an IBI vary across researchers and the systems they study.

For example, Kimberling et al. (2001) used nonparametric Mann-Whitney U tests to determine if metrics could distinguish undisturbed from disturbed sites and Spearman rank correlations to test for correlations between a metric and levels of disturbance. The authors also performed separate tests in each of the 2 years and only used metrics producing similar patterns in both years to increase the chance of choosing consistent metrics. Finally, they performed Discriminate Function Analysis (DFA) to determine if a multivariate statistical approach using disturbance types as categories would rank the sites similarly to the multi-metric IBI. In this study, two metrics were redundant so one was discarded. The DFA ranked sites similarly to the IBI.

O'Connell et al. (2000) used ANOVAs, and Spearman Rank Correlations to determine if metrics generated by their bird surveys varied in value across three categories of wetland rankings (high, medium, low quality) or if the rank of wetlands was correlated with the value of the metric. Furthermore, the authors used Cluster Analysis to determine the maximum number of site categories with statistically distinguishable bird communities. Metrics generated from the bird point count data varied across wetland categories, indicating they could be reliable indicators of disturbance. Furthermore, a number of large-scale variables measured using GIS explained significant amounts of variation in the bird community, suggesting both localized disturbances and larger-scale changes in landscape impact bird communities. The cluster analysis indicated the IBI could distinguish between five categories of disturbance with statistical confidence.

Finally, Fore et al. (1994) provide an excellent example and discussion of distributional considerations and methods for use when evaluating the statistical properties of an IBI. They demonstrate the effectiveness of bootstrapping and power analysis to determine the number of disturbance categories an IBI can distinguish.

Step 4. Validation and verification.

After demonstrating an IBI can discriminate sites with varying levels of disturbance, the final model should be verified. Verification is a process where the model's predictions are tested using a new set of data, independent of those data used to construct the model. In the case of an IBI, one can apply the IBI to an independent set of study sites representative of the study area. If possible, data on biotic and abiotic

condition are collected on the new sites using identical techniques to those used during IBI development. The biotic data from the new sites are then used to generate a ranking of the sites based on the IBI model. This ranking is then compared to the actual level of disturbance at the site derived from the abiotic variables. If the predicted ranking and the actual ranking are similar, the IBI has been successfully validated.

O'Connell et al. (2000) could not collect identical types of abiotic data for new sites; so instead of validating the model, they performed an ingenious analysis they called model "verification". First, they showed the original abiotic data used to rank sites by disturbance and generate the IBI was highly correlated with a new ranking system generated from a GIS using landscape variables such as land use and vegetation type near the bird transects. With this correlation in hand, they then ranked 126 new sites using the GIS methods. Thus, they now had a new set of 126 sites ranked using an algorithm highly correlated to the original ranking method that generated the IBI. They then sampled the bird community at the 126 sites and used the IBI to rank the sites. The IBI model was verified by a strong positive correlation between the ranking of the 126 sites using the GIS approach and the bird-based IBI.

We note that very few of the papers we reviewed on IBI development included the critical phase of model validation. Model validation is expensive, basically requiring a repeat of the same experiment in a new set of locations. Given that after the IBI is developed, statistical tests show the metrics vary across levels of disturbance and the IBI can distinguish sites, it is not difficult to see why the motivation to continue work on the model would wane. However, data collected during a one or two-year IBI development period may not represent the true state of a system, thus model validation through time is critical to a well supported IBI. *The IBI we present here has not been validated.*

How disturbance is measured in IBIs

Ecologists have long known the quality of habitats can be significantly altered by disturbance events. These disturbance events or processes are likely correlated with one another and rarely, if ever, impact an ecosystem in isolation. Disturbance events can be natural events such as fires or they can be anthropogenic, such as livestock grazing, residential and commercial development, and road construction. Disturbance adds to the levels of variability we see in ecosystems and may play a critical part in maintaining biodiversity. For example, some unique plants in prairie systems are found on prairie dog mounds or along the edges of "dirt bath" depressions made by bison. However, anthropogenic disturbances to ecosystems are typically much larger in magnitude and longer lasting than natural disturbances, having far more devastating impacts than historic disturbance regimes. Disturbance events occurring too frequently or fundamentally altering basic life-support systems (such as soil quality, hydrology, and light/shade) can have severe impacts on biological diversity. For example, abnormally high fire frequency in S. CA can result in native shrub replacement by fast growing non-native grasses (Zedler et al. 1983, Haidinger and Keeley 1993). Because the response of natural systems to disturbance is often specific to the ecosystem, habitat type and/or region, managers must select biological indicators (metrics) that respond to human disturbance in detectable and consistent ways.

Measuring the effects of disturbance on habitats and ecosystems in IBI studies requires 1) characterizing disturbance in terms of type (e.g. logging, fire, grazing, etc.),

and 2) establishing the scales at which disturbance will be measured (e.g. landscape, habitat, local). These steps are crucial in the development of an IBI because the disturbance gradient is the foundation on which biological responses are measured and sites are assessed. Measuring several levels of disturbance along a gradient allows for more sensitive detection of biological response to particular levels of disturbance, although in practice this is rare.

When assessing levels of disturbance, professional judgment is commonly involved. Professional judgment typically draws the line between minimally, moderately, and severely impacted sites. Measurements of disturbance are often specific to the site of interest and to the type of disturbance. The majority of disturbance evaluations are entirely qualitative in their assessments (e.g. low, moderate, highly disturbed sites; (Brooks et al. 1996) and only in rare cases do IBIs include a quantifiable scale of disturbance (Kimberling et al. 2001). Brooks et al. (1996) classified wetland sites under three categories (vegetation, water quality, and surrounding landscape condition) and considered a site pristine if it had high rankings for two of three categories, while being mildly disturbed for the third category.

In aquatic systems, disturbance levels are directly measured using both local and larger-scale variables. For example, disturbance is readily estimated at local scales using characteristics of the stream environment such as the presence/absence of channelization, impoundments, and stream bank vegetation, as well as water quality variables such as turbidity, dissolved oxygen, or specific contaminant levels. Larger scale variables include the amount of impervious area (i.e. concrete) within a set distance to the stream, the distance from an upstream point source, or estimates of land-use (urban/agricultural vs. natural) within the watershed. Thus in aquatic stream systems, ranking sites based on their level of disturbance is relatively straightforward. For example, Fore et al. 1996 measured human influences at two scales: 1) at a watershed scale based on USGS data on watershed area, % logged, and road length, and 2) at the riparian scale based on a resident hydrologist's professional assessments of the conditions of the riparian corridor, stream bed, bank stability, and influences of road building and logging on stream channels. They then plotted each measure of human influence against the other (e.g. riparian vs. watershed assessments) to confirm data consistency and identify unexpected outliers.

Measuring disturbance in terrestrial IBIs. Unlike aquatic systems, few IBIs exist for terrestrial systems. A number of authors have discussed the use of biological data as an indicator of either disturbance or biodiversity in terrestrial systems (Kremen 1992, Kremen et al. 1993, Oliver 1993, Weaver 1995, Oliver 1996, McGeoch 1998), yet few terrestrial IBIs exist. As such, there is not yet a prescribed method for measuring disturbance in terrestrial systems.

In terrestrial systems, local disturbances at fairly small spatial scales (1-10 m) can impact local vegetation and hence potentially alter the value of metrics used in an IBI. Thus, developing IBIs for use at spatial scales where many smaller scale management decisions take place; controlled burns, restored sites, invaded patches, road cuts or fire breaks (i.e. 1-100ha), would seemingly require detailed information about current and past disturbance at a fine spatial scale. Thus, ranking sites *apriori* based on non-biological data to develop the biological-dose response curves is potentially problematic because gaining such fine-scale information may be impossible for some sites.

Kimberling et al. (2001) were able to develop disturbance estimates by using past land-use histories for 25 sites at the Hanford Nuclear Reservation. They categorized each site coarsely into undisturbed or disturbed and further subdivided disturbed sites into those with mechanical disturbance, those with past agriculture, sites where buildings once stood, or sites used to dump toxic chemicals. Given the known history of Hanford, each site was also scored based on the extent, time, frequency, and impact to soil of the disturbance. Given this ranking, they successfully developed an IBI using metrics from the arthropod community.

Work by Brooks and O'Connell focused on using bird communities in wetland systems in the Mid-Atlantic region (Brooks et al. 1996, O'Connell et al. 2000). In these studies, the wetlands chosen to create the biological dose response curves were previously ranked during a large effort to assess and protect wetlands in Pennsylvania. In these studies, wetlands were evaluated and ranked in a three-category scale based on soil properties, sediment deposition, vegetation characteristics and amphibian surveys. In addition to these rankings, this successful IBI collected bird data at relatively large spatial scales (up to 2 km transects), allowing researchers to measure disturbance using aerial photography and GIS. They characterized the amount of different land uses (urbanization, agriculture) or vegetation types within a circle (1 km in diameter) surrounding the site where biological metrics were measured, in addition to collecting local vegetation data. Given the large-scale sampling of birds, the metrics responded well to changes in landscape structure caused by urbanization or agriculture.

Finally, Bradford et al. (1998) when developing a bird-based IBI for Great Basin rangelands, used professional opinion from "local range scientists" to categorize sites into low, medium and high levels of impacts from cattle grazing. In addition, low impact sites were protected from grazing for "many decades", while high impacts sites had known heavy grazing.

IBI methods in Southern California

Here we describe both the field methods of our study, and how we developed the gradient of disturbance. Our goal was to develop an IBI with data from multiple taxa (arthropods, birds, herpetofauna, small mammals and vegetation), all sampled simultaneously or nearly so. To date, no IBIs have simultaneously used data from so many different taxa. The goal was to develop an IBI allowing a single summary score across all taxa but in addition, sub-scores for metrics based on each taxa. Thus, a user can first attain an overall score for a site then "drill down" into the IBI to begin determining what aspects of the community show evidence of disturbance relative to intact sites. We feel this hierarchical IBI will create a robust framework for interpreting monitoring data across reserves. Furthermore, because we will have separate metrics for each taxa, a manager can still use a more limited form of assessment if they only have data collected for some of the taxa included in the overall IBI.

Spatial Scale of the IBI

We chose a relatively small spatial scale for the IBI based on discussions with a number of local managers and on the natural patterns of disturbance we observed in various NCCP reserves. Management activities such as weed eradication or restoration

take place at “hillside to hillside” spatial scales, or about what a team could achieve between a day and a week of work. We also noted the natural patterns of variation between disturbed and undisturbed patches of habitat at many reserves are quite small, with patches of relatively intact CSS intergrading with exotic dominated habitat over scales ranging from 10’s to 100’s of meters. In addition to generally matching the scales at which many management activities and the natural patterns of variation occur in the CSS vegetation community, the scale of the IBI sampling plot (50 x 50 m) made it relatively easy to implement in the field. Thus, our IBI measures CSS integrity at spatial scales near ~0.25 ha.

Measuring disturbance

Because many forms of disturbance are associated with increased abundance of exotic plant species in CSS and exotic plant invasion, itself, constitutes a form of disturbance, we used the absolute cover of exotic plant species as a relative measure of disturbance across sample plots. We chose plots to span a gradient of disturbance by sampling plots ranging from intact CSS scrublands, with low cover of exotic species, to plots of former CSS nearly entirely converted to exotic grasslands. Severely degraded plots were judged as former CSS, and not chaparral or native grasslands, based on a combination of local site characteristics and the vegetation of the surrounding landscape. Specifically, a degraded site was judged as former CSS if we found a few CSS shrubs on or very near the plot and CSS occurred as the dominant plant community in surrounding areas of similar slope and aspect. Plots with very low cover of CSS shrubs but relatively high cover of native perennial bunchgrasses were not considered, as such sites likely represented natural native grasslands.

Karr argues against using biotic data to initially estimate disturbance levels (Karr and Chu 1999) although the practice has occurred in other terrestrial IBI’s (Brooks et al. 1996, Bradford et al. 1998, Brooks et al. 1998, O’Connell et al. 2000). Although we would have preferred to use abiotic information to estimate disturbance at each of our sites, we did not. Simply put, it was impossible to establish historic disturbance regimes at each site given the complex histories of fire and land use throughout CSS habitat. Ideally, we would have had perfect knowledge of the frequencies and intensities of all forms of land use and fire history at each plot and selected plots to span a gradient in the level of disturbance.

We justified our selection of the gradient, and inclusion of the taxa based on an extensive literature review conducted during the first year of the project (See Appendix A). Our review of the vegetation literature lead us to conclude levels of non-native annuals in a CSS stand are positively correlated with past or present levels of disturbance (Lozon and Macisaac 1997, Diffendorfer et al. 2002). Thus, although we may not know exactly how many head of cattle grazed a plot or the fire return interval during the last 100 years, we can still gauge the relative level of disturbance based on levels of absolute exotic cover. Indeed, when we asked a number of local plant experts how they recognized a disturbed area, in all cases, they invariably pointed to the level of invasion as the “tell-tale” metric of disturbance. Furthermore, invasion by non-natives does itself constitute a disturbance and is perhaps the most critical threat to the long-term maintenance of native biodiversity in the NCCP reserves (Appendix A). Thus, we focus on invasion for good reason.

In addition to levels of invasion, we have developed other independent measures of disturbance for each plot using a GIS. This work was outside the scope of the DFG contract and we have not completed our analyses with these data. However, we developed this information because we felt the landscape context surrounding our study plots might influence the flora and fauna we sampled. For example, for two plots with identical levels of exotic invasion, a study plot near an urban edge or paved road, may support different species, or different abundances of certain species than a study plot away from human structures. We measured the amounts of general vegetation and land use types (CSS, chaparral, exotic grass, urbanization, etc) in 200m radius circle around each plot. In addition, we measured distances to landscape features such as roads and urbanization.

We chose not to use these landscape variables to initially rank plots for their level of disturbance for 2 reasons. First, because we arrayed our plots across gradients of exotic invasion, and not across gradients of the landscape features, many of the landscape variables had highly skewed distributions and did not clearly separate plots. Second, little is known about the spatial scales at which the flora and fauna in CSS respond to many of the landscape features we measured. For example, we collected data on various taxa on 50 x 50 m grids, yet distance to urbanization varied from 51 to 4890 meters. Because we did not know the spatial scale at which edge effects might attenuate, we could not easily develop an *a priori* ranking system based on distance to urban edge.

Instead, because both literature and professional opinion tie invasion directly to historic disturbance regimes, we ranked plots purely on levels of exotic invasion as the literature and professional opinion ties invasion directly to historic disturbance regimes. To understand the potential impacts of landscape features associated with disturbance on CSS flora and fauna, we will perform separate multiple regression and multivariate statistical analyses. In these cases, we will include both locally measured variables (i.e. data collected on the 50 x 50 m plot) as well as the landscape variables. Furthermore, once we complete the IBI based on levels of disturbance on individual plots, we will perform additional analyses to determine if landscape variables can explain additional variability in the IBI.

Methods

Study areas.

We conducted the study at 3 “sites”: Chino Hills State Park (CHSP), Rancho Jamul Ecological Reserve (RJER), and the area containing Crystal Cove Wilderness Park, Laguna Coast Wilderness Park, San Joaquin Wildlife Sanctuary, and the Aliso and Woods Canyon Wilderness Park. We refer to this area as the Orange County Coastal Reserves (OCCR). We chose these sites because they covered a wide geographic range of Coastal Sage Scrub and because we had additional funding from State Parks and the Nature Reserves of Orange County to conduct small mammal surveys in these areas. By combining these efforts with the funding from CDFG, we increased the overall number of study plots used in the project and enhanced the samples sizes for all projects.

Overall sampling effort.

We began sampling vegetation, mammals, birds, and arthropods during late March of 2002. We initially selected 38 plots throughout the 3 sites, nesting an exotic invasion gradient within each site. We selected plots using 3 criteria. First, in order to obtain herpetofauna data, the plots were located adjacent to sampling arrays used by Dr. Robert Fisher of the United States Geological Service (USGS). Second, proximity to USGS arrays also allowed us to use vegetation data collected at each herpetofauna array (a single 100m transect), along with rapid assessment in the field, to determine the level of exotic grasses at each plot. Third, in order to test for edge effects, we stratified a subset of plots near urban edges in the Orange County Coastal Reserves. Fourth, because we did not want short-term disturbances related to fire affecting our results, we chose only sites that had not burned in at least 9 years. The Laguna Hills fire of 1993 was the most recent known fire at all of our sites.

Resprouting shrubs often seed abundantly during the first post-fire year (Keeley and Keeley 1984, Malanson and Westman 1985, O'Leary and Westman 1988), and CSS shrubs may recover half or more of their pre-fire density and cover within 5 to 7 years after fire (Westman 1981, O'Leary and Westman 1988), though recovery time varies by species (Minnich and Dezzani 1998). Given, these rates of post-fire recovery, we felt our site selection process likely eliminated time since fire as a factor affecting our results.

During the study, we were forced to alter the sampling strategy of 38 plots sampled 4 times per year for two years for a number of reasons. First, 3 different fires burned some of our plots, forcing us to find replacements (Table 1). Seven plots at CHSP burned during the first year; 3 before the first session (no data collected) and 4 more after 3 sessions. In addition, 9 plots at RJER burned in the Cedar Fire. Since 7 of our original 9 plots at CHSP had burned and we had a 1-year contract with State Parks, we stopped work at CHSP after year 1 and transferred the sampling effort to other locations (Table 1). Second, after conducting more thorough plant sampling during the first spring of the study, we identified a gap in the exotic invasion gradient from ~70-90% absolute exotic cover (Figure 4). In year 2, we filled this gap with 10 new plots attained by replacing the 6 plots at CHSP and shifting effort from 4 other plots with repetitive AEC levels (Table 1, Figure 4). While filling the gap, we were forced to choose some new plots away from USGS arrays so these plots have no associated herpetofauna data. However, USGS had stopped sampling herpetofauna by year 2 of the study given their contractual obligations with individual reserves.

In summary, we successfully sampled 21 of the original 38 plots for all 8 sessions and 26 sites for 7 of 8 sessions. We sampled 32 sites for all 4 sessions during year 1, but only 27 sites for all 4 sessions of year 2. We sampled 32 sites for 3 of the 4 sessions in year 2. Table 2 give summary information for each plot.

Table 1. Sampling history on each plot and summary information. S means trapping was stopped at a plot, B means the plot burned. We stopped sampling on plots after burns and transferred our sampling effort elsewhere. Site refers to the geographic region sampled: CHSP = Chino Hills State Park, OCCR = Orange County Coastal Reserves, RJER = Rancho Jamul Ecological Reserve. Reserve refers to the individual reserves sampled in OCCR. AWC = Aliso and Woods Canyon, SJHW = San Joaquin Hills West

Site	Reserve	Plot	Year 1				Total	Year 2				Total	Grand total
			1	2	3	4		5	6	7	8		
CHSP		4	B				0					0	0
CHSP		5	B				0					0	0
CHSP		6	B				0					0	0
CHSP		7		1	1	B	2					0	2
CHSP		8	1	1	1	B	3					0	3
CHSP		10	1	1	1	B	3					0	3
CHSP		11	1	1	1	B	3					0	3
CHSP		17	1	1	1	1	4	S				0	4
CHSP		18	1	1	1	1	4	S				0	4
RJER		4					0	1	1	1	1	4	4
RJER		5	1	1	1	1	4	1	1	1	1	4	8
RJER		6	1	1	1	1	4	1	1	1	1	4	8
RJER		7	1	1	1	1	4	1	1	1	B	3	7
RJER		8					0	1	1	1	B	3	3
RJER		9	1	1	1	1	4	1	1	1	1	4	8
RJER		10					0	1	1	1	1	4	4
RJER		11	1	1	1	1	4	1	1	1	B	3	7
RJER		12					0	1	1	1	B	3	3
RJER		13					0	1	1	1	B	3	3
RJER		14					0	1	1	1	B	3	3
RJER		15	1	1	1	1	4	1	1	1	B	3	7
RJER		18	1	1	1	1	4	1	1	1	B	3	7
RJER		19	1	1	1	1	4	1	1	1	B	3	7
OCCR	AWC	1	1	1	1	1	4	1	1	1	1	4	8
OCCR	AWC	2	1	1	1	1	4	1	1	1	1	4	8
OCCR	AWC	3	1	1	1	1	4	S				0	4
OCCR	AWC	4					0	1	1	1	1	4	4
OCCR	AWC	13	1	1	1	1	4	S				0	4
OCCR	AWC	14	1	1	1	1	4	1	1	1	1	4	8
OCCR	AWC	15	1	1	1	1	4	1	1	1	1	4	8
OCCR	AWC	16	1	1	1	1	4	1	1	1	1	4	8
OCCR	AWC	17	1	1	1	1	4	1	1	1	1	4	8

OCCR	SJHW	4					0	1	1	1	1	4	4
OCCR	SJHW	5	1	1	1	1	4	1	1	1	1	4	8
OCCR	SJHW	6	1	1	1	1	4	1	1	1	1	4	8
OCCR	SJHW	7	1	1	1	1	4	1	1	1	1	4	8
OCCR	SJHW	8					0	1	1	1	1	4	4
OCCR	SJHW	9	1	1	1	1	4	1	1	1	1	4	8
OCCR	SJHW	10	1	1	1	1	4	1	1	1	1	4	8
OCCR	SJHW	11	1	1	1	1	4	1	1	1	1	4	8
OCCR	SJHW	12	1	1	1	1	4	S				0	4
OCCR	SJHW	14	1	1	1	1	4	1	1	1	1	4	8
OCCR	SJHW	16	1	1	1	1	4	1	1	1	1	4	8
OCCR	SJHW	17	1	1	1	1	4	1	1	1	1	4	8
OCCR	SJHW	18	1	1	1	1	4	S				0	4
OCCR	SJHW	19	1	1	1	1	4	1	1	1	1	4	8
OCCR	SJHW	20	1	1	1	1	4	1	1	1	1	4	8
OCCR	SJHW	21	1	1	1	1	4	1	1	1	1	4	8
OCCR	SJHW	22						1	1	1	1	4	4
Total per session			35	36	36	32		36	36	36	27		

Invasion Gradient

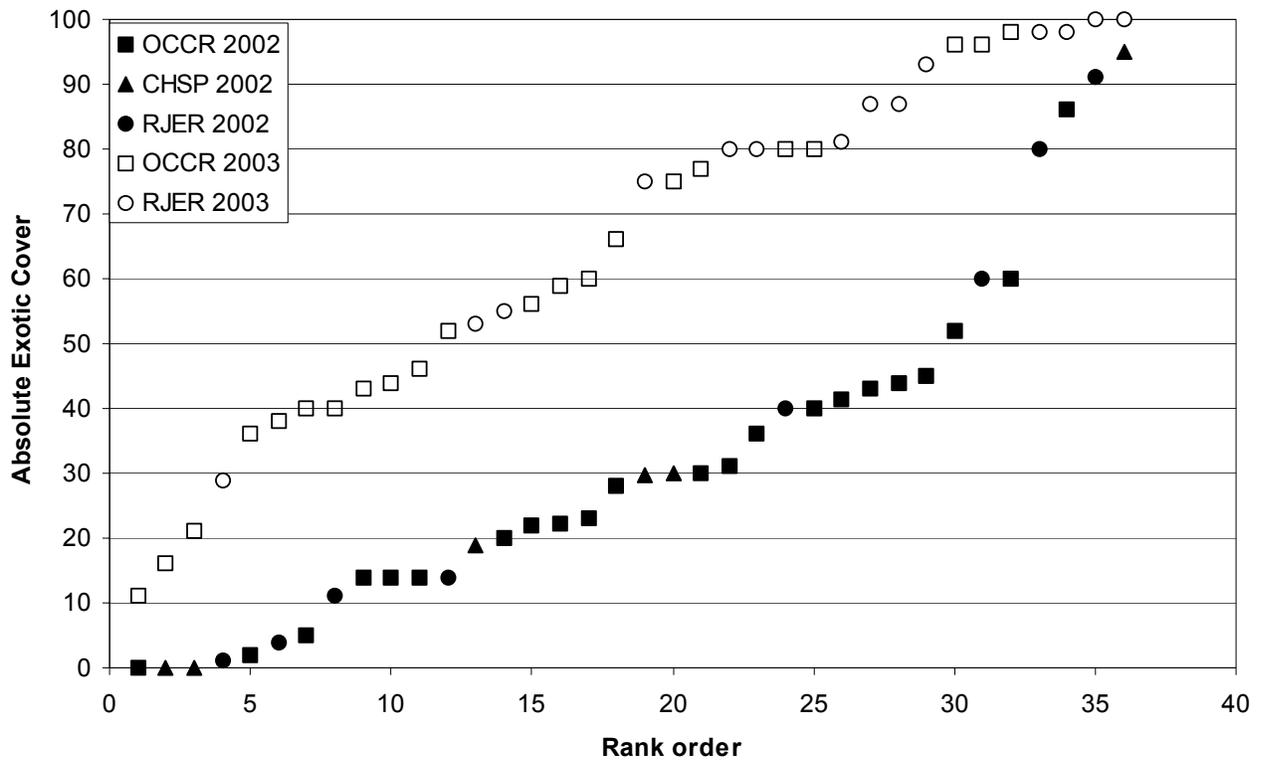


Figure 4. . Gradient of invasion sampled in 2002 and 2003. Plots are coded by site and year. OCCR = Orange County Coastal Reserves, CHSP = Chino Hills State Park, RJER = Rancho Jamul Ecological Reserve. Notice the gap between 60-80% absolute exotic cover in 2002. The low exotic cover in 2002 resulted from a drought.

Table 2. Summary of the 49 study plots spread across three reserve systems. “Site” refers is the 4 letter code we used to designate site: CHSP = Chino Hills State Park, OCCR = Orange County Coastal Reserves, RJER = Rancho Jamul Ecological Reserve. “USGS site-array code” is their label. “Reserve” refers to the individual reserve sampled in OCCR. AWC = Aliso and Woods Canyon, SJHW = San Joaquin Hills West. Location refers to plots at OCCR considered edge plots. “Rel. % Exotic USGS” refers to the USGS estimate of exotic relative cover using a 100m transect. AEC refers to absolute exotic cover estimated in 2002 and 2003 for each grid using our methods.

Running total	Site	Reserve	Plot #	USGS Array code	Location	Rel. % Exotic USGS	AEC 2002 SDSU	AEC 2003 SDSU	LAT	LONG
1	CHSP		4	Chino4		66.04	Burned		437982	3749588
2	CHSP		5	Chino5		72.97	Burned		437864	3749556
3	CHSP		6	Chino6		29.84	Burned		437839	3749436
4	CHSP		7				95		427286	3752495
5	CHSP		8	Chino8		5.88	0		427002	3752561
6	CHSP		10	Chino10		0	0		427021	3752660
7	CHSP		11	Chino11		50	19		425813	3752562
8	CHSP		17	Chino17		11.48	30		424510	3755765
9	CHSP		18	Chino18		27.46	30		424571	3755869
10	RJER		4	Rjer4		99.39		100	512417	3617456
11	RJER		5				91	100	511982	3617526
12	RJER		6	Rjer6		97.97	80	98	512275	3616653
13	RJER		7	Rjer7		11.97	4	55	512373	3615784
14	RJER		8					29	512213	3616161
15	RJER		9	Rjer9		61.82	40	87	513680	3616403
16	RJER		10					93	513457	3616669
17	RJER		11	Rjer11		43.15	1	81	512344	3614337
18	RJER		12					50	512445	3614824
19	RJER		13					75	512888	3615150
20	RJER		14					80	513242	3614241
21	RJER		15				60	98	513579	3614243
22	RJER		18	Rjer18		5.95	14	87	513896	3614222
23	RJER		19				11	80	515111	3614339
24	OCCR	AWC	1	Awc1		7.09	23	21	431374	3709049
25	OCCR	AWC	2	Awc2		23.21	14	39	431655	3709843
26	OCCR	AWC	3	Awc3		100	86		431400	3710232
27	OCCR	AWC	4					98	431453	3711150
28	OCCR	AWC	13	Awc13		34.17	14		430560	3715502

29	OCCR	AWC	14	Awc14	Edge	0	0	16	430362	3716297
30	OCCR	AWC	15	Awc15	Edge	10.87	14	59	429996	3715882
31	OCCR	AWC	16	Awc16		17.42	22	43	429814	3714756
32	OCCR	AWC	17	Awc17		26.72	20	40	429231	3713302
33	OCCR	SJHW	4					80	428913	3714241
34	OCCR	SJHW	5	Sjhw5		14.29	44	52	428408	3715132
35	OCCR	SJHW	6	Sjhw6		22.88	40	44	427402	3714829
36	OCCR	SJHW	7	Sjhw7	Edge	2.01	5	11	427375	3712826
37	OCCR	SJHW	8					96	426714	3714384
38	OCCR	SJHW	9	Sjhw9		52.38	60	66	425690	3714772
39	OCCR	SJHW	10	Sjhw10		58	30	36	425761	3714968
40	OCCR	SJHW	11	Sjhw11	Edge	21.02	2	38	425074	3713576
41	OCCR	SJHW	12	Sjhw12	Edge	37.01	22		424227	3713573
42	OCCR	SJHW	14	Sjhw14		5.21	31	46	426407	3716923
43	OCCR	SJHW	16	Sjhw16		17.61	36	56	426148	3717780
44	OCCR	SJHW	17	Sjhw17		18.75	41	60	426454	3718116
45	OCCR	SJHW	18	Sjhw18		12.36	45		426186	3718280
46	OCCR	SJHW	19	Sjhw19		7.21	28	77	425526	3718536
47	OCCR	SJHW	20	Sjhw20	Edge	47.25	52	80	425359	3717570
48	OCCR	SJHW	21	Sjhw21	Edge	64.03	43	75	423922	3716951
49	OCCR	SJHW	22					96	424259	3715156

Grid design and timing.

At each plot we established 50 x 50 m grids (Figure 5). Each grid contained 49 Sherman traps for sampling small mammals, 6 small bowls for sampling terrestrial arthropods, 4 sticky traps (surrounded by nylon mesh to keep out birds) for sampling flying arthropods, and 4, 50 m vegetation transects. In addition, we conducted bird point counts from the center of each plot. We detail the methods used for each taxa below.

We sampled each plot 4 times a year. We did not sample in 4 evenly spaced sessions across the year. Instead, we timed our sampling to gather as much information on species diversity across all taxa, given a logistical constraint of 4 sampling periods. We discussed sample timing with Dr. Ted Case (University of California, San Diego), Dr. Gerald Braden (San Bernadino County Museum), and Dr. Wayne Spencer (Conservation Biology Institute), who all have extensive experience sampling a wide variety of CSS taxa. We sampled as near as possible the periods described below, given constraints created by the academic calendar (final exams, spring break, etc) and matching the USGS herpetofauna sampling schedule. The following time periods and associated comments summarize our discussions with local experts.

- Late-January to mid-February. Given enough rainfall, amphibians become active. Small mammal captures for some species increase, with peak abundances typically from January through May. Resident bird community is readily sampled.
- Mid-April to early May. Peak bird migration and peak herpetofauna activity. Largest insect biomass and peak in diversity.
- June. Allows insight into patterns of decline as CSS vegetation begins to dry. Final sample before the summer. Sampling during hotter summer months is not productive.
- Mid-September to early October. Return of neotropical migrant birds. A second peak in activity for some herpetofauna and young of the year appear for some species.

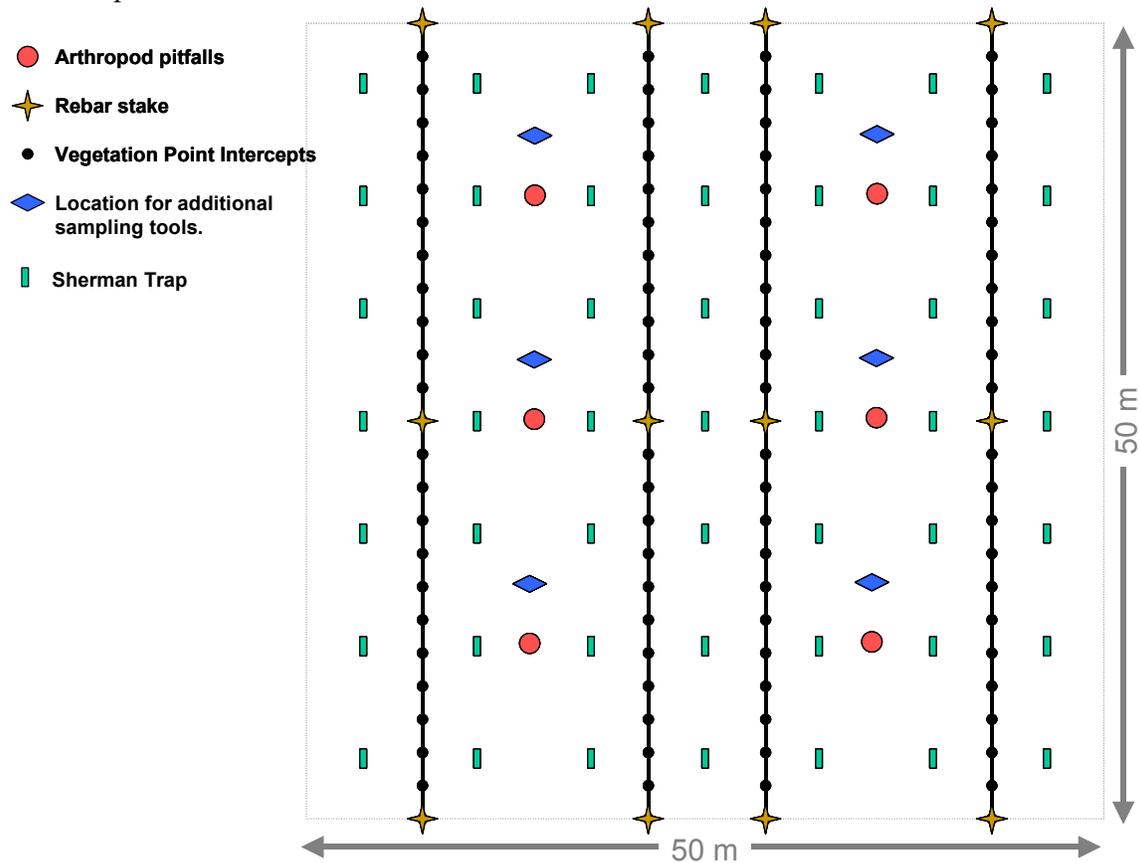


Figure 5. IBI sampling plot. Plot includes 49 sherman live traps for small mammals (7 m spacing), 4 vegetation transects, and 6 pitfall bowls for terrestrial arthropods.

Taxa specific sampling.

Small Mammals.

Sampling sessions lasted 4 days. We typically employed 2 teams, each with 2 or more handlers, to check 8 sites in a morning. We generally completed all plots in 5, 4-day trapping periods (8 sites x 5 trapping periods = 40 sites) over a 2-3 week period,

reducing temporal changes across plots. We coordinated our sampling efforts to coincide within 1 week of USGS herpetofauna sampling.

We trapped grids for 3 consecutive mornings. We opened, set, and baited all traps with roasted (to prevent germination) sunflower seeds at least 1 hour before dusk and closed traps after checking them in the morning. We covered all traps with wooden shingles to protect them from sun and rain. During each capture we recorded date, site, trap location, species, weight, gender, age, sexual characteristics, and a unique toe-clip (assigned on first capture). We distinguished juveniles from adults by the presence of juvenile gray pelage. Female reproductive characteristics included nipple size (large or small), vagina condition (perforate or not perforate), and pubic symphysis condition (open or closed). We considered large nipples, a perforate vagina, and an open pubic symphysis signs of reproductive activity; however, we only counted females as reproductively active if 2 of 3 reproductive characteristics showed signs of activity (McCravy and Rose 1992). We recorded male testes condition and considered descended testes a sign of reproductive activity. After examining and marking, we released each individual at its capture site.

These protocols allowed us to estimate the following variables at each plot: species richness, the number of unique individuals (by species, sex, or age class), sex ratio, age structure (the number of non-adults/all individuals), and average body weight (by species, or age class). Age class was determined by pelage color and by plotting the body weights of reproductively active and non-reproductively active individuals to determine the lowest body weight at which individuals become reproductive in a particular species. Thus, our age class metric divided the population into animals capable of reproduction (adults) and individuals not capable of reproduction (non-adults).

Arthropods.

We sampled terrestrial arthropods using 6, 3-inch diameter cafeteria bowls buried so the lip of each bowl was at ground level. Bowls were placed on plots on the first day of the 4-day small mammal trapping sessions, left open, then collected as small mammal traps were shut down on the fourth day. At each plot, arthropods from all six bowls were placed in a single whirly-pack, placed on ice, and stored in an ultra-cold freezer. Later, arthropods were thawed, sorted to Order, and pinned. The ants are currently being sorted to species and counted by Leticia Sanchez, a PhD student at University of Colorado and former undergraduate volunteer on the project.

We had poor success sampling flying insects with the sticky traps. The traps collected low numbers of arthropods and the majority of individuals captured were small and became so entangled they could not be identified using a dissecting scope. Given these difficulties, we did not use sticky trap data and do not recommend their use.

The arthropod sampling allowed us to estimate ant species richness and abundance (number of individuals captured) for each plot. In addition, we counted the presence/absence and abundance of 3 exotic arthropods European Earwigs (*Forficula auricularia*), the common pill bug (*Armadillidium vulgare*), the Dooryard Sow Bug (*Porcellio laevis*). Much to our chagrin, we discovered many arthropods are extremely difficult to identify to species, requiring hard-to-find experts at museums. Until we develop better taxonomic expertise, the use of arthropods in biologically monitoring may be hampered. Burger et al. (2001) used morphospecies in analyses of CSS atthropod

communities. Unfortunately, morphospecies definitions may vary across researchers, biomonitors, or technicians, so relying on such classification schemes in long-term monitoring programs where different researchers are involved may be unreliable.

Birds:

We conducted point counts at all plots from 30 minutes before sunrise to 10:00 a.m. Point counts consisted of a 7-minute recording session in a 100-meter radius circle. After arriving at the center of a plot, we waited 2 minutes before beginning the count. Flyovers were recorded separately. We recorded any bird species seen or heard, as well as the number of individuals. Point counts generated information on species richness, as well as relative abundance of species at each plot.

Vegetation.

All plots included 4 vegetation transects, each 50m in length, separated by 7 or 14 m (Figure 5). We sampled transects using the point intercept method to estimate vegetation cover by species origin (native or exotic) and growth form (grasses, forbs, and shrubs) for each 2500m² plot. This method estimated cover by noting all species intersecting an imaginary vertical line extending upward from a sample point (i.e. “hits”). We used a thin metal rod to establish the vertical line. Sample points on each transect occurred at 2m intervals along the transect line starting at the 0.5m mark, yielding a total of 100 sample points per plot (25 points per transect). Canopy cover of shrubs and foliar cover of herbs were estimated by noting the tallest hit of each species intercepting the vertical line at each point. Multiple hits of the same species at a given point were not recorded. We recorded height to the nearest 0.1 m for the tallest hit of each species, ground cover, and obvious disturbances at each point.

Vegetation cover on each plot was separated by species origin and growth form. Point intercept data from each sampling year was used to calculate 2 different measures of percent cover on all plots: “absolute cover” and “relative cover”. Absolute cover was the number of individual points across all 4 transects where at least one hit of a particular type of vegetation (e.g. native, non-native, shrub, etc.) was recorded, divided by the total number of points sampled. Absolute cover values estimated the spread of a vegetation type across space. Relative cover was the total number of hits of a particular type of vegetation, regardless of location, divided by the total number of all vegetation hits. While any individual species was only counted once at a given point, multiple hits of an individual origin class or growth form were recorded separately at each point. Thus, relative cover provided an estimate of the extent to which an origin class or growth form dominated the standing vegetation.

Species richness was estimated at 3 scales on each plot. At the smallest scale, species richness was estimated within 48 1m² quadrates on each plot. All unique species alive and rooted within each 1m² quadrat were recorded. Each transect contained 12 quadrats, spaced at varying distances. Species richness was estimated at the 50m² scale by recording all species alive and rooted within a 1m wide belt along the left side of each transect line. Finally, species richness for the entire 2500m² plot was determined using species lists from all 4 transects and any additional species seen alive and rooted within the overall 50m x 50m plot. To ensure equal sampling effort among plots, the search for

additional species was only conducted from the vegetation transects and along the 7 mammal trapping lines.

Shrub structure and recruitment was also assessed along each transect by sampling the size, density, and status (i.e. seedling, live, dead) of each shrub in the 48 1 m² quadrats on each plot.

To verify the sampling protocol and determine levels of sampling error vegetation sampling occurred once per year during the first 2 years of the study. Vegetation sampling occurred during the growing season, independent of the 4 sampling sessions used to sample other taxa.

Data Analysis and IBI development techniques

The data analysis and IBI development process had 3 main steps. First, we performed exploratory data analyses on every variable we extracted from the field data. Second, we combined variables within each taxa showing responses across the gradient into metrics and then used the metrics to develop an IBI using the 1,3,5 scoring system described above and in Figure 1. Third, we conducted statistical analyses on the IBI showing how well it separated sites and how various levels of sampling impact its performance.

Given the impacts of the unplanned fires on the experimental design, we divided our data into 3 groups in order to maximize our sample sizes and our sampling across time: 32 plots sampled all 4 sessions in year 1 “**YEAR 1**”, 36 plots sampled in the first 3 sessions of year 2 “**YEAR 2**”, and 26 plots sampled in 7 of 8 sessions during the entire study “**BOTH**”. These combinations maximized the number of plots, and in so doing, sacrificed the number of sessions sampled. Since our goal was to understand patterns of response across the disturbance gradient, it was more important to adequately sample across the AEC gradient than it was to have repeated samples at the same site through time. As shown in the table 2, using sites sampled all 3 sessions in year 2, instead of sites sampled all 4 sessions, increased the overall sample size from 27 to 36 plots. Furthermore, using plots with 7 of 8 sampling sessions increases the sample size from 21 to 26.

In addition to maximizing sample sizes, the 3 groups also included data from year 1 and year 2 separately, allowing us to check both individual metrics and overall IBI performance across years, as well as examine a combined dataset. Given the known relationship between species diversity observed at a site and the time spent sampling, the combined data gave us the most comprehensive sampling at any give plot. However, 2002 was a drought year, while 2003 approached normal rainfall.

In general, we performed initial metric screening and IBI metric development (see below) on all 3 datasets, checking for major differences between single and combined years. Except for variation in vegetation data with precipitation, general responses across the AEC gradient were consistent across all 3 datasets.

Step 1. Initial variable screening.

We initially explored the data for taxa and species showing discernible responses to increased levels of AEC (absolute exotic cover) using scatter plots (AEC on the x-axis and the variable of interest on the y-axis). Karr and Chu (1999) recommend this

approach, arguing total reliance on statistically significant regressions or correlations may miss metrics with nonlinear responses that express different levels of disturbance.

We focused on two general types of data during variable screening. First, we used presence-absence data to investigate changes in the probability a species is present at a plot along the AEC gradient. Second, we used relative abundance data to determine if the number of individuals of a species, relative to all the other individuals at a plot, changed across the AEC gradient. Relative abundance is considered superior to abundance (Karr and Chu 1997) because it attempts to control for absolute changes in population size across plots or time and measures the proportion of the individuals in a community that are a specific species. For example, 50 individuals of Species X could be captured at site 1 and 10 at site 2. However, at the first site, Species X made up 50 out of the 100 total individuals captured (0.50 relative abundance), while at the second site Species X comprised the same relative abundance ($0.5 = 10 \text{ out of } 20 \text{ total individuals}$).

We used a number of criteria when selecting species for metrics and determining positive, negative and neutral responses. First, rare species were not used. Thus, any species with occurrences at less than 3 plots were not used for metrics based on presence/absence. Similarly, species with low numbers of individuals captured per plot were not used in metrics incorporating abundance. When considering a species for inclusion in a metric, we looked for obvious thresholds in presence and absence across the gradient in exotic cover, or large changes in the proportion of sites where species were present across the gradient. For example, if a species was present at 8 of 10 plots with exotic cover values from 0.0-20%, and only present at 2 of 10 sites with exotic cover values from 70 to 100%, we would conclude the species shows a negative response to exotic cover. In terms of abundance, we included species showing generally negative or positive trends in scatterplots across the gradient. These trends did not have to be statistically significant to include the species in a metric, but instead discriminate between potential categories of disturbance levels. For example, some species showed responses separating highly disturbed plots from all other levels of disturbance, but did not distinguish between low to moderate levels of disturbance. Other species readily distinguished between low to moderate, but not high, levels of disturbance. By combining these results across species into a single metric, we gain the ability to distinguish between many levels of disturbance. We include all of the scatterplots in Appendix B and hope the reader spends time viewing these as some cases were not simply “cut and dry” responses.

Step 2. IBI metric creation and scoring

We generally attempted to create 4 types of metrics from the presence-absence and relative abundance data: First, using presence absence data we calculated the “Proportion of tolerant species” (or genera and orders). This was the number of species showing positive responses in presence-absence across the AEC gradient divided by the total number of species (or genera or orders) on the plot. Second, we calculated the “Proportion of intolerant species”. This was simply the number of species showing negative responses in presence-absence across the AEC gradient, divided by the total number of species (or genera or orders). Third, using relative abundance data, we calculated the “Relative abundance of tolerant species.” This was the sum of the relative abundances across all species showing an increase in relative abundance across the AEC

gradient. Fourth, “Relative abundance of intolerant species” used species showing declines in relative abundance across the AEC gradient. In some cases, we also used overall trends in species richness.

We used several criteria when combining variables screened in step 1 into IBI metrics. First, the variables included had to show either positive or negative responses across the AEC gradient. This was not always easily determined. We initially screened the data using the 2-year “both” dataset, but always checked to see if species showed similar responses across the year 1 and year 2 data sets as well. In addition to checking for consistency across datasets, we also checked for consistency across the reserve units. Shifting plots after the unexpected fires resulted in RJER containing plots with higher levels of exotic cover relative to other reserves. Thus, if a species showed a response across the AEC gradient, variation in the species relative abundance or presence-absence across the reserves, independent of AEC levels, may impact our interpretation. In some cases, we created separate plots of the species response to AEC within each reserve. Though sample sizes were typically small (5-10 plots), we were able to get some sense if the species responded consistently across all reserves.

Second, the resulting metric, when plotted across the AEC gradient, had to have adequate dispersion across the gradient to separate levels of disturbance. In some cases, after excluding species with low sample sizes, throwing out candidate species with inconsistent responses across the datasets or reserves, we were left with just one or two species showing strong, consistent responses. In these cases, the final metric, because it was based on so few species, simply did not adequately distinguish between disturbance categories.

Third, we strongly preferred existing literature or expert opinion to support the inclusion of species, genera, or orders in a metric in addition to the observed response we found. In all cases, we considered the responses we found across the AEC gradient as patterns in need of additional verification. In the case of birds, small mammals, and plants, we typically found supporting literature that verified the observed patterns and added confidence to the metric selection process. However, for other groups, such as herpetofauna, arthropods, and ants, we had a more difficult time established such support. We present a general level of confidence in the species composition of each guild in the results.

Once we created an IBI metric, we scored it using the 1,3,5 scale used in many IBIs. When responses across the gradient were linear, or showed a general increase or decrease, we generally used 1/3's to categorize levels of disturbance. However, in many cases, natural breaks, or clumpiness in the response across the AEC gradient, allowed more refined discernment of disturbance categories. We generally attempted to create categories so that plots at either extreme of the AEC gradient were not incorrectly described by the scoring regime. We describe and illustrate all IBI metric scatter plots and the rules for the 1,3,5 categories in Appendix B and the Results.

Step 3. Final IBI testing

We performed 2 main analyses to understand both the effectiveness of the IBI and how this effectiveness changes with the amount of sampling. First, we performed regressions of the final IBI and AEC across each of our 3 datasets to determine how

much the relationship changed across years and with different levels of data. Second, we used ANOVA's and clustering analyses to understand how many levels of disturbance the IBI could distinguish.

RESULTS

We organized the results into three sections: First, we report and discuss general diversity relationships across the taxa and various plant metrics. This sets the stage for interpreting taxa- and species-specific responses to increases in the level of invasion and losses of shrub cover. Second, we present step 1 (Exploratory data analyses) and step 2 (IBI metric creation and scoring) for each taxa. Third, we present the multi-taxa IBI, and its performance given the different sample regimes we used in the analyses.

Section 1: Overall diversity trends and vegetation relationships

Patterns in species richness across taxa: Do taxa respond similarly to disturbance?

We first explored if sites supporting high levels of species richness in one taxa, also showed high levels of species richness in another. If patterns of richness are positively correlated across taxa, then it may be possible to use a single taxa as an indicator for other taxa or guilds in monitoring programs.

We found no obvious relationships (no statistically significant at $p < 0.05$ Pearson correlations) between taxa in their levels of diversity. Study plots with high levels of diversity for a taxon generally did not have high levels of diversity in another (Figure 6). This indicates taxa responses to invasion vary and indicator taxa, or species will likely not function well as a tool for CSS monitoring programs.

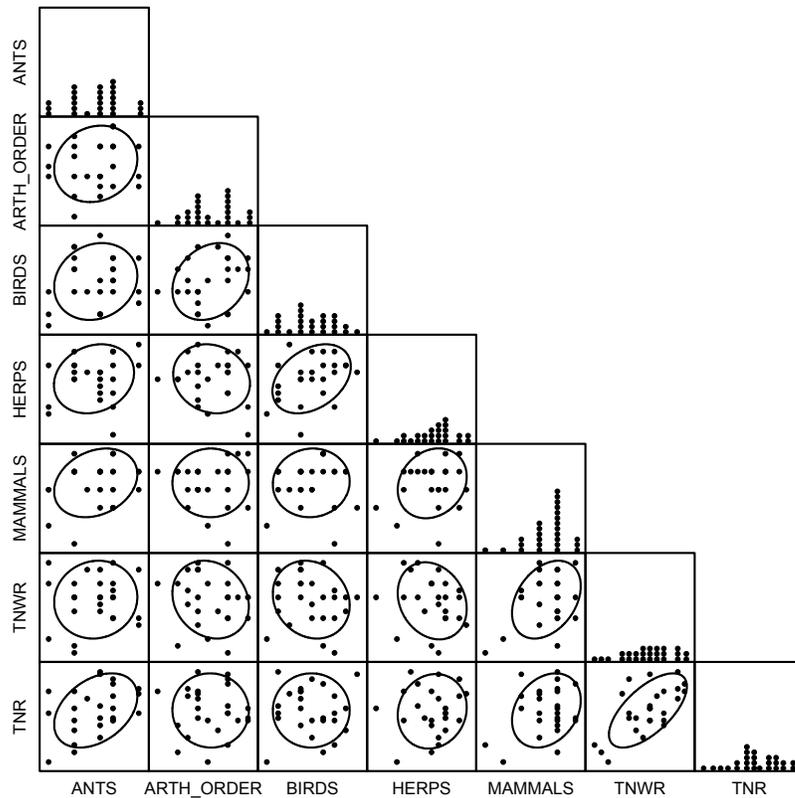


Figure 6. A scatter plot matrix between species richness of ants, arthropod orders (ARTH_ORDER), birds, herpetofauna (“HERPS”), mammals, and vegetation (“TNWR”= Total Native Woody Richness, “TNR” = Total Native Plant Richness). No statistically significant correlations occurred between any of the variables. Spheres represent $p=0.683$ confidence ellipses.

Patterns between native shrub cover and exotic invasion.

Given the drought during 2002, many grasses, forbs and herbs simply did not germinate so these data inadequately described diversity and abundance of native non-woody species and exotic cover. We sampled again in 2003, a year approaching mean rainfall and we used these data in our analyses. In certain analyses when we combined data across years, 4 plots were only sampled in year 1 and not year 2. In these cases, we used the year 1 data.

We found a strong negative relationship between absolute exotic cover (AEC) and both absolute native cover (ANC) and absolute woody cover (AWC) so sites with more total invasion have less total native shrub cover (Figure 7).

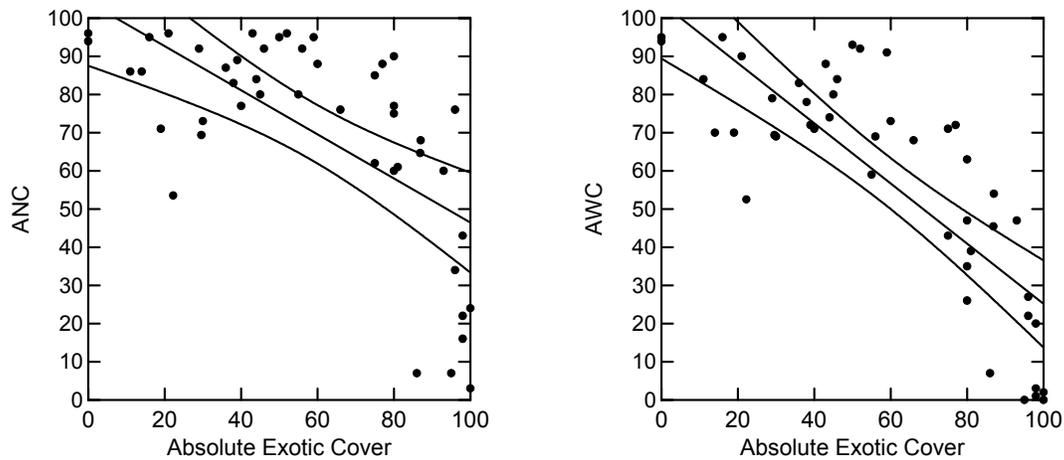


Figure 7. Absolute Native Cover and Absolute Woody Cover vs. Absolute Exotic Cover. All 46 plots included using year 2 plant data except for 4 plots sampled only in year 1. The line represents a linear regression with 95% confidence intervals. (Abs. Nat. Cover = $-0.578(\text{Abs. Exo. Cov}) + 104.7$, $n = 46$, $p < 0.001$, $R^2 = 0.416$. Abs. Woody. Cover = $-0.778(\text{Abs. Exo. Cov}) + 104.0$, $n = 46$, $p < 0.001$, $R^2 = 0.647$).

Given the strong general relationship between AEC and ANC (Pearsons' $r = -0.80$), we expect when species or taxa respond negatively to increased exotic cover, they will respond positively with increased native cover. Biologically, it is difficult to determine if the species is actually responding to 1) more exotic cover, 2) less native cover, or 3) some combination of the two. For example, a species may show a decline in abundance on more invaded plots, indicating the species might be intolerant to invasion. However, if an intact, un-invaded plot of CSS was artificially thinned to have low values of absolute native shrub cover, the species may decline, independent of exotic invasion. Indeed, for many vertebrate consumers the biological mechanism for a decline across the gradient is likely the loss of native vegetation cover and diversity resulting in lower food resources, and less cover from predators, not the actual addition of exotic vegetation. However, native forbs and herbs are likely directly affected by competition with exotic grasses.

Overall taxa response to levels of exotic invasion.

Species richness showed a variety of responses to increasing levels of AEC (Figure 8). Arthropods showed a weak positive increase in the number of orders with increasing AEC. This weak relationship was statistically significant (at $p < 0.05$) in 2 of the three datasets with AEC explaining from ~9-22% of the variation in Order richness (Year 1. Richness = $0.042\text{AEC} + 9.453$, $\text{adj } r^2 = 0.216$, $p = 0.007$, $n = 32$; Year 2. Richness = $0.023\text{AEC} + 8.357$, $\text{adj } r^2 = 0.09$, $p = 0.074$, $n = 36$; Both. Richness = $0.044\text{AEC} + 11.646$, $\text{adj } r^2 = 0.219$, $p = 0.016$, $n = 26$). For birds, though Figure 8 shows a positive trend, this was not statistically significant in any of our data sets. Small mammals showed a reasonably strong decline in species richness with increasing AEC in all 3 datasets (Year 1. Richness = $-0.025\text{AEC} + 7.664$, $\text{adj } r^2 = 0.155$, $p = 0.026$, $n = 32$; Year 2. Richness = $-0.040\text{AEC} + 8.354$, $\text{adj } r^2 = 0.338$, $p < 0.001$, $n = 36$; Both. Richness = $-0.027\text{AEC} + 7.993$, $\text{adj } r^2 = 0.338$, $p < 0.001$, $n = 36$).

$r^2=0.249$, $p=0.010$, $n=26$). Native plant species richness showed a decline with increasing AEC in both of the single year datasets but not the 'both' dataset. (Year 1 Richness = $-0.185\text{AEC}+28.966$, $\text{adj } r^2=0.308$, $p<0.001$, $n=36$, Year 2 Richness = $-0.162\text{AEC}+48.760$, $\text{adj } r^2=0.135$, $p=0.016$, $n=36$). In addition we found a significant nonlinear, negative trend in native plant richness across the AEC gradient. In general, these curvilinear regressions fit the data better than the linear models, and showed total native plant richness tended to peak at low to intermediate levels of AEC with dramatically lower values at high AEC and slightly lower values at low AEC.

The lack of strong changes in species richness across the disturbance gradient does not mean, species, taxa, or the overall system, had little response to increasing levels of AEC. For most taxa, species turned-over across the gradient with different species occurring on invaded sites relative to intact sites, despite the pattern of similar overall richness. Index based approaches are developed specifically to identify, then measure, those suites of species increasing or decreasing with increasing disturbance. Thus, a first look at overall trends in the data strongly argues an index-based approach will be better able to measure responses to disturbance than simple community-level summary statistics.

We feel the lack of a trend for the herpetofauna may result from small sample sizes and the spatial scale of our plots relative to the spatial scales at which herpetofauna respond to disturbance. We obtained data from Dr. Fisher for his sampling arrays placed near our grids for 2002-2003 so we could relate the captures of herpetofauna to the vegetation data collected at each site. This resulted in small data sets, and low numbers of species relative to the actual number of species captured over longer time periods or across the entire suite of arrays at a site. Overall, the herpetofauna data, because it takes time to adequately sample the entire community, may require a longer-term measure of vegetation structure or larger sample sizes (i.e. more arrays with array-specific vegetation data) before we can adequately explore the data for trends. Furthermore, Dr. Fisher has demonstrated changes in the herpetofauna community to urbanization and fragmentation. However, these responses are measured across much larger spatial scales than our 50x50 m plots, and are seen when multiple pitfall arrays are compared across reserves. Thus, forms of anthropogenic disturbance likely impact the herpetofauna, but at spatial scales larger than we could detect.

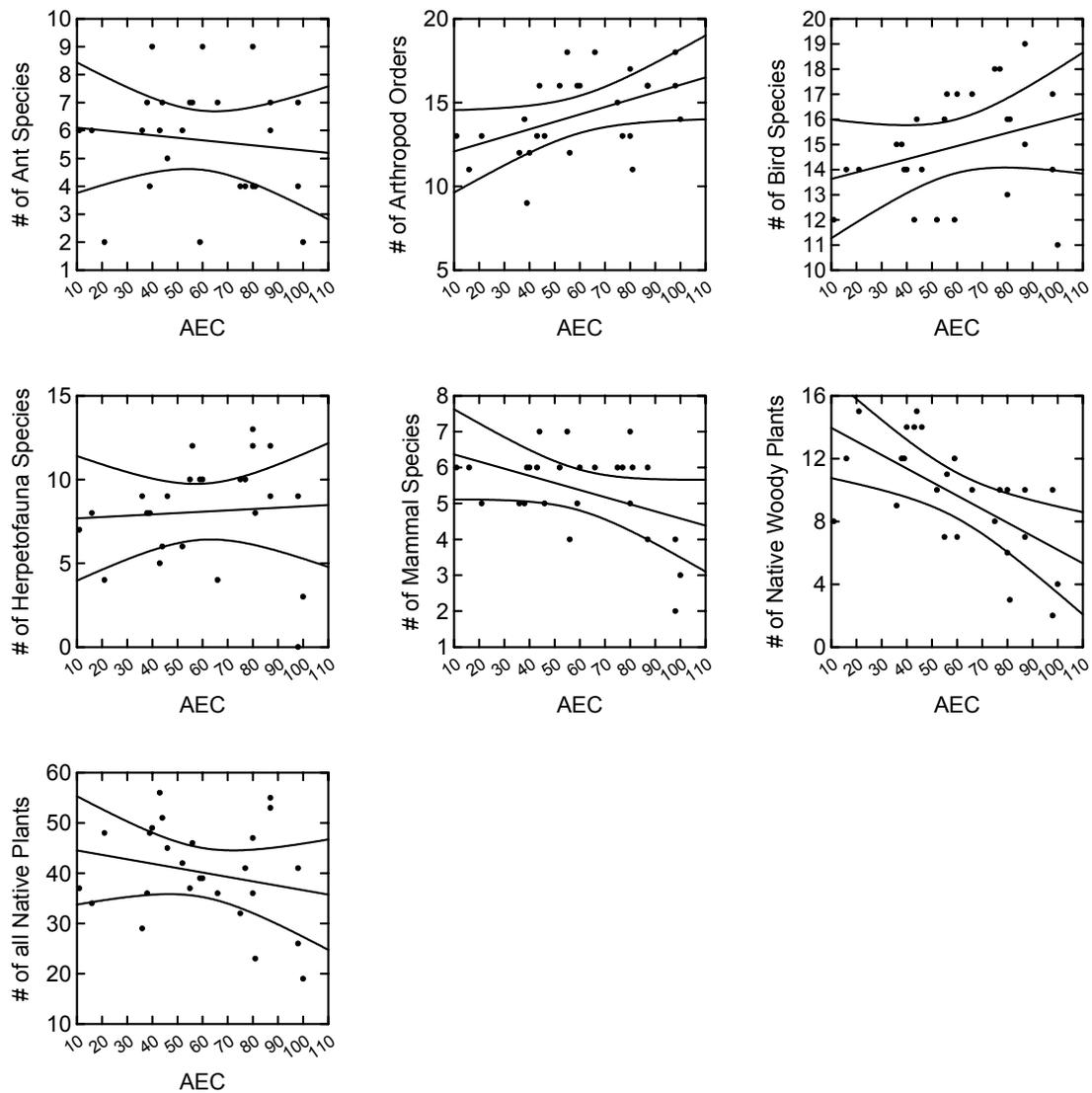


Figure 8. The number of species or orders relative to the level of absolute exotic cover (AEC) for various taxa collected at the IBI study plots. The lines represent linear regressions and their 95% confidence intervals.

Section 2. Exploratory Data Analyses for responses to increased exotic cover, metric selection and scoring.

Methodological overview:

For all taxa we first plotted a metric (presence/absence, abundance, reproductive condition, etc) for each species in the taxa (or orders for arthropods) against levels of

absolute exotic cover for each of the 3 data sets (32 plots sampled all 4 sessions in year 1; 36 plots sampled in 3 of 4 sessions in year 2, and 26 plots sampled all 8 sessions in the 2-year study). We used these plots to categorize species as showing a positive, negative or neutral response to disturbance. We generally considered the 26-plot, 2-year dataset ‘best’ as it contained the fullest assessment of diversity on each plot and the greatest number of samples to assess abundance. We include these scatterplots, for all species as well as a table summarizing responses, in Appendix B. We screened species using the 2-year dataset and then confirmed the trends remained in the other datasets. Once individual species were screened, we combined data, within a single metric, for those species showing either positive or negative responses. We then graphically explored this new metric response to increasing levels of exotic cover and scored the metric using the 1,3,5 categories suggest by Karr and Chu (1997).

We present the scatter plots for all species, labeling those we included in metrics as either “Positive” or “Negative” in Appendix B. We also discuss difficult, or borderline cases, and the level of support for including each species in an IBI metric. This support is based on previous literature and our general level of knowledge of a species. For example, a number of rodents are considered CSS specialists based on large amounts of previous research. Observing these species decline in abundance or occupy fewer sites as exotic invasion increases fits our current understanding of the species and lends credibility to including them in an IBI metric. However, for some species, we observed patterns of increase or decrease across the disturbance gradient, but have found little additional research to explain this pattern. In these cases, we typically included the species in the metric, but feel future research is required to confirm the observed response can be generalized to other areas.

Here, we present scatter plots of IBI metrics created by combining responses across species and the 1,3,5 scoring categories for the metric, while Appendix B contains the individuals species responses.

Ants.

We captured 5154 individuals, identifying 40 distinct types of ant, 16 of these true species. The remaining ants were classified as either morphospecies or considered unidentified. All ants were classified to genus. We did not use morphospecies or unidentified species in the analyses. Our field methods allowed us to detect the presence or absence of ants and count the number of individuals captured at a plot.

The species-specific data generated no reliable metrics. We found species with positive or negative responses in Presence-Absence (Appendix B, figure 1) or Relative abundance (Appendix B, Figure 2) across the AEC gradient. However, the total number of species was small, some species did not show consistent patterns across the 3 datasets, and some species responses varied across sites. When data were combined across species into metrics, the small number of total species in each metric made it difficult to distinguish levels of disturbance. The scatter plots of presence-absence and relative abundance indicate many morphospecies with positive and negative trend. Thus, with better taxonomic resolution, a metric using species-level data is possible.

Since species-level data did not generate metrics, we investigated patterns in genera-level data (Appendix B, Figures 3 and 4). In general, the patterns found using these data were reliable across datasets and we developed 2 metrics (Figure 9). Both

metrics separated low from high levels of AEC but had broad ranges of AEC for the moderate levels of AEC.

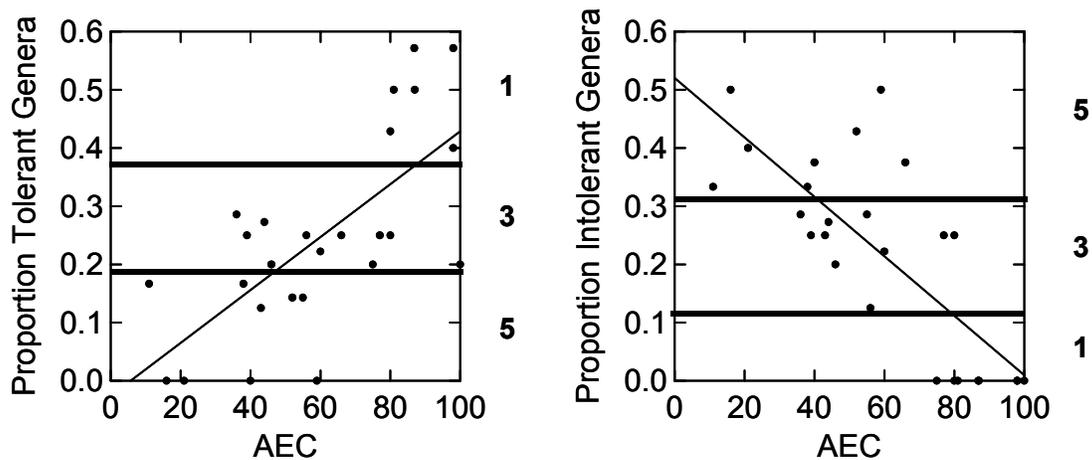


Figure 9. Proportion of tolerant and intolerant ant genera at 26 plots by Absolute Exotic Cover for the 2-year dataset.

Arthropods.

Arthropods, though easy to capture using pitfall traps, are extremely difficult to sort to species. We handled our arthropod data by 1) sorting them to order (except the ants above) and 2) identifying and counting 3 exotic species: earwigs (*Forficula auricularia*) and 2 rollie pollies, *Porcellio laevis* and *Armadillidium vulgare*. The rollie pollies are actually crustaceans but because they were captured using the same methods as arthropods they did not need a separate section in this report. We felt sorting Arthropods to order was a realistic expectation for any large-scale monitoring program implemented in CSS. Identifying many Arthropods to species requires taxonomic expertise held by, in some cases, only a few individuals nationally.

Single orders may contain 100's to 1000's of individual species, each with different responses to anthropogenic disturbance. Given this level of complexity, we chose not to use count data (number of individuals captured) of orders. However, we did use the presence-absence data for orders. The loss or gain of an order across a disturbance gradient represents a potentially large change in the arthropod diversity at a given location.

We found only 1 order with a negative response, and 5 showing positive responses across the AEC gradient (Appendix B, Figure 5). The lack of negative responses was surprising, especially in scorpions, where other studies suggest they may decline in disturbed CSS. Combined presence/absence data across the 5 orders showing increases in occupancy with increasing AEC, generated a clear trend (Figure 10) though this metric did not cleanly separate moderate to highly disturbed sites.

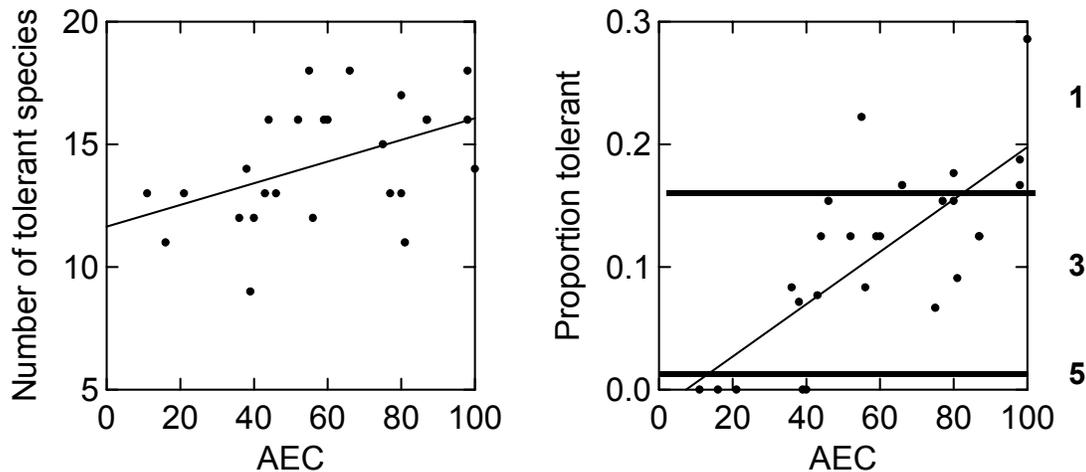


Figure 10. Proportion of intolerant orders within the entire arthropod community against absolute exotic cover (AEC) at 36 sites.

Exotic species. Though we found trends in some of the exotic species, none of these could be used in an IBI framework. Earwigs showed a strong response, appearing only on our most invaded plots (Appendix B, Figure 6). However, because they were only captured at the most extremely invaded locations, plots with absences, ranging from 0 to ~85% AEC would be ranked as either 5 or 3 in an IBI framework, forcing us to rank disturbed sites with high IBI scores. One of the rollie pollies (*Armadillidium vulgare*) appeared to show a decline in abundance with increasing AEC (Appendix B, Figure 6). However, the response was not consistent across sites, as low abundances at all plots at RJER, combined with a decline in abundance with increasing AEC at AWC to produce the overall response. Given these strong site effects, we did not use these data in the IBI.

Birds.

We detected 78 species of birds and counted 3962 individuals during the study. All raptors, crows, ravens and shore birds were removed from the data set since they utilize habitats at large spatial scales. Rare species, those with fewer than 5 detections, were not used in this analysis. We also limited our analyses to birds detected within a 100-meter radius circle from the plot center. This left us with detections of 33 species and 2472 individuals, which we then analyzed using the three data sets described above.

Given the high mobility birds, we detected many species at nearly all study plots after 2 years of sampling, reducing our ability to see threshold responses, even though the

species had strong habitat preferences for either intact CSS or grasslands. We therefore used ‘year 1’, ‘year 2’ and ‘both’ datasets when developing metrics. This differs from the other species, where we used the ‘both’ year dataset to develop metrics then verified these trends remained in the ‘year 1’ and ‘year 2’ datasets. In these cases, we often found threshold trends that matched our pre-existing natural history information using the year-long data. For birds, we present year 1, year 2, and both year datasets in Appendix B.

We combined data across 7 species showing a negative response in presence/absence across the AEC gradient (intolerant species) and 7 species showing positive responses (positive species, Appendix B Figures 7-9). For both proportion of intolerant species (number of intolerant species/total number of species), and the proportion of positive species (number of positive species/total number of species) we found reasonably strong relationships with increasing exotic cover (Figure 11). Proportion intolerant separated 5 high AEC plots from all other plots, but did not give high levels of resolution between low and intermediate levels of AEC. Similar to proportion intolerant, proportion positive separated 5 sites with AEC levels greater than 80% while plots with intermediate to low levels of disturbance were difficult to distinguish.

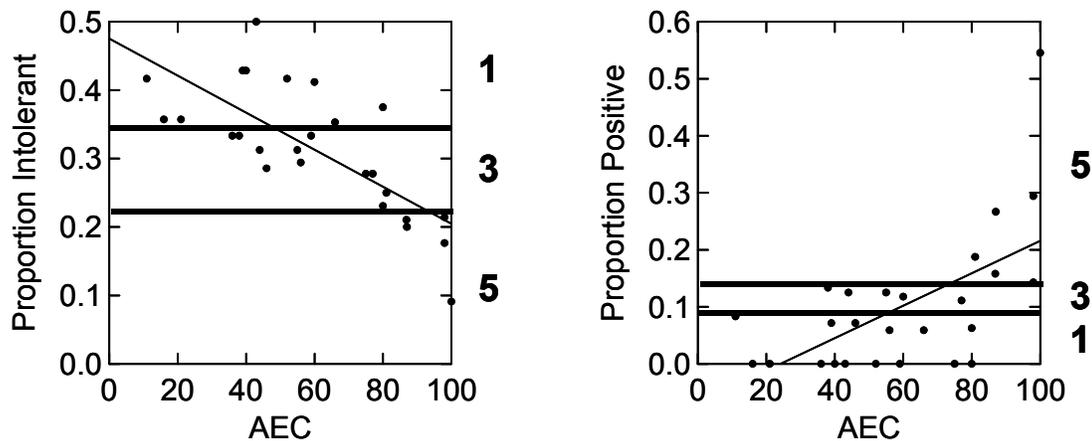


Figure 11. Proportion of 7 intolerant and 7 positive responding species in the entire bird versus absolute exotic cover (AEC). The data from the 26 plot, 8-session, 2-year data set.

We found a variety of responses in bird relative abundance across the gradient of invasion (Appendix B, Figures 10-12). Based on natural history and the patterns we observed across the AEC gradient, we selected 5 species for an intolerant species metric and 8 species for a metric based on increases in relative abundance with increasing AEC. The relative abundance of intolerant species showed a strong linear decline with increasing AEC and separated all disturbance categories (Figure 12, left panel). The relative abundance of positive responding species was generally low (0-0.4, Figure 12) but one high AEC plot had a value of 0.8. Given the shallow, and variable relationship, the metric did not cleanly distinguish between all disturbance categories.

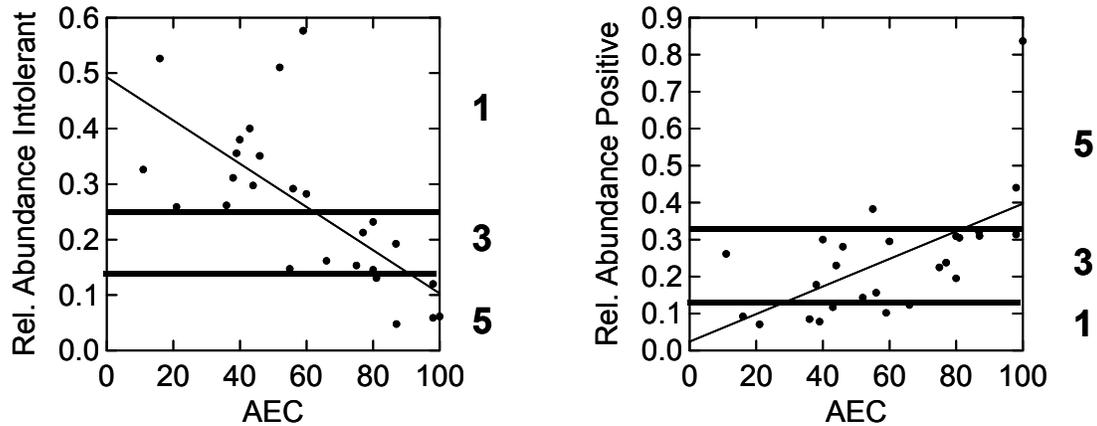


Figure 12. Relative abundance of 4 intolerant species and 7 positive species versus absolute exotic cover (AEC). The data from the 26 plot, 8-session, 2-year data set.

Herpetofauna.

Data collected by the United States Geological Survey near our plots was not perfectly synchronized with our vegetation samples. Our experimental design was opportunistic in this regard. We did not have funding to pay for additional sampling of herpetofauna and USGS had ongoing contracts at the reserves we selected for the project. We sampled in the Spring of 2002 and 2003, while USGS collected data from 1998 to 2003 at our sites.

We first attempted to only use USGS data collected from Spring 2002 through Spring 2003 to tighten the relationship between herpetofauna captured and vegetation present. However, doing so resulted in small sample sizes in nearly all cases and at some plots, no data at all. Instead, we chose to use all of the herpetofauna data available at a plot, to maximize sample sizes. We realize levels of exotic cover, and herpetofauna activity vary from year to year but feel the level of variability in the vegetation community across the study plots was greater than temporal fluctuations at any given plot. Thus we compared all of the herpetofauna data collected at a plot to the absolute exotic cover at the plot in 2003 and assume the vegetation community at any given plot did not change considerably from 1998-2003.

27 species were collected during the study. Of these, 4 showed obvious positive threshold responses in presence/absence to increased levels of exotic cover while 5 showed negative responses (Appendix B, Figure 13). The remaining species showed no obvious response across levels of exotic cover, being captured in both closed canopy CSS and open grasslands, or were too rare for further analyses.

We combined data across species with positive and negative responses to create potential IBI metrics. The number, and especially proportion, of intolerant species declined with levels of absolute exotic cover (Figure 13). In addition, the number and

proportion of tolerant species showed a strong positive relationship with AEC (Figure 13).

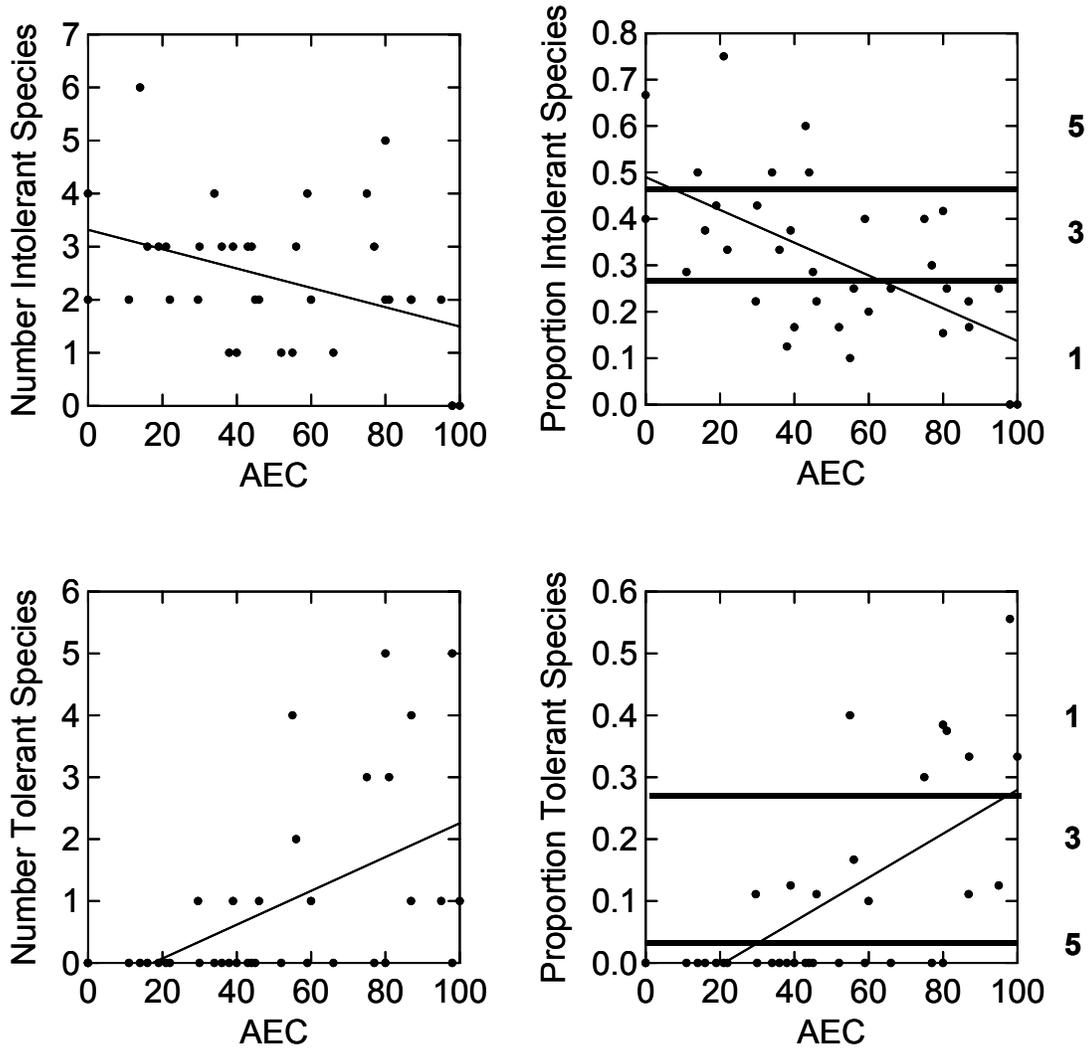


Figure 13. Number and proportion of intolerant and tolerant species within the entire herpetofauna community against absolute exotic cover (AEC2003) at 36 sites.

Herpetofauna abundances were generally low across plots. In 21 of the 27 species, 5 or fewer individuals were captured at a site, and in most cases, only one individual was captured (Appendix B, Figure 14). In these cases, we could not reliably estimate changes in abundance across the gradient. Of the remaining 6 species, 2 had unimodal relationships with AEC, showing highest abundances at medium levels of AEC and lower abundances at both higher and lower levels of AEC (Appendix B, Figure 14). 4 species showed negative responses in abundance with increasing AEC (Appendix B, Figure 14). We combined the abundances of these 4 species into an intolerant species group and also calculated the relative abundance (Figure 14). Despite showing a strong, general decline with increasing disturbance, relative abundance did not clearly separate sites with high to moderate levels of AEC. Plots with ~0.7 to 1.0 AEC had a wide range

of relative abundance values (~0.25-0.8) making them hard to distinguish from other sites.

We created 1,3,5 scoring lines to separate intact sites from the remaining locations. If we had placed the line separating 1 from 3 at 0.55 to include the 3 sites with low AEC (see Figure 14), we would have then miscategorized ~11 sites with moderate to high levels of disturbance. In general, this metric is of lower quality than others, but somewhat separates relatively intact sites from the rest.

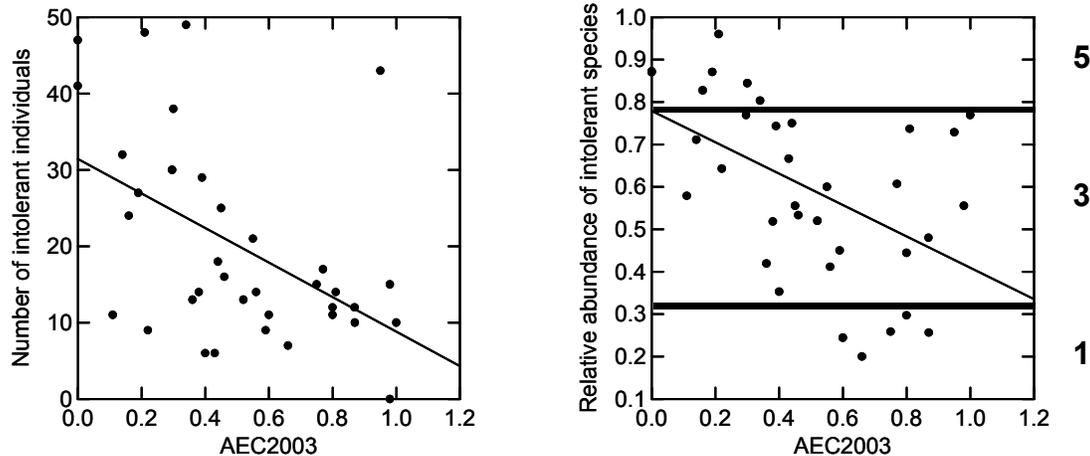


Figure 14. Scatter plot of the number of individuals of intolerant species and their relative abundance across absolute exotic cover in 2003 (AEC2003) at 36 plots.

Mammals.

Our capture-recapture methods allowed us to estimate not only abundance and species richness at each site, but also aspects of reproduction (Proportion of males and females reproductively active), average body weights of males and females, and the size structure of the population (i.e. proportion of juveniles). Karr and Chu (1997) consider metrics related to population processes and individual health important aspects of IBIs. Thus, for small mammals, we explored these individual and population metrics in addition to the community level metrics.

Unfortunately, we found no usable metrics from the information we gathered on reproduction, body weights and size structure across the disturbance gradient for the small mammals (Appendix B, Figures 15-19). Demographic processes varied by season and through time for each species, and in different temporal patterns from site to site, obfuscating trends across the disturbance gradient.

We did find a potential metric using male body weight. Nearly all mammal species showed a decline in average male body weight with increasing levels of exotic invasion (Appendix B, Figure 17). The strength of this relationship varied across species but was generally weak, and statistically non-significant in many cases. It may be possible to combine standardized body weights across species to develop an IBI metric. However, we could not hypothesize a mechanism causing this general trend in males across so many species. Furthermore, the relationship was weak and statistically not significant in many cases. As such, we did not have enough confidence in this metric to

use it in our IBI, but feel the pattern we observed is worth further investigation and validation.

Patterns of presence-absence across AEC allowed us to create 2 metrics. In general, a suite of species showed declines in the chance of occupying a site as disturbance increase while another group of species was tolerant to such changes (Appendix B, Figure 20). We could capture this response using first, overall mammal species richness, which declined with increasing exotic cover (Figure 15, left). This metric separated highly disturbed sites from other sites, but poorly separated sites with low to moderate levels of disturbance. We also used the proportion of 4 tolerant species in the community as a metric (Figure 15, right). This metric showed positive relationship with exotic cover and separated all disturbance categories reasonably well.

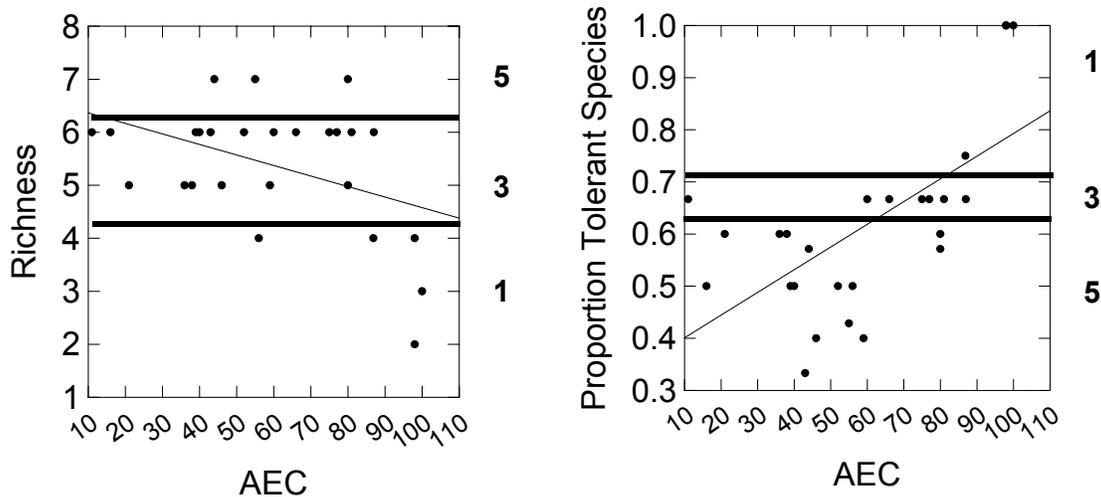


Figure 15. Species richness the proportion of intolerant species within the entire small mammal community against absolute exotic cover (AEC) at 36 sites. The line represents a linear regression through the points.

We investigated two potential IBI metrics based on the small mammal abundance data. Three species declined in relative abundance with increasing AEC while 3 increased (Appendix B, Figure 21). We combined data across these two groups to calculate the proportional abundance of intolerant species (negative responses, *Microtus californicus*, *Neotoma Lepida* and *Peromyscus californicus*) and tolerant species (positive responses, *Peromyscus maniculatus*, *Reithrodontomys megalotis*, and *Dipodomys simulans*). The metric based on intolerant species separated low and high levels of disturbance reasonably well (Figure 16, left). Unfortunately, the metric based on tolerant species, though a positive trend, had high levels of variability and low values at all levels of AEC (Figure 16, right). This metric did not adequately separate disturbance categories and we did not use it in the final IBI.

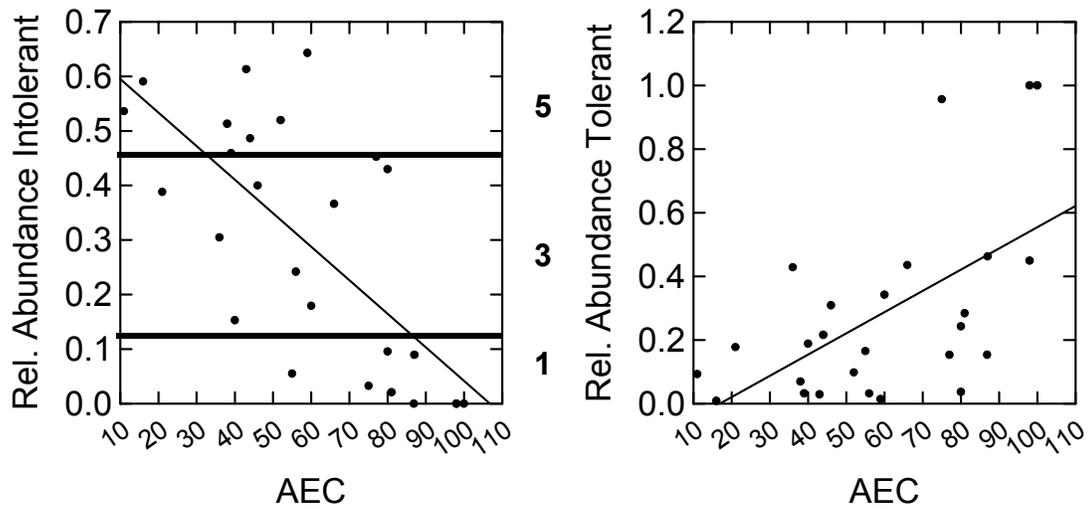


Figure 16. Proportion of intolerant and tolerant individuals from 3 tolerant and 3 intolerant species against absolute exotic cover (AEC) at the 26 study plots. We did not use proportion tolerant as a metric in the IBI. For the intolerant metric: $Y=0.545+$

Vegetation.

The vegetation sampling created the most diverse, complex dataset in the study. Additional vegetation analyses are being done by Genie Fleming, a MS student at SDSU. We could not use exotic plant species in any IBI metrics because we used these data to rank plots. Using the same data to both create the gradient and measure responses creates circular reasoning.

Instead, we focused on the response of the native plant community and individual native species across the gradient and developed 3 IBI metrics. We searched for responses across a wide array of summary variables (e.g. total native species richness, forb cover, etc) and performed species-specific screening of presence-absence data for 214 plant species, as we did for other taxa. Appendix B, figures 22 and 23 and Tables 3 and 4, illustrate these explorations.

The woody plant community responded strongly and negatively to our disturbance gradient. Our literature review (Appendix A) found a number of studies showing scrub cover (i.e. woody species) declined with increased grazing, agriculture, and decreased fire return interval. Thus, we expected the woody component to decline across our gradient and we saw this response in both the cover of woody species and woody species richness. Two IBI metrics described this response (Figure 17). First, absolute woody cover declined sharply across the gradient, separating all three levels of disturbance, but particularly high from low levels. Second, total native woody species richness also declined but this metric did not separate levels of disturbance as well as Absolute Woody Cover.

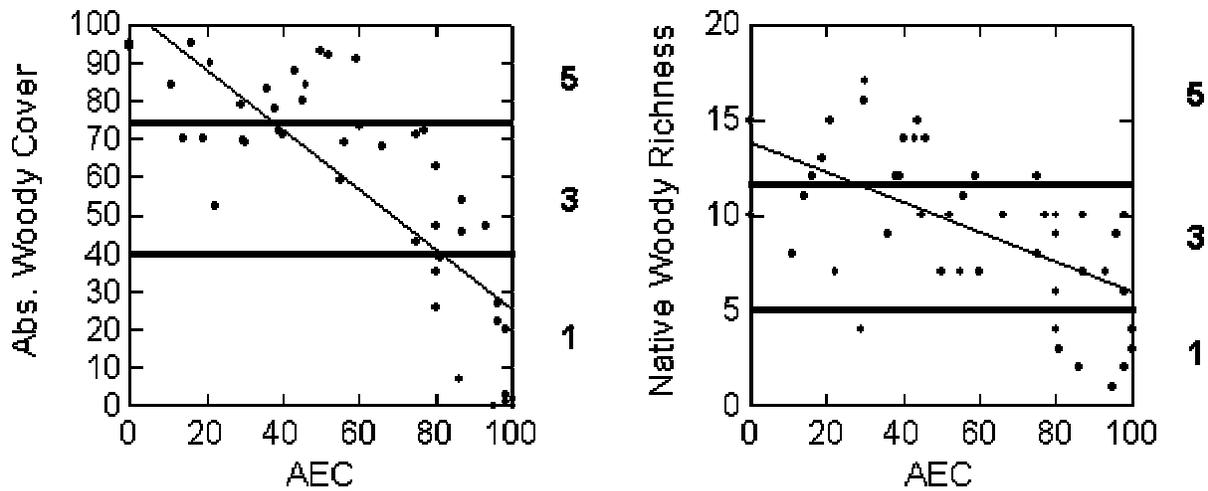


Figure 17. Absolute Woody Cover and Native Woody Richness versus Absolute Exotic Cover.

In addition, we found 28 species with reasonable biological justification, showing positive or tolerant responses in presence-absence across the AEC gradient (Appendix B, Figure 23, Table 4). We used these species in a Proportional Tolerant metric, which weakly separated disturbance categories with rare miscategorizations (Figure 18).

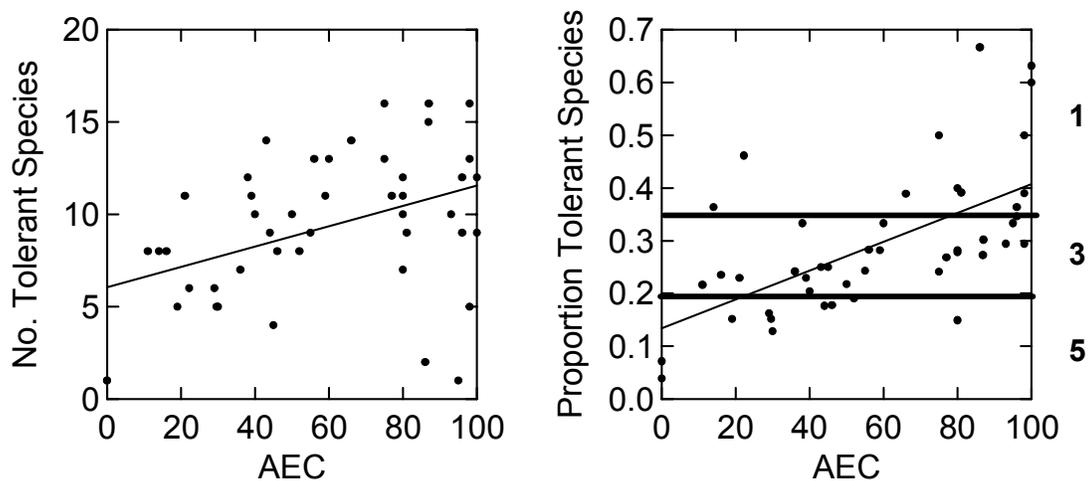


Figure 18. Number and Proportion of Tolerant Species versus Absolute Exotic Cover.

An IBI for CSS:

We combined the 16 metrics into a single IBI by simply summing the 16 individual metric scores (See Table 3, following page, for metric summaries) for each plot (Appendix B, Tables 5 and 6-11). The IBI ranged from 22 to 72. To understand how the CSS-IBI was related to the AEC gradient we ran regressions between the CSS-IBI and AEC for each data set (Figure 19). Across all datasets, AEC explained a large percent of the variation in the IBI (60- 78%). Data combined across years, despite a smaller sample size, showed a better fit than either year 1 or year 2 datasets (Both: $IBI = -0.546AEC + 82.142$, $adj. r^2 = 0.775$, $n = 26$, $p < 0.001$; Year 1: $IBI = -0.446AEC + 74.217$, $adj. r^2 = 0.589$, $n = 32$, $p < 0.001$; Year 2: $IBI = -0.433AEC + 74.975$, $adj. r^2 = 0.704$, $n = 36$, $p < 0.001$). Interestingly, the year 1 data, collected during a drought, showed the lowest fit, perhaps indicating how incomplete sampling may influence IBI performance.

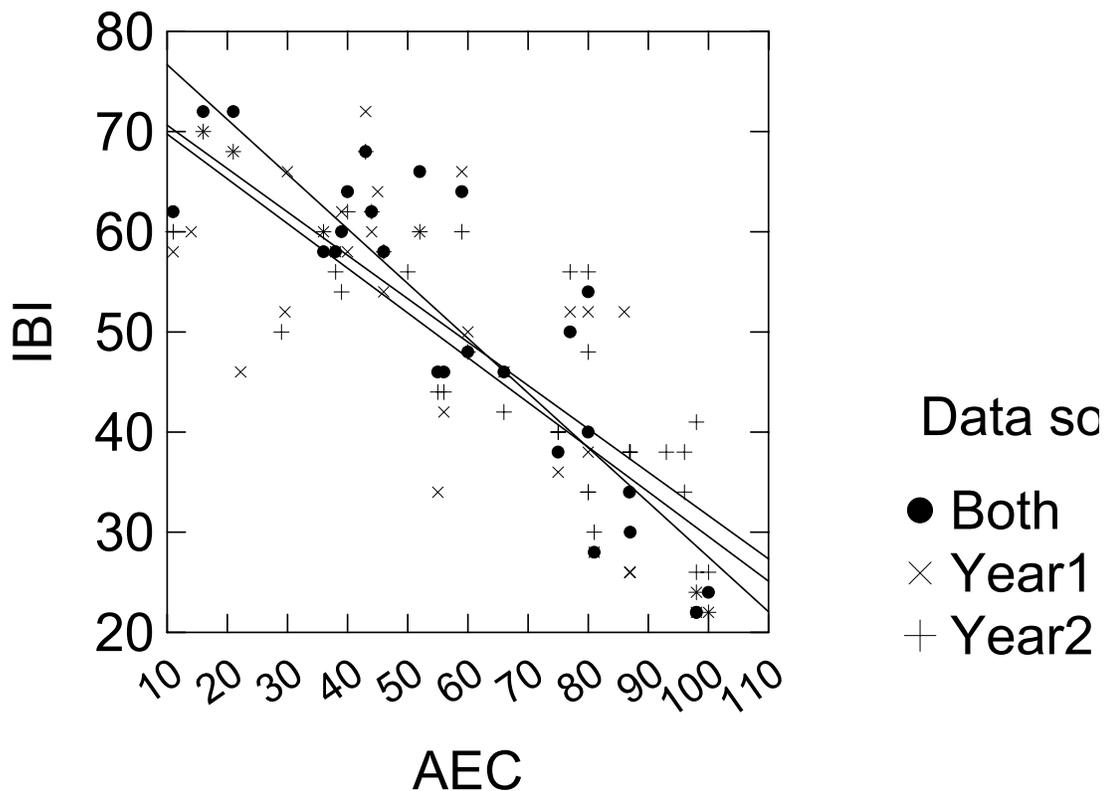


Figure 19. CSS-IBI versus AEC for the three datasets used in analyses.

To further investigate the IBI and its ability to distinguish levels of disturbance, we performed a series of cluster analyses on each of the data sets. After using the clustering algorithms to separate plots based on their suite of individual metric scores, we

the performed one-way ANOVA's with Tukey post-hoc tests to determine if the overall IBI score differed across categories created by the clustering algorithms.

Table 3. 16 IBI metrics. Summary of metrics, the 4 letter codes used in Appendix B, and the rules for creating 1,3,5 scores for each metric. Table 1 in Appendix B, gives species names associated with each species code.

Taxa	Metric	Code	Metric Description	1,3,5 rules			Species or order in Metric
				1	3	5	
Ants	Proportion of Tolerant Genera	APTG	# tolerant guild genera/total # genera	>0.38	0.2-0.38	<0.20	MESS, POGO, TETR, PHEI
	Proportion of Intolerant Genera	APIG	# intolerant guild orders/total # genera	0	>0-0.3	>0.3	DURY, FORM, LINE, MYRM
Arthropods	Proportion of Tolerant Orders	APTO	# tolerant guild orders/total # orders	>0.16	>0-0.16	0	DIPLO, DIPT, ISOPT, OPII, THYSAN
Birds	Proportion Tolerant species	BPTS	# tolerant guild species/total # species	>0.14	0.05 - .14	<0.05	BLGR, BRBL, GRSP, LASP, WCSP, WEKI, WEME
	Proportion Intolerant Species	BPIS	# intolerant guild species/total # species	<0.22	0.22-0.34	>0.34	ANHU, COBU, NOMO, SOSP, SPTO, WESJ, WREN
	Relative Abundance of Tolerants	BRAT	# tolerant guild individuals/ total # individuals	>0.31	0.16-0.31	<0.16	BLGR, BRBL, CALT, GRSP, LASP, WCSP, WEKI, WEME
	Relative Abundance of Intolerants	BRAI	# intolerant guild individuals/ total # individuals	<0.16	0.16-0.34	>0.34	ANHU, SOSP, SPTO, WREN, WESJ
Herpetofauna	Proportion Intolerant Species	HPIS	# intolerant guild species/total # species	<0.26	0.26-0.49	>0.49	BANI, CRVI, ELMU, HYRE, LAGE, MALA
	Proportion Tolerant species	HPTS	# tolerant guild species/total # species	>0.28	>0-0.28	0	LEHU, SCHA, TAPL, UTST, PHCO
	Relative Abundance of Intolerants	HRAI	# intolerant guild individuals/ total # individuals	<0.3	0.3-0.8	>0.8	CNTI, ELMU, MALA, SCOC
Mammals	Native Species Richness	MNSR	# native species whose ranges overlap the entire study area and were equally sampled on all plots	<4.6	4.6-6.5	>6.6	DISI, MICA, NEFU, NELE, PECA, PEER, PEMA, REME
	Proportion of Tolerant Species	MPTS	# tolerant guild species/total # species	>0.72	0.71-0.61	<0.61	DISI, PEER, PEMA, REME
	Relative Abundance of Intolerants	MRAI	# intolerant guild individuals/ total # individuals	<0.12	0.13-0.453	>0.453	MICA, NELE, PECA
Plants	Absolute Woody cover	AWC	Proportional cover of all woody species	>0.60	0.25-0.59	<0.24	All woodyspecies, see Table 1, Appendix B
	Total Native woody richness	TNWR	# of native woody species	>12	5-11	<4	All woodyspecies, see Table 1, Appendix B
	Proportion of Tolerant Species	PPTS	# of tolerant species/total number of species	>0.34	0.2-0.34	<0.2	See Table 1, Appendix B

Metric Clustering Methods and results

Data sets from each year of sampling (Year 1, Year 2 and both years) were screened for multivariate outliers on the set of 16 metric scores using Mahalanobis distance. No outliers were detected in any year. Multicollinearity among the 16 metrics was also examined. Correlation coefficients ranged from $<0.01 - 0.8$. High correlations among sets of variables can substantially influence cluster results when the number of variables in each set are substantially different. Because the largest set of variables with consistently high correlations ($r \sim 0.7$ or greater) across all years (1,2, and both) was small (3 metrics: HPTS, WRA, and PII), no metrics were excluded from the cluster analyses.

Cluster analyses were performed using squared Euclidean distance as a proximity measure between metric scores. No data standardization procedures were used as all metric scores occur on the same scale. Different agglomeration methods were initially explored, but Ward linkage produced the best clustering results and was used in the final analyses. Evaluation of clustering results focused on the number of clusters in the “optimal” solution(s) (where the optimal solution was determined by the location of uniquely large gaps in distance coefficients and the presence of outliers (i.e single- or few-plot clusters). Theoretical considerations regarding distinguishing disturbance levels made a 3 to 5 cluster solution with few outliers desirable. After cluster solutions were determined, the nature of the clusters was explored by examining cluster means on the individual metric scores used in the clustering procedure.

Year 1

Analysis of Year 1 data suggested potential 4- and 5-cluster solutions (Figure 20). The 4-cluster solution contained similarly sized groups, while the 5-cluster solution contained one small cluster with only 3 members. The properties of both solutions were explored.

Dendrogram for Year 1 Data

Rescaled Distance Cluster Combine

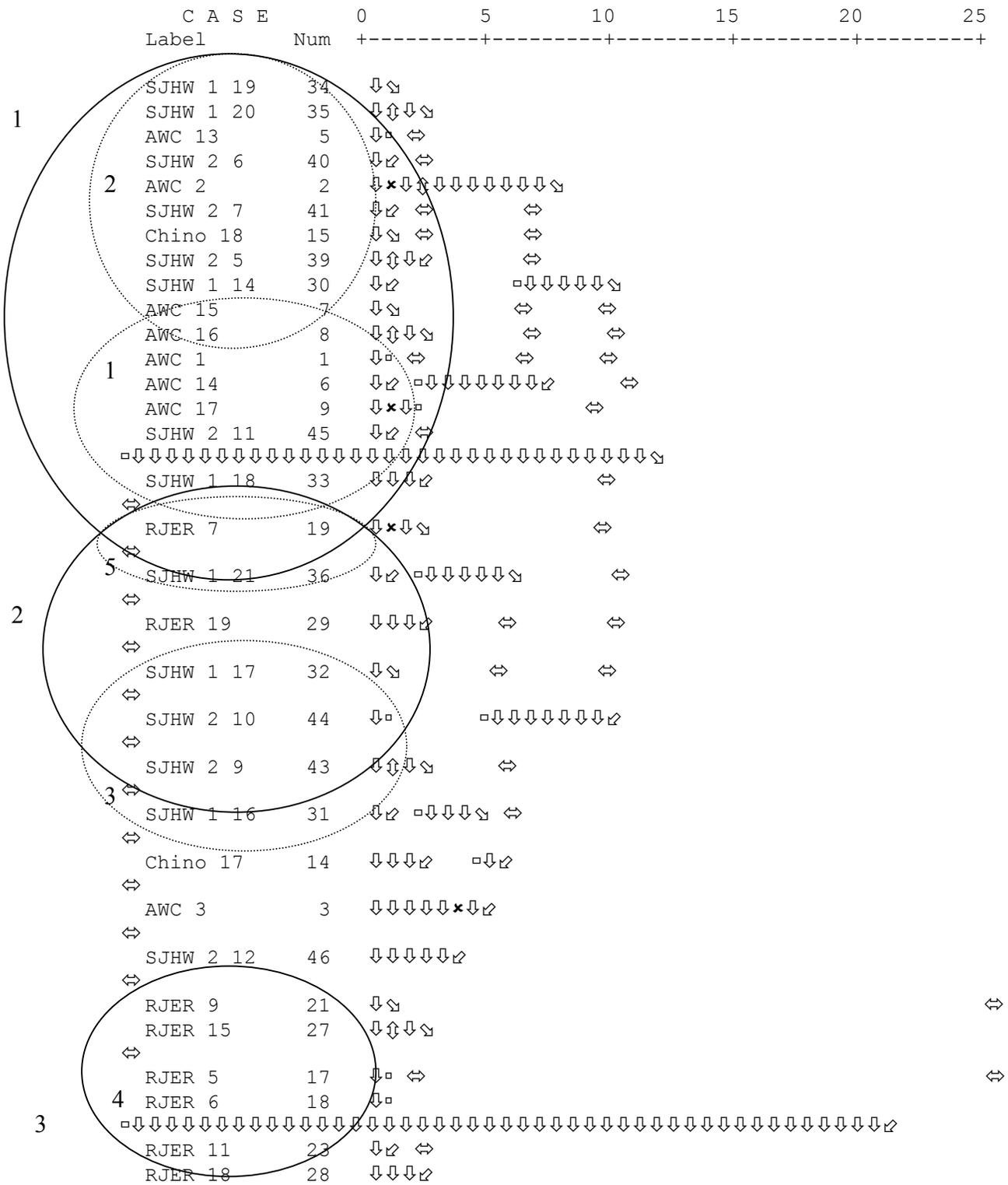


Figure 20. Clustering diagram for year 1 data showing 4 and 5 cluster solutions.

In the 4-cluster solution utilizing Year 1 data, group means differed significantly (according to univariate omnibus F-tests) across clusters for all metrics except PPTS, TNWR, and APTG (Table 4). In general plots in cluster 4 had low scores on average on most metrics; plots in cluster 3 had low to moderate scores on metrics; and plots in clusters 1 and 2 had moderate or high scores on metrics.

Table 4 Year 1 mean metric scores for 4-cluster solution.

Cluster Statistics Year 1

Cluster	n	PPTS_S1	AWC_S1	TNWR_S1	PTS_S1	PII_S1	HRAI_S1	NSR_S1	HPIS_S1	HPTS_S1	GRA_S1	WRA_S1	GPA_S1	WPA_S1	APTG_S1	APTO_S1	APIG_S1
1	7	2.7	4.7	3.6	5.0	3.9	3.6	3.3	3.0	5.0	3.6	4.4	4.7	3.9	4.4	5.0	4.4
2	10	3.0	3.6	3.4	4.4	4.0	3.4	3.0	3.4	4.8	3.2	4.2	5.0	3.8	3.6	1.0	4.4
3	9	2.8	3.0	3.2	2.8	1.9	2.1	1.9	1.4	3.4	2.6	3.0	4.6	3.4	3.9	1.4	2.8
4	6	1.7	2.0	2.0	1.0	1.0	2.3	1.3	1.0	1.3	1.3	1.7	1.0	1.0	4.0	1.0	1.0

In the 5-cluster solution utilizing Year 1 data, group means differed significantly across clusters for all metrics except TNWR and APTG (Table 5) The 5-cluster solution differed from the 4-cluster solution by the splitting of cluster 3 into 2 separate clusters (3 and 5). Again, plots in cluster 4 had low scores on average on most metrics, while plots in clusters 1 and 2 had moderate or high scores on metrics. Plots in clusters 3 and 5 had low to moderate scores on most metrics, but the scores in cluster 5 were generally lower than those in cluster 3.

Table 5 Year 1 mean metric scores for 5-cluster solution.

Cluster Statistics Year 1: 5-cluster solution

Cluster	n	PPTS_S1	AWC_S1	TNWR_S1	PTS_S1	PII_S1	HRAI_S1	NSR_S1	HPIS_S1	HPTS_S1	GRA_S1	WRA_S1	GPA_S1	WPA_S1	APTG_S1	APTO_S1	APIG_S1
1	7	2.7	4.7	3.6	5.0	3.9	3.6	3.3	3.0	5.0	3.6	4.4	4.7	3.9	4.4	5.0	4.4
2	10	3.0	3.6	3.4	4.4	4.0	3.4	3.0	3.4	4.8	3.2	4.2	5.0	3.8	3.6	1.0	4.4
3	6	3.7	3.0	3.3	3.7	2.3	2.0	2.3	1.0	4.0	2.7	3.0	4.7	3.7	4.0	1.0	2.7
4	6	1.7	2.0	2.0	1.0	1.0	2.3	1.3	1.0	1.3	1.3	1.7	1.0	1.0	4.0	1.0	1.0
5	3	1.0	3.0	3.0	1.0	1.0	2.3	1.0	2.3	2.3	2.3	3.0	4.3	3.0	3.7	2.3	3.0

SJHW 2 5 35 ↓↗ □↓↓↓↓↗
 RJER 8 16 ↓↓↓↗

Figure 21. Clustering diagram for year 2 data showing a 4 cluster solution.

Table 6. Year 2 mean metric scores for the 4-cluster solution.

Cluster Statistics Year 2: 4-cluster solution

Cluster	n	PPTS_S2	AWC_S2	TNWR_S2	PTS_S2	PII_S2	HRAI_S2	NSR_S2	HPIS_S2	HPTS_S2	GRA_S2	WRA_S2	GPA_S2	WPA_S2	APTG_S2	APTO_S2	APIG_S2
1	10	3.4	4.2	4.0	4.6	3.8	3.4	3.2	3.2	5.0	3.0	3.6	4.4	3.8	3.8	4.2	4.4
2	10	3.0	3.8	2.8	4.6	3.6	2.6	2.8	1.8	4.2	3.2	3.6	4.2	3.4	2.6	3.6	2.0
3	12	1.7	1.5	2.2	1.4	1.0	2.7	1.3	1.2	1.5	2.7	1.8	4.5	2.3	2.8	2.5	1.7
4	4	3.5	3.0	3.0	4.5	2.0	2.0	3.0	1.0	1.0	2.0	1.5	5.0	2.0	1.5	2.0	1.5

Both years

Data from only plots sampled in both years, using year 2 values, yielded a 4-cluster optimal solution (Figure 22, Table 7). Most cluster sizes were similar except for one small 3 plot cluster. In the 4-cluster solution, group means differed significantly across clusters for all metrics except PPTS, HPIS, and GRA (Table 7). In general cluster 1 had high scores on most metrics; cluster 2 had moderate to high scores; cluster 3 had low scores; and cluster 4 had moderate or low metric scores.

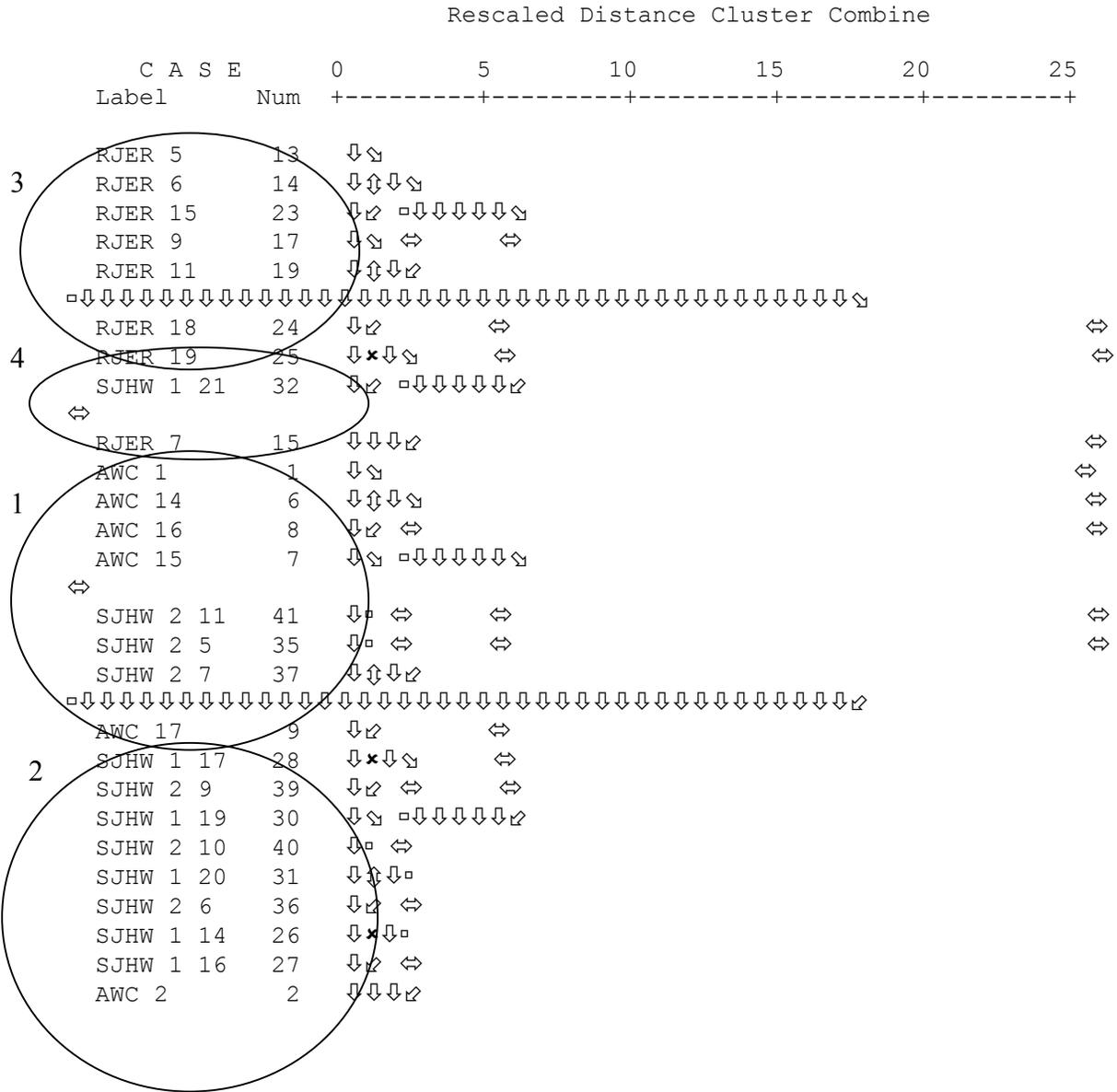


Figure 22. Clustering diagram for Both data showing 2 and 5 cluster solutions.

Cluster Statistics Both Years: 4-cluster solution

Cluster	n	PPTS_SB	AWC_SB	TNWR_SB	PTS_SB	PII_SB	HRAI_SB	NSR_SB	HPIS_SB	HPTS_SB	GRA_SB	WRA_SB	GPA_SB	WPA_SB	APTG_SB	APTO_SB	APIG_SB
1	8	3.3	4.8	3.8	4.8	4.5	3.5	3.0	2.8	5.0	3.5	4.3	4.5	4.5	5.0	4.0	4.8
2	9	3.2	3.4	3.4	4.3	3.4	2.6	3.2	2.3	4.6	3.4	3.4	3.2	3.9	3.0	2.8	3.2
3	6	1.7	1.7	2.0	1.7	1.0	2.3	1.7	1.0	1.3	4.3	1.3	1.0	1.3	1.3	2.0	1.0
4	3	3.0	3.0	3.0	4.3	1.0	1.7	3.7	1.7	1.0	3.7	1.0	4.3	3.0	3.0	2.3	1.7

Table 7. Both year mean metric scores for 2-cluster solution.

To examine the stability of cluster membership for individual plots across years, we ranked clusters from low to high based on IBI metric scores as described for each cluster solution. We then examined Spearman rank correlations among cluster ranks for each plot across the 3 data sets (Year 1, Year 2, Both) to determine the extent to which year influenced the ranking and grouping of plots. Only the 4-cluster solutions from each data set were used. In addition, we only selected plots with IBI scores across each of the datasets. Correlations of cluster rankings among years for individual plots were high (Table 8), indicating plots were consistently grouped with other plots of similar quality in all years.

Table 8. Spearman rank correlations for cluster ranks across years for individual plots.

Cluster Rank Correlations

		Cluster rank year 1	Cluster rank year 2	Cluster rank both years
Cluster rank year 1	Spearman's rho	1.000	0.778	0.931
	Sig. (2-tailed)	.	0.000	0.000
	N	32	26	26
Cluster rank year 2	Spearman's rho	0.778	1.000	0.775
	Sig. (2-tailed)	0.000	.	0.000
	N	26	36	26
Cluster rank both years	Spearman's rho	0.931	0.775	1.000
	Sig. (2-tailed)	0.000	0.000	.
	N	26	26	26

ANOVA results on Clusters.

The regression analyses indicated the IBI score and AEC are strongly related, however, our goal was to determine how many different categories of disturbance the overall IBI score could distinguish. We performed one-way ANOVA's using the unique groups of plots created by the cluster analyses as categories in the ANOVAs. Thus, we tested if the overall IBI score could distinguish between groups of plots clustered based on the pattern of individual metric scores. We did this because 2 plots could have identical overall IBI scores, yet different patterns of scores across individual metrics. For example, in a 5 metric case, a plot could have scores of 1,1,1,5,5 across the metrics, while another 5,5,1,1,1, and another 3,3,3,3,1. All plots have the same overall IBI score (13), yet they represent very different patterns of response to disturbance. If the overall IBI does not differ among the categories of plots distinguished by the cluster analyses, then situations like the example above have occurred and the overall IBI score would be less

effective at distinguishing responses to disturbance. Since our goal was to distinguish as many levels of disturbance as possible, we only present the results for the largest number of clusters within each dataset (Year 1, 5 clusters, Year 2, 4 clusters, and Both, 4 clusters).

Year 1.

The IBI differed across all 5-plot clusters ($F_{4,27}=64.313$, $p<0.001$) though statistical significance was 0.07 for the comparison of clusters 1 and 2 while groups 3 and 5 were not significantly different ($p=0.16$,

Figure 23). All other Tukey-post hoc comparisons showed statistically significant differences across clusters at $p\leq 0.014$. The order of the clusters is trivial in these figures.

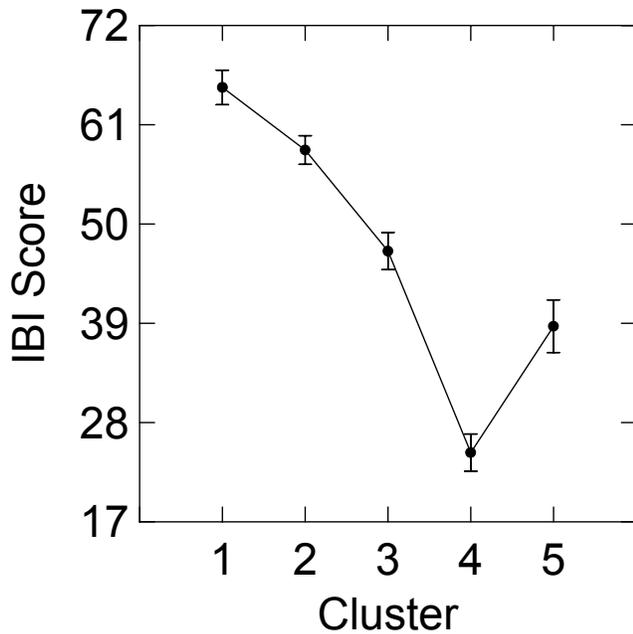


Figure 23. Mean IBI score for 5 different clusters.

Year 2.

The IBI differed across 3 of the 4 plot clusters ($F_{3,32}=47.948$, $p<0.001$) though groups 3 and statistical significance was 0.058 for the comparison of clusters 1 and 2

(Figure 24). Clusters 3 and 4 were not statistically different. All other Tukey-post hoc comparisons showed statistically significant differences across clusters at $p\leq 0.001$.

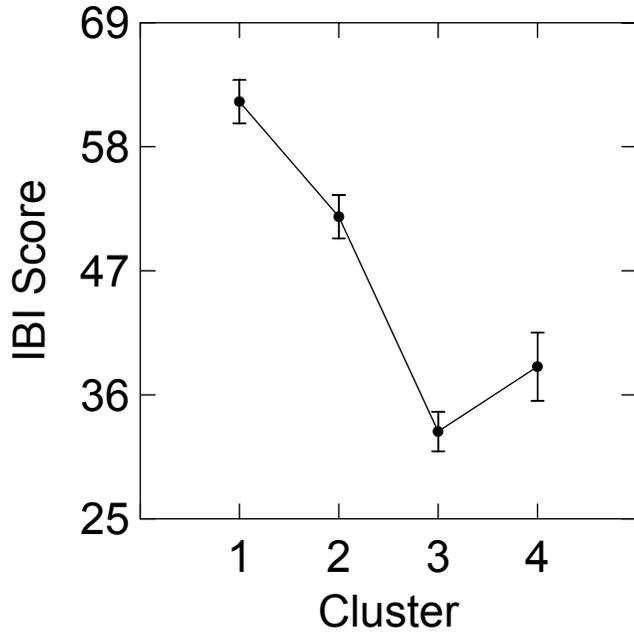


Figure 24. Mean IBI Score for 4 different clusters

Both years.

The mean IBI score differed across all 4 of the plot clusters ($F_{4,21}=65.488$, $p<0.001$) as all Tukey-post hoc comparisons showed statistically significant differences across clusters at $p\leq 0.001$.

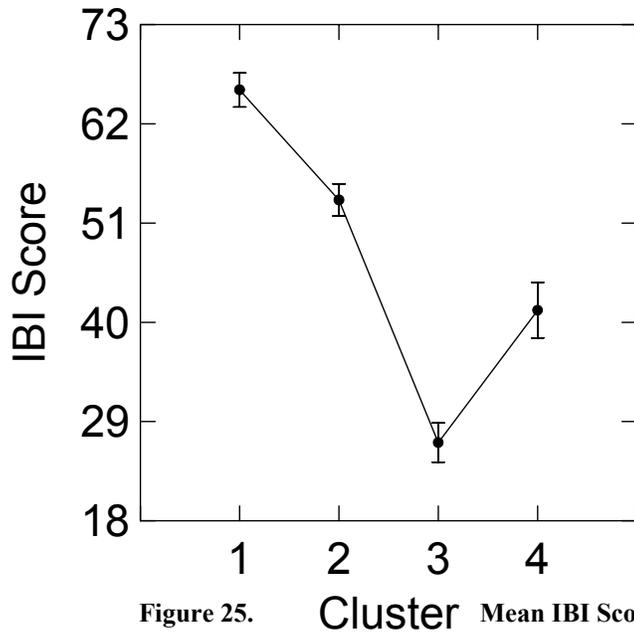


Figure 25. Mean IBI Score for 5 different clusters.

Summary.

These analyses indicate the IBI tracks change across the disturbance gradient well and is reasonably robust across years. In all three datasets, the IBI could distinguish either all, or all but one, of the categories created by the clustering algorithms. Thus, the overall IBI score is highly correlated with metric-specific patterns of response across the AEC gradient and can distinguish 3 to 5 levels of disturbance with confidence. The similar and strong regressions between IBI and AEC across years (Figure 19) and the tight correlations in plot rankings (Table 8) indicates the IBI performed well even with different levels of sampling and across drought years.

CONCLUSIONS AND FUTURE WORK.

Here we discuss topics related to the development, validation, and implementation of an IBI in CSS based on the research we performed. Instead of concluding with a list of recommendations, we place our recommendations within the supporting text, but offset and labeled for easier identification. We occasionally interject conclusions regarding the ecological response of CSS taxa to disturbance. Designing our study to allow IBI development required measuring ‘ecological dose response curves’ (Karr and Chu 1997). These curves, the ~300 scatter plots in Appendix B, provide insights into how CSS species and communities are impacted by historical land use and exotic invasion. These insights should not be ignored or utilized by focusing solely on IBI development. We will pursue the details of these responses in the next few years through additional analyses (2 MS theses, for example), and publications. Thus, the detailed ‘story’ of this research is a work in progress. However, a number of patterns are noteworthy and have consequences for the conclusions we draw regarding IBI and CSS management.

We focus the discussion loosely around three questions. First, “*Is an IBI for CSS possible?*” The short answer is “Yes”. Second, “*Is an IBI for CSS ready for use?*” We reluctantly answer “Not yet” but offer recommendations and examples for refinement and validation. Third, “*Is an IBI for CSS necessary?*” Here we answer “perhaps” and discuss potential applications of the IBI we developed, other index-based approaches for larger spatial scales, and discuss how an IBI might fit into a larger adaptive management framework in Southern California.

Is an IBI for CSS possible?

Yes. Our research indicates tractable responses to disturbance can be found in a wide variety of taxa in CSS. These responses reflect real changes in both CSS community composition and its inherent ecological processes. In addition, our study suggests the scoring system advocated by Karr and Chu (1997) allows discrimination of 4 to 5 categories of disturbance with statistical confidence.

We observed multifaceted responses to disturbance in CSS ranging from declines in overall species richness in mammals and woody plants, to no changes in others, to patterns of either increase or decrease in abundance across the gradient. In plants, many species and community level summary variables had unimodal responses, with highest

values at moderate levels of disturbance. Furthermore, we found almost no correlation in the responses of taxa and many species to disturbance. If one species or taxa increased across the gradient, another showed no response or declined.

The causes of these responses are likely two-fold and strongly advocate for an index-based approach to estimating status and trend in the overall system. First, intense disturbance in CSS leads to the loss of shrubs and the increase in exotic cover and results in not just species turnover, but in large changes to the abiotic environment and the replacement of a shrub-dominated ecosystem with one dominated by grasses. In so doing, turnover in species composition occurs and entirely different suites of species interact affecting the outcomes of ecological processes such as competition and predation. By measuring so many species and taxa across the gradient we were bound to see a wide array of responses as the changing abiotic conditions and species-interactions impacted different species in different ways.

Measuring these complex changes in CSS is difficult for two reasons. First, as mentioned above and shown in our data, the changes themselves are complex and not correlated across taxa. Second, the biotic and abiotic conditions within CSS show high levels of variation. Through time the system responds strongly to rainfall, showing much higher levels of productivity with higher rains. In our case, we generally saw increases in abundance or occurrence across many species, including exotic grasses from 2002 (drought) to 2003 (almost normal rainfall). CSS also varies considerable across space. Changes occur at large scales caused by moisture and rainfall gradients with distance from the coast, while mosaics of shrubs and patches of exotic grasses occur at scales of 50-100m. These patterns of variability add difficulty to monitoring because they require us to select species and sampling procedures that show responses at the spatial and temporal scales at which we perform management. For example, using raptors to inform us of the condition of a hillside stand of CSS would not be useful.

Index-based approaches are helpful in these situations because, based on empirical data like those presented here, we discover species showing responses at scales of interest/need and then use them to estimate status and trend. In a nutshell, we collected data for just 2 years, screened ~370 species, genera, or orders for their response to disturbance in CSS and found enough responses to the AEC gradient to develop 16 metrics across 6 taxa. Furthermore, when the metrics are combined, the resulting IBI can distinguish 4 or 5 levels of disturbance in our AEC gradient.

RECOMMENDATION 1. Our findings convince us index-based methods for assessing status and trend of overall CSS condition show great promise and are likely superior to general fauna surveys and comparisons of community level variables such as species richness or diversity. We recommend continued efforts to develop index-based methods in CSS, including indices for use at larger spatial scales.

RECOMMENDATION 2: Avoid loosing CSS shrub structure in HCP reserves. Loss of shrubs basically means the conversion of one entire ecosystem (CSS), to another (exotic grasslands) and greatly impacts the ability of a reserve to maintain viable populations of covered species or native diversity.

This recommendation may seem obvious. However, the single largest threat to loss of shrubs in reserves is short fire return intervals caused by anthropogenically-inflated rates of fire ignition. Our data indicate a wide array of species disappear with the loss of shrub cover and 12 of our 38 sites (31%) burned in just two years. After the fires

of 2003, it became obvious most HCPS have no detailed fire management plan. Furthermore most reserves have not adequately considered management strategies for: 1) reducing the chances of fires starting within reserve boundaries, 2) reducing the ability of a burn to enter a reserve from outside and 3) enhancing the ability to reduce the spread of a fire within a reserve. *Developing and implementing strategies that reduce fire return intervals in reserves should be a top priority for NCCP management and may directly counter the controlled burns advocated by some agencies.*

Is an IBI for CSS ready for use?

Not yet. Two main reasons exist for proceeding with caution and the need for additional work before implementing a CSS-IBI. First, as presented in the original grant application, this project covers only 3 of 4 steps in IBI creation. Validation using an independent dataset is step 4 and must be done before we have confidence the IBI can function. Second, the IBI we developed needs further refinement on a number of fronts. We discuss each of these issues below.

CSS-IBI Validation.

The literature regarding IBI development emphasizes the need for IBI validation after the initial IBI development. The best example of IBI validation in a terrestrial system comes from O'Connell et. al (1997). They developed a "landscape indicator of ecological condition" for the Central Appalachians using bird community data. Their research program had 2 main steps. First, they used 34 reference sites with known levels of disturbance to create an index. Second, O'Connell et. al then sampled 126 new locations (across 3 states) and compared the index developed from the bird community data to the expected rankings of plots based on landscape variables associated with disturbance. The second part of their research program was essential to showing their index could measure and rank the ecological condition of sites across large spatial scales.

The research we've done to date on a CSS-IBI mimics only the first step of work performed by O'Connell et al. We developed a disturbance gradient, sampled across this gradient, and developed a promising 16-metric IBI. However, the IBI has not been validated. Validation would require selecting new study plots based on their level of AEC then sampling these locations to calculate the IBI. Each plot would have an expected IBI based on the relationship between IBI-score and AEC developed from the data presented in this report (i.e. the regression equations associated with Figure 19). These predicted IBI scores would then be compared to the observed IBI scores. An ideal sampling strategy for this work would span the entire AEC gradient and contain new plots both in and outside the reserves previously sampled. By sampling new plots in the reserves used to generate the IBI as well as plots in new reserves we can estimate how well the IBI can function outside the geographic areas in which it was originally developed.

RECOMMENDATION 3. Validate the IBI before proceeding further.

IBI Refinement.

The IBI we generated remains a work in progress and requires refinement in 2 main areas: Metric verification and optimal sampling strategies. Metric verification is most pressing, as biological knowledge should confirm the inclusion of species, genera, and orders in particular IBI metrics. Sampling strategies impact both cost of implementation and the statistical properties of the IBI. Typically cheaper, less robust sampling results in an IBI less capable of distinguishing disturbance levels.

Metric Verification. In developing IBI metrics, we explored the data for species, Genera, or Orders showing responses across the AEC gradient. When species showed responses across the gradient, we attempted to determine if pre-existing information about the species supported the trends we observed. If so, we had some confidence in our inclusion of the species. Though we feel our sampling techniques were adequate, the responses of some species to the gradient may be surprising or unexplainable by experts knowledgeable about a species. We would prefer to only include species in the IBI where the response has a reasonably good biological explanation and no contradictory data.

We have reasonably high confidence in the metrics for mammals, birds and to a lesser degree plants because 1) our research team has experience in these taxa and the level of preexisting research on these species is relatively high. Thus, the species we included in these three metrics: 1) showed responses across the gradient; 2) had pre-existing information on habitat preferences, diets, response to disturbance, or other life history information that coincided with the observed responses; and 3) were consistent with our general observations and expertise of these species.

However, we have less confidence in our metrics derived from Ant, Arthropod and Herpetofauna data. We discussed the observed ant responses with Dr. David Holloway at UCSD and he indicated the results were consistent with knowledge for some genera, but that information regarding other genera we included in the metric was sparse. We have little pre-existing information supporting the metrics surrounding the Arthropod orders. Finally, we have not yet discussed our results with Dr. Fisher and his USGS herpetofauna experts.

Recommendation. Verify the patterns we observed using additional datasets, and discuss metric development with more taxa experts. Alter metrics if and when necessary. Large amounts of pre-existing data might be useable to verify the patterns we observed across the AEC gradient. For example, many bird point counts have occurred throughout CSS. If those counts have associated vegetation information, we should be able to look for relationships between levels of AEC and the birds. Furthermore, the USGS sampling arrays all have an associated vegetation transect (we used these data for initial site selection). These data could be used to verify if the patterns we observed for ants, arthropods, mammals and herpetofauna. It may even be possible to generate a modified IBI for use with these data.

Optimal Sampling Strategies. Figure 19 compares the IBI across all three of our datasets. In general, the IBI performed similarly whether it was created with 2002, 2003 or a combination of both years of data. However, we know for some taxa, one year of sampling and perhaps even 2 years of sampling may not give a complete census of the species richness or accurately estimate relative abundance at a plot. We used

proportional or relative metrics in attempts to correct for incomplete sampling or variability through time. The congruence across the IBIs for each dataset suggests the combination of strong biological responses across the gradient, and the use of proportional metrics, overwhelms potential pitfalls caused by under-sampling the community.

However, a wide range of additional analyses could be performed to further understand how different levels of sampling impact the ability of the IBI to distinguish between disturbance categories. For example, we can take each metric individually and ask how often the 1, 3, 5 plot rankings change under different sampling scenarios. For some metrics, it might be possible to sample these taxa less frequently than others and still generate robust metric scores.

In addition, we could ask how well the IBI performs if we remove specific metrics. For example, monitoring ants and arthropods may be difficult if taxonomic expertise is not available. Thus, we could remove the three ant- and arthropod-based metrics the reanalyze to determine the new IBIs relationship to AEC and how well it discriminates levels of disturbance. It may be possible to create a suite of IBIs based on different combinations of taxa so that managers at different reserves can use ‘whatever data they could get’ to compare their plots to known reference conditions.

All IBI metrics we developed assume our ability to detect the presence of species, genera, or orders, and the ability to census individuals are not impacted by levels of AEC or at least impacted at much smaller levels than the biological response we measure across the gradient. For example, if birds were easier to detect in open vs. closed canopy CSS, then all else being equal, we would expect to find more species in the open CSS. We generally feel the magnitude of the biological signal we measured with each metric is quite large and likely overwhelms patterns caused by differences in detectability. In addition, our use of proportional data may help correct for changes in detection probabilities across AEC. Thus, though we feel investigating the impacts of sampling bias on the IBI is important, we do not consider it as critical as other efforts.

Our data is structured so we can estimate detection probabilities for species and capture probabilities for individuals. Doing so would allow us to understand if, and quantify, the impacts of sampling biases on the IBI metric. Ultimately, IBI metrics could be generated using combinations of sampling and statistical techniques that allow one to estimate detection probabilities and, in a sense, correct the final IBI metric for false absences. However, this would likely require theoretical statistical modeling and is not directly applicable in the short-term.

Finally, we often did not capture species at sites even though they were captured at other sites with similar levels of AEC. These “zero’s” in the data reduced our ability to include these species in metrics because the absences tended to occur across the entire AEC gradient. Multiple explanations for these zeros exist. For example, variables not included in our analyses, such as distance to urban edge, may influence a species resulting in absence on plots with low levels of invasion. However, for difficult to detect, or rare species, inadequate sampling may result in many “zero’s” in the data even when the species is present, degrading the utility of an IBI. We were able to develop a reasonable set of metrics despite this issue. However, better sampling methods would allow us to include additional species in the metrics, perhaps increasing robustness of the IBI.

Recommendation. Perform analyses focused on how individual metrics and the overall IBI perform with different levels of sampling. Investigate if and how detection probabilities change across the AEC gradient.

Is an IBI for CSS necessary?

Perhaps. We are convinced a CSS-IBI can be created, validated and implemented as an effective tool for assessing overall ecological condition in the NCCP. However, the use of an IBI must be placed in a broader context of how data and science will be used to drive management decisions in NCCP reserves. We feel HCPs must generate data that: 1) allows jeopardy assessments for the species covered by the plans, 2) indicate the status of overall ecological condition, and 3) fill gaps in information necessary for effective management decisions.

Carefully designed monitoring programs can simultaneously meet many aspects of all three requirements. For example, if we continued to monitor our ~40 plots through time, the data generated would certainly meet #2, perhaps assist in #1 for some covered species, and because the plots are arrayed across a gradient of AEC, generate information pertinent to #3. However, for covered species and to understand the impacts of specific threats or management strategies, focused studies with robust experimental designs must be employed beyond simply re-sampling the same suite of locations. If this additional information regarding mechanisms is not generated, then a monitoring program will generate trends, but no one will be able to explain them or make management decisions that might change the directions these trends take.

The issue then is resource allocation. How much funding and time should we put towards IBI implementation, monitoring covered species, or focused studies of particular covered species? The answers to these questions will depend in part on how large the information gaps are for management in NCCP reserves and how critical it is to monitor status and trend. Our aim here is not describe a framework for complete monitoring and management in NCCP reserves. Instead, we discuss issues related to the IBI that may assist decision-makers when considering its implementation.

First, we address the question, “What do the IBI numbers mean?” If the information our IBI produces does not assist in effective decision-making, then the IBI should not be used. Our IBI estimates ecological condition at a scale of approximately 2500 m². The estimate of ecological condition is based on the patterns we observed in Ants, Arthropods, Birds, Herpetofauna, Small mammals and Vegetation across a disturbance gradient. A low IBI score indicates the flora and fauna at a plot are typical of what would be found in highly disturbed, or degraded CSS habitat while a high IBI score indicates a suite of species similar to those found in intact CSS. Thus, our IBI creates a relative measure of CSS integrity at somewhat small spatial scales.

We chose the small spatial scale for two main reasons. First, managers indicated many of their activities, such as controlling exotics, or restoring habitat take place a small spatial scales. We wanted an IBI capable of determining if their activities impacted ecological condition. Second, we felt the size of the study plots were both large enough to detect responses of many species, while small enough to assure logistical feasibility.

The spatial scale at which our IBI measures ecological condition impacts how it can be used in a management context. For example, the small scale of our IBI may limit its application for large-scale assessment of ecological condition. For example, if NCCP managers wanted to compare ecological condition across Mission Trails Regional Park, the FWS Sweetwater Reserve, and DFG's Rancho Jamul Ecological Reserve, our IBI would be limited. First, our IBI only applies to CSS, or former CSS habitat. Thus, it could not gauge the quality of other habitat types found in these areas. Furthermore, because we use species in our metric that responded to disturbance at small spatial scales, we may miss biological response to larger scale features in reserves or larger-scale disturbance regimes. For example, Crooks (2002), showed carnivores respond to landscape features such as patch size and isolation. These types of disturbances could affect reserves, but our IBI would not likely detect them.

However, extrapolating from the small plots to larger spatial scales is feasible if the goals are related to tracking large-scale changes through time in CSS and one recognizes the limitations discussed above. Repeated plots spaced either randomly or stratified-randomly across a reserve could be used to compare average IBI scores across reserves. Furthermore, it may be possible to extrapolate the IBI across large areas using remote sensing and GIS. If we can detect signatures in the disturbance categories using remotely sensed imagery, then it may be possible to model 'predicted' IBI scores across the extent of CSS through the reserve system.

However, a different, and perhaps more cost-effective solution to assess overall condition at larger spatial scales is to develop an IBI based on species who integrate multiple habitat types in their daily activities. Raptors, corvids, medium and large carnivores, snakes, and perhaps some larger mammals such as deer likely show strong responses to landscape features related to urbanization, but may not respond to patterns and types of disturbance our smaller-scale IBI measures. Thus, there is no reason why different IBIs cannot be used to answer different questions. For example, the bird-based index developed by O'Connell et al. (1997), described ecological condition at approximately 1km².

Despite limitations for particular questions at larger spatial scales, our IBI should be effective in a number of applications. First, when arrayed across a single reserve, or multiple reserves and sampled every 2-5 years, the IBI would measure both spatial variation and temporal trend in CSS. Changes in the proportion of plots with different IBI scores would produce a good assessment of changes in the overall system.

Second, the IBI is ideal for gauging the impacts of particular management activities such as restoration or weeding. These comparisons could be particularly powerful when utilized in the context of monitoring areas before and after a management activity. In this context, the IBI could be used to gauge the impacts of a wide array of management actions.

Third, the IBI can likely be used to study the impacts of, and recovery from, particular disturbances such as habitat edges, or fire. However, the relative manner by which the IBI measures ecological condition must be considered in these situations. The IBI was created by sampling across a gradient of AEC, which we felt adequately measured the sum of historic disturbance regimes at a plot. It may be possible habitat edges, light pollution, or fire, alone or in combination, impact CSS food webs and their

components differently than the disturbance regimes we measured. If so, the IBI score may not detect certain changes in the system.

Why not measure just the plants? If one steps away from the IBI focus of the project and simply asks if the information we generated can be used in monitoring, one quickly realizes a measure of shrub cover or levels of exotic plant invasion may offer managers significant levels of information regarding the status of the overall CSS community. Our work shows a strong relationship between 1) exotic plant cover and native shrub cover and 2) massive shifts in species composition and relative abundance take place across gradients of exotic plant and shrub cover. As such, measuring just the plant community should generate data strongly correlated with the community structure of consumers and act as a general estimate of ecological condition. Our research team has discussed this idea repeatedly and the following issues nearly always arise.

First, we used a time consuming, multi-pronged approach to measure the plant community. In general, it took one entire day in the field for a highly trained plant ecologist to complete vegetation sampling at a single plot. We used this method to gain an accurate and precise measure of the plant community and gather data on overall richness and shrub density as we felt this would reduce sampling error and enhance our abilities to construct an IBI. Thus, as currently implemented, plant sampling is time consuming and requires hard to find and difficult to train plant identification skills. However, if management only required a simple measure of exotic and/or shrub cover, sampling times would decrease 2 to 3 fold using point intercept methods.

Our team worried cheaper, faster, and more simplistic plant sampling methods (i.e. relevee methods or less detailed transects) may not generate quality estimates of shrub or exotic plant cover. For example, with the year 1 data, the mean AEC values for the 5-cluster solution were 35.2, 48.6, 55.9, 70.0, and 91.8. We doubt less detailed methods could readily distinguish between these levels of AEC and would result in coarse assessments of habitat quality or ecological condition. However, sampling the same location with different levels of effort then comparing results could test fast, economical plant sampling methods.

While we feel it is possible to accurately and precisely measure exotic and shrub cover using faster methods than we employed, this decision must be considered as a trade off with the collection of other data. Measuring only plants might allow sampling at two or three times as many plots for the same effort. However, no other data would exist for these plots. In our project, 2 individuals, with perhaps 1 or 2 undergraduates helping, could sample 8 IBI plots for small mammals, arthropods, insects, herpetofauna, and birds in 4-5 days. A plant only approach could sample roughly 16 plots but only garner vegetation data. In addition, with refinement the final IBI may require sampling fewer taxa with even less effort. If so, the difference in the level of effort between consumers vs. plant only sampling may be reduced.

Second, we felt a more robust estimate of ecological condition is more likely using multiple metrics. The CSS system is complex, highly variable, and prone to large fluctuations in rainfall. Furthermore, our study is only the first to explore relationships between disturbance and food web structure and our results should be considered preliminary and with little verification. Given this level of variability, and uncertainty,

we felt relying solely on measures from the plant community may result in high levels of variability in plant metrics and a reduced ability to measure ecological condition.

In addition, given the tenuous understanding of how many species respond to changes in the vegetation community (i.e. the numerous borderline cases or unverified patterns we observed), we cannot be entirely sure what plant community data may indicate about the community composition of consumers and the status of many species. Thus, a metric based on plants only may not be easy to interpret biologically. Bottom line, by focusing only on plants we risk putting all our eggs in a basket full of holes.

Third, by measuring multiple taxa, we have a direct estimate of their status instead of inference based on correlation. By measuring multiple taxa for IBI estimation, one not only estimates general ecological condition, one also gets large amounts of species-specific data. Thus, implementing IBI protocols by measuring many taxa, may actually meet multiple requirements of a monitoring program because species level data generated by IBI sampling may supplement or allow status assessments for individual species as well.

However, at coarse spatial scales, we feel the relationships we've discovered between community composition and vegetation structure will allow some levels of comparison. For example, our results create a context for gauging the ecological impacts of general measures such as the extent of grasslands vs. scrublands in reserves. Our data strongly suggest that, in general, losses of shrub cover and increases in exotic grasses reduce overall ecological condition in reserves away from the conservation target of intact CSS ecosystems. The issue is measuring status. If managers feel a two-category (CSS or Grass only) approach is useful then remotely sensed data and coarse visual assessments will likely serve monitoring purposes.

Additional items worth mentioning.

Finally, we discuss the issue of disturbance and what sorts of disturbance our IBI likely measures. Disturbance is an over-used, ill-defined word in management circles and a wide variety of conservation oriented literature. We fully admit to using the term just as loosely in this document.

We developed our IBI based on levels of exotic cover at 2500m². Thus, our IBI primarily measures disturbances that strongly impact both the vegetation and the consumers at this scale. We feel these forms of disturbance include historic land use such as fire, agriculture, grazing, mowing, and mechanical impacts to the soil such as grading, or heavy machinery. The responses of our taxa to these types of disturbance could be either direct or indirect. For example, historic grazing could impact soil structure so that specific arthropod orders and perhaps some small mammal species are excluded from the site because they cannot find enough refugia or locations for burrows. In this case, the disturbance directly impacts the taxa in question. In contrast, because grazing impacts vegetation, the arthropods and small mammals could be indirectly affected by the disturbance if they respond to vegetation changes. In this case, vegetation responses to disturbance mediate the responses of the arthropods and small mammals.

We know other forms of disturbance affect CSS plant and animals. For example patch size and isolation impact levels of diversity on canyon fragments in urbanized San Diego (Soule et al. 1992, Suarez 1998) and habitat edges impact bird nesting success (Bolger et al. 1997). Diffendorfer, in collaboration with D. Bolger (Dartmouth) currently

has a study to determine how urbanization alters predator-prey dynamics in CSS. These larger scale disturbance regimes do impact CSS ecological condition and minimizing their impacts should be considered a management goal.

When we developed our IBI, we realized we could not set up a fully replicated sampling design to simultaneously tease apart the impacts of historic disturbance at small spatial scales with any number of larger-scale disturbance regimes. Ideally we could have replicated our study at different distances from urban edges, in reserves of different sizes and even at different distances from the coast.

Since we could not perform such sampling, we developed a GIS for our plots so we could measure larger scale disturbance regimes, such as distance to urban edge, road and trail density around our plots, and the levels of general habitat types or urbanization at larger distances. The GIS development was done independently of DFG funding and was not considered a deliverable of the project. Our future goal is to see if these large-scale variables explain any of the residual variation in the IBI.

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APPENDIX A.

Coastal Sage Scrub response to disturbance. A literature review and annotated bibliography

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Table of Contents

Executive Summary	4
Introduction	5
Background: Developing Indices of Biological Integrity	5
Description of task	6
Overview of review process, scope of work, and limitations.	7
Results	10
Vegetation	10
Abstract	10
Introduction	10
Methods	11
Results	11
The CSS vegetation community.	11
Disturbance Effects on Coastal Sage Scrub Vegetation	12
Fire	12
Grazing	14
Mechanical Disturbance	15
Air Pollution	17
Habitat Fragmentation	19
Discussion	20
Conclusions	22
Mammals	23
Introduction	23
Fire	23
Invasive species	24
Habitat Fragmentation and Isolation	24
Edge Effects	26
Conclusion	26
Placing Small Mammal monitoring data in an IBI.	27
Data Gaps and Recommendations for further research	27
Birds	28
Avian Response to Habitat Fragmentation and Urbanization	28
Bird Response to CSS Fire	30
Birds as Single Species Indicators	30
The Use of Avian Guilds as Indicators of Habitat Quality	31
Developing a bird IBI for CSS	34
Data Gaps and Recommendations for further research	42
Herpetofauna	42
Introduction	42
Amphibians	43
Reptiles	44

Conclusion	46
Data gaps and recommendations for future research	46
Arthropods	51
Introduction	51
Arthropods as Indicators of biological condition.....	51
Existing Ecological information	52
The role of Arthropod in CSS food webs.....	54
Responses to disturbance	55
Conclusion	56
Data gaps and recommendations for future research	56
Conclusions	57
Annotated Bibliography	59
Literature cited	83

Executive Summary

This report was completed in February, 2002. The bibliography, except for plants, has not been updated since then. The plant review was updated in September, 2004 while creating a publication. We reviewed literature pertaining to how the vegetation, mammals, birds, arthropods, and herpetofauna of Coastal Sage Scrub (CSS) respond to disturbance. During the review process, we looked for evidence indicating these taxa, or members of the taxa, would show responses to disturbance and therefore contribute useful metrics to an Index of Biological Integrity (IBI) for CSS. As expected, the level of research, and hence the current state of knowledge regarding the natural history, ecology, and response to disturbance is highly variable across the taxa. For all taxa, a long-history (10-20 years) of well-designed research on responses to disturbance simply does not exist (if it did, this project would likely not be needed). This variation in past research and state of knowledge caused our taxa reviews to vary in structure and content.

Despite the poor state of knowledge in some cases, the review indicates responses to disturbance are measurable, obvious and quantifiable with standard field protocols for all the taxa. As such, we are encouraged about deriving a multi-taxa IBI in the next two years as our field studies produce new data.

Of all taxa, vegetation is best studied, with a large and varied literature. Responses to fire have been well documented. Unfortunately, vegetation response to other forms of disturbance is less studied. Our review indicates invasion of CSS by nonnative grasses and forbs is highly correlated with disturbance. Furthermore, these nonnative species likely persist in CSS stands even after the shrub component has recovered. The response of the native, herbaceous understory to invasion is basically unknown. However, our review indicates a stand of CSS could appear intact from a distance, yet have a reduced herbaceous component replaced by nonnative grasses.

Unlike the other taxa, a number of researchers have successfully created IBI like measures from bird survey data. These studies, when combined with studies of CSS bird community response to habitat fragmentation and urbanization, suggest the development of an IBI for birds will be straightforward. We include a preliminary conceptualization of a guild structure for a bird IBI since so much pre-existing information exists for birds.

We found limited information on mammals and arthropods. In small mammals, a handful of studies suggest processes structuring small mammal communities in CSS are similar to those in well-studied systems such as the desert Southwest and Australian heathlands. These studies, in concert with a few directed studies of small mammals in CSS, strongly suggest CSS mammals will show direct and rapid responses to disturbances, particularly those altering vegetation composition or structure.

Arthropods, given their massive levels of diversity, are poorly studied, yet our review found evidence suggesting Arthropod diversity and composition would change with disturbance. Methodological challenges associated with identifying species must be overcome to use Arthropods in monitoring programs. The use of morphospecies may solve this problem. If so, the large diversity of species (or morphospecies), the relative ease sampling, and the highly specified life histories of many arthropods may make it possible to discriminate between more levels of biological integrity in CSS than with the other taxa.

Our review resulted in three general recommendations. First, we need more studies elucidating responses of taxa to the most threatening forms of disturbance in reserves. In general, we do not understand how specific forms of disturbance impact CSS species. Thus, we strongly recommend directed studies on how target species, specific taxa, and food webs in CSS respond to both disturbances and possible management practices. These studies can be implemented by setting up monitoring locations across gradients in the level of disturbance (e.g. distances from edges, time since fire, level of restoration, amount of invasion, etc), or by designing experimental manipulations in particular reserves designed to

answer specific questions or test hypotheses. Second, we found little support for the use of indicator species as a method for understanding or predicting community-level patterns in CSS. Indicator species may be useful in particular situations (such as a host plant for a particular species), but they rarely show a strong correlation with the rest of the community. As such they have little value in helping understand responses to disturbance or guiding management. Third, given the unique nature of CSS and the realities of reserves in an urban landscape, there are a limited number of management actions available to reserve managers. These should be prioritized and studied for their utility and cost-effectiveness.

Introduction

Rapid and sustained human population increases during the last century have increased displacement and fragmentation of southern California's (S.CA) native systems. Coastal sage scrub's (CSS) tendency to occur on relatively fertile coastal lowlands has made it particularly vulnerable to both agricultural and urban displacement. Estimates of the extent of coastal sage scrub loss range from 36% to 85% (O'Leary 1990a, 1995), with most experts agreeing with higher levels of destruction. Efforts to conserve remaining CSS have resulted in complex, large-scale efforts at reserve design and management. These efforts, driven almost exclusively by the Natural Communities Conservation Plan, are precedent setting and have received recognition at the highest levels of government.

Given: 1) the heterogeneous mix of land use patterns surrounding most of the NCCP reserves, 2) the extent to which many existing reserve lands were historically disturbed by grazing, altered fire regimes, agriculture and invasive species, and 3) strong public pressure to use these lands as parks instead of reserves for endangered and threatened species, the success of the NCCP program will rest almost solely on the ability of local reserve managers to maintain the *in-situ* biological integrity of each reserve in face of worsening external pressures. This task, though difficult, is not insurmountable if managers are given the correct tools, have high quality data to make science-based decisions, and if we understand the response of CSS food webs to both disturbance and the limited number of possible management actions feasible.

The overarching goal of the "IBI project" (i.e. the CDF&G grant to Dr. Diffendorfer "Creating and Index of Biological Integrity for Coastal Sage Scrub: A tool for habitat quality assessment and monitoring") is to develop a conceptual framework for analyzing multi-taxa monitoring data in CSS habitats. The approach, if successful, will create a method for comparing the biological integrity of different CSS sites or the same site before and after management actions. Developing a successful IBI will also help direct efforts to create and implement comprehensive, economical, and biological relevant monitoring programs in CSS throughout the NCCP reserve system.

Background: Developing Indices of Biological Integrity.

In this section, we familiarize the reader with the Index of Biological Integrity (IBI), the function it plays, and how it is developed. This general background is necessary to fully understand the information we attempted to gather during our literature review. The final report describes the development of IBI's and the measurement of disturbance in more detail (see pages 10-19). Readers wanting additional information on IBI's should look there.

The IBI was first developed for fish communities in midwestern streams of the US (Karr 1981, Fausch et al. 1984), and is predominately used to evaluate aquatic ecosystems. Since then, the use of IBI's and sophistication of IBI's has grown primarily on two fronts. The first is the statistical methods used to determine the level of discrimination of a final IBI. The second, which we describe below, is the organizational structure used for creating the guilds, or species for which response curves are derived

from monitoring data. We present this methodology here because we report potential guild structures for birds and small mammals, and felt it important to give an overview.

Herricks and Schaffer (1985) presented 6 general criteria for data useful in IBI development. All attempts to build IBI's we have reviewed generally meet these criteria. 1) *The measures used must be biological.* Abiotic responses to disturbance are not included in IBI's. 2) *The measures must be interpretable at several trophic levels, or provide a connection to other organisms not directly involved in the monitoring.* An IBI should measure both composition and function in a system. Thus, IBI metrics should relate to an array of ecological processes. If so, then a change in the index reflects a broader, more complex change in the system. 3) *The measure must be sensitive to the environmental conditions being monitored.* The key point is that the taxa used and the measures chosen (i.e. species richness, abundance, or community overlap, etc) must show true responses to human induced changes of the ecosystem. 4) *The response range (i.e., sensitivity) of the measure must be suitable for the intended application.* In the case of CSS habitat, measures used in an IBI should be able to distinguish healthy CSS from disturbed CSS. If the IBI cannot distinguish disturbed from undisturbed sites, it will not be useful. 5) *The measure must be reproducible and precise within defined and acceptable limits for data collection over space and time.* A measure will be useful to an IBI if, when using a standardized protocol, similar values are generated from repeated samples at the same site. If different samples yield different estimates of the variable being measured, the results are not reproducible and an IBI based on these data would be unreliable. 6) *The variability of the measure(s) must be low.* Any variable used in an IBI will have some inherent level of variability caused by seasonal, temporal, or spatial variation in ecological processes. However, the inherent variability must be significantly less than changes in the variable caused by human disturbance to the ecosystem.

Given these 6 general criteria, the actual variables measured and used in IBI's varies from system to system. Karr and Chu (1999) indicate metric selection is an iterative process, where a specific metric is chosen and then exploratory data analyses used to determine if the metric shows a response curve to disturbance. Karr and Chu (1999, Table 5) recommend four broad categories of metrics for use in and IBI: Taxa richness, Tolerance vs intolerance (e.g. taxa richness of disturbance tolerant vs. intolerant species, etc), trophic structure (e.g. relative number of predators, etc), and individual health (e.g. growth rates, relative abundance of individuals with deformities, etc). More recent IBI's, particularly bird-based IBI's, create "guilds" of species based on rather complex categorization schemes. Generally, 3 broad divisions are initially used (composition, structure, and function) and then more specific categories are derived within each of these categories. We describe this method in the bird section.

Description of task

This deliverable represents the first phase of a multi-year effort. The goal of this phase is to assess our current state of knowledge of CSS food webs and their response to disturbance, and then use this knowledge to design and implement the field studies crucial for developing the species response curves for the IBI. The original contract states this document will review and synthesize scientific and gray literature, as well as expert opinion regarding the response of Coastal Sage Scrub ecosystems to disturbance. In addition, this document was to focus on four items while summarizing the literature and it's relevance to the development of an Index of Biological Integrity: A) identify functional/taxonomic groups or unique species within these groups that are sensitive to human impacts in CSS; B) describe any data gaps regarding how CSS ecosystems respond to human disturbance; C) contain an annotated bibliography for those reference materials that contain valuable information (Included at the end of this document) and; D) make recommendations regarding additional areas of research that are needed to better understand how CSS habitat, and the species it supports, respond to human disturbance. Here, and with

the attached annotated bibliography, we address each of the above points. After discussions with CDF&G, we focused the review on 5 major groups because these groups will likely be included in long-term monitoring plans required by the NCCP process. They include Plants, Birds, Mammals, Arthropods, and Herpetofauna.

Overview of review process, scope of work, and limitations.

General Approach. Dr. Diffendorfer oversaw the literature review and synthesis while graduate students and a post-doctoral researcher (Dr. Rosalie del Rosario) performed the majority of the work. Graduate students only worked on taxa for which they had expertise. They interacted closely with Dr. Rosario to screen articles and reports. Students met with Dr. Diffendorfer bi-weekly to discuss the progress of the reviews and key publications from March – December, 2001. The entire research group met weekly to discuss key papers, drafts of taxa reviews, and the scope of the project. Each graduate student wrote the initial review for a taxa. This draft was then edited by Drs. Diffendorfer and Rosario and reviewed by the entire group. The considerable experience of the graduate students (at the time of the review, 2002) is summarized below.

Vegetation: Genie Flemming, MS student. Ms. Fleming has been involved with ecological research for over 6 years and has extensive experience working in coastal sage scrub and a variety of other plant communities in S. CA and elsewhere. Since 1995, Ms. Fleming has been employed with Dr. Paul Zedler as a research assistant, and has been responsible for collection of field data, plant identification, data management, and writing project reports. Dr. Zedler's lab has conducted several research projects, including long-term vegetation trend monitoring on Marine Corps Base Camp Pendleton. This project is primarily concerned with investigating the responses of vegetation communities on the Base, including CSS, to major disturbances associated with military training activities. Other projects include measuring the response of riparian vegetation to changes in groundwater, long-term ecological monitoring on Twenty Nine Palms Marine Corps Base, and monitoring populations of rare or sensitive plant species such as *Pinus torreyana*, *Cupressus forbesii*, *Brodiaea filifolia*, and *Dudleya multicaulis*. In addition, Ms. Fleming worked on a multi-taxa research project conducted by Dr. Guy McPherson and Dr. Bob Steidl at the University of Arizona from September 1999 until October 2000. This project investigated the responses of both the flora and fauna of semiarid grassland to varied fire regimes. These experiences have provided Ms. Fleming with a strong understanding of plant community ecology and, in particular, the responses of communities to anthropogenic disturbance.

Mammals: Jenny Duggan, MS. Student. Jennifer Duggan attended the University of Wisconsin-Madison as an undergraduate and earned Bachelors degrees in both zoology and psychology. Since entering the Masters program in Ecology at SDSU under the advisorship of Dr. Diffendorfer, she has handled thousands of small mammals belonging to 18 species found throughout S. CA. Ms. Duggan is currently involved in a variety of local ecological projects, including her thesis research (The effects of food supplementation on small mammal communities over chaparral-grassland ecotones), surveys with USGS, and surveys with the U.S. Fish and Wildlife Service for the endangered Pacific Pocket Mouse. Her thesis work has given her considerable experience in the S. CA small mammal literature as well as an in-depth background in mammalian population and community ecology.

Avifauna: Robert Chapman, MS student. Mr. Chapman has been involved with birding for more than 20 years in California, Arizona, and Nevada. His background as an electronics engineer provides him with mathematical and analytic tools and skills that fit nicely into ecological studies of birds in a variety of habitats. His field experience includes working with the Yellow-billed Cuckoo and with the Global Climate Change team in Alaska. He has mist netting, banding, and survey experience. His Masters program work will apply GIS techniques and spatial analysis tools to the use of avifauna as indicators of habitat health.

Herpetofauna: Milan Mitrovich, Ph.D. Student. Mr. Mitrovich's Ph.d. applies a diverse set of methods to the study of the decline and conservation of the coachwhip snake in S. CA. His research focuses on: the behavior, movement ecology, and habitat use of the California whipsnake and coachwhip snake in fragmented and non-fragmented habitats of S. CA.; conservation genetics and evolutionary history of the coachwhip snake in Western U.S. and Mexico; and the ecological assessment of the population-level response of the California whipsnake and coachwhip snake to habitat fragmentation. He has been conducting field research on the herpetofauna of S. CA. since 1996 and has extensive experience in animal behavior, population and community ecology, and habitat conservation planning and monitoring. From 1995 through 1998, he worked with the U.S. Forest Service on development of an assessment of habitat and species conservation issues associated with the mountains and foothills of S. CA. Since 1998, Mr. Mitrovich has worked with the USGS. Biological Resources Division on development and implementation of a long-term multi-taxa ecological monitoring and research program measuring the response of coastal sage scrub plant and animal communities to anthropogenic disturbance in S. CA.

Arthropods: Matthew Rahn, Ph.D Student. Mr. Rahn's Ph.d focuses on the ecological assessment of ground obligate beetle diversity in CSS and their response to anthropogenic disturbance. He has extensive experience in arthropod ecological research, community ecology, and large-scale habitat conservation planning and monitoring. He investigated patterns of bat community assemblage and it's application to the Clark County, Nevada HCP for his MS. He has published work on arthropod community structure on isolated dune habitats in the Great Basin and has been conducting field research on terrestrial arthropods since 1994. Most recently he has been working on a review of various Habitat Conservation Plans and their inventory/monitoring plans across a range of taxonomic groups.

Scope of work. We reviewed published material to help us understand how different species or taxa in CSS respond to anthropogenic impacts (see a list of disturbances below). Because few studies were designed to directly assess a particular species' or communities' response to a particular form of disturbance, we "cast a broad net" when selecting material to review. We reasoned a broader approach may 1) reveal information which, while not directly showing responses to disturbance, could lead to hypotheses about possible responses, and 2) help develop a mechanistic understanding regarding observed responses to disturbance or predict how these responses may impact other taxa in CSS. For example, papers showing habitat preferences of CSS birds, diets of rodents, or indirect evidence for trophic cascades between top predators, mesopredators, birds and small mammals, were reviewed, as well as publications reporting more direct responses of CSS taxa to disturbance.

Disturbance. We defined disturbance as any human-caused factor potentially affecting the integrity of CSS. The following variables were considered disturbances. Fire, Grazing, Edge, Roads, Mechanical disturbance, Agriculture, Air pollution, Light pollution, Habitat Fragmentation, and Recreation. In each case the impact on CSS will depend on the intensity, time since, history of, frequency, and size of the disturbance. We recognize the presence of any one type of disturbance is likely correlated with many of the other disturbance types, making it difficult to easily separate effects of each form of disturbance. For example, urban sprawl likely increases the incidence of all disturbance types in a given area except grazing and agriculture. Given these complexities, it was impossible, based on the literature review, to ascertain the unique effect of each disturbance type on particular taxa. When possible, we noted the responses of CSS taxa to those unique forms of disturbance studied.

The reviews. In general, publications were found by searching library databases, including government document searches, asking local experts for papers, and general Internet searches. Publications with potentially interesting titles were scanned and read further if they contained pertinent information. Bibliographies of pertinent papers were researched in detail to check the original sources of information. As publications were gathered, they were categorized as follows. The annotated bibliography contains papers in categories A and B. We chose to limit the bibliography to those papers focusing on responses to disturbance in CSS, because CDF&G already has an extensive bibliography on the ecology and natural history of CSS provided by Dr. John O' Leary, San Diego State University.

- A) Critical paper: Papers specifically addressing the goals of the review and/or show responses of CSS taxa to disturbance. These often included specific studies of a species or taxa's response to a specific disturbance type.
- B) The study focused specifically on responses to disturbance but did not carry the significance of critical papers. This may be due to experimental flaws, small sample sizes, a lack of a relationship, or that the work was carried out in habitat similar, but different from CSS (i.e. chaparral, or Mediterranean scrub habitat).
- C) The study was not designed to specifically study responses of CSS to disturbance, but information regarding CSS response to disturbance could be extrapolated from the findings.
- D) The publication was not focused on CSS and disturbance, but does give pertinent information on the natural history of CSS.

Taxa reviews. The depth of any given taxa review depended solely on the level of past research, which was highly variable. For example, the literature on CSS vegetation is enormous, with large reviews and bibliographies already in place, including one funded by California Department of Fish and Game (i.e. at <http://www.ceres.ca.gov/CRA/NCCP/>). In stark contrast, only a handful of papers have been published on both arthropods and mammals in CSS and even fewer on how these taxa respond to disturbance. Birds, like vegetation, are reasonably well studied. Finally, our review of the herpetofauna is somewhat terse. We choose not to emphasize this taxa given the tremendous amount of research and data collected by Dr. Robert Fisher (USGS). Dr. Fisher's data, as it is analyzed over the next 1-5 years, will quickly outdate any historic information we may have discovered. Nonetheless, we make some interesting observations about the herpetofauna, which we hope are evaluated using Dr. Fisher's data. Overall then, the structure of each taxa review is quite different, emphasizing different key points illustrated by past research. We conclude each taxa review with a section entitled "Data gaps and recommendations for future research."

Limitations. As with any literature review and synthesis, this review has limitations. The most obvious limitation is simply the rarity of research in CSS. The exception to this point is vegetation, where a large literature exists primarily on responses to fire. In general, studies on many of the animal taxa in CSS are limited in number and only a few specifically address responses to disturbance. As such, conclusions from this review should be viewed as working hypotheses. Second, scientific literature, especially when gray literature is included, is vast and difficult to completely cover. As such, we undoubtedly missed relevant publications. We attempted to minimize this by doing a thorough review, and we feel the vast majority of publications available through numerous digital databases were covered. However, obscure reports by local consultants or academics to government agencies are likely missing.

Results

Vegetation

Abstract

In southern California, coastal sage scrub (CSS) habitat is a major focus of conservation efforts. Many reserves have been created specifically to protect CSS habitat, but even in reserves this habitat is or has been subject to disturbance. We reviewed literature detailing disturbance effects in CSS to determine whether it could help inform CSS conservation management decisions. We conclude, despite over 3 decades of research, our current knowledge is still insufficient to make predictions about the effects of any single disturbance type or larger disturbance regime on the CSS community, as a whole, or properly guide management actions. Causes include: 1) a scarcity of studies focused specifically on disturbances other than fire within CSS; 2) variation in post-disturbance recovery patterns of CSS vegetation across studies; and 3) the focus of research primarily on the dominant shrub species and a dearth of studies on understory responses to disturbance. Nonetheless, many studies indicate as disturbance, in general, becomes more frequent or more severe, the abundance of exotic (or non-native) plant species increases. Because exotics may alter community dynamics and interfere with post-disturbance recovery, disturbance-facilitated invasion may represent the greatest long-term threat to the CSS community. We recommend devoting future research in CSS towards a better understanding of disturbance-facilitated invasion in CSS and the processes that determine CSS recovery in the face of such invasion, as well as the development of effective methods for reducing post disturbance invasion.

Introduction

Preserving sensitive habitat in the presence of encroaching human impact is one of the major challenges currently facing land managers and conservation scientists. This is particularly true in coastal southern California where undeveloped land is scarce and many small parcels are included among the region's system of nature reserves. These small reserves, as well as many larger ones, are completely surrounded by human development. Some reserves also include land previously used for grazing or agriculture. As a result, many forms of past and present anthropogenic disturbance impact reserves. Effective conservation and management will require understanding and minimizing the effects of these disturbances on the natural habitat.

In southern California, Coastal Sage Scrub (CSS) habitat is of primary conservation concern and the focus of three large multiple species habitat conservation plans (MSHCPs). CSS habitat supports ~100 rare and endangered plants and animals (O'Leary et al. 1994), but is disappearing at a rapid pace. The developing MSHCPs, as well as other protected lands, will conserve parts of the remaining CSS, but many reserve areas are (or will be) surrounded by development and some have a history of human resource use. Impacts may include (but are not limited to) alterations in the natural fire regime, past or

current livestock grazing, a variety of mechanical disturbances associated with development and recreation, increasing air pollution, and continued fragmentation of the remaining habitat area.

We reviewed literature related to the effects of these disturbances on CSS vegetation to 1) synthesize our current understanding of how the disturbances impact various vegetation life forms in CSS; 2) develop hypotheses regarding if and how post-disturbance recovery occurs; and 3) discover shortcomings or gaps in our knowledge regarding human impacts on CSS. Our motivation arose from conversations with reserve managers in southern California concerned with evaluating disturbance effects and maintaining the integrity of CSS stands in their reserves.

From these conversations, we found different land managers held conflicting opinions regarding CSS responses to disturbance and its ability to recover from disturbances. We realized the research literature relevant to CSS was large and full of many excellent studies. Indeed, during the planning of the San Diego County Multiple Species Conservation Plan, O'Leary, et al. (1994) produced a bibliography containing 51 pages of references related to CSS. Much of this work focuses on basic research questions, and we wanted to know how this large body of knowledge could inform current management issues in CSS.

Methods.

Given our focus on disturbance effects, we did not attempt to review all research articles published on CSS. Instead, we developed our list of references by performing searches using "CSS" and "disturbance", or the specific disturbances listed above, as keywords. We reviewed titles and abstracts for over 100 papers gathered in this manner, broadly selecting papers with references to disturbance effects on vegetation, as well as papers on successional dynamics, within CSS and related Mediterranean-type shrubland communities such as chaparral. We included papers published in refereed journals, "gray literature" (government reports, graduate theses and dissertations, etc.), and past literature reviews on CSS and some disturbances (O'Leary 1990a, Saunders et al. 1991, D'Antonio and Vitousek 1992, White 1995).

Results.

The CSS vegetation community.

The distribution of CSS extends from San Francisco, CA south to El Rosario, Baja California (Kirkpatrick and Hutchinson 1977, Westman 1981b, O'Leary 1990a, DeSimone and Burk 1992), primarily occupying the coastal plains and foothills of the Transverse and Peninsular Ranges and the California Channel Islands (Westman 1981b, O'Leary 1990a). The vegetation community is dominated by facultatively drought-deciduous and seasonally dimorphic shrubs and subshrubs, along with stem and leaf succulents in some areas. An herbaceous "understory" of perennial and annual grasses and forbs occurs primarily in the open spaces between shrubs. Plant species composition varies at both local and regional scales, giving rise to several floristic associations and subassociations (Kirkpatrick and Hutchinson 1977, Axelrod 1978, Westman 1981b, DeSimone and Burk 1992). In southern California, common dominant shrub species include *Artemisia californica* (California sagebrush), *Eriogonum fasciculatum* (California buckwheat), *Encelia californica* (brittlebush), *Salvia mellifera* (black sage), and *Salvia apiana* (white sage). Some stands may also include hard-leaved, evergreen shrubs, such as *Malosma laurina* (laurel sumac) and *Rhus integrifolia* (lemonadeberry), though dominance by evergreen shrubs typifies chaparral vegetation with which CSS sometimes intergrades. Local species composition

and shrub cover are influenced by local climate, substrate, and other environmental variables (Westman 1981b, DeSimone and Burk 1992), as well as disturbance (Kirkpatrick and Hutchinson 1980).

The dominant plants of the CSS vegetation community have traits allowing rapid post-disturbance recovery. Thus, the community exhibits some resilience to disturbances. Axelrod (1978) argued the current distribution of CSS is due, in part, to anthropogenic disturbance regimes, which resulted in the conversion of grassland and chaparral to CSS. This suggests the CSS community may be disturbance dependent or successional in nature. On some sites, such as recently burned chaparral, CSS may, in fact, represent a successional stage. Nonetheless, relatively stable CSS communities do exist on undisturbed sites (e.g. Westman 1982). These climax CSS communities are typically found at lower elevations and/or more xeric sites than tolerated by chaparral (Westman 1981a, 1982, Mooney 1988) and with soils more basic, shallow, and rocky than those found in native perennial grasslands (Keeley 1993).

Disturbance Effects on Coastal Sage Scrub Vegetation

Fire

Periodic fire is common in mediterranean-type shrublands like CSS and may be considered a natural, or even necessary, disturbance. We found more publications on how CSS vegetation responds to fire than any other form of disturbance. Though post-fire patterns of recovery are well studied, little is known about pre-European settlement fire regimes (i.e. size, season, and frequency of fire) in CSS or how long-term variation in fire regime affect the vegetation community. Since both the exclusion of fire over very long intervals and high fire frequency can impact fuel loading and wildfire risk, as well as CSS shrub recruitment and species diversity, shrubland fire regimes in southern California have become a topic of intense debate and management concern.

O'Leary (1990a) and White (1995) reviewed much of the literature concerning post-fire recovery in CSS following a single fire, so we only briefly summarize here. Post-fire recovery patterns are highly variable and depend on factors such as geographic location (e.g. coastal vs. inland sites), slope aspect, season of burn, and fire intensity (Westman 1981a, Keeley and Keeley 1984, Malanson 1984, Westman and O'Leary 1986, O'Leary 1988, O'Leary and Westman 1988, O'Leary 1990b). In general, many shrubs and herbs in CSS recover rapidly after fire and are considered "fire-adapted". Most notably, many CSS shrub species, as well as some perennial herbs, resprout from underground rootstocks or re-establish, to a lesser extent, from seeds in the soil (Westman 1981a, Malanson and O'Leary 1982, Westman 1982, Zedler et al. 1983, Keeley and Keeley 1984, O'Leary and Westman 1988, O'Leary 1990b). The extent of post-fire shrub recovery due to resprouting versus seedling recruitment varies geographically, with lower resprouting rates at inland sites compared to coastal sites (O'Leary and Westman 1988). However, resprouting shrubs often seed abundantly during the first post-fire year (Malanson and O'Leary 1982, Keeley and Keeley 1984, O'Leary and Westman 1988), and CSS shrubs may recover half or more of their pre-fire density and cover within 5 to 7 years after fire (Westman 1981a, Westman and O'Leary 1986, O'Leary and Westman 1988), though recovery time varies by species (Minnich and Dezzani 1998).

Cover of herbaceous plants (i.e. forbs and grasses) peaks following fire and declines as shrub cover increases (Westman 1981a, Keeley and Keeley 1984, O'Leary 1990b). Some "fire-following" annual species show enhanced germination from dormant seed-banks following fire (Keeley et al. 1985, Keeley 1991) and are often only visibly present in the first few years after a burn (Keeley and Keeley 1984, O'Leary and Westman 1988). Species richness and equitability generally peak in the first few years after fire, primarily due to the diversity of herb species (Westman 1981a, Keeley and Keeley 1984, O'Leary 1990b). Though both herbaceous cover and diversity generally decline after the first few years post-fire, declines in diversity may differ according to slope aspect (O'Leary 1990b) and a second

temporary peak may occur after 20 years with the appearance of a different suite of herbs than those seen immediately post-fire (Westman 1981a).

Despite the relatively large body of knowledge regarding mediterranean-type shrubland responses to a single fire event, the natural fire regime and the full impact of variations in regime are poorly understood. Pre-European fire return intervals for CSS are unknown, estimates of average fire return times within the last century are highly variable (e.g. Keeley 1982, Westman 1982, Minnich and Chou 1997, Keeley et al. 1999), and researchers actively debate the existence and effects of long-term fire suppression in southern California shrublands and how fire management in these shrublands should proceed (Keeley 1982, Minnich 1983, Zedler 1995, Minnich 1998, Keeley et al. 1999, Keeley and Fotheringham 2001a, b, Minnich 2001). Very long fire intervals may, debatably, increase fuel loading, fire size, and fire intensity in shrublands (Minnich 1983, Minnich and Chou 1997, Minnich 2001) and could lead to stand senescence with reduced diversity and postfire recovery (Zedler 1995, Franklin et al. 2001). Very short intervals, on the other hand, may reduce perennial resprouting ability and seed banks (Zedler et al. 1983, Haidinger and Keeley 1993, Zedler 1995), also preventing post-fire recovery.

If occasional fire maintains the CSS vegetation community, scientists do not yet know the ideal fire interval. The reported stability of the CSS shrub community in the absence of fire varies. For example, in the absence of fire and other disturbances for 60 years or more, chaparral and oak woodland has partially replaced CSS in some areas (Callaway and Davis 1993). However, structurally intact stands of CSS free of fire for over 60 years also exist (Westman 1982). Such discrepancies likely arise from the successional nature of CSS on some sites and climax status on others. Where CSS exists as a successional stage, shorter than natural fire return intervals might arrest succession towards the true climax community and favor persistence of the CSS vegetation, suggesting a need for fire to maintain CSS shrubs. However, stability of CSS vegetation as a climax community in the absence of fire is likely related to environmental factors at a given site (Westman 1981a, Mooney 1988, Keeley 1993) and to the ability of CSS shrubs to produce new basal shoots without fire (Malanson and Westman 1985). Some CSS shrub species also recruit seedlings between fires, though this recruitment may depend on small mammal consumption of the herbaceous understory, which provides suitable gaps for seedling establishment (DeSimone and Zedler 1999).

While CSS shrubs can survive long fire-free intervals, the viability and diversity of understory herbs may require periodic fire. For example, some herbs lack any form of seed dormancy (Keeley 1991) and, with fire suppression, could be eliminated from a CSS stand through shading when the shrub canopy closes. Maintenance of such species as part of the community would then depend on dispersal from nearby areas when the stand finally burns, which could be problematic if the fire is large or the stand is isolated. Alternatively, other herbs produce dormant seeds that may remain viable in the soil for up to 100 years (Keeley 1991). In addition, some late-successional taxa, such as lichens, may require very mature, undisturbed CSS stands for population expansion (Bowler and Riefner 2000). As a result, overall diversity may be negatively impacted by fire suppression only under the most extreme scenarios.

On the other hand, short fire intervals (or high fire frequencies) might have dramatic impacts on community composition. Frequent burning over short time intervals, such as 1 to 3 years, can eliminate or reduce shrub species such as *Eriogonum fasciculatum* (Zedler et al. 1983), *Salvia mellifera* (Haidinger and Keeley 1993) and *Lotus scoparius* (California broom, deerweed) (Zedler et al. 1983). Though a few CSS shrubs, such as *Artemisia californica*, may be less negatively affected by frequent burning (Zedler et al. 1983), shrub diversity and shrub cover will likely decline with frequent fires. In fact, frequent fire has been associated with CSS vegetation community transitions to grassland (Keeley 1990, Callaway and Davis 1993, Keeley 1993).

The transition of shrubland to grassland involves both direct damage to native shrub populations by fire and the invasion of shrublands by exotic annual grasses and forbs following fire. Haidinger and Keeley (1993) compared chaparral sites burned 1, 2, or 3 times in a six year period and found the density and percent composition of exotic annuals increased while species richness and percent composition of natives decreased with increased fire frequency. In contrast, sites with only one burn in 20 years had an absence or low abundance of exotic annuals. Similarly, Giessow (1997) found a positive relationship between fire frequency and abundance of exotic herbs in CSS. Thus, frequent fire likely favors disturbance-tolerant exotics over potentially less tolerant shrubs. Once established, exotics may further decrease the fire return interval by increasing fuel loading and continuity (Zedler et al. 1983, D'Antonio and Vitousek 1992), while decreasing the typical high fire intensities that might otherwise exclude them (Keeley 2001).

CSS conversion to a landscape dominated by exotic plants may be favored with fire return intervals of less than 5-10 years (O'Leary 1990a, Keeley 2004), but this will likely vary with the pre-burn condition of the CSS community, the size of the stand, and the species composition of stand edges. A single fire may provide the opening for aggressive invaders and result in more rapid vegetation type conversion, particularly in small habitat fragments. For example, Zedler and Scheid (1988) found *Carpobrotus edulis* (ice plant) invaded a 2500m² chaparral plot after a single controlled burn and had the second highest cover of any perennial plant three years after the fire. Prior to the fire, *C. edulis* appeared only on the disturbed edges of the plot. As noted by the authors, this illustrates the importance of edge effects in determining community response to disturbance. Edges with an abundance of exotic species, as well as presence of exotic plants within the stand prior to burning, will likely increase the potential for and rate of CSS conversion following fire.

Grazing

Grazing has occurred in many southern California plant communities since the arrival of early European settlers (Dodge 1975), though very few studies directly examine the effects of grazing on CSS. Nonetheless, alternative hypotheses regarding the effects of grazing on CSS exist. One argument contends grazing may benefit CSS and may have contributed to CSS expansion historically. Alternatively, heavy grazing may promote the conversion of CSS into exotic annual grassland.

Dodge (1975), Axelrod (1978), and, to a lesser extent, Kirkpatrick and Hutchinson (1980) argued heavy livestock grazing following Spanish settlement allowed shrub invasion into grasslands by reducing competition from grasses and reducing fuels and fire frequency in grasslands. Axelrod (1978) also argued overgrazing caused erosion of the soils that support grasses, further facilitating the expansion of CSS into grassland. Though not addressing the direct effects of grazing within CSS specifically, some evidence suggests grazing might facilitate the persistence, if not the expansion, of CSS shrubs. Callaway and Davis (1993) found grazing slowed the transition of CSS to chaparral quantified from aerial photos acquired in 1947 and 1989, suggesting grazing, like fire, might help maintain an otherwise successional CSS community in some circumstances.

However, strong evidence for expansion of CSS since European settlement is lacking. Mensing (1998) analyzed pollen cores from 2 sites near Santa Barbara and found only a slight (~2-3%) increase in *Artemisia* pollen since 1820, providing weak support for grazing-related increases in CSS or chaparral shrubland. Other studies suggest grazing does not facilitate expansion of CSS into grassland. For example, McBride (1974) observed cattle foraging on shrub seedlings and found actively grazed grasslands remained relatively free of shrubs. Freudenberg et al. (1987) also reported an absence of shrub seedlings in transition zones between CSS and grassland in grazed areas, indicating shrubs were not

invading the grassland. Finally, Callaway and Davis (1993) found grazing in grasslands decreased transition rates from grassland to CSS.

Oberbauer (1978) argued theories of grazing-facilitated shrub expansion into grassland were based on the expansion of spiny and unpalatable shrubs into grazed desert grasslands, whereas most CSS shrubs lack such defenses and are more likely damaged by grazing. Oberbauer (1978) also reported a decline in CSS and expansion of grassland in San Diego County between 1928 and 1975, while Freudenberger et al. (1987) found a similar overall increase in grasslands between 1928 and 1980 in the Los Angeles basin. Since grazing was widespread in both regions, these studies suggest grazing may actually facilitate expansion of grassland vegetation into CSS rather than the expansion or maintenance of CSS. Furthermore, Freudenberger et al. (1987) found the soils of grasslands near grazed transitions with CSS were more similar to those within CSS than to other grassland soils, supporting CSS replacement by grasses.

Additional evidence for grazing related replacement of CSS by grasses, particularly exotic annual grasses, comes from studies on the California Channel Islands where grazing by domestic and feral livestock has been widespread and sometimes intense. In a comparison of historical photographs with present-day vegetation on Santa Catalina Island spanning over 80 years of grazing, Minnich (1982) showed areas previously covered by chaparral and CSS converted to grasslands with only scattered shrubs. As with frequent fire, grazing related transitions from shrubland to grassland likely result from direct damage to shrubs combined with exotic grass invasion. Coblenz (1980) and Van Vuren and Coblenz (1987) showed grazing by feral goats and sheep on some islands directly and significantly damaged mature shrubs, as well as shrub seedlings, and was associated with the creation of large patches of bare ground. Many of the common exotic grasses and forbs in California are fast-growing, disturbance-adapted species (Oberbauer 1978, Groves 1986) well equipped to invade the openings created by grazing. Indeed, Westman (1983) found CSS stands on the inner Channel Islands had a higher percentage composition of exotic species compared to mainland sites, which had a less intense grazing history, and suggested this was due to increased colonization by exotics on the islands following reduced shrub cover from grazing.

While grazing can damage CSS shrubs and may promote exotic invasion in CSS, the grazing intensity required for conversion to grassland and the permanency of the conversion remain unclear. McBride (1974) found shrubs reinvaded grasslands after grazing ended, and other studies (Oberbauer 1978, Freudenberger et al. 1987) found evidence of shrub recruitment in grasslands even where some grazing still occurred. Yet even with the successful re-establishment of shrubs on grazed sites, the exotic species established during grazing may persist as permanent components of the vegetation (e.g. Westman 1983) and could impact the integrity and long-term viability of the CSS community.

Mechanical Disturbance

We define “mechanical disturbance” as disturbance that damages or completely destroys aboveground vegetation by mechanical means. Mechanical disturbances may occur as a result of urban development, agriculture, road and fuel break construction, or recreation and, in all cases, cover a range of intensities. Despite the varied sources of mechanical disturbance in CSS, few studies exist detailing its effects.

A small number of studies suggest CSS shrubs can recover from even intense mechanical disturbance. Westman (1976) found native shrubs successfully re-established in an area bulldozed and planted with exotic species 13 years earlier. *A. californica* dominated the site, primarily due to seedling establishment, despite its low representation prior to disturbance. Similarly, Zink et al. (1995) studied vegetation recovery following underground pipeline construction through a nature reserve. The

construction affected four major plant communities: grassland, CSS, chaparral, and oak woodland. After the disturbance, the construction site was seeded with a mixture of native and exotic species. Surveys over 10 years later revealed *E. fasciculatum* and *A. californica* had successfully re-established along most of the disturbance corridor, and shrub cover was similar between disturbed and undisturbed CSS. Finally, Narog et al. (2000) examined recovery of an artificially created stand of CSS shrubs subjected to mechanical removal of all aboveground vegetation. The stand was originally planted with *Artemisia californica*, *Eriogonum fasciculatum*, *Encelia farinosa*, *Salvia mellifera*, and *S. apiana* maintained in equal numbers prior to the disturbance. Four years after the disturbance, all shrub species had re-established on the site from both resprouting individuals and seedling recruitment. Of all shrub species, *A. californica* and *E. fasciculatum* had the highest density, percent composition, and relative cover, again due to successful seedling establishment.

Despite some evidence of limited localized seed dispersal (Chalekian 2002), the relative success of *A. californica* and *E. fasciculatum* by seedling recruitment suggests these shrub species can readily colonize openings created by mechanical disturbance and, therefore, may generally have more resilience to disturbance than other CSS shrubs. The lightweight, potentially wind-dispersed seeds of these shrubs may allow rapid colonization of disturbance-related gaps suitable for germination and seedling establishment. In fact, these species are well-known colonizers of disturbed areas such as road cuts and old fields within a variety of vegetation communities (Zedler 1981, Stylinski and Allen 1999). Resilience of these shrubs may aid post-disturbance recovery, but changes in local shrub species composition may result due to differential resilience among CSS shrub species.

Even for these seemingly resilient shrub species, however, reported levels of post-mechanical disturbance recovery vary. Stylinski and Allen (1999) compared a chronosequence of relatively undisturbed CSS and chaparral with near-by shrublands that had experienced severe mechanical disturbance between 2 to 71 years previously. While *E. fasciculatum* and *Baccharis sarothroides* (broom baccharis) occurred on disturbed sites, disturbed plots had lower native shrub cover than undisturbed plots and shrub cover did not increase with age of the disturbed sites.

Site-specific conditions such as water or nutrient availability during periods of seedling recruitment might explain differences in CSS shrub recovery across studies. For example, in the studies conducted by both Westman (1976) and Narog et al. (2000), some supplemental water was provided after the disturbance, which may have contributed to successful re-establishment of native species. In fact, Westman (1976) showed both irrigation and fertilizer addition generally enhanced reinvasion and increases in cover by native species.

Despite the potential resilience of some shrub species, two studies suggest the overall CSS community may fail to fully recover following mechanical disturbance, even where some shrubs do recover. Zink et al. (1995) found higher indices of similarity between mechanically disturbed CSS and other disturbed vegetation types (i.e. oak woodland and chaparral) than between disturbed and undisturbed CSS, despite similar shrub cover between disturbed and undisturbed CSS. Similarly, Stylinski and Allen (1999) found no relationship between time since disturbance and percent similarity of severely disturbed and undisturbed CSS, indicating a failure of CSS community recovery even after 43 years without subsequent disturbance.

The apparent failure of CSS plant community recovery in these two studies relates to exotic species invasion and a lack of native understory species recovery following disturbance. Zink et al. (1995) found exotic species had significantly higher cover and native herbs had significantly lower cover in all disturbed communities compared to undisturbed vegetation. While direct seeding of exotics following construction likely aided the establishment of some species, the most abundant exotics ten years later were annual grasses not included in the original seed mix but present in areas surrounding the

reserve. The abundance of these grasses long after the disturbance suggests they successfully invaded following disturbance and became persistent components of the post-disturbance community, while native herbs did so less successfully. Styliniski and Allen (1999) also found high exotic plant cover and low native herb cover and richness (see Davis 1994) on disturbed CSS plots.

These studies of mechanical disturbance in CSS show, though at least some shrub species may recover from intense disturbances, the resilience of a few shrubs within the CSS community does not guarantee recovery of the entire plant community. In particular, the low cover and richness of native herbs noted on mechanically disturbed sites suggests they may not share the resilience of native shrubs. Also, like frequent fire and grazing, mechanical disturbance in CSS is associated with a potentially permanent increase in exotic plant species, even in cases where shrubs recover. It is still not clear whether the decrease in native herbs on mechanically disturbed sites results from the disturbance itself, or is a function of the exotic invasion associated with disturbance.

Air Pollution

Urbanization and industrialization often increase atmospheric pollutants such as ozone, sulfur dioxide, and oxides of nitrogen. These pollutants may have harmful effects on plant tissues, but documenting the direct effects of pollutants on natural plant communities is difficult given possible time lags in response to pollution and the number of potentially confounding variables in natural systems. Nonetheless, studies have associated high pollution levels with decreases in cover of native shrubs and increases in exotic plant abundance in CSS.

The southeastern Riverside basin often has higher air pollution concentrations than elsewhere in southern California, and this may influence the successional and floristic differences seen between CSS in this region and low pollution regions (Westman 1979, 1981a, O'Leary and Westman 1988). Using data from 67 southern California CSS sites, Westman (1979) found increases in the mean annual concentration of oxidants accounted for decreased native cover at some sites, such as those in the Riverside basin, better than other explanatory variables. Increased oxidant levels also related to decreased species richness and equitability (Westman 1979, 1981a). Several studies have also found CSS in the Riverside basin has a higher cover of exotic species compared to other regions, and the authors have offered air pollution as a potential causal factor (Westman 1981a, Westman 1981b, O'Leary and Westman 1988, Allen et al. 1997, Minnich and Dezzani 1998). For example, O'Leary and Westman (1988) hypothesized pollution damage to CSS shrubs and perennial herbs may decrease growth and resprouting ability, providing space for pollution resistant exotics. However, others (e.g. Haidinger and Keeley 1993, Keeley 2004) have argued simple correlations between atmospheric pollution and CSS decline are confounded by the effects of other disturbances, such as frequent fire, common in high pollution areas like the Riverside basin.

Some studies have demonstrated a negative effect of atmospheric pollutants on CSS shrubs and increase in exotics while controlling for other disturbances. In particular, Preston (1988) examined the effects of sulphur dioxide (SO₂) emissions on physiologic and growth changes in *Salvia mellifera* by comparing sites along an SO₂ gradient downwind of an oil refinery with relatively pollution-free upwind sites. Stomatal resistance, intranodal stem length, and heights of *Salvia* shrubs were lower overall in the downwind, polluted sites compared to upwind sites. In addition, at the most polluted sites close to and downwind of the refinery, defoliation rates increased, leaf size and perennial cover were lower, and exotic cover was higher compared to sites further away or upwind.

While these results suggest high levels of SO₂ pollution may directly affect CSS shrubs, Allen et al. (1997) and Padgett and Allen (1999) argued the seasonal fluctuations in pollution loading may mitigate the potentially harmful effects of some atmospheric pollutants. In particular, they argued atmospheric pollution loads during the active spring growing season are low in the Riverside basin. The

highest pollution loads occur during the summer when the leaves of most drought deciduous, perennial species in CSS begin to senesce and, therefore, atmospheric pollutants may have minimal effects on these plants.

On the other hand, nitrogen deposition from atmospheric pollution increases inorganic nitrogen loads in the soil, which may affect plants independently of seasonal changes in atmospheric levels (Allen et al. 1997, Padgett and Allen 1999). Allen et al. (1997) found qualitatively reduced native shrub cover and increased exotic annual grass cover in areas with higher nitrogen levels and hypothesized high nitrogen deposition may not only directly and negatively affect native shrubs but also positively affect the growth and competitive abilities of exotic annual grasses. However, their experiments showed both exotic grasses and native shrubs had similar growth responses to nitrogen fertilization in the field and greenhouse, suggesting nitrogen addition does not necessarily harm or favor one plant group over the other. In addition, greenhouse experiments by Padgett and Allen (1999) showed some native shrubs responded to nitrogen fertilization with increased growth well beyond the levels of some common exotic annuals. Similarly, Westman (1976) found fertilization with nitrogen and phosphorous enhanced the ability of native CSS shrubs to successfully re-invade disturbed sites.

Thus, greater nitrogen uptake by exotics compared to shrubs does not appear as a causal mechanism of CSS replacement by exotics in high pollution areas. Instead, competition with exotic grasses for resources other than nitrogen may influence native shrub survival. For example, Allen et al. (1997) found *Artemisia californica* seedlings grew larger and had increased survival in plots with exotic annuals removed compared to seedlings in unweeded plots, regardless of nitrogen addition.

Despite initial increases in shrub seedling growth following fertilization, chronic high levels of soil nitrogen may, nonetheless, contribute to shrub mortality. Zink and Allen (1998) suggested increased nitrogen availability decreased survival of CSS shrubs on a restoration site, but potentially confounding factors such as water availability and exotic abundance were not apparently controlled. Allen et al. (1997) noted early senescence and mortality of *A. californica* seedlings under high nitrogen fertilization conditions in the greenhouse, suggesting a toxic effect of nitrogen at very high levels. However, Padgett & Allen (1999) hypothesized increased shrub growth in response to nitrogen fertilization may prevent plants from restricting growth and preparing for dormancy leading to increased mortality of shrubs.

High levels of nitrogen deposition could also indirectly influence shrub survival through effects on beneficial symbionts, such as mycorrhizal fungi. Egerton-Warburton and Allen (2000) evaluated changes in diversity and abundance of arbuscular mycorrhizae (AM) in CSS along a nitrogen deposition gradient and in experimental nitrogen fertilization plots. Plots with increased nitrogen had lower abundances of larger-spored *Scutellospora* and *Gigaspora* species, increased abundance of some small-spored *Glomus* species, and reductions in AM species richness and diversity. The authors argued the abundance of *Glomus* spp. in high nitrogen plots may contribute to declines in CSS shrubs over time as these AM species exhibit a more parasitic relationship with host species than the AM species decreased by nitrogen addition.

The overall effects of atmospheric pollution on the CSS community are not entirely clear. While some evidence suggests high pollution levels can negatively impact CSS shrubs either directly or indirectly, impacts to other native growth forms have not been well studied. O'Leary and Westman (O'Leary and Westman 1988) hypothesized atmospheric pollution might negatively impact native herbaceous perennials since the growing season for these species extends further into the summer when pollution loads near maximum values, though to our knowledge studies of such impacts have not been conducted. In addition, because high pollution levels are associated with increases in exotic annuals, declines in some native CSS species with increased pollution may primarily be a function of increased

competition with exotics. Ultimately, more studies are needed to determine the exact nature and mechanisms of these relationships.

Habitat Fragmentation

Habitat fragmentation is often associated with losses in native species diversity, particularly losses of rare species (Saunders et al. 1991). CSS vegetation contains many rare species, which may be vulnerable to the negative impacts of fragmentation (e.g. O'Leary 1990a). Although fragmentation and isolation of CSS has occurred rapidly over the last 50 years, few studies document the effects of fragmentation on the CSS vegetation community.

Alberts et al. (1993) conducted surveys for native and exotic plant species (excluding grasses) in CSS canyon fragments (0.4 to 102.7 ha) in San Diego County, CA. Native species richness in fragments declined with fragment size (i.e. area and/or perimeter) and as fragments aged, suggesting natives are excluded from fragments both initially and over time. Overall exotic species richness, on the other hand, increased as canyon fragments aged primarily due to an increase in ornamental species. In addition, fragments with longer perimeters, or increased edge, had more exotic, ruderal weeds, while supplemental water related to increased numbers of escaped ornamental species.

While initial species loss in CSS fragments is likely a direct result of reductions in habitat area and the number unique of micro habitats, the increase in exotics and decline of natives with increasing age suggests exotics competitive exclusion may explain native species loss over time. However, in Alberts et al.'s (1993) study, neither ruderal nor ornamental exotic species richness were significant predictors of native species richness in regression analyses also including size and age, among other variables. In addition native species richness did not significantly differ between groups of fragments with and without a strongly competitive exotic tree. According to the authors, this suggested the competitive interactions between exotic and native species may be less important to shifts in species composition in fragments over time than the effects of other variables such as habitat alterations.

While habitat alterations in fragments likely do contribute to losses of some species, Alberts et al.'s (1993) conclusion about the relative importance of competitive effects is problematic for a variety of reasons. First, because ruderal exotic species richness, like native richness, was highly correlated with fragment size (Alberts et al. 1993), it is not surprising it did not emerge as a significant predictor of native richness when entered into a regression model simultaneously with fragment size, given the study's small sample (N = 25). In essence, the study likely lacked sufficient power to separate the effects of exotic richness from those of fragment size. Second, the authors did not include exotic annual grasses among their inventory of exotic species. However, these exotics are common invaders (e.g. O'Leary and Westman 1988, Haidinger and Keeley 1993, Zink et al. 1995, Giessow 1997, Stylinski and Allen 1999) and may strongly compete with native species (Allen et al. 1997, Eliason and Allen 1997, Chalekian 2002). Thus, inclusion of exotic grasses might (or might not) have produced different results. Finally, because competition occurs at scales smaller than the size of the fragment, the number of exotic individuals (i.e. exotic abundance) within the fragment area might be a better indicator of competitive effects than the number or simple presence of exotic species.

One study of coastal sage-succulent scrub in Baja California by Escofet and Espejel (1999) has shown exotic annual grasses can be abundant and persistent invaders in habitat fragments. Escofet and Espejel (1999) found the exotic grass *Bromus madritensis ssp. rubens* accounted for most of the annual cover in habitat fragments of all ages, while unfragmented control areas contained no *Bromus*. While not directly examined or discussed by the authors, their presented data also suggested species loss occurred in fragments over time since older habitat fragments contained only ~52% of the species found in the unfragmented control while newly fragmented patches contained ~83% of those species.

Though evidence suggests native species are lost following CSS fragmentation, the exact mechanisms of loss are unknown. Ultimately, the process of fragmentation typically results in additional forms of disturbance, including exotic invasion. The combined effects of both fragmentation and the additional disturbances likely contribute to localized native extinctions in fragments over time. Whatever the cause of species loss, we lack the pre-fragmentation data to determine which CSS plant species or growth forms are most vulnerable to local extinction. Thus, more long-term studies are needed in CSS fragments to elucidate relative extinction risk, as well as the interrelationships between fragmentation, invasion, other forms of disturbance, and species loss.

Discussion

Our review does not, in all cases, indicate predictable responses of native CSS vegetation to specific disturbance types, but some generalities across disturbances do emerge. First, while many disturbances have the potential to directly damage or kill native CSS species, the vegetation exhibits a large potential for recovery once disturbances are reduced. In particular, the dominant CSS shrub species possess some resilience to a wide range of physical disturbances (e.g. fire, grazing, & mechanical disturbances), since they may quickly regenerate from resprouting individuals or widely dispersed seeds. Yet, resilience appears to vary across shrub species and the overall level of shrub recovery documented across studies is variable. The critical factors that promote high levels of shrub recovery have not been clearly identified. In addition, we do not know the extent to which other vegetation life forms, such as the understory herbs, share the resilience noted for shrub species. Furthermore, mechanisms like resprouting, which impart resilience to shrubs following physical disturbances, may be of less consequence in the face of increasing air pollution or habitat fragmentation. Additional work must be done to determine which factors favor or limit overall CSS community recovery and the mechanisms at work during the recovery process.

Our review also indicates a strong association between many forms of disturbance in CSS and exotic plant invasions. The association between disturbance and invasion has been discussed, in general, by several authors (e.g. Hobbs 1987, 1989, Rejmanek 1989, Hobbs 1991, Lepart and Debussche 1991, Hobbs and Huenneke 1992), and has been documented in other plant communities (e.g. Hobbs 1989, Burke and Grime 1996, Abensperg-Traun et al. 1998). Because invaders may: a) compete directly with native species; b) alter food-web structure by attracting additional predators (Mack et al. 2000); or c) disrupt community dynamics and natural disturbance regimes (Zedler et al. 1983, D'Antonio and Vitousek 1992, Keeley 2001), levels of post-disturbance invasion may be one critical factor determining community recovery. Therefore, understanding and effectively managing the impact of disturbance in CSS, or any community, will require a better understanding of the relationship between disturbance and invasion and the specific impacts of invaders within the community.

In any plant community, disturbance likely facilitates the invasion process by creating openings for colonization and/or altering resource availability (Rejmanek 1989, Hobbs 1991, Hobbs and Huenneke 1992). However, the amount of disturbance, if any, necessary to facilitate invasion likely varies by invading species (Lepart and Debussche 1991, Rejmanek et al. 1991). Invasive species with strong competitive abilities and/or high dispersal likely require low levels of disturbance and may spread rapidly through a community. Other invaders may depend on relatively intense disturbances, which create large openings or substantial resource modifications to initially invade new sites (e.g. Giessow 1997). Subsequent small and natural disturbances creating suitable microsites for population expansion may then facilitate the spread of these invaders into adjacent areas (e.g. Lambrinos 2000), while large-scale

disturbances, such as fire, may allow very rapid proliferation (e.g. Zedler and Scheid 1988). Thus, various disturbances may interact to promote invasion.

Determining the impacts of post-disturbance invasion within a community requires separating the effects of invasion from the effects of the disturbance itself. Some studies in CSS suggest increased competition, between natives and exotics, is one impact of invasion separate from any disturbance effects. Presence of dense exotic grasses decreases germination (Eliason and Allen 1997) and first season growth (Allen et al. 1997, Eliason and Allen 1997) of *Artemisia californica* seedlings, as well as seedling establishment and survival of *Eriogonum fasciculatum* and *Salvia apiana* (Chalekian 2002). Yet, despite this apparent competition, CSS shrub recruitment can occur in areas dominated by exotic species after disturbance is reduced (McBride 1974, Westman 1976, Oberbauer 1978, Minnich 1982, Freudenberger et al. 1987, Zink et al. 1995, Narog et al. 2000). Successful recruitment may be, in part, explained by the decreased impact of competition on future shrub survival if seedlings survive the first season of growth (Eliason and Allen 1997). However, it may be the full extent of post-disturbance shrub recovery in invaded areas depends, again, on additional water availability (e.g. heavy rain years) to offset competition during periods of seedling recruitment, as well as the abundance and distribution of invader-free microsites suitable for seedling establishment relative to a shrub species' seed-dispersal ability (Chalekian 2002).

Recovery of native herb species may also face risks from post-disturbance invasion, though few studies have examined this directly. For example, post-disturbance competition between native and exotic herbs might explain the declines in cover and richness of native herbs reported by Davis (1994) and Zink (1995). Though more studies are needed to verify this hypothesis, some evidence for competition between herbs does exist. Presence of exotic annuals in grassland habitat can decrease seedling establishment and reproductive output of the native perennial bunchgrass *Nassella pulchra* (purple needlegrass) (Dyer and Rice 1999, Hamilton et al. 1999), a common component of the CSS understory. Compared to the deep rooting, tall shrubs some native herbs could face stronger competition with annual exotic grasses and forbs given their similar growth forms and rooting patterns. Annual herbs, which must both survive and reproduce in a single season, might be particularly vulnerable to competitive exclusion by exotics, though no studies have examined this directly in CSS.

Ultimately, predicting or actively managing for CSS recovery following disturbance will depend on a more effective separation of the impacts of disturbance versus the full impacts of post-disturbance invasion. Alternative scenarios are possible. First, despite direct injury and mortality to natives following disturbance, CSS will recover if: 1) seed banks are not completely destroyed or some adult plants remain, 2) the area disturbed is small enough for colonization by new seeds; and 3) subsequent disturbances are small and/or occur at intervals longer than the time required to build new seed banks and resprouting rootstocks. When disturbances are so intense that all possible in-situ reproduction is lost or are much larger than typical seed dispersal distances, recovery may still occur but may be extremely slow. Colonization by exotics, even in abundance, might reduce rates of recovery, but recovery may still occur provided there are enough post-disturbance years with good rainfall (or other sources of water) and processes which create invader-free microsites for native seedling establishment. Eventually, if recovery proceeds to the extent of shrub canopy closure, exotics may be effectively eliminated from the CSS stand (Keeley 2001). Alternatively, once exotics become established in CSS following disturbance, added competition and changes to a variety of community dynamics may permanently alter the community if exotic spread is not held in check. The spread of exotics may be slow, at first, until some threshold is reached or until some large-scale disturbance or several small-scale disturbances accelerate invasion and community type conversion. Even without full-scale type conversion, differential resilience among natives in the face of exotic spread might produce a "new" CSS vegetation community dominated by just

a few shrub species and largely devoid of native herbs. Under this scenario, exotic spread is the single largest determinant of CSS recovery following disturbance.

While the likelihood of either scenario depends, again, on the specific invaders at any site, we still do not truly know which scenario is most relevant to the common European annual invaders. With respect to these exotics, the truth probably lies somewhere between the two scenarios and is dependent, in addition, on the larger landscape within which the CSS stand is embedded.

Conclusions

Despite the rich literature on CSS and related mediterranean-type shrublands, several questions remain unanswered with respect to disturbance effects and invasion. Additional studies are needed to aid the management and conservation of CSS habitat. In particular, studies are needed to:

1. Examine the effects, including both pattern and process, of various disturbances on the non-dominant vegetation forms in CSS. While the dominant shrubs may be the primary players in overall system dynamics, ignoring understory species yields an incomplete picture of community response to disturbance. Long-term studies of recovery, which are lacking for many disturbances, will be needed to fully assess the relative risks to different CSS components.
2. More effectively disentangle the effects of disturbances from the effects of invasion following those disturbances. This is particularly relevant for disturbances, such as air pollution and habitat fragmentation, with unclear direct effects. Such studies will likely require experimental manipulations that control post-disturbance invasion levels and may prove difficult to perform. Yet these studies should provide some indication of the mechanisms through which recovery from these disturbances occurs.
3. Examine whether and how different disturbances interact to affect rates of invasion, and whether particular disturbances or disturbance regimes favor specific invaders. These studies could involve experimental manipulations or exploration of large, long-term data sets, and will not only aid our understanding of invasion dynamics in CSS, but will inform management decisions aimed at minimizing post-disturbance invasion.
4. Further examine how invaders interact or compete with native species, including both dominant shrubs and understory herbs, which have been largely ignored in CSS. Such studies will indicate if and how post-disturbance invasion interferes with CSS community recovery, and may elucidate additional recovery mechanisms or suggest potential invasion control procedures. In addition, any variability in rates of exclusion among individual or groups of native species may be useable as a potential indicator of the community degradation in invaded CSS habitat.

Answering these questions will provide practical information to land managers and aid the success of CSS conservation programs such as the MSHCP's being developed in southern California. Additionally, such research might shed light on more general ecological questions regarding disturbance effects, exotic species invasions, and ecosystem function applicable to other vegetation communities.

Mammals

Introduction

There are many reasons small mammals might be useful indicators of disturbance in CSS systems. Because of their abundant numbers, high reproductive rates, and broad diets, small mammals play a vital role in nearly every ecosystem in which they have been studied. For example, small mammal herbivory and granivory can influence rates of production, alter species composition (Mills 1983, DeSimone and Zedler 1999), and change patterns of post-disturbance succession in vegetation communities (Mills 1986, Ostfeld and Canham 1993), Davidson 1993. Work by J. Brown and his students show small mammal seed predation maintains desert scrub communities in the arid southwest. In the absence of this predation, the areas would convert to arid grasslands (Brown and Heske 1990). Other studies indicate small mammal herbivory and predation limits population sizes of terrestrial arthropods (Batzli & Pitelka 1970). Additionally, small mammals are key prey items for snakes, raptors, and medium-sized mammals such as coyotes and bobcats and may play a role in limiting the population sizes of such predators (Bowyer et al. 1983). As both predators and prey items, small mammals interact with numerous trophic levels within a food web and thus play a critical role in determining species diversity and composition within an ecosystem.

In addition to crucial roles in many food webs, numerous studies demonstrate small mammals often respond directly to disturbances. Small mammals are well known for responding directly to both physical structure and species composition in plant communities. Indeed, many studies suggest small mammals may show large responses to fire, invasive species, habitat fragmentation and isolation, and edge effects.

Fire

Fire is recognized as a primary factor affecting both the distribution and species composition of CSS flora and fauna. The density of plant cover, in combination with southern California's arid Mediterranean climate, makes CSS one of the most fire-prone vegetation types in the world (Hanes 1971). Despite many studies on the responses of CSS vegetation to fire and its consequent succession, little research has been done on the responses of CSS mammals to fire and post-fire faunal succession.

Fire can directly kill individuals through incineration and asphyxiation. Fire mortality is most common among small, relatively immobile mammals that lack access to refuge during fire, but even fast moving large mammals, such as deer, can be killed in intense, fast moving fires (Quinn 1994). Sources of refuge during fire can include below ground burrows, rock crevices, rock outcroppings, and spaces beneath logs or stones, where temperatures remain below 138° F (Howard, et al. 1959).

Post-fire impacts on small mammal communities are well studied in a Mediterranean region of Australia similar to southern California. By sampling areas with different ages since fire (substituting space for time), and by sampling a small number of locations for over 20 years, Fox (1982, 1993, 1996) has shown a predictable post-fire pattern of succession in the small mammal community. In the Australian heathlands, small mammal succession occurs because species-specific habitat requirements and interspecific competition varies with temporal changes in vegetation composition (Fox 1996). Fox (1981) also showed succession in heathland small mammals occurs as a replacement of dominant species rather than as a replacement of guilds as sometimes seen in other taxa. Although small mammal responses to fire are not nearly as well studied in southern California's CSS, we feel post-fire succession patterns are likely similar. We describe our reasoning below.

In CSS, fire causes a reduction in cover and food resources and may cause a shift from species preferring dense shrubs and high amounts canopy cover to those that live in more open habitats (Lawrence 1966). Generally, early post-fire shrub sites, such as CSS, are colonized by grassland species such as *Microtus californicus* (California vole), *Reithrodontomys megalotis* (harvest mouse), *Peromyscus maniculatus* (deer mouse), and sometimes the non-native *Mus musculus* (house mouse), rather than species more often found in mature CSS such as *Peromyscus eremicus* (cactus mouse) and *Neotoma lepida* (desert wood rat) (Cook 1959). Typically, as shrub density and food availability increase in the months after a fire, CSS species re-invade the site (Lawrence 1966, Schwilk and Keeley 1998).

In a study of post-fire responses of five small mammal species in CSS, Price and Waser (1984) were able to successfully predict the post-fire relative abundances of co-existing species by understanding their microhabitat use (open, brush, debris, and rock microhabitats). They found *Dipodomys agilis* (pacific kangaroo rat) was the only species that specialized on open microhabitat. *Peromyscus maniculatus* showed a slight preference for debris and *Chaetodipus fallax* (San Diego pocket mouse) preferred rock. *Peromyscus eremicus* and *N. lepida* used both rock and bush microhabitats with approximately equal frequency.

In the burned CSS, fire significantly increased open microhabitat and significantly decreased bush microhabitat. Immediately after the fire, debris was much less abundant in the burned area than in the unburned area. Fire had no impact on the rock microhabitat available. Given these changes in microhabitat, *D. agilis* became consistently more abundant in burned than unburned sites, while the other species were generally more abundant in unburned sites. Thus in CSS, like in the Australian heathlands, a predictable post-fire rodent succession may occur because of species-specific habitat requirements associated with changes in vegetation through time.

It is likely, however, that there is a delay between the time when plant succession creates suitable habitat for an absent species and the establishment of a viable population of that species. The length of the delay will depend on the dispersal rate and the distance to the nearest source of immigrants (Quinn 1994). This implies the recovery rate of the mammal community will depend on both the characteristics of a species, the characteristics of the matrix surrounding a burn, and the size of the burn (Schwilk & Keeley 1998).

Invasive species

Additionally, changes in fire frequency and intensity, often associated with human disturbance, may alter post-fire plant succession and even promote non-native invasion. Frequent burning, for example, has been associated with increased transition rates from CSS to grasslands and the invasion of CSS by exotic annual forbs and grasses (Callaway and Davis 1993). Price (1994) found that small mammal responses to shrub removal in CSS differed by species. While the abundance of *Dipodomys stephensi* (Stephen's kangaroo rat), *Perognathus longimembris* (little pocket mouse), *C. fallax*, and *P. maniculatus* increased in response to shrub removal, the abundance of *Dipodomys agilis* (pacific kangaroo rat) and *P. eremicus* decreased. Given the specific diets (Meserve 1976) and habitat requirements (Price and Waser 1984) of small mammals, it is likely that CSS small mammal species will respond uniquely to changes in post-fire plant succession and the establishment of non-native plant associations.

Habitat Fragmentation and Isolation

Habitat fragmentation is a process that degrades a continuous habitat into smaller patches. Though habitat fragmentation can occur through natural processes such as fire or windfall, the most large-

scale cause is the expansion of human land use (Andren 1994). Habitat fragmentation has three major impacts; loss of original habitat, reduced habitat patch size, and increased isolation of habitat patches.

Habitat loss generally results in decreases in native species abundance. Reduced patch size and increased patch isolation typically results in decreases in species richness in habitat fragments. This pattern is well established from both empirical studies of habitat fragmentation and a long history of research into the theory of island biogeography (MacArthur and Wilson 1967, Rosenweig 1995)). Decreases in patch size might also result in smaller effective populations and decreased genetic diversity. Additionally, habitat fragmentation may affect interspecific interactions and ecological processes, such as competition and predation, possibly intensifying any original fragmentation effects (Debinski and Holt 1998).

Ecologists have studied small mammal responses to habitat fragmentation both experimentally and in naturally patchy systems more so than many other taxa (Lidicker 1995, Barrett and Peles 1999). Small mammals are relatively easy to study given the small spatial scale at which populations exist, and the ability to mark and recapture individuals. In a review paper, Debinski and Holt (2000) concluded studies have indicated some support for the expected decreases in species abundance and richness with decreased fragment size, but have also demonstrated changes in movement rates, distances, and spatial patterns, as well as changes in home range sizes and overlap.

Though remaining CSS is highly fragmented, the impacts of this patchy landscape on small mammals, as well as many other organisms, have not yet been thoroughly studied. In one study (Bolger et al. 1997a), small mammal distributions were surveyed by live-trapping at 25 fragmented CSS/chaparral sites and 3 relatively unfragmented CSS/chaparral sites in coastal San Diego County. Isolated sites varied in percent shrub cover, size, time since isolation, and distance from a mainland, but were all completely surrounded by human-modified habitat. Rodents were trapped from 2 to 5 times at each site during 1986-87 and then 12 of the 25 sites were surveyed again in 1992 to assess extinction and recolonization events. The authors found an association between high abundances of native small mammal species (*Neotoma fuscipes* (dusky-footed woodrat), *Peromyscus californicus* (California mouse), *P. eremicus*, *C. fallax*, *N. lepida*, and *R. megalotis*) and more recent time since isolation; higher percentages of shrub cover; and shorter distances from unfragmented sites; and the absence of *Rattus rattus* (Black rat).

Small mammals were not found on 13 of the 25 fragment sites. These 13 sites were generally smaller than fragments that did support populations of native small mammals. The size of a fragment site was the primary determinant of species diversity. In addition, fragments supported fewer species than equivalently sized plots in unfragmented habitat and older fragments (isolated for longer periods of time) supported fewer species than younger fragments. The isolation distance of a fragment had no relationship to species diversity. These results indicate that relatively rapid local extinctions occur within the small, urban fragments and are predictable based on species abundances in non-fragmented areas (i.e. species that are typically abundant everywhere are less likely to go extinct when isolated in a fragment). Also, the area of shrub vegetation at a site was a better predictor of the number of native small mammals at that site than the fragment's total area. This indicates that small mammals native to CSS may not often use non-native herbaceous vegetation and emphasizes, once again, the importance of microhabitat availability. Lastly, high numbers of native small mammal species were associated with the absence of the non-native *Rattus rattus* (Black rat) at a site, indicating competition between native and non-native small mammals for resources in CSS systems.

Edge Effects

Because habitat fragmentation inevitably creates a matrix of qualitatively different habitats, edge area is increased. Increasing the amount of edge can have enormous impacts on a system. Increased edge can subject a system to changes in abiotic conditions such as wind and temperature. Changes in abiotic conditions can result in altered vegetation communities and processes such as nutrient cycling. Increased edge can also make a system vulnerable to biotic changes such as invasion by non-native species. Changes in biotic and abiotic conditions resulting from increased edge can profoundly alter ecological processes such as competition and predation (Debinski & Holt 2000).

In general, small mammals seem to respond strongly to habitat alteration and exhibit species-specific responses depending on their habitat requirements (Sauvajot et al. 1998). Because small mammals respond directly to both physical structure and species composition in plant communities, as well as to disturbances within those communities, species abundance often varies both with distance to an edge and with type of edge (Weber 1995, Sauvajot et al. 1998). Potential effects of edges include changes in resource availability (food or habitat) and competition, changes in predation or other types of direct mortality such as road mortality, (Adams and Geis 1983), and avoidance behavior (i.e. avoidance of anthropomorphic lights, sounds, or smells).

CSS is, to a large extent, surrounded by an urban matrix. Evidence suggests that native small mammals may respond strongly to urban edges. Non-native species often associated with urban development, such as the Argentine ant, may compete with small mammals for food resources and infest nesting areas (Suarez 1998). Some studies also suggest that native species may compete with non-native species such as *R. rattus* and *M. musculus* near urban edges and possibly be displaced by them (King 1957, Bolger et al. 1997b).

Probably the strongest edge effect confronted by CSS small mammals is predation by domestic cats. Crooks & Soule (1999) found evidence for “mesopredator release” throughout the fragmented CSS canyons of southern California. Decreases in large mammalian carnivores such as coyotes and mountain lions in CSS fragments allow for increases in medium-sized predators (mesopredators) such as domestic cats. Domestic cats are recreational hunters subsidized by humans and are extremely effective predators of small mammals. It is likely that the presence of domestic cats greatly decreases small mammal abundances near urban edges.

Conclusion

The above review suggests mammals, and small mammals in particular, may show measurable responses to certain forms of disturbance and may be effective in an IBI for CSS. CSS mammals will exhibit responses primarily to changes in the vegetation community, and perhaps also to alterations in levels of predation and levels of competition caused by disturbance. While many forms of disturbance will impact mammals directly, perhaps most of the impacts will result from complex, difficult to understand, indirect effects. For examples, urban edges may directly impact small mammal communities because some species are intolerant to the increased light and noise levels. However, with urban edges come domestic cats, non-native small mammals, altered hydrology, argentine ants, invasive plants, and perhaps altered fire regimes. The literature review leads us to cautiously conclude disturbance to CSS will likely have larger indirect effects on small mammals, mediated by changes to CSS vegetation, than direct effects. As discussed in the vegetation section above, a fundamental effect of nearly any form of disturbance in CSS is an alteration in the vegetation community.

Given the high levels of habitat selection CSS small mammals perform and the degree of microhabitat preferences exhibited by some species, well-designed monitoring programs should be able to

measure responses of small mammals to changes in vegetation caused by disturbance. These responses will likely manifest themselves as changes in the numbers and types of species present in a system. Studies in other systems indicate processes such as fire, predation and competition can create predictable assemblages of small mammals (Fox 1987, Kotler and Brown 1988). Furthermore these assemblages can be predicted by the presence or absence of factors related to these processes (i.e. rainfall, soil type, or particular competing or predatory species).

Fortunately, the few studies of CSS small mammal communities were well-designed and implemented (Meserve 1972, 1976, Price and Waser 1984). These documented the diets, foraging behaviors and microhabitat use of CSS small mammals and suggest the presence of particular species can be associated with specific processes. Because the diets, foraging behaviors, and microhabitat use of CSS small mammals are somewhat species-specific, it is likely that some members of the community will respond uniquely to disturbances within the CSS system. For example, we hypothesize *N. lepida* and *P. eremicus* presence in older, closed stands of CSS, and absence in disturbed CSS dominated by grasses. Furthermore, we expect *M. Californicus* presence in moderately to highly disturbed sites with high amounts of grasses. Finally, we expect the presence of non-native *M. musculus* and *R. Rattus* only at disturbed sites near urban edges.

Predation and competition, particularly at edges, may influence the species composition of CSS small mammal communities. However we currently lack a good understanding of how, and to what extent, these two processes actually determine the presence or absence of species in specific areas. Dr. Douglas Kelt, at UC Davis, recently completed a three-year study of CSS small mammal communities. When analyzed, these data should indicate the relative strength of competition between CSS small mammal species and its role in structuring the community.

Placing Small Mammal monitoring data in an IBI.

Unlike other taxa that contain many species, we expect monitoring programs in CSS to never catch more than 16 species. This small number of species makes placing them in a typical IBI guild framework with functional, compositional and structural components somewhat difficult because false absences (not detecting a species, when it is truly there) will have a large impact on a guild with only 3 species and potential cause large amounts of error around an IBI score for small mammals. We will likely develop an alternative scoring mechanism for small mammals or combine some guilds. Our field studies will allow us to determine the effectiveness of the standard IBI scoring protocols and make necessary modifications.

Data Gaps and Recommendations for further research

Though thorough, high quality scientific research on mammals in CSS exists, the total number of studies is small and hence our current understanding of the processes structuring mammalian communities in CSS and their response to disturbance is underdeveloped. Given the strong evidence showing small mammal microhabitat preferences, differences in diets, and habitat selection, we expect small mammal species composition to track changes in vegetation, as it does in similar systems around the world. Thus, understanding how, over time scales of 10-30 years, small mammal communities track (or do not) changes in CSS vegetation as it is disturbed and recovers (i.e. succession after fire), or is invaded by non-native species, will play a key role in our ability to understand and predict responses to disturbance. Furthermore, we must also understand how both individual species and entire communities of small mammals respond to the little studied processes such as predation and interspecific competition and how

the strength of these processes varies with disturbance, in particular distance from edges. Once we better understand the interactions affecting the presence and absence of species in CSS, we will be able refine the use of small mammal monitoring data in IBI's and better understand processes affecting small mammal communities through time or across space and make more informed management decisions.

Birds

In terrestrial systems, birds have been utilized more than other taxa when relating biological metrics to habitat quality and ecological condition (Croonquist and Brooks 1991, Lynam 1996, Bradford et al. 1998, Chase 1998, Fleury 1998, Chase et al. 2000, O'Connell et al. 2000). Birds are considered good indicators of habitat quality for a number of reasons (Lovio 2000, O'Connell et al. 2000). First, many studies show rapid and large responses in bird communities to habitat fragmentation. Second, community structure is often strongly tied to habitat type. Third, birds are known to feed at different trophic levels (arthropods and other invertebrates versus fruits, nectar, and seeds) depending upon seasons and conditions.

The presence or abundance of single species, rare species, and guilds of various combinations have all been used to relate data collected on birds to habitat quality. In many cases, researchers studied the effect of habitat condition, such as fragmentation, on birds (Bolger et al. 1991, Soule et al. 1992, Lovio 2000). Other studies considered the use of particular species as indicators of total species richness and/or abundance within a habitat (Boulinier et al. 1998, Chase 1998, Nichols et al. 1998, 2000). However, only a handful of studies have attempted to connect the richness or abundance of birds with the health of a habitat. Several very encouraging studies illustrate the promise of using birds as one of the primary indicators of habitat health and are discussed below (Croonquist and Brooks 1991, Bradford et al. 1998, Canterbury et al. 2000, O'Connell et al. 2000).

Avian Response to Habitat Fragmentation and Urbanization

Given the relative ease of sampling species presence and absence, avian response to large-scale habitat fragmentation is relatively well studied when compared to other taxa. Studies done across disparate habitat types, countries, and levels of fragmentation nearly all show a decline in native birds as landscapes become more fragmented. In addition, a number of studies have begun untangling the complex causes of bird declines in fragmented landscapes. These vary from a loss of habitat, to increased predation and nest parasitism along patch edges, to changes in species composition and local or regional extinction of species (Renjifo 1999). Interestingly, in many systems studied, a subset of species responds to fragmentation. For example, fragmenting of shrubsteppe habitats by human-caused disturbance or by fire affects obligate bird species, while species not specializing on the shrubsteppe habitat specifically, fare better in fragmented landscapes (Knick and Rotenberry 1995). In another example, loss or conversion of ~94% of the original habitat to agricultural use in the wheatbelt region of S. W. Australia caused a decrease in range and/or abundance in 34 of 109 species of birds and an increase in 9 species (Saunders 1993) Our literature review strongly suggests the native bird community inhabiting CSS likely shows similar, generally negative responses to habitat fragmentation and will be a key element in the development of an IBI.

Landscape patterns of habitat fragmentation and urban/habitat edge are important determinants of birds in coastal southern California. When housing developments, and their associated roads, shopping complexes, and industrial/business sites replace CSS, fragments and edges are created. Two studies suggest an interesting response of CSS avifauna to habitat fragmentation. First, the overall diversity of

birds in suburban communities does not change relative to the diversity found in the area prior to development (Guthrie 1974, Vale 1976). However, the *composition* of species is altered as native CSS species are replaced by both non-native exotics and a set of species that persist in urban environments.

This result is caused by species-specific responses to habitat fragmentation. Bolger, et al, 1997, found unique responses of 20 of the most common bird species to fragmentation and edges in the urban landscapes of southern California. Species ranged in response from reductions to increases in abundance with increased levels of edge and/or fragmentation. In approximately half the species, abundance in natural habitats was positively correlated with position in the landscape relative to urban development. Furthermore some species were abundant, widely distributed and showed no sensitivity to landscape position or shrub habitat type. Edge/fragmentation enhanced species such as House Finch, Northern Mockingbird, Lesser Goldfinch, and Anna's Hummingbird, tended to make up the losses of edge/fragmentation sensitive species, such as Black-chinned Sparrow, Sage Sparrow, Lark Sparrow, Rufous-crowned Sparrow, and Costa's Hummingbird, creating the mechanism by which overall diversity does not change yet species composition does -- as found by (Guthrie 1974, Vale 1976).

Ten of the species were considered insensitive to edge or fragmentation effects (Bolger et al. 1997c). Some, like the Rufous-sided Towhee, Wrentit, California Quail and Bewick's Wren, are mainly restricted to the shrub habitats in the region while others, such as the Scrub Jay, Mourning Dove, California Towhee, Common Bushtit and California Gnatcatcher, are known to commonly reside and breed in residential areas, but with lower abundances near edges.

Work by Lovio (2000), also shows species-specific responses to fragmentation of CSS habitat. Lovio studied breeding bird assemblages in 36 CSS fragments ranging from 0.3 - 420 ha. Fragments selected were undisturbed, included less than 5% woody non-native vegetation (by area), and generally separated from other CSS fragments. In some cases areas of disturbed vegetation, including some CSS species, connected sites. Lovio (2000) found the bird assemblages exhibited a low response threshold to fragmentation. Area-sensitive species quickly dropped out of patches, even relatively large patches. A larger proportion of species in the assemblage appeared relatively unaffected by habitat fragmentation, though this study did not focus on detailed demographic responses. Additionally, some generalist species were enhanced on small fragments.

Bolger, et al.'s work (1991) and work done by Crooks and Soulé (1999), further elaborate on potential mechanisms causing declines in bird communities. Bolger et al (1991) showed the rapid extinction of resident birds in fragmented chaparral was a function of population density. There was a strong positive correlation between relative persistence ability and population density. Furthermore, population density was influenced by patch size. Thus, processes such as demographic stochasticity, perhaps caused by large fluctuations in reproductive output associated with rainfall, may lead to local extinctions of birds from smaller, isolated patches. We assume similar processes could occur in CSS habitat.

Crooks and Soulé (1999) found fragmented coastal sage scrub and chaparral habitats resulted in the local extinction of top mammalian predators (coyotes) as patch sizes reached minimum thresholds. This loss of top predators was correlated with an increase in avifaunal presence, and negatively correlated with the presence of mesopredators (skunks, domestic cats, possums, etc.). These results suggest mesopredators may regulate populations of birds in CSS patches. Furthermore, in highly fragmented systems, a loss of top predators may release mesopredators from competition and/or predation and greatly increase the predation pressure on birds, leading to their extirpation from patches. (see also Soule et al. 1988).

In summary, fragmentation of, and edge effects in, CSS habitat tends to cause a reduction in obligate avian species and often attracts species adapted to suburban environments and to the boundaries

of such environments. The result is often an increase, or at least no decrease, in species richness in CSS habitat, particularly near edges of fragmented patches. However, species composition changes as CSS obligate species drop out of the system and are replaced by urban and edge tolerant species.

Bird Response to CSS Fire

Fire has a substantial effect on CSS vegetation structure and therefore should impact avian communities. Unfortunately, few studies have documented the response of CSS birds to fire. Furthermore, we did not find any studies focusing on the effects of fire history or burn frequency on bird communities. Moriarty et al. (1985) showed an initial drop in species richness and abundance as a result of a fire in CSS on the campus of California State Polytechnic University. The bird community recovered quickly, with 70 – 90% of the original species richness and abundance of birds intact after one year. A study of controlled burning in chaparral by showed a decrease in chaparral birds up to four years after the fire, but an increase in grassland birds as well as a temporary increase in predator birds in both the grassland and chaparral areas (Lawrence 1966). Fire will certainly influence the composition of bird species in CSS immediately after a fire, with recovery to pre-burn status likely following vegetation successional trajectories and timescales. Given the lack of studies, we currently cannot be sure how the avian community responds to the conversion of CSS to grasslands caused by frequent fires. However, given the large differences in species composition between grassland and CSS bird communities, and the apparent obligate nature of some CSS birds, we expect frequent fires to drastically alter CSS bird communities to those associated with grasslands.

Birds as Single Species Indicators

A number of authors have investigated the use of single bird species as indicators of overall species richness, or ecosystem function at a location (Williams et al. 1996, Boulinier et al. 1998, Chase et al. 2000). Because birds are relatively well studied, we chose to review the utility of a single species approach to estimating habitat quality, or indicating some aspect of overall ecosystem health. Our review has convinced us single species approaches work poorly, especially when compared to the guild or community-based approaches we describe below. A number of studies have come to similar conclusions. For example, Chase, et al, 2000 performed a study of bird and small mammal community structure in CSS. They attempted to discover if the presence of specific birds and small mammals provides information about the overall species richness or composition in a given area. Using 37 species, they found the presences of specific species was not related to overall species richness, but in some cases, was related to the composition of species at a site. They also found the use of rare species as an indicator of species richness varied with geographical location and spatial scale. The lack of correlation between bird and mammal species richness led them to conclude that the use of a diverse suite of taxa may be a better method for conservation planning because it allows representation across the range of variation found in CSS. In addition, they found several of the bird species were sensitive to environmental changes. Birds such as the Cactus Wren, Sage Sparrow, Black-chinned Sparrow, and Rufous-crowned Sparrow were sensitive to habitat fragmentation.

The California gnatcatcher, an endangered CSS obligate, has also been studied to determine if it's presence at a location is correlated with the presence of other species. Chase, et al, 1998, conducted a study to determine if the California gnatcatcher made a good indicator of species richness. In a two year study at 17 sites in Riverside, Orange and San Diego counties they found a very small correlation between presence of the California Gnatcatcher and species richness in one year and no correlation in the following year. As a result, they found that the California Gnatcatcher is not a “particularly good indicator of species richness in coastal sage scrub”.

Fleury, et al (1998), analyzed the effectiveness of reserves designed around a single umbrella species as a means of protecting a wide variety of plants and animals. Using 40 sensitive plant and animal species in the Otay Mesa region of San Diego, they found the gnatcatcher functioned as a suitable umbrella species for less than half of the species evaluated. Four criteria were used to place each species into one of three protection categories (good, marginal, poor). The criteria used included: 1) the presence of large unfragmented blocks habitat for a species within the reserve; 2) an abundance of at least several hundred individuals in the reserve; 3) at least 3 large blocks of habitat within an area and 4) the ability of an area to allow dispersal between sites. The best-protected species had area requirements that were equal to or less than the Gnatcatcher's with similar habitat requirements; primarily habitat generalists at lower trophic levels.

Besides having no empirical support, we find the indicator species approach unsatisfactory because it does not allow informed management decisions in many management situations. Simply knowing the presence of one species is associated with the presence of another may reduce the cost of monitoring, and may allow some level of informed decision-making in areas of land prioritization or reserve design. However, having an indicator species fails to allow insights into ecological processes controlling a system. For example, it does not indicate potential causes of decline for both the indicator and the target species (which could be different) and therefore does not allow effective management decisions in situations requiring information about species responses to extrinsic factors (i.e. some form of disturbance).

In terms of understanding overall ecosystem performance or health, the indicator species approach generally fails because species have specific and often complex responses to disturbance making it difficult for any one species to adequately predict the response of the entire system. Instead, studies have shown (Chase, et al, 2000) inferences are more likely using a mixture of species representative of the community being studied.

The Use of Avian Guilds as Indicators of Habitat Quality

A number of studies document the successful application of Avian-based IBI's in understanding and predicting the relative state of an ecosystem. Indeed, research done on birds is leading the way in converting IBI methods developed for freshwater streams to terrestrial landscapes. In general, these studies convince us of the utility in a guild-based, IBI approach to understanding system responses to disturbance. Here we briefly describe 6 studies using an IBI, or IBI-like approach, then discuss how a bird-based IBI for CSS might be structured.

Brooks and Croonquist 1990, and Croonquist and Brooks 1991. Here, the authors assigned all bird and mammal species in Pennsylvania values for response guilds that reflected sensitivity to disturbances (Brooks and Croonquist 1990). High guild scores represented birds and mammals with specific habitat requirements and therefore low tolerances to disturbance. Croonquist and Brooks (1991) then sampled birds and mammals across 24 locations in two watersheds of central Pennsylvania and simultaneously estimated the degree of habitat alteration caused by human land use (agriculture, residential and commercial). Birds with high response guild scores decreased in abundance as the intensity of habitat alteration increased while the composition of mammal guilds showed no consistent pattern associated with habitat disturbance. Thus, specific guilds of birds were highly sensitive to habitat disturbance and could easily be tracked using this method. Interestingly, mammals were not impacted by disturbance. It is possible the large-scale nature of the research was too coarse to find a signal in small mammals, who respond to habitat disturbance at much smaller spatial scales than birds. This work showed that using the

available, published information, such as foraging, breeding, functional characteristics, response and other attributes of birds will provide more insight into both the functional characteristics of an avian community and their subsequent response to disturbance than will similarity coefficients or simple estimates of species diversity. In other words, development of functional guilds will provide more valuable information than Shannon diversity or other more traditional techniques.

Brooks, et al. In the late 1990's, Brooks et al. (1998) proposed a regional index of biological integrity and put forth six principles to guide the development of such an index. Their six principles are: "1) biological communities with high integrity are the desired endpoints; 2) indicators can have a biological, physical or chemical basis; 3) indicators should be tied to specific stressors that can be realistically managed; 4) linkages across geographic scales and ecosystems should be provided; 5) reference standards should be used to define target conditions; and 6) assessment protocols should be efficiently and rapidly applied." They found traditional measures of species richness and diversity were not sensitive to the stressors of management concern (invasive species, increased fragmentation, loss of habitat). At the same time, they determined it is not cost effective to collect detailed population, health or genetic data across many species of management concern across an entire region of a state.

Instead, they chose to develop response guilds and functional groups specifically to address the stressors of interest. They went on to empirically verify and document such a regional IBI in a forest riparian system by integrating four bioindicator communities (macroinvertebrates, amphibians, and avian communities and avian productivity of the Louisiana Waterthrush). These indicators are directly related to the ecological condition of associated habitats at one or more levels. Use of the Louisiana Water Thrush productivity, density and abundance acted as a bridge to the other three elements and spans the widest range of habitat scale. This bird is an area-sensitive species and as such may be used as a surrogate measure of the status of other species.

Bradford et al. Not all attempts to develop an IBI work to the level initially anticipated. For example, Bradford et al. (1998) demonstrated the application of an index of biological integrity in Great Basin rangelands in Idaho and Utah/Nevada using bird assemblages. They developed an IBI to understand and predict ecosystem responses to grazing in nonriparian rangeland. The metrics used were species richness, Shannon-Weaver diversity index, the percent of species considered shrub-obligate, total bird abundance, dominance by a single species, and relative abundance of individual species. Birds were sampled by point-count on 9 plots at 20 sites in each of 2 study areas, for a total of 360 plots. One study area consisted of sagebrush-steppe vegetation and the other area consisted of salt-desert shrub. Vegetation samples were taken at two predetermined points at each of the 9 plots. Data included location of nearest shrub, perennial grass, and perennial forb from the point, and then, species, distance from point to the main stem, canopy diameter, and maximum height. Percent cover for each of the three classes was also estimated. Unlike other attempts using birds in IBI's, the IBI here was not a strong indicator of rangeland habitat health in the intermediate stages of degradation; it could only distinguish between high and low levels of grazing.

Bradford et al., concluded the poor performance of the IBI was caused primarily by the low species diversity in the sagebrush (8 vs. 64-121 species forest studies mentioned herein) and the presence of a dominant avian species. They also suggested the relatively poor performance of bird metrics in sagebrush-steppe habitats may be caused by nonlinear relationships between vegetation characteristics and grazing practices and argued grazing affected shrub-obligate in ways other than those reflected in the vegetation variables measured. They concluded bird species assemblages can serve as indicators of the

extremes in rangeland condition in the sagebrush-steppe habitats and perhaps can act as indicators of landscape-level effects on biological integrity resulting from human derived fragmentation.

Canterbury et al. Canterbury, et al. (2000) developed and tested community-level environmental indicators for monitoring forest bird populations and their associated habitat. Birds were grouped into three assemblages based on diet and foraging strategies, nesting location, and the common habitat used (described by canopy cover and tree basal area). They sampled 197 plots in loblolly-shortleaf pine forests from Georgia to Virginia for the number of bird species and the number of deciduous and coniferous stems in seven diameter categories. Bird species were also classified into four habitat assemblages grouped as disturbance-sensitive species (mature forest (MF) assemblage), disturbance-tolerant species (Shrubland (SL) and forest-edge (FE) assemblages), and neutral species (habitat generalist assemblages). The number of species in each of the three assemblages was counted at each plot to estimate the bird-community-index. In addition, a habitat index, based on levels of disturbance, was developed as a predictor of the bird-community index. The resulting bird-community index provided a broad-based indicator of how the bird-community responded to forest disturbance. The bird-community index detected avifaunal responses to major human-created disturbances such as clearcutting and fragmentation occurring from forest conversion to agricultural and residential uses. The work showed a habitat index could be used as an independent measure of local disturbance affecting the bird community, and index values could be easily generated by relatively inexperienced field technicians.

O'Connell et al. In perhaps the best example of bird-IBI development, O'Connell, et al. (2000) categorized forest bird communities into eight mutually exclusive guilds based on behavior and responses to habitat disturbance. The response guilds were chosen to reflect different aspects of each species' breeding season life history traits. They included such items as trophic level and foraging behavior, whether resident or migratory, location of nests and nest placement, number of broods per season, and whether they are generalists or obligate forest species. Data on 112 total species was gathered from published literature and was used to assign the birds to 32 behavioral and physiological response guilds. This number was subsequently reduced to 16 guilds in 8 categories.

A reference gradient of ecological condition for 34 sites sampled in 1994 was established using best professional judgment and they were ranked in a three category scale of human disturbance. The scale went from pristine, to moderately disturbed to severely disturbed. A ranking of the same sites using only the bird community profile data was made. This information was used to develop a Bird Community Index (BCI), which was then used to rank forested areas in the central Appalachian Mountains into four broad categories; excellent, good, fair and poor. A second phase of research was then initiated to field verify the BCI. Here, birds and vegetation was sampled at 120 sites in a 168,420 square kilometer area. Samples were made at points along transects of up to 21 kilometers in length at separations of 50 to 200 m. Birds were sampled in 10 minute, unlimited-radius point counts along each transect. At each bird-sampling plot, a suite of vegetation variables was recorded. The variables included percentage herbaceous cover of graminoids, forbs, mosses, and ferns in three 5-meter radius subplots 15 meters away from the plot center. The percentage shrub cover in 3 categories was also recorded at each subplot as well as percent cover of the overstory trees.

O'Connell et al., used multiple regression analyses to discover landscape variables explaining variation in the BCI. A four-variable model consisting of landscape-level diversity (i.e. the amount of urbanization in a 1km radius around the plot), the percentage of forested land, canopy height, and slope best predicted the overall biotic integrity. The regression approach allows managers to develop a mechanistic understanding of the factors affecting the integrity of bird communities. In this case, the

amount of urbanized land at fairly large scales, most impacted bird communities, which also showed natural levels of variation also explained by the amount and age of the forest, and slope. Overall, the BCI could reliably separate four categories of habitat quality. This work indicates a relatively inexpensive annual survey of birds can now effectively categorize forest habitat quality over large areas of Pennsylvania.

Developing a bird IBI for CSS.

Because so much is known about the natural history and response to disturbance in CSS birds, and other studies have successfully developed bird-based IBI's, we were able to develop a general, "working" framework for the development of an IBI for CSS birds. We emphasize our framework for IBI develop may change. Our upcoming field studies will verify/modify this framework and collect data on other species, which we will integrate to create a multi-species IBI. The approach described here for birds will be somewhat similar for all taxa.

Table 1 represents a proposed method for developing an IBI for birds in CSS (after O'Connell et al. 2000). In this approach, biological integrity is divided into three elements: function, composition, and structure. Function refers to particular processes occurring in an ecosystem or food web. As such, species are categorized in this element based on particular roles they carrying out within the ecosystem the occupy. In this case, we categorized species based on the role they play as predators. Composition refers to the actual assemblage of species in a system, and their origin (often times native vs. or exotic) relative to an intact example of the focal ecosystem. Finally, structure refers to how species making up the IBI use different structural characteristics of an ecosystem. The three elements are chosen based on an anticipated response to anthropogenic disturbance. Disturbance may impact any or all of the three elements. For example, Frequent fire will cause changes in the prey base of many predators, impacting the functional aspect of integrity while simultaneously impacting the presence or absence of exotic species (compositional element), and removing particular structural features of an ecosystem (i.e. shrubs for nesting or insect gleaning species).

Each Integrity Element has one or more Guild Categories and associated Response guilds (Table 1). Each guild is categorized as a specialist or generalist based upon their relationship to the specific elements of the ecosystem structure, function and composition. Response guilds are groups of species that are similar when categorized trophically, by origin relative to southern California, and by nesting or foraging habitat. The Trophic category includes omnivores; birds that are feeding generalists with neither plant nor animal food comprising less than one-third of their diet; and carnivores; birds that generally eat invertebrates and vertebrates for more than one-half their diet. The carnivores are subdivided into groupings associated with where they feed in the vertical dimension. Origin levels represent the basic origin of the bird. Exotics we define as birds not normally found in CSS. This includes birds found in human-modified habitats, such as edge specialists. Resident birds use CSS on a year-round basis, not just in the breeding season, while migratory birds use CSS seasonally. Nest placement is for the two major nesting methods, ground and shrubs, which is typical of CSS. Primary habitat is based upon the breeding habitat for resident birds and the foraging habitat for all others.

The metric ratings of 5, 3, and 1 are categorical representations of deviations from a relatively undisturbed CSS habitat. The value 5 indicates a guild is within the range found in undisturbed CSS: 3 indicate some deviations, and 1 deviates strongly from an undisturbed or reference CSS habitat. The sum of all metrics taken over all the sample plots within a site provides a measure of the integrity of the habitat at that site (Brooks and Croonquist 1990, Croonquist and Brooks 1991, O'Connell et al. 2000). The

aggregate of scores from all sites gives a measure of the health of the total CSS habitat within a study area.

Table 1. Potential Avian Guild Attributes for Measuring CSS Habitat Disturbance*

Integrity Element	Guild Category	Response Guild (%)	Specialist	Generalist	Metric Rating		
					5	3	1
Functional	trophic	omnivore		x			
		carnivore, bark prober	x				
		carnivore, ground gleaner	x				
Compositional	origin	carnivore, foliage gleaner	x				
		Exotic		x			
		resident	x				
Structural	nest placement	migratory		x			
		Shrub nester	x				
	primary habitat	ground nester	x				
		CSS generalist			x		
		CSS obligate	x				

*Adapted from O’Connell et al. 2000, Croonquist & Brooks 1991, and Karr 1991. Metric scoring to be determined.

Guild data for coastal sage scrub candidate avifauna are provided in Table 2. We expect to find each species in the table within or near CSS sampling locations. Some species are more associated directly with CSS while others accommodate fragmented patches and edges, while still others accommodate suburban areas surrounding CSS, and finally, some breed in chaparral or riparian areas but forage within CSS. The birds listed on the Primary and Secondary Species list in the draft Coastal Scrub and Chaparral Conservation Plan (Lovio 2000) will certainly be on the final list.

The avifauna of Table 2 are listed again in Table 3, but assigned to one of the 11 possible guilds possible in Table 1. We consider the guild assignments in Table 3 preliminary and will verify these categories with local experts. However, based on these categories, we see that for each of the 11 possible guilds, the maximum number of species possible varies from X – X. We emphasize a site does not need to have the maximum number of species in a category to receive a 5, but instead, it must within natural levels of variation, close to the number of species found in undisturbed sites.

Table 2. Guild Data for Candidate Avifauna in Coastal Sage Scrub and Environs

Common Name	Habitat	Diet	Foraging Substrate	Foraging Technique	Nest Site	Residence
Cooper’s Hawk	Mature forest, open woodlands, wood edges, river groves	Mostly birds & small mammals.	Air, ground.	Aerial pursuit.	In tree @ 25-50 ft.	Mig. Sp – F.

Red-tailed Hawk	Open country, woodlands, prairie groves, mountains, plains, roadsides.	Mammals, some birds and reptiles.	Ground	High patrol, swoops.	In trees, cliff ledges towers, etc. @ up 120 ft.	Res.
California Quail	Broken chaparral, woodland edges, coastal sage scrub, parks, farms.	Mostly seeds & leaves, some berries, flowers, arthropods.	Ground	Ground glean.	On ground under cover.	Res.
Mourning Dove	Farms, towns, open woods, roadsides, grasslands, edges.	Seeds	Ground	Ground gleans, foliage.	In shrubs and on ground.	Res.
Greater Roadrunner	Deserts, open country with scattered brush	Arthropods, reptiles, rodents, birds.	Ground	Ground Glean	Dense bush or low tree or cactus.	Res.
Lesser Nighthawk	Arid scrub, dry grassland, desert washes	Flying arthropods.	Air	Aerial foraging.	On level ground.	Mig. Sp – F..
Common Poorwill	Dry hills, open brush, chaparral.	Flying arthropods.	Air	Hover and glean.	On ground.	Res. (?)
Black-chinned Hummingbird	Semi-arid country, river groves, chaparral, suburbs.	Nectar, arthropods.	Air	Hover and glean, hawk.	Tree or shrub.	Mig. Sp – F.
Anna's Hummingbird	Gardens, chaparral, open woods.	Nectar, arthropods.	Air	Hover and glean, hawk.	On tree branch.	Res.
Costa's Hummingbird	Deserts, washes, mesas, sage scrub, arid hillsides.	Nectar.	Air	Hover and glean.	Sparsely leaved shrub.	Mig. Sp – F.
Pacific Slope Flycatcher	Moist woods, mixed forests, shady canyons	Arthropods.	Air	Hawk, hover and glean.	In trees near ground.	Mig. Sp – F.
Black Phoebe	Shady streams, walled canyons, farmlands, towns; near water.	Arthropods.	Air	Hawk, hover and glean.	Mud nests on cliffs, bridge supports, etc.	Res.

Say's Phoebe	Scrub, canyons, ranches, open areas.	Arthropods.	Air	Hawk, hover and glean.	Rocky ledge or crevices in cliffs.	Res.
Ash-throated Flycatcher	Semi-arid country, deserts, brush, mesquite, pinyon-juniper, dry open woods.	Arthropods, some berries.	Air	Hawk, hover and glean.	Cavities.	Mig. Sp – F.
Cassin's Kingbird	Semi-open high country, pine-oak mountains, groves.	Arthropods, some berries.	Air, ground	Hawk, hover and pounce.	In large tree.	Mig. Sp – F.
Western Kingbird	Semi-open country, farms, roadsides, towns.	Arthropods, some berries.	Air, ground	Hawk, hover and pounce.	In tree, 15-30 feet.	Mig. Sp – F.
Cliff Swallow	Open to semi-open land, farms, cliffs, river bluffs, lakes.	Arthropods.	Air	Aerial foraging.	On vertical surface with overhang.	Mig. Sp – F.
Western Scrub Jay	Foothills, oak-chaparral, river woods, pinions, junipers, suburbs.	Omnivorous.	Ground	Ground glean.	Low in tree or shrub.	Res.
Bushtit	Oak scrub, chaparral, mixed woods, pinions, junipers.	Arthropods, some berries.	Foliage, bark	Foliage, bark glean.	Tree or shrub.	Res.
House Wren	Open woods, thickets, towns, gardens.	Arthropods.	Ground, foliage.	Ground, foliage glean.	Cavities.	Mig. Sp – F.
Bewick's Wren	Thickets, underbrush, chaparral, gardens.	Arthropods, some berries.	Ground, foliage.	Ground, foliage glean.	Cavities.	Res.

Table 2, Cont. Guild Data for Candidate Avifauna in Coastal Sage Scrub and Environs.

Common Name	Habitat	Diet	Foraging Substrate	Foraging Technique	Nest Site	Residence
Canyon Wren	Cliffs, canyons, rockslides, stone buildings, boulder fields.	Arthropods.	Ground, foliage.	Ground, foliage glean.	Hole or crevice in rocks.	Res.
Cactus Wren	Chollas, prickly pear @ <450 m elev.	Arthropods, some fruits and seeds.	Ground, foliage.	Ground, foliage glean.	Hole in cactus.	Res.
Wrentit	Dense chaparral, coastal sage, parks, garden shrubs.	Arthropods and berries.	Foliage, bark.	Foliage, bark glean.	Dense, low shrub.	Res.
California Gnatcatcher	Low elevation coastal sage scrub with broken canopy.	Arthropods.	Foliage, air.	Foliage glean, hover and glean.	Dense, low shrub.	Res.
Northern Mockingbird	Dense, low shrubs and open ground, towns, farms, roadsides, thickets, brushy areas.	Arthropods and berries.	Ground, foliage.	Ground, foliage glean.	Dense shrub or tree.	Res.
California Thrasher	Chaparral, foothills, valley thickets, parks, gardens.	Arthropods and berries.	Ground, foliage.	Digging, foliage glean.	Dense shrub or thicket.	Res.
Orange-crowned Warbler	Brushy clearings, chaparral, aspens, undergrowth.	Arthropods, some berries.	Foliage	Foliage glean.	On ground under overhanging vegetation.	Mig. Sp – F.
Common Yellowthroat	Swamps, marshes, wet thickets, edges.	Arthropods. few seeds.	Foliage, air.	Hover glean, bark glean.	Low on tussocks of briers, weeds, grasses.	Mig. Sp – F.

Black-headed Grosbeak	Deciduous and mixed woods.	Arthropods, seeds, and berries.	Foliage.	Foliage glean.	In tree or large shrub.	Mig. Sp – F.
California Towhee	Brushy areas, chaparral, coastal sage scrub.	Seeds and arthropods.	Ground	Ground glean.	In dense shrub.	Res.
White-crowned Sparrow	Thickets, chaparral, gardens, parks.	Arthropods and seeds.	Ground, foliage.	Ground, foliage glean, hawks.	On ground or in shrubs.	Res.
Bell's Sage Sparrow	Dry brushy foothills, semi-open, evenly spaced chaparral, coastal sage.	Seeds and arthropods.	Ground, foliage.	Ground, foliage glean.	Low shrub.	Res.
Black-chinned Sparrow	Brushy mountain slopes, open chaparral, sagebrush.	Seeds and arthropods.	Ground, foliage.	Ground, foliage glean.	Low shrub.	Mig. Sp – F.
Rufous-crowned Sparrow	Grassy or rocky slopes with sparse low bushes, chaparral, coastal sage scrub, drier, moderate to steep slopes.	Arthropods and seeds.	Ground, foliage.	Ground, foliage glean.	On ground at base of bush or grass clump.	Res.
Song Sparrow	Thickets, brush, marshes, roadsides, gardens.	Arthropods and seeds.	Ground, foliage	Ground, foliage glean.	On ground under clump of trees.	Res.
Western Meadowlark	Open fields, pastures, meadows, prairies.	Arthropods and seeds.	Ground	Ground glean.	On ground in dense grass.	Mig. Sp – F.
Brown-headed Cowbird	Farms, fields, prairies, wood edges, river groves.	Seeds and arthropods.	Ground	Ground glean.	Parasite.	Res.
Hooded Oriole	Open woods, shade trees,	Arthropods, berries,	Foliage	Foliage glean.	In palm or yucca.	Mig. Sp – F.

	palms.	nectar.				
House Finch	Cities, suburbs, farms, canyons, semi-open chaparral and brush.	Seeds, buds, berries.	Ground, foliage	Ground, foliage glean.	Wide variety of sites.	Res.
Lesser Goldfinch	Open brushy country, open woods, gardens.	Seeds, some arthropods.	Foliage	Foliage glean.	Vertical fork of tree or shrub.	Res.

Table 3. Preliminary Guild Assignments for CSS Candidate Avifauna

Common Name	Scientific Name	Probable Guild**												
		1	2	3	4	5	6	7	8	9	10	11		
Cooper's Hawk	<i>Accipiter cooperii</i>													
Red-tailed Hawk	<i>Buteo jamaicensis</i>													
California Quail	<i>Callipepla californica</i>													
Mourning Dove	<i>Zenaida macroura</i>													
Greater Roadrunner	<i>Geococcyx californianus</i>													
Lesser Nighthawk	<i>Chordeiles acutipennis</i>													
Common Poorwill	<i>Phalaenoptilus nutallii</i>													
Black-chinned Hummingbird	<i>Archilochus alexandri</i>													
Anna's Hummingbird	<i>Calypte anna</i>													
Costa's Hummingbird	<i>Calypte costae</i>													
Pacific Slope Flycatcher	<i>Empidonax difficilis</i>													
Black Phoebe	<i>Sayornis nigricans</i>													
Say's Phoebe	<i>Sayornis saya</i>													
Ash-throated Flycatcher	<i>Myiarchus cinerascens</i>													
Cassin's Kingbird	<i>Tyrannus vociferans</i>													
Western Kingbird	<i>Tyrannus verticalis</i>													
Cliff Swallow	<i>Hirundo pyrrhonota</i>													
Western Scrub Jay	<i>Aphelocoma californica</i>													
Bushtit	<i>Psaltriparus minimus</i>													
House Wren	<i>Troglodytes aedon</i>													
Bewick's Wren	<i>Thryomanes bewickii</i>													
Canyon Wren	<i>Catherpes mexicanus</i>													
Cactus Wren	<i>Campylorhynchus brunneicapillus</i>													
Wrentit	<i>Chamaea fasciata</i>													
California Gnatcatcher	<i>Polioptila californica</i>													
Northern Mockingbird	<i>Mimus polyglottos</i>													
California Thrasher	<i>Toxostoma redivivum</i>													
Orange-crowned Warbler	<i>Vermivora celata</i>													
Common Yellowthroat	<i>Geothlypis trichas</i>													

ensures that no disturbance can proceed without ultimately eliciting a response from some component of the community. 2) Amphibians and reptiles play significant ecological roles in both terrestrial and aquatic ecosystems. In terrestrial ecosystems amphibians and reptiles are the primary vertebrate predators on invertebrates as well as the primary prey base for a number of bird and mammal species. This unique trophic position makes them a major “conveyor belt” for invertebrate energy sources to predatory animals higher up the food chain (Stebbins and Cohen 1995). In addition, in many terrestrial ecosystems the herpetofaunal community makes up a large part of the vertebrate biomass. With twenty-five fold greater conversion efficiencies (the proportion of the energy consumed that is converted to new animal tissue) than birds and mammals, amphibians and reptiles populations are able to reach and sustain much greater densities than their endothermic counterparts (Pough 1980, 1983). Because this ability to sustain high population densities even when overall ecosystem resources are low, ectotherm populations are able to act as energy reserves for other predators present in the system (Stebbins and Cohen 1995). 3) Detecting herpetofaunal response to disturbance is facilitated by amphibians’ and reptiles’ tendencies to be long lived and to maintain relatively stable population densities through time.

We initially used electronic search engines and species names of the S. CA herpetofauna, as well as different forms of disturbance to initiate the searches. Unfortunately, the lack of published work on CSS herpetofauna became readily apparent. Given the lack of studies on CSS herpetofauna, we chose to make inferences concerning how the local CSS herpetofaunal community may respond to anthropogenic disturbance by looking at trends and associations documented outside S. CA. Even though this situation is not ideal, we believe some significant insights came from the literature review. However, we feel analyses of data collected by Dr. Robert Fisher will build a strong understanding of herpetofaunal responses to disturbance. We summarize our findings of herpetofaunal response to different forms of disturbance in Table 4.

Amphibians

Amphibians are sensitive to changes in landscape structure, the presence of introduced aquatic predators, and exposure to atmospheric pollutants (see references cited in Table 4). Included within the category “landscape structure” are a number of studies that document amphibian sensitivity to differences in habitat patch size and isolation, road density, and wetland permanency. Amphibians’ significant response to differences in road densities makes intuitive sense when considering amphibian biology. Because many amphibians breed, forage, and overwinter in more than one habitat, these species must make seasonal movements to and from different habitat types. When these movements occur across roads, mortality can be substantial (Fahrig et al. 1995). In Hels and Buchwald (2001), the authors estimated that roughly ten percent of a local adult population (which represents individuals from all ponds located within 250 meters of the road) of common spadefoot toads (*Pelobates fuscus*) and brown frogs (*Rana temporaria* and *Rana arvalis*) were killed annually by vehicular traffic over a five-year period.

Amphibians’ negative association with permanent wetlands is likely due to the presence of exotic predators in these systems. Permanent wetlands (perennial streams, reservoirs, and ponds) typically harbor a greater number of exotics predators and

competitors than the more ephemeral aquatic habitats (Adams 2000). Because native amphibian populations are significantly impacted by predation from introduced fish, bullfrogs, and crayfish, and by competition from introduced bullfrog larvae (Gamradt and Kats 1996; Lawler 1999; Goodsell and Kats 1999; Knapp and Matthews 2000; Kiesecker et al. 2001) it becomes obvious why there is a negative correlation between exotic predator and competitor presence and native amphibian abundance. In one clear example of this relationship, Fisher and Shaffer (1996), using broad-scale field sampling and historical analyses of museum records to quantify amphibian declines in California's Great Central Valley, documented a strong correlation between the presence of introduced predators and the absence of native amphibian species. Throughout the surveyed regions native amphibians were found more frequently at higher elevations while exotic species were found primarily in lowland areas. Evidence exists for some of the native species that suggests their present distributions represent a significant restriction to higher elevation sites from a formerly broader distribution.

Amphibians are sensitive to atmospheric pollutants present in biological systems (references cited in Stebbins and Cohen 1995; Read 1998; Marco et al. 1999; Davidson et al. 2001). Specific aspects of amphibian biology that might explain why amphibians are likely more susceptible to the presence of chemical pollution than any other vertebrate group are: (1) adult amphibians feed primarily on small invertebrate animals, making them susceptible to the effects of biomagnification of chemical contaminants in the food web; (2) amphibian larvae are susceptible to the deleterious effects of ingesting and absorbing chemicals easily transported and accumulated in aquatic mediums; (3) amphibian adults, larvae, and eggs all possess absorptive surfaces designed to be permeable to gases and liquids, the basic nature of which facilitates the absorption of chemical contaminants present in the system (Stebbins and Cohen 1995). To some researchers, the evidence of amphibian sensitivity and opportunity for exposure to chemical contaminants is so great they have suggested that chemical pollution may be the single most important factor contributing to present day worldwide amphibian declines (Stebbins and Cohen 1995).

Reptiles

Reptiles, specifically lizards, snakes, and turtles, respond to changes in land use, fire history, and landscape structure (Table 4). For lizard species, studies have documented significant effects of logging, grazing, agricultural activity, mining, human recreation and wildfire. Across this diverse suite of disturbance types, the one common element is the disturbances' shared tendency to simplify microhabitat structural and compositional diversity. Because lizard species exhibit extensive habitat specificity in natural systems, reducing a habitat's natural heterogeneity will significantly impact lizard species dependent upon the specific microhabitats eliminated by the disturbance (Pianka 1989). This microhabitat simplification eliminates critical habitat components necessary for lizards to carry out their most basic biological activities such as, thermo-regulation, predator avoidance, breeding activities, and juvenile dispersal (Hecnar and M'Closkey 1998; Germaine and Wakeling 2001). As evidence of this phenomenon, two studies conducted in Australia's Eucalyptus forests, document that changes to forest understory density, amount of woody litter, percent shrub cover, and structural heterogeneity all

contribute significantly to explaining patterns of lizard abundance and species richness (Smith et al., 1996; Brown 2001).

For snakes, a few studies suggest they are sensitive to changes in patch size, road density, human recreation, and fire history. In one study, snake diversity was shown to decrease with decreasing patch size (Kjoss and Litvaitis, 2001). In support of these findings, preliminary analyses of data collected from ongoing S. CA herpetofauna surveys identify a suite of snakes species believed to be sensitive to the effects of habitat fragmentation (Fisher and Case, 2000; R.N.F. unpublished data). In both cases the underlying causes of sensitivity are unknown, however, the incidence of road mortality in snakes has been documented by a number of studies (Rosen and Lowe 1994; Bernardino and Daryruple 1992; Klauber 1931, 1939) and increases in road density in fragmented landscapes are assumed to significantly impact a number of snake populations (Greene 1997). Because of the difficulty in estimating snake populations densities, good quantitative estimates of the impact road mortality has on local snake populations is rare. In one of the few good examples, Rosen and Lowe (1994) estimated highway mortality for snakes encountered during four years of sampling along a 45km stretch of highway in the Sonoran Desert of southern Arizona. The authors estimated the highway mortality was equivalent to the removal of five square kilometers of snake populations from the surrounding region. Additionally, the authors noted the number of snakes observed on the roads during the survey was an order of magnitude less than the number of snakes observed decades earlier by surveys conducted along the same stretch of road.

The effect of fire on local snake populations has rarely been studied (Table 4) and with mixed results. However, generally it does appear snake species favoring more open habitats are found in greater densities following fires, while more secretive species and those using ambush predatory tactics are more negatively impacted (Smith et al., 2001).

Freshwater turtle populations appear sensitive to increases in human activity in areas surrounding their streamside habitats (Garber and Burger 1995). Direct removal, road kills, handling by recreationists, increased predation, and disturbance by dogs are impacts typically associated with increased human use of surrounding upland areas. In one clear example of this relationship, the decline of North American wood turtles (*Clemmys insculpta*) in a southern New England wildlife reserve was inextricably linked to increases in the use of the reserve for human recreation (primarily fishing and hiking). Previous to the reserve being opened to human recreational use, the turtle populations in the reserve remained stable. With the commencement of human recreational use, the turtle populations in the reserve decreased steadily over a ten-year period (declining by nearly 100% in ten years). During the decline, the turtle populations experienced recruitment failure and a disproportionate loss of females from the populations (Garber and Burger, 1995).

In a similar situation, ongoing mark-recapture studies of a southwestern pond turtle (*Clemmys marmorata*) population in the Sacramento Valley of California have documented recruitment failure and a disproportionate loss of females from the population (Phil Spinks and H. Bradley Shaffer, unpublished data). As in case described above, this population appears to be sensitive to increases in human activities in the surrounding terrestrial landscape that have occurred over the last few decades. In both turtle studies, the sensitivity of female turtles to disturbance is believed to be due to their

increased vulnerability while moving through upland areas in search of appropriate sites to deposit their eggs.

Conclusion

Based on the findings of the reviewed published literature, it appears likely that measuring local CSS amphibian and reptile populations will provide insight into the types and levels of disturbance that occur within CSS ecosystems. Specifically, measuring local amphibian abundance is likely to provide researchers with insight into changes to local landscape structure. As local aquatic-breeding amphibians (i.e. western toad (*Bufo boreas*), western spadefoot toad (*Spea hammondi*), and California newt (*Taricha torosa*)) make seasonal movements through the surrounding habitat matrix, disturbances that fragment and alter this landscape will surely be reflected in changes to their local population densities.

Measuring the presence or absence of introduced aquatic predators (i.e. bullfrog (*Rana catesbiana*), mosquito fish (*Gambusia* spp.), and crayfish) will provide insight into the health and productivity of freshwater habitats for native CSS amphibian species such as the California treefrog (*Hyla cadaverina*) and the arroyo toad (*Bufo californicus*) and other native freshwater species. Likewise, severe and sudden declines or unexplained absences of even the most common CSS amphibian species may be indicative of chemical contamination occurring with the system.

Measurements of the local CSS lizard diversity and abundance, specifically the abundance of species with specific habitat requirements like western whiptail lizards (*Cnemidophorus tigris*), Gilbert's skinks (*Eumeces gilberti*), coast horned lizards (*Phrynosoma coronatum*), and banded geckos (*Coleonyx variagatus*), should provide researchers with insight into recent land use history. Lizard diversity appears to be a good metric for measuring changes to a local landscape's microhabitat structural and compositional diversity.

Monitoring changes to the local abundance of highly active wide-ranging CSS snake species, such as the coachwhip snake, *Masticophis flagellum*, will provide researchers with information concerning changes in road densities in the local region. Lastly, the abundance, sex ratio, and age structure of southwestern pond turtle populations should provide insight into the level of human activity occurring within surrounding upland habitats.

In sum, because amphibians and reptiles respond to a variety of disturbances, play critical ecological roles in both terrestrial and aquatic systems, and maintain relatively stable population densities through time, these species and species assemblages should be considered as significant candidates for inclusion in any IBI.

Data gaps and recommendations for future research

In general, the lack of published information on CSS herpetofauna makes predicting responses to disturbance difficult. As mentioned above, we strongly feel the work by Dr. Fisher will greatly increase our understanding of the CSS herpetofauna on a number of fronts. His large-scale, long-term monitoring program will allow us to address

systems. An IBI approach was adapted to terrestrial systems by focusing on arthropods at the Department of Energy Hanford site in Washington (Kimberling et al. 2001). They observed patterns in species richness along disturbance gradients with the following groups: *Diptera*, *Tachinidae* (*Diptera*), *Acarina*, *Tenebrionidae* (*Coleoptera*), parasitoids, decomposers, and predators. The relative abundance of *Eleodes* (*Tenebrionidae*) decreased and dominance increased along the gradient. Their integration of these measures into an index provided an effective measure of relative biological condition useful in evaluating site restoration. Thus previous studies suggest the use of arthropods as components of IBI's is feasible and yields productive results.

There are several difficulties inherent in using arthropods in a multi-species monitoring plan. These problems revolve around the tremendous diversity seen in arthropods and the difficulties in sorting and identifying species. Thus, this diversity requires a focused and structured approach to inventory and monitoring (New 1999). One approach to these problems involves first defining the goals and criteria necessary for a surrogate species, and performing a pilot study to validate the choice (Caro 1999). This method reduces the total number of species used in the study greatly reducing logistic and monetary constraints. Another approach useful in large-scale monitoring efforts is the use of morphospecies (Oliver 1996). This also reduces both the time and cost. However, given the potential for high endemism and rare species, we recommend full identification to species whenever possible in the initial phases of arthropod monitoring and IBI development. Based in part of the observed levels of endemism, Redak (2000) estimated the true number of threatened or endangered arthropods is at least an order of magnitude larger than the current number listed. Thus, the use of just morphospecies, without additional taxonomic separation, will likely mask the true diversity in a system.

Existing Ecological information

Arthropod surveys within coastal sage scrub (CSS) in San Diego County have occurred, with one of the more comprehensive single-site reports from Point Loma (Bruyey 1994). Unfortunately, these types of surveys generally have little information or recommendations for conservation. Recent work focused on an inventory and analysis of arthropods coupled with conservation implications and/or recommendations. Of key importance to the investigation of arthropods and CSS is identifying a response to anthropogenic changes in the landscape (fragmentation, isolation, patch size, edge effects, reserve design, overgrazing, introduced species, and fire). Here we first discuss general ecological knowledge and arthropod responses to disturbance for main taxonomic groups. We then briefly describe the role arthropods play in CSS food webs. Finally we summarize a handful of studies showing more general responses of all arthropods to fire, mechanical disturbance, and habitat fragmentation.

Ants

There are approximately 200 species of ants identified from California (Powell and Hogue 1979). Some of the major invasive species in California include the red imported fire ant (*Solenopsis invicta*), the Argentine ant (*Iridomyrmex humilis*), the Pharaoh ant (*Monomorium pharaonis*) and the pavement ant (*Tetramorium caespitum*), with a continuing expansion of their distribution within the state (Knight and Rust 1990).

Suarez et al. (1998) and Suarez et al. (2002) investigated the influence of habitat fragmentation and exotic ant invasion on native ants in CSS patches in San Diego County. Greater times since isolation, smaller fragment sizes, patches with relatively more edge, and higher percent native vegetation, were all positively correlated with the relative abundance of exotic Argentine ants. They also found the abundance of Argentine ants, fragment size, and time since isolation most strongly influenced the presence of native ants.

Beetles

The families Carabidae (predatory ground beetles) and Tenebrionidae (darkling ground beetles) contain over 800 and 400 species throughout California, respectively (Powell and Hogue 1979). The beetles lack wings, and live on the ground or in burrows, making them ideal for pitfall trapping and potential indicators of land use, fragmentation, disturbance, and change (Luff and Rushton 1988, Luff and Woiwod 1995). Also, biogeographic investigations into fragmented dune systems indicated the distribution of sand-obligate beetles was nested, resulting from isolation and fragmentation of their habitat (Rahn and Rust 2000).

The Carabid beetles are well studied across many regions of the globe with regards to their application for environmental studies and conservation (Stork 1990). Carabids and Tenebrionids may be extremely useful species in an IBI because they are reasonably understood in California, and the numerous species are both widespread and narrowly distributed. By focusing on the beetles that are flightless, with either reduced or absent wings, the impact of fragmentation and isolation should be seen through the resulting assemblage of these species found at each study site.

Although many studies indicate beetles show a response to natural and anthropogenic alterations, only a few studies have occurred on this group of arthropods in CSS (Bolger 2000). A comprehensive assessment of the beetles of San Diego County was conducted, which may provide insight into changes in species richness over the past several decades (Moore 1937). There have also been taxonomic studies on the Coleoptera of Baja, California (Horn 1894) along with an assessment of the genus *Eleodes* (Triplehorn 1996).

Butterflies/moths

Approximately 240 species of butterflies are identified in California; with well over 3,000 species of moths and many others not yet described (Powell and Hogue 1979). Some consider them suitable bio-indicators given their sensitivity to micro-climate and light level changes, and their interactions with plants (Kremen 1992). Brown (1997) discussed the use of arthropods as indicators or "early warning" systems based on their short lifecycle characteristics. In this study, butterflies responded well as indicators of environmental heterogeneity, species richness, and (natural) disturbances. Butterfly populations responded best to levels of disturbance near the natural levels, with both species and genetic diversity being lower at unnatural levels of disturbance (Brown 1997).

In his examination of the ability to use California gnatcatchers as an umbrella species for 2 butterflies and 1 moth, Rubinoff (2001) found that the presence of arthropods in CSS was dependent on fragment size, whereas gnatcatcher distribution was independent of

patch size. All 3 arthropods were specialist feeders on *Eriogonum fasciculatum*, a common CSS plant, and the arthropods' limited mobility allowed for a conservative estimate of habitat fragmentation effects on invertebrates.

Spiders

There are multiple compilations of described spider species from California, with estimates from 150 to 1,029 (see (Prentice et al. 2001) for a complete description). The most comprehensive inventories of spiders in CSS habitats in S.CA has revealed 200 species; 35 of which were new records for the county, 4 were new records for the state, 20 were undescribed, and 7 were non-native (Prentice et al. 2001). Other studies of CSS communities in San Diego County suggest non-native spiders have no significant influence on the native ground-dwelling species, with the productive lowland coastal region having both the highest number of invasive spiders and the highest number of spider species overall (Burger et al. 2001). The authors also suggest the impact of non-native spiders on the local community is largely dependant on the availability of resources. The highly linked CSS communities typically had more invasive species, but their higher diversity and resource availability increased their resistance to the deleterious impacts of exotic spiders.

The role of Arthropod in CSS food webs

Arthropods as prey items

A few studies suggest arthropods play critical roles in the CSS food web. If so, the presence or absence of arthropods with trophic links to other taxa within the food web could be a major component of an IBI. Surveys suggest arthropods may play a key role in determining habitat quality of insectivorous birds (Burger et al. 1996). For example, the habitat of California gnatcatchers was not associated with vegetation type, but rather with food abundance and diversity (Redak et al. 1996), primarily, leafhoppers (*Homoptera*) and spiders (*Araneae*), with a lesser dependence on true bugs (*Hemiptera*) and wasps, bees, and ants (*Hymenoptera*) (Burger et al. 1999). Recently, work by Ted Case and Andrew Suarez shows strong evidence for cascading impacts of argentine ants through the CSS food web (Suarez and Case 2002). Their work shows argentine ants reduce populations of native ants and result in declines in horned lizards, a predator specializing on native ants.

Pollination

There are 80 species of bees, 7 beetles, 3 butterflies, 10 muscoid flies, 4 syrphid flies, 7 bee flies, 7 wasps, and 1 sphyngid moth identified as pollinators of CSS vegetation (Moldenka 1976). There is often an important link between these arthropods and the multitude of host plants, rare and endemic plants, and endangered plants. The unique flora of southern California has led to a high number of mutualistic relationships between plants and arthropods; of the 1,200 native bee species, almost 800 are considered specialists (with 172 species identified in coastal dunes and sage, and 520 species identified in the S. CA coastal ranges, Moldenka 1976). The insect pollinators also provide a substantial economic service for California's agricultural economy. The introduction of the numerous non-native species on the pollinators is not well understood.

However, the introduced honey bee (*Apis mellifera*) is now integrated into the pollination biology of California plants, and has potentially caused the extirpation or extinction of many native species of insect pollinators (Moldenka 1976).

Responses to disturbance

Coastal Sage Scrub Fire Ecology and Arthropods

A tremendous amount of work has been done on the response of the chaparral and CSS community to fire. However, post-fire arthropod succession is less understood. In chaparral, the arthropod community follows predictable patterns related to the succession of plants, suggesting the initial phase of arthropod succession is largely influenced by the presence of plants, while later succession is influenced by the presence of other arthropods (Force 1981). Pollen/nectar feeders and predatory species can be abundant the spring after the fire, while other feeders and parasitic arthropods colonize later (Force 1982).

Depending on the type of ecosystem, fire may be influential on many species but may not be necessarily harmful (Samways 1994). In fact, some species are dependent on a regular fire regime. Some species of beetles have developed highly specialized tactile responses to smoke, aiding in their ability to locate and lay eggs in burnt trees (Mitchell and Martin 1980). There is an obvious link between fire management practices and the potential impacts on the native arthropod community.

Restoration of Coastal Sage Scrub

Restoration is becoming a common tool for conserving habitat in S. CA. Unfortunately, restoration efforts are often very diverse, and lack any measures of success beyond simple plant survival. Little attention is given to restoring the ecosystem services and functions beyond what is observed in the plant community. Longcore (1999) conducted the most comprehensive study on the use of arthropods as indicators of CSS restoration success. Here, the use of arthropods as bioindicators was employed as an additional measure of the success for restored CSS. Overall arthropod diversity and evenness were significantly lower at restored sites than undisturbed control sites, even though the vegetation was similar. Non-native species of arthropods (Argentine ants, European Earwigs, and Sowbugs) were more commonly associated with restored sites. Longcore (1999) concluded arthropods are important indicators of restoration success, and should be included future monitoring efforts.

Other studies have shown physical disturbance to the landscape can produce declines in family richness (Kimberling et al. 2001). Both studies indicate we should expect both different responses from specific species/groups of arthropods to disturbance and potential complex ramifications through the CSS food web (Longcore 1999, Kimberling and Karr 2001). Therefore, we should be able to detect a response to disturbance by the invertebrate community, and use it in the development of an IBI.

Habitat Fragmentation

The effects of habitat fragmentation on the overall arthropod diversity and abundance have been observed in the CSS community (Suarez 1998, Bolger 2000).

Arthropod diversity and abundance were positively correlated to fragment size, and negatively to time since isolation; spider diversity and abundance were enhanced by fragmentation, exotic ants reduced the richness and abundance of other arthropods; spiders and carabid beetles increased in abundance in older fragments, being positively correlated with the abundance of Argentine ants and exotic Isopods, Dermaptera, and Blattaria (Bolger et al. 2000).

Conclusion

The characteristics of arthropods make them ideal candidates for inclusion in an IBI. Within CSS, studies have found quantifiable responses to fire, mechanical disturbance and habitat fragmentation. Likewise, our ability to rapidly and efficiently assess the arthropod community in CSS provides a cost effective tool for evaluation. A host of other studies in different systems indicates substantial support for the use of arthropods in assessing ecological risk. Thus, we feel the arthropods will make up a key component of a terrestrial IBI for CSS.

Data gaps and recommendations for future research

Given the sparseness of studies and the complexity of the taxa, there are numerous directions for future research in S.CA arthropods. Significant data gaps exist in the overall distribution of many of the key taxa. The natural history of these organisms is also not well understood. We should also focus future efforts on the effects of fire on the arthropod community, how diverse interactions result in food web dynamics, the effects of pesticide use and exotic introductions on native taxa, and the role insect pollinators play in the CSS plant community. Regardless of the project's focus, all future research should incorporate measurement of the abiotic factors at study sites. Many of the previously mentioned studies show the arthropod community assemblage is significantly influenced by the local abiotic conditions. We therefore recommend a minimal amount of abiotic data be gathered in addition to an biological data (e.g. temperature, relative humidity, and precipitation); along with a characterization of the site (e.g. elevation, distance from coast, vegetation type, soil type, etc.).

Table 5. Threatened or endangered species listed in California.

Status (Threatened = T; Endangered = E)	Common Name	Scientific Name
T	Beetle, delta green ground	<i>Elaphrus viridis</i>
E	Beetle, Mount Hermon June	<i>Polyphylla barbata</i>
T	Beetle, valley elderberry longhorn	<i>Desmocerus californicus dimorphus</i>
T	Butterfly, bay checkerspot	<i>Euphydryas editha bayensis</i>
E	Butterfly, Behren's silverspot	<i>Speyeria zerene behrensii</i>
E	Butterfly, callippe silverspot	<i>Speyeria callippe callippe</i>

E	Butterfly, El Segundo blue	<i>Euphilotes battoides allyni</i>
E	Butterfly, Lange's metalmark	<i>Apodemia mormo langei</i>
E	Butterfly, lotis blue	<i>Lycaeides argyrognomon lotis</i>
E	Butterfly, mission blue	<i>Icaricia icarioides missionensis</i>
E	Butterfly, Myrtle's silverspot	<i>Speyeria zerene myrtleae</i>
T	Butterfly, Oregon silverspot	<i>Speyeria zerene hippolyta</i>
E	Butterfly, Palos Verdes blue	<i>Glaucopsyche lygdamus palosverdesensis</i>
E	Butterfly, Quino checkerspot	<i>Euphydryas editha quino</i>
E	Butterfly, San Bruno elfin	<i>Callophrys mossii bayensis</i>
E	Butterfly, Smith's blue	<i>Euphilotes enoptes smithi</i>
E	Fly, Delhi Sands flower-loving	<i>Rhaphiomidas terminatus abdominalis</i>
E	Grasshopper, Zayante band-winged	<i>Trimerotropis infantilis</i>
T	Moth, Kern primrose sphinx	<i>Euproserpinus euterpe</i>
E	Skipper, Laguna Mountains	<i>Pyrgus ruralis lagunae</i>
E	Tiger beetle, Ohlone	<i>Cicindela ohlone</i>

Conclusions

The final task of this deliverable was to make recommendations regarding further research necessary to understand how CSS responds to disturbance. We have largely covered specific gaps in knowledge within the individual taxa reviews. We conclude with three main points.

1) Focused studies on taxa responses to disturbance are needed. Across all taxa, only a few studies were specifically designed to address CSS responses to disturbance. Furthermore, the majority of studies cover specific forms of disturbance for specific taxa. For example, plant ecologists have largely focused on CSS vegetation response following fire or to decreased fire interval, while studies of other impacts, such as invasive species are rare. Studies of animals have focused primarily on responses to patch size and isolation. These animal studies have primarily focused on the relatively small, urban canyons within the city of San Diego. Unfortunately, many of the NCCP reserves exist in much larger areas of land, making it difficult to extrapolate the findings of these smaller scale studies to the larger reserves. In conclusion, there are massive gaps in our knowledge of how many species, functional groups, and taxa, within CSS respond to disturbance.

Two general approaches could be used to begin filling in these data gaps. First, monitoring plans required by the NCCP should be designed to sample CSS taxa across gradients in various forms of disturbance. By randomly placing monitoring sites, or by focusing on “good” patches of CSS, reserve managers are missing an opportunity to gain critical information regarding responses to disturbance. By using “natural” variation in the level of disturbance, USFWS and CDFG managers can begin to infer how taxa respond to anthropogenic impacts. Experimental designs may include monitoring at various distances from edges, or across different types of edges (i.e. urban vs,

agriculture); sampling in sites with different fire histories ranging from high to low fire average fire return intervals.

The second approach is to experimentally manipulate existing stands of CSS. Small amounts of CSS could be “sacrificed” to understand how particular disturbance regimes impact (or not) various CSS species or taxa. The degree of impact to CSS could be substantial if studies such as mechanical disturbance are implemented. We recommend experiments with less direct impacts, designed to study potentially subtle responses. For example, placing artificial lighting in an area to simulate light pollution would likely have little long-term impacts on a site, but could indicate those particular species highly sensitive to changes in light regimes.

2) Single species approaches to understanding community-level responses are not well supported. Our review indicates the use of indicator species in CSS has not been successful and given the wide array of ecological specialization and response to disturbance already seen by species in CSS, we are doubtful of any future successes. Unfortunately, studies attempting to show relationships between single species and higher order metrics, such as overall species richness, have generally failed to find relationships.

Furthermore, our review indicates the remaining CSS is likely facing sustained levels of a variety of anthropogenic and biological (invasive species) impacts. As such, we feel relatively high levels of focused, active management will likely be required to sustain current levels of biodiversity in CSS. As such, we are not convinced an indicator species approach is useful for directing the types of management we envision as critical. For example, suppose an indicator species is discovered in the next two years. What should a manager do if the species is absent, or disappears at specific locations in their reserve? Without a more complete understanding of CSS responses to disturbance and the effectiveness of potential management practices, at this point in time, a reserve manager would be hard pressed to invoke new policies. We emphasize the need to first understand the ecological responses, which we currently do not yet know, then focus on finding efficient forms of monitoring.

3) A prioritization of possible management actions and simultaneous studies regarding their utility and cost-effectiveness is critical. As indicated in #2, our review convinces us active management will more likely be the rule than self-sustaining, easily managed systems. Furthermore, the number of economically, and logistically possible management actions possible at a site is relatively short. Broad categories in a list would likely include: altering fire regimes, minimizing human intrusion, restoring habitat, and controlling invasive species. As in #1 above, we need studies showing how effective different forms of these four broad management actions may be. In most cases, well-designed experiments will provide the greatest amount of information for the least cost. A current example is the CSS restoration study occurring on Metropolitan Water District at the Shipley reserve. This experiment is simultaneously testing the effectiveness of burning, grazing, and water addition on invasive grass control and CSS restoration. In addition to specific experiments, monitoring programs should be integrated into investigations of potential management actions by placing monitoring sites within manipulated and un-manipulated management units. To begin the process of selecting

and studying the utility of management actions, we recommend a symposium focused specifically on 1) developing a “laundry list” of possible management actions and 2) synthesizing the current state of knowledge regarding which strategies work.

Annotated Bibliography

Vegetation

Alberts, A. C., A. D. Richman, D. Tran, R. Sauvajot, C. McCalvin, and D. Bolger (1993). “Effects of habitat fragmentation on native and exotic plants in southern California coastal scrub. Interface Between Ecology and Land Development in California.” J. E. Keeley. Los Angeles, Southern California Academy of Sciences: 103-110.

Surveys for native and exotic plant species (excluding grasses) in 25 canyon fragments with CSS vegetation showed that fragment size, perimeter, age, and presence of supplemental water were important predictors of native and exotic species richness. Larger and more recently isolated patches had greater numbers of native plant species while older fragments had higher numbers of exotic plant species. Interestingly, neither ruderal nor ornamental species richness, nor presence of *Schinus* spp., a strong resource competitor, were strong predictors of native species richness. According to the authors, this suggests competitive interactions may be less important than habitat alterations in explaining the shifts in species composition over time. However, the exclusion of grasses in the analyses may be important to this hypothesis. Exotic annual grasses are often abundant, widespread invaders, and may be compete more effectively with natives on a larger scale.

Allen, E. B., P. E. Padgett, A. Bytnerowicz, and R. B. Minnich (1997). “Nitrogen deposition effects on coastal sage vegetation of southern California.” Proceedings of the International Symposium on Air Pollution and Climate Change Effects on Forest Ecosystems, February 5-9, 1996, Riverside, California. A. Bytnerowicz, M. J. Arbaugh and S. L. Schilling. Berkeley, CA, Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture.

The study reports on a series of greenhouse and field experiments designed to examine causal factors of reduced native shrub cover and increased exotic annual grass cover in CSS in areas with high nitrogen (N) deposition. Experiments showed that both exotic grasses and native shrubs respond similarly to N fertilization, and that competition with exotic grasses appeared to be more important to survival of *Artemisia californica* seedlings than N fertilization. The authors also argue that ozone might not be as important in CSS decline as previously thought since spring concentrations of ozone are low, stomates are closed, and many leaves have senesced on CSS shrubs during peak summer concentrations of ozone.

Axelrod, D. I. (1978). "The origin of coastal sage vegetation, Alta and Baja California." American Journal of Botany **65**(10): 1117-1131.

In describing a variety of factors that have resulted in the current distribution of CSS, Axelrod argues that CSS currently occupies former grassland over much of its range, in part, as a result of repeated burning and overgrazing, both of which result in erosion of the soils that support grasses. Axelrod also argues that fire promotes sage over grasses due to turf removal and exposure of bare ground. This view is also expressed by Dodge (1975), but other authors have argued against it (e.g. Oberbauer 1978).

Callaway, R. M. and F. W. Davis (1993). "Vegetation dynamics, fire, and the physical environment in coastal central California." *Ecology* **74**(5): 1567-1578.

The authors compared vegetation communities identified on aerial photographs between 1947 and 1989. "Plots" were established on the photographs and transition rates among vegetation communities were measured for plots that were: a) undisturbed (no fire or grazing); b) grazed only between 1947 and 1989; c) burned at least once, but not grazed, during the study period; and d) both burned and grazed. The major findings with respect to CSS were a) the transition from CSS to grassland increased in burned plots (compared to undisturbed plots), and the transition from grassland to CSS decreased; b) grazing also decreased the transition from grassland to CSS and decreased the transition from CSS to chaparral or oak woodland; and c) there was no transition from CSS to grassland in grazed plots.

Coblentz, B. E. (1980). "Effects of feral goats on the Santa Catalina Island ecosystem." The California islands: proceedings of a multidisciplinary symposium. D. M. Power. Santa Barbara, CA, Santa Barbara Museum of Natural History: 167-170.

This paper provides a brief, historical overview of goats as domestic grazers, their foraging behaviors, and introduction to Santa Catalina Island. Some results, but little methodology, from a study comparing vegetation in goat inhabited areas and goat free areas on Santa Catalina Island are briefly presented. Grazing by feral goats resulted in reduction of total vegetative cover by about 40% and elimination of "sagebrush" in goat inhabited areas compared to areas free of goats for 15 years. However, goats are described as potentially more destructive than other grazers due to their foraging habits, including consumption of coarser and more bitter vegetative material.

Davis, C. M. (1994). "Succession in California shrub communities following mechanical anthropogenic disturbance." M.S. Thesis. Biology. San Diego, CA, San Diego State University: 65.

Davis examined a chronosequence of undisturbed and mechanically disturbed CSS and chaparral sites. Results showed that exotic herbs dominated disturbed sites with only scattered shrubs of *Eriogonum fasciculatum* and *Baccharis sarothroides*. Shrub cover and richness of native forbs were greater on undisturbed plots, and exotic species richness and percent cover of exotic forbs were greater on disturbed plots. Davis found no evidence that exotic species abundance was decreasing or that shrub cover was increasing on disturbed sites even more than 70 years after disturbance.

Dodge, J. M. (1975). "Vegetational changes associated with land use history in San Diego County." Ph.D. Dissertation. Geography. Riverside, CA, University of California, Riverside: 216.

Dodge examined various types of historical records related to land use and vegetation change since pre-European settlement. He argued that introduction of livestock by Spanish settlers and subsequent heavy grazing allowed shrub invasion into historic grasslands via reduced competition from grasses and reduced the fire frequency due to lack of fuels. He also argued that fires in shrublands were historically more frequent due to both natural and intentional ignitions and that current fire suppression policies have led to a buildup of fuel resulting in larger and more intense fires in shrublands. Both lines of argument have other supporters (e.g. Axelrod 1978 and Minnich 1983) and critics (e.g. Oberbauer 1978 and Keeley 1999).

Egerton-Warburton, L. M. and E. B. Allen (2000). "Shifts in arbuscular mycorrhizal communities along an anthropogenic nitrogen deposition gradient." *Ecological Applications* **10**(2): 484-496.

This study examined changes in arbuscular mycorrhizal (AM) species diversity and abundance in CSS along a gradient of nitrogen deposition and in plots fertilized with nitrogen. Increased nitrogen was associated with decreased AM species richness and Shannon-Wiener diversity, the displacement of *Scutellospora* spp. and *Gigaspora* spp., and proliferation of *Glomus* spp. in both fertilized plots and along the pre-existing gradient. The authors suggest that abundance of *Glomus* spp. and lack of larger-spored *Scutellospora* and *Gigaspora* spp. may be early indicators of "eutrophication" since shifts in species composition occurred within less than 2 years on fertilized plots. The authors also argue that the changes in AM community composition in nitrogen enriched areas may be, in part, responsible for declines in CSS shrub cover and replacement by exotic annuals because the *Glomus* species which flourish in nitrogen enriched sites have a more parasitic relationship with host shrubs than other AM species.

Eliason, S. A. and E. B. Allen (1997). "Exotic grass competition in suppressing native shrubland re-establishment." *Restoration Ecology* **5**(3): 245-255.

This study examined the effect of annual grasses on germination and survival of *Artemisia californica* by manipulating grass density around seedlings. Germination, first season growth, and seedling survival were all negatively related to annual grass density. Depletion of soil water by annual grasses was proposed as the most likely causal factor contributing to reduced seedling growth. Interestingly, the negative effects of annual grasses on shrub seedling growth did not persist into the second growing season, suggesting shrubs are only vulnerable at the youngest stages but are successful competitors once past the seedling phase. As noted by the authors, the results indicate that exotic annual grasses may interfere with post-disturbance recovery of CSS by reducing shrub seedling recruitment.

Escofet, A. and I. Espejel (1999). "Conservation and management-oriented ecological research in the coastal zone of Baja California, Mexico." *Journal of Coastal Conservation* **5**(1): 43-50.

This study examined the effects of fragmentation on coastal sage-succulent scrub vegetation and the effects of urbanization on Snowy Plovers. With respect to succulent scrub fragmentation, the authors concluded that "83 % of the original species assemblage persisted in the fragments". However, this was only the case when both new and old fragments are considered together. Though not discussed by the authors, a comparison of old fragments with the control area (Table 2) shows that only ~52% of species are

maintained in old fragments. In addition, the authors' claim that "invasion by opportunistic exotic species may not progress beyond certain limits" is not supported by evidence other than no apparent increase in *Bromus madritensis* cover in old fragments compared to new. Instead, the data shown suggest that fragmentation of coastal sage-succulent scrub is associated with species loss over time and an increase in exotic species cover compared to unfragmented areas.

Freudenberger, D. O., B. E. Fish, and J. E. Keeley (1987). "Distribution and stability of grasslands in the Los Angeles basin." *Bulletin Southern California Academy of Sciences* **86**(1): 13-26.

This study utilized aerial photographs from 1928 to 1936 and 1980 to evaluate changes in the distribution of grassland in the Los Angeles Basin and examine transitions in vegetation types. While transitions between shrubland and grassland were noted in both directions, the overall trend was an increase in grassland and a decrease in CSS and chaparral, particularly in areas disturbed by frequent fires and grazing. In addition, field sampling of transition zones between CSS and grassland revealed an absence of shrub seedlings where grazing was present. These results argue against the theory of grazing-facilitated CSS expansion into grassland, and suggest, instead, that grazing, might promote conversion of CSS to grassland. Interestingly, transitions from grassland to CSS were often noted in heavily grazed areas where some sort of grazing exclusion was implemented suggesting that CSS may be capable of recover from grazing over time.

Giessow, J. H. (1997). "Effects of fire frequency and proximity to firebreak on the distribution and abundance of non-native herbs in coastal sage scrub." M.S. Thesis. Biology. San Diego, CA, San Diego State University: 76.

In this study plots were establish in CSS either adjacent to or more than 500 meters away from firebreaks. Plots were also grouped according to time since last fire, fire interval (time between 2 most recent fires), and fire frequency (average fire return time). Biomass and richness of exotic forbs and grasses, richness of native forbs and grasses, and cover of native shrub were measured. Essentially, fire frequency was the most important variable related to abundance of non-native forbs and grasses. Proximity to firebreak did not affect the abundance of non-natives other than immediately adjacent to the firebreak (within 10m). However, some exotic species were found only near firebreaks suggesting that disturbance associated with firebreak construction may facilitate invasion by some exotic species.

Haidinger, T. L. and J. E. Keeley (1993). "Role of high fire frequency in destruction of mixed chaparral." *Madroño* **40**(3): 141-147.

This study compared mixed chaparral sites dominated by *Adenostoma fasciculatum* (chamise) and *Salvia mellifera* (black sage, a common CSS species) that had burned 1, 2, or 3 times over a 6 year period. Results showed resprouting and/or seedling recruitment of shrubs decreased as fire frequency increased. Density of exotic species, particularly *Brassica nigra* (black mustard) and *Bromus* spp., was highest on sites burned 3 times and lowest on recently burned sites that had not burned in 20+ years. Overall percent composition of natives decreased with increasing fire frequency, although native fire-following annuals did not appear to be negatively affected by frequent fire. The study supports the relationship between high fire frequency and increases in exotic species

abundance in California shrublands and the potential role of high fire frequency in conversion of shrubland to landscapes dominated by exotic annuals.

Keeley, J. E. and S. C. Keeley (1984). "Post-fire recovery of California coastal sage scrub." *The American Midland Naturalist* **111**(1): 105-117.

This study demonstrates some general aspects of post-fire recovery in CSS, as well as variability in recovery patterns due to environmental factors. The authors sampled two burned CSS sites in the first and second growing seasons after fire. Most shrubs resprouted in the first post-fire growing season though there were some differences due to slope aspect (e.g. north facing slopes had fewer resprouts). Seedlings were scarce in the first year but increased in the second year due to flowering of resprouting shrubs, though some species flowered more abundantly. Herbaceous cover was highest in the first growing season and declined thereafter, but composition changed between years, and composition varied among different slope aspects. In the second year, and on some sites the first year, the herbaceous component was dominated by resprouting perennial herbs, rather than annuals, which differs from chaparral where annuals dominate the post-fire herbaceous cover.

Lambrinos, J. G. (2000). "The impact of the invasive alien grass *Cortaderia jubata* (Lemoine) Stapf on an endangered mediterranean-type shrubland in California." *Diversity and Distributions* **6**(5): 217-231.

This study compared plots of uninvaded and relatively undisturbed maritime chaparral with matched plots that had been invaded and were dominated by *Cortaderia jubata* (pampas grass). Plant species composition and structure were significantly different in invaded areas. Shrub cover and native species richness were lower and perennial herb cover was higher in *C. jubata* plots. In addition arthropod abundance and diversity and rodent activity was lower in *C. jubata* plots. The author suggests changes in the vegetation composition and structure are likely related to the changes in arthropod and small mammal diversity and abundance after invasion. This study provides an example of the potential effects of exotic plant invasions on food web dynamics.

Malanson, G. P. and W. E. Westman (1985). "Postfire succession in Californian coastal sage scrub: the role of continual basal sprouting." *American Midland Naturalist* **113**(2): 309-318.

The authors investigated the ability of *Artemisia californica*, *Salvia leucophylla*, and *Salvia mellifera* to produce new basal sprouts from root crowns in the absence of disturbance or in response to disturbance other than fire. Findings indicate these shrubs can continually produce basal sprouts in the absence of fire and resprout in response to disturbance other than fire (clipping). It was estimated that branches of shrubs persist around 30 years. *A. californica* and *S. mellifera* were also found to recruit by seeding between fires. The authors argue that continual basal sprouting is important in maintenance of the shrub community and may play a role in determining long-term species composition. This study is relevant to the question of how frequent fire should be in CSS, and supports the hypothesis that CSS can tolerate long fire free intervals.

McBride, J. R. (1974). "Plant succession in the Berkeley Hills, California." *Madroño* **22**(7): 317-380.

This study examined vegetation succession following the elimination of heavy livestock grazing. Replacement of grassland by *Baccharis pilularis* (chaparral broom, coyote brush) scrub after elimination of grazing was reported, but actively grazed grasslands were relatively free of shrubs, and cattle were observed foraging on shrub seedlings. These results are relevant to the potential negative effects of grazing on CSS and lend support to criticism of the theory that grazing facilitates CSS expansion.

Mensing, S. A. (1998). "560 years of vegetation change in the region of Santa Barbara, California." *Madroño* **45**(1): 1-11.

Pollen evidence from two sites in the Santa Barbara area are used to investigate vegetation changes following European settlement in California. This analysis of the pollen record is relevant to assumptions about how historical disturbances, such as grazing, have affected CSS and chaparral. Pollen cores from 2 sites near Santa Barbara, the Santa Barbara channel and Zaca Lake, indicate only a slight (2-3%) increase in *Artemisia* pollen since 1820, providing only weak evidence for any increases in CSS since European settlement. Pollen percentages of chaparral-type species show no consistent trends, thus providing little support for theories suggesting that grazing has promoted an increase in shrubland since European settlement.

Minnich, R. A. (1982). "Grazing, fire, and the management of vegetation on Santa Catalina Island, California." *Dynamics and Management of Mediterranean-Type Ecosystems*. C. E. Conrad and W. C. Oechel. Berkeley, CA, Pacific Southwest Forest and Range Experiment Station: 444-449.

This study compared historical photographs with present-day vegetation on Santa Catalina Island and showed areas previously covered by chaparral and CSS were covered mainly by grass and large, scattered shrubs after decades of grazing. This finding is relevant to the impacts of grazing on CSS vegetation and the potential role of grazing in vegetation-type conversion. The paper also discusses the potential impact of heavy grazing on fire regimes and the implications for future management.

Minnich, R. A. and R. J. Dezzani (1998). "Historical decline of coastal sage scrub in the Riverside-Perris Plain, California." *Western Birds* **29**(4): 366-391.

Relocation and resampling of vegetation inventory plots established in 1932 revealed large declines in cover of CSS shrub species in the Riverside-Perris Plain and replacement of CSS stands by exotic annual grasses. *Encelia farinosa* appeared to be the most resilient CSS shrub species to replacement by exotics, as it declined in cover very little and showed increases on some sites. Minnich suggests the relatively low replacement of *Encelia* may be due to its predominance on "xeric southern exposures where herbaceous cover is limited" (p. 383). The potential roles of fire, grazing, and air pollution in the spread of exotics are discussed in general, but, other than fire occurrence, complete disturbance histories were not provided for the study sites.

Narog, M. G., J. L. Beyers, T. E. Paysen, B. M. Corcoran (2000). "Recovery of coastal sage shrub species after mechanical disturbance." *2nd Interface Between Ecology and Land Development in California*. J. E. Keeley, M. Baer-Keeley and C. J. Fotheringham, U.S. Geological Survey: 263-269.

Sampling of an artificially created stand of CSS shrub species 4 years after mechanical removal of all aboveground vegetation showed that *Artemisia californica*, *Eriogonum fasciculatum*, *Encelia farinosa*, *Salvia mellifera*, and *Salvia apiana* all reestablished from both sprouting and seeding. The original stand consisted of equal numbers of all 5 species. Four years after mechanical disturbance *Artemisia californica* and *Eriogonum fasciculatum* were the most abundant shrub species primarily due to seedling establishment, suggesting these species may be better able to invade openings created by disturbance and may be more resilient to disturbance than the other 3 species studied. Interestingly, non-native grasses and forbs initially dominated the disturbed site, but shrub seedling establishment was still successful. Ability to generalize from these results is limited, however, due in part to supplemental watering of the site for "a few months" after the disturbance (p. 265).

O'Leary, J. F. (1988). "Habitat differentiation among herbs in postburn Californian chaparral and coastal sage scrub." *American Midland Naturalist* **120**(1): 41-49.

This study examined habitat differentiation of herbs in post-burn CSS. Results showed that in the first post-burn spring herbs accounted for 84% of the total plant cover, and that many dominant herb species exhibited slope and substrate preferences. While similar studies and analyses have been undertaken with the shrub component of CSS, this study demonstrates the variability of the post-fire herbaceous component according to different habitat variables.

O'Leary, J. F. (1990). "California coastal sage scrub: general characteristics and considerations for biological conservation." *Endangered plant communities of southern California*. A. A. Schoenherr. Claremont, CA, Southern California Botanists: 24-41.

The paper is a good review of the major characteristics of coastal sage scrub with a focus on those features most relevant to conservation considerations. There is also some discussion of CSS response to disturbance, particularly fire and air pollution.

O'Leary, J. F. (1990). "Post-fire diversity patterns in two subassociations of Californian coastal sage scrub." *Journal of Vegetation Science* **1**(2): 173-180.

This study examined recovery of CSS in 2 aspect-related subassociations and at two scales (1m² and 625m²) for five years following a fire. Results showed differences related to slope aspect in post-fire species richness and equitability. For example, north facing slopes showed higher richness and equitability 10 & 28 years post-fire at the 1m² scale (but not at the larger 625m² scale) compared to south-facing slopes, and these variables were more or less stable on north facing slopes over the study period at the 1m² scale (but not at the 625m² scale). South-facing slopes showed decreasing richness and equitability within 4 years following fire. This study demonstrates some of the variability in post-fire recovery patterns in CSS.

O'Leary, J. F. (1995). "Coastal sage scrub: Threats and current status." *Fremontia* **23**(4): 27-31.

The paper is a briefly reviews threats to coastal sage scrub from disturbances such as grazing, fragmentation, and exotic species invasion. The review is similar to O'Leary's 1990 review, but includes some new information and is written for a mixed scientific and "lay" audience.

O'Leary, J. F. and W. E. Westman (1988). "Regional disturbance effects on herb succession patterns in coastal sage scrub." *Journal of Biogeography* **15**: 775-786.

This study examined differences in post-burn recovery of CSS between coastal and inland sites. The authors found that shrubs & subshrubs recovered foliar cover to near pre-burn levels within 5 years post-fire in coastal sites. Herb cover reached pre-burn levels within the first year after fire and remained high for 5 years thereafter. Inland sites recovered more slowly due to dominance by weak resprouting subshrubs before the fire and high mortality of strong resprouters due to high fire intensity. Higher herbaceous cover at inland sites after fire may have also impeded seedling establishment of subshrubs. Compared to coastal sites, inland herb composition contained a higher percentage of exotic species before and after fire. Differences between coastal and inland sites in pre-burn species composition and post-burn succession were hypothesized to be due to differences in grazing history and pollution loads, both of which are higher at inland sites.

Padgett, P. E. and E. B. Allen (1999). "Differential responses to nitrogen fertilization in native shrubs and exotic annuals common to Mediterranean coastal sage scrub of California." *Plant Ecology* **144**(1): 93-101.

This study continued the investigations reported on by Allen, et al. (1996), and found that *Eriogonum fasciculatum* and *Encelia farinosa*, as well as *Artemisia californica*, responded to high levels of nitrogen (N) with increased growth well beyond the N levels that produced increased growth in exotic annuals except for *Hirschfeldia incana*. Thus, greater "nitrophilly" of exotics compared to native shrubs does not appear to be causal mechanism explaining increased cover of exotics and decreased cover of native shrubs in CSS occurring in regions with high atmospheric pollution levels. Competition for light and water are cited as potentially more important, but it is also hypothesized that the response by shrubs to increased N represents an inability to restrict growth and prepare for dormancy under high N conditions, which may lead to increased mortality of shrubs.

Preston, K. P. (1988). "Effects of sulfur dioxide pollution on a Californian coastal sage scrub community." *Environmental Pollution* **51**(3): 179-196.

Preston examined the effects of sulphur dioxide (SO₂) emissions on stands of CSS, focusing particularly on the physiologic and growth changes in *Salvia mellifera*. Sites along an SO₂ pollution gradient downwind of an oil refinery were compared with relatively pollution-free, upwind sites. Negative physiologic changes were noted for *S. mellifera* in downwind sites compared to upwind sites. Decreased perennial cover and increased exotic species cover were also found in the most polluted sites close to and downwind of the refinery compared to sites further away or upwind. Pollution levels along the downwind gradient sites or at upwind control sites were not actually measured, but the distances to maximum pollution levels were assumed to be consistent with calculations performed by other authors. Thus, it is not clear how different far-away downwind sites really were from upwind sites with respect to pollution levels.

Van Vuren, D. and B. E. Coblenz (1987). "Some ecological effects of feral sheep on Santa Cruz Island, California, USA." *Biological Conservation* **41**(4): 253-268.

The authors studied the effects of grazing by feral sheep on plant communities on Santa Cruz Island. Sheep were found to be generalist herbivores, grazing and browsing a

variety of plants, including some considered unpalatable, according to availability. Sheep browsing resulted in partial defoliation of large chaparral shrubs, complete defoliation of low growing shrubs such as those found in CSS, and reduced shrub regeneration. In addition, sheep grazing was associated with reduced herbaceous cover and increased bare ground. The study illustrates some potential effects of intense grazing on CSS.

Westman, W. E. (1976). "Vegetation conversion for fire control in Los Angeles." *Urban Ecology* **2**: 119-137.

This study evaluated the success, after 13 years, of various treatments designed to convert CSS into less flammable vegetation. Treatments varied across the experimental area but included clearing of vegetation followed by planting of exotic shrub and/or herb species and, in some cases, irrigation and fertilization. Results suggest that conversion was generally unsuccessful, as native species re-established on all sites, though to different degrees among various treatments. In general, irrigation and fertilization enhanced the ability of natives to re-establish and increase in cover on sites. Of native shrubs, *Artemisia californica* was found to be the most aggressive invader via seedling establishment and was the dominant shrub at the time of the study despite low representation in the stand prior to conversion treatments. Westman stated that occasional weeding of natives might result in more successful conversion, suggesting that frequent disturbance is necessary to achieve vegetation type conversion of CSS.

Westman, W. E. (1979). "Oxidant effects on Californian coastal sage scrub." *Science* **205**(7): 1001-1003.

The study utilized data collected from 67 southern California CSS sites to determine potential causes of reduced cover of native species in certain sites. Analyses suggested that, of the 43 variables examined, increased oxidant levels were the most likely causal factor relating to decreased cover of natives. Increased oxidant levels were also found to be related to decreased species richness and equitability. This study provides relatively weak correlational evidence for a link between high pollution levels and CSS decline.

Westman, W. E. (1981). "Diversity relations and succession in Californian coastal sage scrub." *Ecology* **62**(1): 170-184.

Westman utilized data collected from CSS at several sites in southern California to examine the floristic diversity of CSS and its relationship to environmental factors. Post-fire CSS succession is also examined, and the potential effects of frequent and intense fires are discussed.

Westman, W. E. (1981). "Factors influencing the distribution of species of Californian coastal sage scrub." *Ecology* **62**(2): 439-455.

The distributions of CSS species were examined in relation to several environmental variables and some disturbance variables, such as fire, grazing, and air pollution. Of disturbance variables, only air pollution was correlated with floristic composition. Some variables likely to be changed by disturbance such as soil nutrients and litter mass are also examined and could be useful for predicting a given species' response to disturbance. However, such predictions will be complicated by the strong influence of environmental variables on local species composition.

White, S. D. (1995). "Disturbance and dynamics in coastal sage scrub." *Fremontia* **23**(4): 9-16.

White effectively reviewed previously published literature concerning the effects of fire on coastal sage scrub. The review emphasizes the variability in post-fire recovery patterns depending on geographic location, slope aspect, season of burn, fire intensity, stand age, etc. The review also discusses the opposing views regarding the natural fire regime in California shrublands and points to gaps in the understanding of fire ecology in CSS.

Zedler, P. H. (1988). "Invasion of *Carpobrotus edulis* and *Salix lasiolepis* after fire in a coastal chaparral site in Santa Barbara County, California." *Madroño* **35**(3): 196-201.

During a study designed to evaluate the effects of a controlled burn on *Eriodictyon capitatum*, invasion of the burned area by *C. edulis* was noted. Data collected on the invader showed seedling density of *C. edulis* was high 1 year after the fire, and survivorship 3 years later was also high (70%). Three years after fire, *C. edulis* had the second highest cover of any post-burn, perennial plant. Shrub cover was dense prior to burning and no *C. edulis* was noted in the area to be burned. However, *C. edulis* was present along the disturbed edge of the site and probably dispersed seeds into the site before the fire. As noted by the author, this study illustrates the importance of edge effects on invasion, and the need to consider these effects in relation to fire management practices.

Zedler, P. H. (1995). "Fire frequency in southern California shrublands: biological effects and management options." *Brushfires in California Wildlands: Ecology and Resource Management*. J. E. Keeley and T. Scott. Fairfield, WA, International Association of Wildland Fire: 101-112.

This paper deals more explicitly with the fire ecology of chaparral species, but includes generalizations about fire in shrublands that can be applied to CSS. The paper describes the potential risks to shrub communities associated with various fire regimes and uses data on chaparral species to evaluate the likelihood of those risks. Estimates of historical fire intervals in chaparral are also reviewed. Ultimately it is argued that a) the major risk to southern California shrublands comes from fire intervals that are too short and that deplete soil seed banks and kill adults before they can reproduce; b) historical fire intervals were probably toward the long end; and c) these factors should be considered when developing fire management plans for biological objectives.

Zedler, P. H., C. R. Gautier, and G. S. McMaster (1983). "Vegetation change in response to extreme events: the effect of a short interval between fires in California chaparral and coastal scrub." *Ecology* **64**(4): 809-818.

This study documents changes in chaparral and CSS shrub composition in plots that burned twice over a 2-year period. The second fire was partly fueled by a high cover of deliberately seeded introduced grass species, illustrating the potential alteration of fire regimes by exotic species invasion. In CSS, two fires resulted in elimination or reduction of many shrub and subshrub species, though the extent of reduction varied among species producing changes in species composition. *Artemisia californica* increased in abundance relative to pre-fire levels due to higher seedling survival of the second burn. Introduced annual species were present after both fires, suggesting tolerance to frequent burning.

The potential for type conversion of chaparral vegetation after frequent fire is also discussed.

Zink, T. A., M. F. Allen, B. Heindl-Tenhunen, and E. B. Allen (1995). "The effect of a disturbance corridor on an ecological reserve." *Restoration Ecology* 3(4): 304-310.

The authors examined recovery of grassland, CSS, chaparral, and oak woodland vegetation from mechanical disturbance associated with underground pipeline construction. A survey of the disturbance corridor more than 10 years after the disturbance revealed that, although native shrub cover was not significantly different between disturbed and undisturbed CSS, percent similarity between the two was relatively low. Exotic species had significantly higher cover, and native herbs had significantly lower cover in disturbed compared to undisturbed vegetation. These results suggest that CSS herbs may be more vulnerable to disturbance than some shrub species. The results also support the relationship between disturbance and exotic species invasion in CSS.

Mammals

Bolger, D. T., A. C. Alberts, R. M. Sauvajot, P. Potenza, C. McCalvin, D. Tran, S. Mazzoni, and M. E. Soulé (1997). "Response of rodents to habitat fragmentation in coastal southern California." *Ecological Applications*, 7:552-563.

The authors wanted to determine whether small fragments of coastal sage scrub and chaparral, isolated by urbanization, could support viable populations of native rodent species. Rodents were surveyed at 25 fragmented sites (mostly canyons) and 3 mainland sites in coastal San Diego County. Isolated sites varied in percent shrub cover, size, time since isolation, and distance from a "mainland", but were all completely surrounded by human-modified habitat. Analyses indicated that high native rodent species numbers are associated with young fragments having high percentages of shrub cover, which are not far from mainland sites (this is correlated with time since isolation), and which do not support *Rattus rattus*. Analyses also indicated that the size of a fragmented site is the primary determinant of species diversity. When comparing equal sized mainland plots to fragmented sites, it was found that fragments supported fewer native rodent species than mainland sites. This may be evidence for local extinctions following fragmentation.

Chase, M. K., W. B. Kristan III, A. J. Lynam, M. V. Price, and J. T. Rotenberry (2000). "Single species as indicators of species richness and composition in California coastal sage scrub birds and small mammals." *Conservation Biology*, 1: 474-487.

The authors searched for any association between the presence of potential indicator species and the species richness and composition of the bird or small mammal community in which it was found. Point counts and live trapping were used to quantify the distribution of birds and small mammals, respectively, in 3 counties. Two groups of taxa were chosen as potential indicators of species richness and species composition: species of conservation concern and species occurring commonly in samples. In general, species of conservation concern were not more frequently associated with species richness than common species and should not be assumed to be indicators of "hot spots". Though the presence of a species from one taxon was sometimes associated with species

richness in another taxon, overall there was a lack of correlation between bird and mammal species richness. This indicates that multiple taxa should be targeted in conservation planning.

Crooks, K. R. and M. E. Soulé (1999). “Mesopredator release and avifaunal extinctions in a fragmented system.” *Nature*, 400:563-566.

It is believed that mammalian carnivores are vulnerable to extinction in fragmented landscapes, and that their extinctions can lead to increased numbers of small carnivores that prey on birds and small vertebrates. This ‘mesopredator release’ is believed to play a part in the decline and extinction of small vertebrates in fragmented landscapes. To test this idea, the authors determined presence and relative abundance of coyotes, mesopredators, and scrub-breeding birds in 28 coastal sage scrub fragments surrounded by urbanization in coastal southern California. Fragment area, age, and isolation were variables used to describe each fragmented site. Analyses indicated that fragment size was a positive indicator of coyote abundance. Mean total mesopredator abundance was more than twice as high in fragments in which coyotes were not detected during the study. Though the positive effect of fragment area and the negative effect of fragment age were the strongest determinants of bird diversity, the negative effect of total mesopredator abundance on bird diversity persisted even after accounting for age and area effects, thus providing evidence for mesopredator release.

Price, M. V., and N. M. Waser (1984). “On the relative abundance of species: post-fire changes in a coastal sage scrub rodent community.” *Ecology*, 65:1161-1169.

Mac Arthur’s “Q-Minimization” theory of competition-mediated community structure was used to predict species abundance (based on knowledge of species’ resource use and resource availability) and was compared to actual species abundances over burned and unburned coastal sage scrub sites. Trapping was done at Motte Rimrock Reserve, south of Riverside, CA. Four distinct microhabitats (open, bush, rock, and debris) were measured at each trapping station on the grid. *Dipodomys agilis* was the only species that specialized on “open” microhabitat. *Peromyscus maniculatus* showed a slight preference for “debris” and *Perognathus fallax* preferred “rock”. *Peromyscus eremicus* and *Neotoma lepida* used “rock” and “bush” microhabitats with approximately equal frequency. *Dipodomys agilis* was consistently more abundant in burned than unburned sites, though most of the other species were more abundant in unburned sites. This was not because of persistence of resident individuals, since turnover of individuals between censuses was high.

Sherburne, F. Cook, Jr. (1959). “The effects of fire on a population of small rodents.” *Ecology*, 40:102-108.

This study examined the post-fire recovery of rodent populations in Tilden Regional Park, one mile east of Berkeley, California, where on October 21, 1953, a wild-fire burned 600 acres of grassland and brush habitat. Rodent densities and species composition were compared between one burned and one unburned brush site. After initial annihilation, rodents in the burned brush exhibited a population increase that surpassed that of the control area throughout the second year following the burn.

Schwilk, D. W., and J. E. Keeley (1998). "Rodent populations after a large wildfire in California chaparral and coastal sage scrub." *The Southwestern Naturalist*, 43:480-483.

This study examined the post-fire recovery of rodent populations in the Santa Monica Mountains of southern California. The fire occurred in late October 1993 and consumed 16,215 ha in Ventura Co. CA. Rodent densities and species composition were compared between two sites distant from the unburned vegetation, two sites adjacent to unburned areas, and two intermediate sites. Variation in species composition among sites was not consistently correlated with either vegetation type or distance from unburned brush. Lack of correlation between rodent density and distance from unburned vegetation might be explained by rapid migration from unburned areas before the beginning of the study, rodent survival through intense fires, or lightly burned habitat patches in canyons acting as refuges and sources of recolonization. Additionally, several species may show different distributions relative to unburned brush depending on the type of post-fire vegetation.

Birds

Bolger, D. T., A. C. Alberts, and M. Soulé (1991). "Occurrence patterns of bird species in habitat fragments: Sampling, extinction, and nested species subsets." *American Naturalist* **137**(2): 155-166.

The authors compared the species-area relationship for birds in unfragmented chaparral habitat with that in urban chaparral fragments. The results confirmed that rapid population extinction of resident bird species has occurred in these fragments. A strong positive correlation between the relative persistence ability of a species and its density remains even after correcting for the sampling effect of area. They concluded that abundant species persist longer in fragments than do less abundant species and that this creates a pattern that results in species poor fragments being a nested subset of species rich fragments.

Bolger, D. T., T. A. Scott, and J. T. Rotenberry (1997). "Breeding bird abundance in an urbanizing landscape in coastal Southern California." *Conservation Biology* **11**(2): 406-421.

The authors showed that, of 20 focal bird species, there was a marked difference in response to landscape gradients in coastal San Diego County. Groups responded to edge/fragmentation in one of three ways: enhanced, reduced, or unaffected densities. Therefore they propose that landscape heterogeneity is an important consideration in the design and management of nature reserves.

Boulinier, T., J. D. Nichols, J. R. Sauer, J. E. Hines, and K.H. Pollack (1998). "Estimating species richness: The importance of heterogeneity in species detectability." *Ecology* (Washington D C) **79**(3): 1018-1028.

This paper is one of several that address the issue of determining species richness when there are species detectability problems. The authors develop a well-reasoned argument that use of the Mh model is a better choice when addressing heterogeneous detection probabilities.

Bradford, D. F., S. E. Franson, A. C. Neale, D. T. Heggem, G. R. Miller and G. E. Canterbury (1998). "Bird species assemblages as indicators of biological integrity in Great Basin rangeland." *Environmental Monitoring and Assessment* **49**(1): 1-22.

This study evaluated the potential for bird species assemblages to serve as indicators of biological integrity of rangelands in the Great Basin. The rangeland impact under consideration was the result of grazing. Their selection of metrics and species was not sufficient to distinguish impact gradients. Only the extremes of light and heavy impact were detectable. A more robust approach similar to that of O'Connell, et al, 2000 may be worth pursuing.

Brooks, R. P. and M. J. Croonquist (1990). "Wetland, habitat and trophic response guilds for wildlife species in Pennsylvania." *Journal of the Pennsylvania Academy of Science* **64**(2): 93-102.

The authors developed a database of response guilds for all wildlife species in Pennsylvania. They ranked the species to emphasize those that are sensitive to anthropogenic disturbances in aquatic habitats. Metrics were established and species were assembled into guilds that would provide a way to compare structural and functional changes in wildlife communities affected by environmental impacts. This data was used in the development of a regional index of biological integrity as described in Brooks, et al, 1998 and in the application of response guilds to impacts in riparian-wetland areas in Croonquist and Brooks, 1991.

Brooks, R. P., T. J. O'Connell, D. H. Wardrop, and L. E. Jackson (1998). "Towards a regional index of biological integrity: The example of forested riparian ecosystems." *Environmental Monitoring and Assessment* **51**(1-2): 131-143.

The authors proposed six principles to guide development of any RIBI: 1) biological communities with high integrity are the desired endpoints; 2) indicators can have a biological, physical, or chemical basis; 3) indicators should be tied to specific stressors that can be realistically managed; 4) linkages across geographic scales and ecosystems should be provided; 5) reference standards should be used to define target conditions; and 6) assessment protocols should be efficiently and rapidly applied. As an illustration of how an RIBI might be developed, the authors showed how four integrative bioindicators can be combined to develop a RIBI for forest riparian ecosystems in the Mid-Atlantic states: 1) macroinvertebrate communities, 2) amphibian communities, 3) avian communities, and 4) avian productivity. This paper, however, lacked the data required to actually evaluate the concept.

Canterbury, G. E. and D. E. Blockstein (1997). "Local changes in a breeding bird community following forest disturbance." *Journal of Field Ornithology* **68**(4): 537-546.

This paper shows that substantial changes in the composition and population of breeding birds in northern Minnesota changed as a result of local disturbances in forested areas. The authors showed that the changes were related to increased density of ground vegetation and increased light levels. Ground foragers were more affected than arboreal species. While CSS is structurally different than the forests discussed, some CSS ground nesters could have a similar response.

Canterbury, G. E., T. E. Martin, D. R. Petit, L. J. Petit, and D. F. Bradford. (2000). "Bird communities and habitat as ecological indicators of forest condition in regional monitoring." *Conservation Biology* **14**(2): 544-558.

This paper provides support to the application of bird guilds and habitat variables to the measure of forest condition. The authors developed a bird community index and a habitat index that was then applied to loblolly-shortleaf pine forests in the southern U.S. across a disturbance gradient that varied from undisturbed to clear-cut forests. Canopy cover and basal area were measures of disturbance that correlated with probability of occurrence of bird guilds. The authors state that the cumulative distribution function of the bird-community and habitat indices provide a strong tool for evaluating forest condition over regional scales.

Chase, M. K., J. T. Rotenberry and M. D. Misenhelter (1998). "Is the California Gnatcatcher an indicator of bird-species richness in coastal sage scrub?" *Western Birds* **29**(4): 468-474.

The authors' paper suggests that the California Gnatcatcher is not a good indicator of bird-species richness for CSS. The sample may be too small to draw any far ranging conclusions, however this paper is one of many that suggest that a single species does not provide a good indication of species richness.

Chase, M. K., W. B. Kristan, III, A. J. Lynam, M. V. Price, and J. Rotenberry (2000). "Single species as indicators of species richness and composition in California coastal sage scrub birds and small mammals." *Conservation Biology* **14**(2): 474-487.

The authors evaluated 40 species of birds and small mammals, including 11 species of conservation concern, as potential indicators of species richness and species composition in southern California coastal sage scrub habitats. Of the few species they found associated with species richness, some were associated with higher species richness and others with lower richness, and species of conservation concern were not more frequently associated with species richness than were common species. Their results suggest that efforts to conserve bird and small-mammal biodiversity in coastal sage scrub should not focus exclusively on rare species or on locations with the highest species richness, but instead should focus on a diverse suite of species that are representative of the range of variation in communities found in coastal sage scrub habitats.

Crooks, K. R. and M. E. Soulé (1999). "Mesopredator release and avifaunal extinctions in a fragmented system." *Nature (London)* **400**(6744): 563-566.

This study shows that removal of top predators, in this case the coyote, increases the presence of mesopredators with the attendant reduction of avian species. Its importance is associated with urban CSS fragments.

Croonquist, M. J. and R. P. Brooks (1991). "Use of avian and mammalian guilds as indicators of cumulative impacts in riparian-wetland areas." *Environmental Management* **15**(5): 701-714.

The authors developed a method of assessing the cumulative effects of human activities on bird and mammal communities in riparian-wetland areas through the use of response guilds to reflect how species theoretically respond to habitat disturbance at a landscape

level. All bird and mammal species of Pennsylvania were assigned values for each response guild to reflect their sensitivity to disturbances; high guild scores corresponded to low tolerance toward habitat disturbance. The percentage of bird species with high response-guild scores remained relatively stable through the protected watershed. As intensity of habitat alteration increased through the disturbed watershed, percentage of bird species with high response-guild scores decreased. Species at the edge and exotic guild classifications (low guild scores) were found in greater percentages in the disturbed watershed. The authors found that the composition of mammalian guilds showed no consistent pattern associated with habitat disturbance while avian response guilds reflected habitat disturbance in a more predictive pattern.

Fleury, S. A., P. J. Mock and J. F. O'Leary (1998). "Is the California Gnatcatcher a good umbrella species?" *Western Birds* **29**(4): 455-467.

The authors analyzed 40 out of 120 listed species (threatened, endangered, etc.) to see if they can be protected under the umbrella of protection of the California Gnatcatcher. They found that it is suitable for less than half of those 40 species. Best protected are those whose area requirements are less than or equal to the California Gnatcatcher and which have similar habitat requirements. It also supported species that are at lower trophic levels and that are not habitat specialists.

Franklin, J. (1998). "Predicting the distribution of shrub species in southern California from climate and terrain-derived variables." *Journal of Vegetation Science* **9**(5): 733-748.

The author applied generalized additive, generalized linear, and classification tree models to predict the distribution of 20 species of chaparral and coastal sage shrubs within the southwest ecoregion of California. Mapped explanatory variables included bioclimatic attributes related to primary environmental regimes and topographically distributed potential solar insolation of the wettest quarter (winter) and of the growing season (spring). Models were parameterized and evaluated based on species presence/absence data from 906 plots surveyed on National Forest lands. The author showed that all variables were significant in at least one of the species' models, but those models based only on the bioclimatic variables predicted species presence with 3 - 26% error. All three methods produced models with similar accuracy for a given species; GAMs were useful for exploring the shape of the response functions, GLMs allowed those response functions to be parameterized and their significance tested, and classification trees, while sometimes difficult to interpret, yielded the lowest prediction errors (lower by 3 - 5%).

Guthrie, D. A. (1974). "Suburban bird populations in Southern California." *American Midland Naturalist* **92**(2): 461-466.

Guthrie found that species diversity and numbers of birds increased within suburbs when compared with undisturbed natural surroundings. Although many ground dwelling birds associated with coastal sage were eliminated, they were replaced with birds normally found in wooded canyons of local foothills

Knick, S. T. and J. T. Rotenberry (1995). "Landscape characteristics of fragmented shrubsteppe habitats and breeding passerine birds." *Conservation Biology* **9**(5): 1059-1071.

The authors examined the influence of local and landscape-level attributes of fragmented habitats in shrubsteppe habitats on the breeding distributions of five bird species in the Snake River Plains of southwestern Idaho. They developed habitat (resource) selection models for each species by combining bird counts conducted from 1991 through 1993 with local vegetation characteristics and landscape attributes derived from satellite imagery. They found that typical grassland species were not sensitive to landscape features, only on the amount of grassland or shrub cover. However, shrubsteppe species depended upon local vegetation cover and landscape features such as patch size or spatial similarity of sites. Their results demonstrate that fragmentation of shrubsteppe influenced the presence of shrub-obligate species.

Lovio, J. C. (1996). "The Effects of Habitat Fragmentation on the Breeding-Bird Assemblage in California Coastal Sage Scrub." Masters Thesis, San Diego State University.

Lovio showed that the bird assemblage found in and around CSS exhibits a low threshold response to habitat fragmentation. Area-sensitive species are quickly lost at fairly large scales of fragmentation. The larger proportion of relatively ubiquitous species is generally unaffected. A few generalists seem to be enhanced by fragmentation.

Lovio, J. C. L. A. (2000). "The draft coastal scrub and chaparral bird conservation plan: a strategy for protecting and managing coastal scrub and chaparral habitats and associated birds in California." California Partners in Flight, Point Reyes Bird Observatory.

The authors have developed a plan for conservation of birds utilizing a subset of the natural shrub lands of California: those low-elevation shrub lands west of the state's major mountain axis. The goals of the plan are to emphasize what is needed to conserve both populations of species, and species assemblages, to synthesize and summarize current scientific knowledge of the requirements of birds in shrub land habitats, to provide recommendations for habitat protection, restoration, management, monitoring, and policy to ensure the long-term persistence of birds and other wildlife dependent on shrub land ecosystems, and to support and inform efforts to increase the overall acreage and effectiveness of shrub land habitat conservation efforts in California by funding and promoting on-the-ground conservation projects. The plan contains good bird, vegetation data.

Lynam, A. J., M. K. Chase, W. B. Kristan III, J. T. Rotenberry and M. V. Price (1996). "Geographic variation and habitat associations of coastal sage scrub bird and small mammal communities." Coastal Sage Scrub Natural Community Conservation Plan Annual Report

Using rapid survey techniques at 11 study sites in southern California, the authors documented the distributions of birds and small mammals across part of the NCCP study region. Detrended correspondence analyses were used to identify major correlated patterns of species turnover in both taxa. Analysis of vegetation data collected at each of the 125 locations where birds and small mammals were jointly censused showed that plant species turnover showed a similar trend. Canonical correspondence analysis was used to identify the habitat associations of birds and mammals. Regression analyses of bird presence/absence and mammal abundance data with vegetation structure and floristics variables supported the findings from the ordination.

The degree to which a number of designated "target" and "sensitive" birds and mammals are useful indicators of the area of high species richness or unique species composition

was assessed using randomized procedures. California gnatcatchers were good indicators of places of high species richness for birds and mammals. Gnatcatchers along with Cactus Wren, Sage and Rufous-crowned sparrows and San Diego pocket mouse were useful indicators of places with unique and distinct assemblages of birds and mammals. San Diego woodrats were not associated with distinct species assemblage but might be generally useful for conservation by indicating the occurrence of coastal sage habitats.

MacNally, Ralph C. (1994). "Habitat-specific guild structure of forest birds in southeastern Australia: a regional scale perspective." *Journal of Animal Ecology* **63**: 988-1001.

In this study, foraging information is used to produce a guild classification for birds of forests and woodlands of central Victoria, Australia. Four replicate sites of five forest and woodland classes were censused. Guild structures in two habitat classes were distinct from each other and also from those of three other habitat types. There was little differentiation between the three habitat types. Much the same guild structure occurred in replicate sites of each habitat class, indicating that there is a systematic basis for guild structure that can be broadly related to habitat structure. In some habitats, maintenance of guild structure from replicate to replicate is mediated by similar arrays of species, whilst in other habitat types, there are significant differences in the actual species occupying guilds even though the numbers of species in guild are similar. Thus, use of replicate sites provides important additional information on how guilds are composed in different habitats.

Moriarty, D. J., R. E. Farris, D. K. Noda, and P. A. Stanton (1985). "Effects of fire on a coastal sage scrub bird community." *The Southwestern Naturalist* **30**(3): 452-3.

The authors show that fire has a substantial effect on the structure of the avian communities in coastal sage scrub. Species richness declines and resident species vary considerably in their use of the burned area. Recovery to prefire conditions is affected by the time of the fire relative to the onset of winter rains. In their case, 70-90% similarity was achieved within one year.

Nichols, J. D., T. Boulinier, J. E. Hines, K. H. Pollack, and J. R. Sauer (1998). "Estimating rates of local species extinction, colonization, and turnover in animal communities." *Ecological Applications* **8**(4): 1213-1225.

This is another of the papers that looks at modeling the state of species in a community. In this case the authors extend the probabilistic capture-recapture model approach to the development of estimators useful for studying rates of local species extinction, turnover and colonization. Assuming closed animal populations that permit heterogeneity in detection probabilities they developed a computer program (COMDYN) that computes many of these estimators. Their approach was tested against data from the North American Breeding Bird Survey and performed reasonably well.

Nichols, J. D., T. Boulinier, J. E. Hines, K. H. Pollack, and J. R. Sauer (1998). "Inference methods for spatial variation in species richness and community composition when not all species are detected." *Conservation Biology* **12**(6): 1390-1398.

This paper carries their previous paper one step farther by applying their techniques and the COMDYN program to those situations where it is clear that not all species are detected.

O'Connell, T. J., L. E. Jackson, and R. P. Brooks (2000). "Bird guilds as indicators of ecological condition in the central Appalachians." *Ecological Applications* **10**(6): 1706-1721.

This paper is important for the purposes of the CSS IBI. The authors developed an index of biotic integrity based on bird communities in the central Appalachians. The index is intended to indicate landscape-scale stressors to upland environments in the central Appalachians. The Bird Community Index (BCI) ranks bird communities according to the proportional representation of 16 behavioral and physiological response guilds. The index was developed from 34 sites in central Pennsylvania that represented a gradient of human disturbance from near pristine to degraded. Upon satisfactory demonstration that the BCI could discriminate between categories of biotic integrity identified from the human disturbance gradient, it was applied to an independent, probability-based sample of 126 sites across the study area. Their assessment indicates that 16% of the area is in "excellent" condition, 27% is in "good" condition, 36% is in "fair" condition, and 21% is in "poor" condition. Either urban or agricultural bird communities dominated sites in poor condition, but these communities could not be numerically distinguished from each other by BCI score. Forested sites in good and excellent condition supported different bird communities and ground-level vegetation attributes but could not be separated by land cover composition alone. In general, the shift from medium to poor ecological condition defined by bird communities coincided with a shift in land cover composition from forested to nonforested.

Parish, T., K. H. Lakhani, and T. H. Sparks (1994). "Modelling the relationship between bird population variables and hedgerow and other field margin attributes. I. Species richness of winter, summer and breeding birds." *Journal of Applied Ecology* **31**(4): 764-775.

Regression models were used to relate measures of bird species richness, in both winter and summer over several years, to field boundary attributes, including adjacent land use in a farming area. Bird variables were positively correlated with the physical size of hedges, the number and height of trees and the adjacent permanent pasture. These attributes accounted for most of the observed variation in bird variables. Ditch and verge dimensions played a statistically significant but relatively less important role.

Parish, T., K. H. Lakhani, and T. H. Sparks (1995). "Modelling the relationship between bird population variables and hedgerow, and other field margin attributes. II. Abundance of individual species and of groups of similar species." *Journal of Applied Ecology* **32**(2): 362-371.

The earlier paper related bird species richness to hedgerows and other field boundary attributes including adjacent land use while the present paper deals with similar relationships, but the bird variables used are the abundance of individual species, of groups of species (all finches, all raptors, etc.) and of all birds, as well as Simpson's index of diversity. Like bird species richness in the earlier paper, the abundance of many species was strongly influenced by the land use, and by the main vegetation variables (tree height and number, hedgerow length, height and width). Land use was important to a larger majority of species. Verge width appeared important for small insectivores and was particularly important for seed eating birds. Ditch dimensions were associated with the abundance of some seedeaters and insectivores. Raptors, corvids, waders and aquatics were associated with large ditches. Though the general management

prescriptions to benefit the majority of farmland birds were found to be similar to those in the earlier paper, the detailed species specific modelling made it possible to examine the habitat requirements of particular species.

Soule, M. E., D. T. Bolger, A. C. Alberts, J. Wright, M. Sorice, and S. Hill (1988).

"Reconstructed dynamics of rapid extinctions of chaparral-requiring birds in urban habitat islands." *Conservation Biology* **2**(1): 75-92.

The distribution of native, chaparral-requiring bird species was determined for 37 isolated fragments of canyon habitat ranging in size from 0.4 to 104 hectares in coastal, urban San Diego County, California [USA]. The area of chaparral habitat and time since isolation of the habitat fragment explains most of the variation in the number of chaparral-requiring bird species. In addition, the authors found that the distribution of native predators may influence species number. There is statistical evidence that coyotes control the populations of smaller predators such as foxes and domestic cats. The absence of coyotes may lead to higher levels of predation by a process of mesopredator release. The best predictors of vulnerability of the individual species are their abundances (densities) in undisturbed habitat and their body sizes; together these two variables account for 95 percent of the variation in canyon occupancy. A hypothesis is proposed to account for the similarity between the steep slopes of species-area curves for chaparral-requiring birds and the slopes for some forest birds on small islands or in habitat fragments. The authors indicate that a provision for corridors appears to be the most effective design and planning feature for preventing the elimination of chaparral-requiring species in a fragmented landscape.

Soule, M. E., A. C. Alberts, and D. T. Bolger (1992). "The effects of habitat fragmentation on chaparral plants and vertebrates." *Oikos* **63**(1): 39-47.

The authors found that the effects of fragmentation in a scrub habitat in California [USA] on three taxa (plants, birds, and rodents) are concordant. Extinctions within the habitat remnants occur quickly and the sequence of species disappearances of birds and rodents is predictable based on population density in undisturbed habitat. Distance effects on species diversity are weak to non-existent, and habitat area effects are strong. Edge effects and cumulative habitat loss following isolation of the remnants are correlated with loss of species diversity. Recolonization in these taxa occurs rarely. Rodents appear to be extremely susceptible to extinction. Small, old patches retain a predictable subset of bird and rodent species, reinforcing the principle that larger reserves are generally superior.

Herpetofauna

Gamradt, S. C. and L. B. Kats (1997). "Impact of chaparral wildfire-induced sedimentation on oviposition of stream-breeding California newts (*Taricha torosa*)." *Oecologia* (Berlin) **110**(4): 546-549.

The effects of chaparral wildfire on stream breeding California newts (*Taricha torosa*) were examined in a perennial Santa Monica Mountain stream following a 1993 wildfire. Major changes in stream morphology and composition were produced from erosion following the 1993 fire. Even though the average density of adult newts did not differ

among pre and post fire years, the researchers documented approximately one-third of the total number of newt egg masses observed in pre-wildfire surveys were observed in surveys following the fire. Previous to the wildfire, pools and runs, stream habitat preferred by California newts for egg laying, compromised 40-50% of the pre-fire stream area. Following the fire, pools and runs, consisted of less than 20% of the stream area. The researchers conclude that fire-induced landslides and siltation eliminated many of the stream's pools and runs, necessary habitat for successful California newt egg oviposition.

Davidson, C., H. B. Shaffer, M. R. Jennings (2001). "Declines of the California red-legged frog: Climate, UV-B, habitat, and pesticides hypotheses." *Ecological Applications* **11**(2): 464-479.

Patterns of decline across the entire range of the federally threatened California red-legged frog (*Rana aurora draytonii*) showed a strong positive association with elevation, percentage upwind from agricultural land use, and local urbanization. Due to the absence of a latitudinal gradient in declines, the previous documentation of both transport and deposition of pesticides to the Sierra Nevada from the Central Valley agricultural region, and the presence of pesticide residues in the bodies of the frog species, *Rana muscosa* and *Hyla regilla*, the authors conclude that wind-borne agrochemicals may be an important factor in this species decline.

Goodsell, J. A. and L. B. Kats (1999). "Effect of introduced mosquitofish on Pacific treefrogs and the role of alternative prey." *Conservation Biology* **13**(4): 921-924.

Data from field surveys revealed sixty-five percent of stream-caught Mosquitofish (*Gambusia affinis*) in the Santa Monica Mountains contained Pacific treefrog (*Hyla regilla*) tadpoles in their stomachs. In support of these observations, both laboratory and field experiments documented mosquitofish affinity to prey on treefrog tadpoles, even when high densities of mosquito larvae were presented as alternative prey.

Gamradt, S. C. and L. B. Kats (1996). "Effect of introduced crayfish and mosquitofish on California newts." *Conservation Biology* **10**(4): 1155-1162.

Both field survey data and experimental evidence implicates the introduced crayfish (*Procambarus clarkii*) and mosquitofish (*Gambusia affinis*) predators as a possible cause of decline of the California newt (*Taricha torosa*). Ten streams in the Santa Monica Mountains surveyed between 1981 and 1986 were found to contain California newts. 1994 surveys these same ten streams documented in three streams the presence of mosquitofish and crayfish and the absence of California Newts and in the other seven streams the absence of crayfish and mosquitofish and the presence of California newts. Field and lab experiments indicate that crayfish consume California newt egg masses and that both crayfish and mosquitofish consume larval newts. Predation rates on newts by both predators were high.

Kiesecker, J. M., A. R. Blaustein, C. L. Miller (2001). "Potential mechanisms underlying the displacement of native red-legged frogs by introduced bullfrogs." *Ecology* (Washington D C) **82**(7): 1964-1970.

The presence of Bullfrog larvae had strong negative effects on the performance of red-legged frog larvae (reduced survivorship to metamorphosis and mass at metamorphosis) when food resources in experimental ponds were clumped rather than scattered.

“Behavioral observations indicate that a passive interference mechanism is likely to be responsible to for the outcome of interactions between bullfrogs and red-legged frogs.”

Lawler, S. P., D. Dritz, T. Strange, M. Holyoak (1999). "Effects of introduced mosquitofish and bullfrogs on the threatened California red-legged frog." *Conservation Biology* **13**(3): 613-622.

The effects of mosquitofish and bullfrog tadpoles were tested on red-legged frog (*Rana aurora draytonii*) tadpoles in spatially complex, speciose communities. In the experiments hatchling red-legged frogs were added to earthen ponds. Ponds were either maintained as controls, stocked with bullfrog tadpoles, stocked with adult mosquitofish, or stocked with both bullfrogs and mosquitofish. The survival of red-legged frogs in the presence of bullfrog tadpoles was less than 5%; survival was 34% in control ponds. Mosquitofish did not affect red-legged frog survival, even though fish became abundant. Red-legged frog tadpoles did, however, suffer more injuries in ponds with fish and weighed 34% less at metamorphosis. “The growth decrease could have been caused by injuries or by lower foraging levels in the presence of fish. Laboratory results showed that young tadpoles were less active in the presence of mosquitofish. Although both mosquitofish and bullfrogs affected red-legged frogs, the impact of bullfrogs on the survival of red-legged frogs may contribute more strongly to their decline.”

Suarez, A. V., J. Q. Richmond, T. J. Case (2000). "Prey selection in horned lizards following the invasion of Argentine ants in southern California." *Ecological Applications* **10**(3): 711-725.

In both field and laboratory studies the coastal horned lizard (*Phrynosoma coronatum*) displayed an aversion to feeding on the invasive Argentine ant (*Linepithema humile*). “In non-invaded areas the coastal horned lizard diet consisted predominately of ants (>94% by prey item in three reserves examined), particularly harvester ants in the genera *Pogonomyrmex* and *Messor*. In invaded areas, most native ants were displaced, and remaining horned lizards incorporated more non-ant arthropods and smaller ants into their diets.” In laboratory prey preference experiments, the coastal horned lizard confirmed patterns observed in the field by repeatedly choosing to prey upon native ants when presented with both native ants and Argentine ants as potential prey.

Fisher, R. N. and H. B. Shaffer (1996). "The decline of amphibians in California's Great Central Valley." *Conservation Biology* **10**(5): 1387-1397.

Broad-scale field sampling and historical analyses of museum records were used to quantify amphibian declines in California's Great Central Valley. Overall, all amphibian species surveyed showed an unambiguous pattern of decline. The species most affected by the decline were *Bufo boreas* and *Rana aurora*. *Pseudacris regilla* was the least affected. Introduced predators (mosquito fish, other fish, and bullfrogs) appear to be the primary threat to native amphibian species. Native amphibians tended not to co-occur with exotics. Native amphibians were more frequently found in higher elevations sites while exotic species were found primarily in lowland sites. For some native species, evidence exists that suggests their present distributions represent a significant restriction to higher elevation sites from a formerly broader distribution.

Arthropods

Bolger, D. T., A. V. Suarez, K. R. Crooks, S. A. Morrison, and T. J. Case (2000).

"Arthropods in urban habitat fragments in southern California: Area, age, and edge effects." *Ecological Applications* **10**(4): 1230-1248.

This paper is linked to the Suarez et al. 1998 paper, focusing on the same habitat patches, but on non-ant arthropods. Individual arthropods were identified to order and Recognizable Taxonomic Unit (RTU) or morphospecies. They found: arthropods are influenced by fragmentation with their diversity and abundance correlated to fragment size, and negatively to age; spider diversity and abundance are enhanced by fragmentation, exotic ants influence the richness and abundance of other arthropods, spiders and carabid beetles increased in abundance in older fragments, being positively correlated with the abundance of Argentine ants and exotic Isopods, Dermaptera, and Blattaria. This paper emphasize the effects that fragmentation, isolation, and percent native vegetation have on the non-ant arthropod community in coastal sage scrub.

Burger, J. A., M. A. Patten, T.R. Prentice, and R. Redak (2001). "Evidence for spider community resilience to invasion by non-native spiders." *Biological Conservation* **98**: 241-249.

Native and non-native spiders were sampled across 60 sites in undisturbed coastal sage scrub and chaparral across San Diego County. They determined that although the presence of many non-native species has caused detrimental impacts to the CSS community, invasive spiders have produced no obvious negative impacts on their native counterparts. However, non-native spiders are more typically found in areas of high biodiversity. Their findings are contrary to the commonly held notion that introduced species are more likely to invade areas with low biodiversity.

Burger, J. C., M. A. Patten, J. T. Rotenberry, and R. A. Redak (1999). "Foraging ecology of the California gnatcatcher deduced from fecal samples." *Oecologia* **120**(2): 304-310.

This paper provides an interesting link between the idea of studying arthropods in coastal sage scrub, and the potential implications of the distribution of the California gnatcatcher. The authors suggest that further studies on the food sources of the gnatcatcher may provide insight into the presence or absence of this species in what appear to be suitable areas of coastal sage scrub. Surveys for the arthropod food items found in gnatcatcher fecal samples may provide useful indicators of suitable habitat for the bird.

Burger, J. C., R. A. Redak, E. E. Porter, J. T. Rotenberry, and T. A. Scott (1996). "Habitat preservation in southern California coastal sage scrub communities." *Bulletin of the Ecological Society of America* (**3** Suppl. Part 2) 77: 60.

The paper is focused on surveys on Mirimar Naval Base and Camp Pendelton. They used vacuum sampling and pitfall traps to collect the samples. The results suggest that inland areas have differing species richness for certain orders of insects than coastal sites. These results may provide a useful tool for assessing the habitat quality of coastal sage scrub, with emphasis on the impacts on the insectivores.

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

This paper comments on the observation that postfire insect succession parallels the successional patterns of the plants. This suggests that insects follow similar patterns that plants do, whereby the initial insect migrators into a burned area are largely influenced by the presence of plants, while later migrating insects are influenced by the presence of other insects. The richness and diversity of plants after a fire influences the insect community. The resulting plant communities that provide food for the sap and foliage feeding insects will in turn allow for other insects to enter the community.

Force, D. C. (1982). Postburn insect fauna in southern California chaparral. Berkeley, CA, Pacific Southwest Forest and Range Experiment Station, U.S. Forest Service: 234-239.

This publication describes a 4-year post fire study of the insects in chaparral communities. Pollen/nectar feeders and predatory insects can be abundant the spring after the fire, while other feeders and parasitic insects move in later. The results found in the chaparral community can be translated into similar effects in the coastal sage scrub vegetation type.

Moldenke, A. R. (1976). "California pollination ecology and vegetation types." *Phytologia* **34**: 305-361.

Surveys across a variety of vegetation types (including coastal sage scrub) were conducted to determine the type of pollinators found on the plants. There are 80 species of bees, 7 beetles, 3 butterflies, 10 muscoid flies, 4 syrphid flies, 7 bee flies, 7 wasps, and 1 sphingid moth identified as pollinators of CSS vegetation. There is often an important link between these insects and the multitude of host plants, rare and endemic plants, and endangered plants. The unique flora of southern California has led to a high number of mutualistic relationships between plants and insects; of the 1,200 native bee species, almost 800 are considered specialists (with 172 species identified in coastal dunes and sage, and 520 species identified in the southern California coastal ranges). The insect pollinators also provide a substantial economic service for California's agricultural economy. Unfortunately, the agricultural and urban use of pesticides inflicts a heavy cost on the arthropod community. The introduction of the numerous non-native species on the pollinators is not well understood. However, the introduced honey bee (*Apis mellifera*) has become heavily integrated into the pollination of California plants, and has potentially caused the extirpation or extinction of many native species.

Prentice, T. R., J. C. Burger, W. R. Icenogle, and R. A. Redak (2001). "Spiders from Riversidian coastal sage scrub with comparisons to diegan scrub fauna (Arachnida: Araneae)." *Pan-Pacific Entomologist* **77**(2): 90-122.

This is the most comprehensive study on spiders of San Diego County, California. They have provided a comprehensive species list of Araneae collected from Diegan coastal sage scrub (200 species). They found 35 species that are new records for the county, 20 undescribed species, and 3 species that they considered to be "rare". Additionally, they discovered 7 non-native species of spiders, whose presence may show a competitive displacement of the native fauna.

Redak, R. A. (2000). "Arthropods and multispecies habitat conservation plans: Are we missing something?" *Environmental Management* **26**(1): 97-107.

This article explains the current status and knowledge of arthropods in North America. They suggest that most land management, conservation, and reserve design occurs without the basic taxonomic and life history associated with majority of the organisms present. They suggest that the number of federally listed arthropods is highly underestimated. The lack of knowledge of this group provides a significant obstacle for their treatment in conservation plans. They suggest that the multispecies habitat conservation plans cannot prevent extinction, only minimize it.

Suarez, A. V., D. T. Bolger, and T. J. Case (1998). "Effects of fragmentation and invasion on native ant communities in coastal southern California." *Ecology* **79**(6): 2041-2056.

This article investigated the influence of habitat fragmentation and exotic ant invasion in on native ants in CSS patches in San Diego Co. They focused on time since isolation, fragment size, edge, percent native vegetation, isolation, and relative abundance of exotic Argentine ants. They found the abundance of exotic ants, fragment size, and time since isolation most strongly influenced the presence of native ants. Argentine ants strongly influence native ant communities. The research and study sites provided the groundwork for the later publication and evaluation by Bolger et al. (1998).

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Appendix B. Presentation of data used in IBI metric development

Overview.....	2
Species Information.....	3
Ants.....	6
Species level data.....	16
<i>Presence-Absence Data</i>	16
<i>Relative Abundance Data</i>	17
Genera Specific.....	17
<i>Presence-Absence Data</i>	18
<i>Relative Abundance Data</i>	20
Arthropods.....	21
<i>Presence-Absence Data</i>	21
<i>Exotic species</i>	23
Birds.....	24
<i>Presence-Absence Data Year 1</i>	24
<i>Presence-Absence Data Year 2</i>	25
<i>Presence-Absence Data BOTH</i>	26
<i>Relative Abundance Year 1</i>	28
<i>Relative Abundance Year 2</i>	29
<i>Relative Abundance Both</i>	30
Herpetofauna.....	32
<i>Presence/Absence Data</i>	32
<i>Relative Abundance Data</i>	34
Mammals.....	36
<i>Proportion adult</i>	36
<i>Female Body Weight</i>	37
<i>Male Body Weights</i>	38
<i>Male Reproduction</i>	39
<i>Female Reproduction</i>	40
<i>Mammal presence absence</i>	41
<i>Mammal Relative Abundance</i>	43
Vegetation.....	45
<i>Summary metrics</i>	45
<i>Presence-Absence Data</i>	48
Summary Tables.....	64

Overview

Results are ordered alphabetically by taxa to match the main report. We begin with a summary table giving the species name, a common name if possible, and code we used when labeling scatterplots and discussing IBI metrics. This table can be used to look up particular species codes when viewing the figures.

Within each taxa, we first present presence-absence data, then relative abundance data. In the case of small mammals, life history variables are also presented. For ants, we present both species level and genera level data. For plants we present community-level summary variables and presence-absence data.

For each data type, we present the scatterplots for the “BOTH” dataset unless otherwise noted, though we generated and screened similar scatterplots for both the YEAR 1 and YEAR 2 datasets. If the label above the figure indicates “We generated no metrics from these data”, then the scatterplots are present with no additional information. If metrics were produced from a data type, then the species, orders, or genera included in the metric were labeled as positive or negative within each scatterplot figure. Below each set of scatterplots, we briefly describe the positive and negative responses.

In some cases, the inclusion of a species was based on patterns also found in the YEAR 1 and YEAR 2 data and pre-existing information on the species. Thus, the scatterplot for the BOTH data may not be particularly revealing. In addition, we discuss other possible metric candidates and why they were not included. Finally, we end each section with a brief description of existing support for the inclusion of species within a metric. The end of the appendix contains summary tables.

Species Information.

Table 1. Scientific name, common name, and 4 or 5 letter species code for animal taxa analyzed in the study and used in Figures.

Species, Genera, or Order	Common name	Code
Ants species		
<i>Crematogaster californicus</i>		CRCA
<i>Crematogaster spp.</i>		CRSP
<i>Cyphomyrmex spp.</i>		CYSP
<i>Dorymyrmex insanus</i>	pyramid ant	DOIN
<i>Dorymyrmex spp.</i>		DOSP
<i>Forelius mccooki</i>		FRMC
<i>Formica moki</i>		FOMO
<i>Temnothorax andrei</i>		LEAN
<i>Linepithema humile</i>	Argentine ant	LIHU
<i>Messor andrei</i>		MEAN
<i>Myrmecocystus spp.</i>		MYSP
<i>Nievamymex nigrescens</i>		NONI
<i>Pheidole californica</i>		PHCA
<i>Pheidole clementensis</i>		PECL
<i>Pheidole hyatti</i>		PHHY
<i>Pheidole vistana</i>		PHVI
<i>Pogonomyrmex rugosus</i>		PORI
<i>Pogonomyrmex spp</i>		POGO
<i>Solenopsis amblychila</i>		SOAM
<i>Solenopsis molesta</i>	thief ant	SOMO
<i>Solenopsis xyloni</i>	native fire ant	SOXY
<i>Tapinoma sessile</i>	odorous house ant	TASE
<i>Tetramorium spinosum</i>		TPSP
Ant Genera		
<i>Campanotus</i>	carpenter ants	CAMP
<i>Crematogaster</i>	acrobat ants	CREM
<i>Cyphomyrmex</i>	fungus-growing ants	CYPH
<i>Dorymyrmex</i>		DORY
<i>Forelius</i>		FREL
<i>Formica</i>	wood ants	FORM
<i>Hypoponera</i>	crypt ants	HYPO
<i>Temnothorax</i>		LEPT
<i>Linepithema</i>		LINE

<i>Messor</i>	harvester ants	MESS
<i>Myrmecocystus</i>	honeypot ants	MYRM
<i>Neivamyrmex</i>	new world army ants	NEIV
<i>Paratrachina</i>	crazy ants	PARA
<i>Pheidole</i>	big-headed ants	PHEI
<i>Pogonomyrmex</i>	harvester ants	POGO
<i>Solenopsis</i>	fire ants	SOLE
<i>Stenamma</i>		STNN
<i>Tapinoma</i>	odorous ants	TAPI
<i>Tetramorium</i>	pavement ants	TETR
Arthropod Order		
<i>Acarina</i>	mites	
<i>Araneae</i>	spiders	
<i>Archaeognatha</i>	bristletails	
<i>Blattodea</i>	cockroaches	
<i>Chilopoda</i>	centipedes	
<i>Coleoptera</i>	beetles	
<i>Collembola</i>	springtails	
<i>Dermaptera</i>	earwigs	
<i>Diplopoda</i>	millipedes	
<i>Diplura</i>	dipluran	
<i>Diptera</i>	flies	
<i>Embiidina</i>	webspinners	
<i>Hemiptera</i>	true bugs	
<i>Homoptera</i>	aphids, hoppers	
<i>Hymenoptera</i>	bees, wasps, ants	
<i>Isopoda</i>	sowbug	
<i>Isoptera</i>	termites	
<i>Lepidoptera</i>	Butterflies, moths	
<i>Lepidoptera larva</i>	butterfly larvae	
<i>Opiliones</i>	daddy longlegs	
<i>Orthoptera</i>	grasshoppers, crickets	
<i>Pseudoscorpionida</i>	pseudoscorpions	
<i>Psocoptera</i>	book lice, bark lice	
<i>Scorpionida</i>	scorpions	
<i>Siphonaptera</i>	fleas	
<i>Solpugida</i>	sun spiders	
<i>Thysanoptera</i>	thrips	
<i>Thysanura</i>	silverfish	
Bird Species		
<i>Calypte anna</i>	Anna's Hummingbird	ANHU
<i>Myiarchus cinerascens</i>	Ash-throated Flycatcher	ATFL

<i>Thryomanes bewickii</i>	Bewick's Wren	BEWR
<i>Guiraca caerulea</i>	Blue Grosbeak	BLGR
<i>Euphagus canocephalus</i>	Brewer's Blackbird	BRBL
<i>Icterus bullockii</i>	Bullock's Oriole	BUOR
<i>Psaltiriparus minimus</i>	Bushtit	COBU
<i>Polioptila californica</i>	California Gnatcatcher	CAGN
<i>Callipepla californica</i>	California Quail	CAQU
<i>Toxostoma redivivum</i>	California Thrasher	CATH
<i>Pipilo crissalis</i>	California Towhee	CALT
<i>Tyrannus vociferans</i>	Cassin's Kingbird	CAKI
<i>Petrochelidon pyrrhonota</i>	Cliff Swallow	CLSW
<i>Geothlypis trichas</i>	Common Yellowthroat	COYE
<i>Ammodramus savannarium</i>	Grasshopper Sparrow	GRSP
<i>Carpodacus mexicanus</i>	House Finch	HOFI
<i>Chondestes grammacus</i>	Lark Sparrow	LASP
<i>Carduelis psaltria</i>	Lesser Goldfinch-	LEGO
<i>Zenaida macroura</i>	Mourning Dove	MODO
<i>Colaptes auratus</i>	Northern Flicker	NOFL
<i>Mimus polyglottos</i>	Northern Mockingbird	NOMO
<i>Picoides nuttallii</i>	Nuttall's Woodpecker	NUWO
<i>Phainopepla nitens</i>	Phainopepla	PHAI
<i>Agelaius phoeniceus</i>	Red-winged Blackbird	RWBL
<i>Amphispiza bellii</i>	Bell's Sage Sparrow	SAGS
<i>Melospiza melodia</i>	Song Sparrow	SOSP
<i>Pipilo maculatus</i>	Spotted Towhee	SPTO
<i>Tyrannus verticalis</i>	Western Kingbird	WEKI
<i>Sturnella neglecta</i>	Western Meadowlark	WEME
<i>Aphelocoma californica</i>	Western Scrub Jay	WESJ
<i>Zonotrichia leucophrys</i>	White-crowned Sparrow	WCSP
<i>Chamaea fasciata</i>	Wrentit	WREN
<i>Dendrioca coronata</i>	Yellow-rumped Warbler	AUWA
Herpetofauna species		
<i>Batrachoseps nigriventris</i>	black-bellied slender salamander	BANI
<i>Batrachoseps pacificus</i>	Pacific slender salamander	BAPA
<i>Batrachoseps spp</i>	slender salamander	BASP
<i>Bufo boreas</i>	western toad	BUBO
<i>Cnemidophorus hyperythrus</i>	orange throated whiptail	CNHY
<i>Cnemidophorus tigris</i>	western whiptail	CNTI
<i>Crotalus ruber</i>	red diamond rattlesnake	CRRU
<i>Crotalus viridis</i>	southern Pacific rattlesnake	CRVI
<i>Diadophis punctatus</i>	western ringneck snake	DIPU
<i>Elgaria multicarinatus</i>	southern alligator lizard	ELMU
<i>Eumeces skiltonianus</i>	western skink	EUSK
<i>Hyla regilla</i>	Pacific treefrog	HYRE

<i>Hypsiglena torquata</i>	Night snake	HYTO
<i>Lampropeltis getula</i>	California kingsnake	LAGE
<i>Masticophis lateralis</i>	striped racer	MALA
<i>Phrynosoma coronatum</i>	coast horned lizard	PHCO
<i>Pituophis melanoleucus</i>	San Diego gopher snake	PIME
<i>Scaphiopus hammondi</i>	western spadefoot toad	SCHA
<i>Sceloporus occidentalis</i>	western fence lizard	SCOC
<i>Uta stansburiana</i>)	side-blotched lizar	UTST
Small Mammal species		
<i>Chaetodipus californicus</i>	California pocket mouse	CHCA
<i>Chaetodipus fallax</i>	San Diego pocket mouse	CHFA
<i>Dipodomys simulans</i>	Dulzura kangaroo rat	DISI
<i>Microtus californicus</i>	California vole	MICA
<i>Mus musculus</i>	house mouse	MUMU
<i>Neotoma fuscipes</i>	dusky-footed woodrat	NEFU
<i>Neotoma lepida</i>	desert woodrat	NELE
<i>Notiosorex crawfordi</i>	desert shrew	NOCR
<i>Peromyscus californicus</i>	California mouse	PECA
<i>Peromyscus eremicus</i>	cactus mouse	PEER
<i>Peromyscus maniculatus</i>	deer mouse	PEMA
<i>Rattus rattus</i>	black rat	RARA
<i>Reithrodontomys megalotis</i>	harvest mouse	REME
<i>Thomomys bottae</i>	Botta's gopher	THBO

Table 2. Summary information for native plant species analyzed in the study.

Native Plant Species	Common name	Code	Family	Life history	Growth form 1	Growth form 2	Full Form 1
<i>Adenostoma fasciculatum</i>	chamise	ADEFAS	ROSACEAE	P	S	W	NPS
<i>Artemisia californica</i>	coastal sagebrush	ARTCAL	ASTERACEAE	P	S	W	NPS
<i>Baccharis pilularis</i>	coyote bush	BACPIL	ASTERACEAE	P	S	W	NPS
<i>Baccharis salicifolia</i>	mule fat, seep willow	BACSAL	ASTERACEAE	P	S	W	NPS
<i>Baccharis sarothroides</i>	broom baccharis	BACSAR	ASTERACEAE	P	S	W	NPS
<i>Bebbia juncea</i>	rush sweetbush	BEBJUN	ASTERACEAE	P	S	W	NPS
<i>Cneoridium dumosum</i>	coast spice bush; bushrue	CNEDUM	RUTACEAE	P	S	W	NPS
<i>Encelia californica</i>	California encelia	ENCCAL	ASTERACEAE	P	S	W	NPS

<i>Ericameria</i>		ERICA	ASTERACEAE	P	S	W	NPS
<i>Eriophyllum confertiflorum</i>	golden-yarrow	ERICON	ASTERACEAE	P	HS	W	NPHS
<i>Eriogonum fasciculatum</i>	California buckwheat	ERIFAS	POLYGONACEAE	P	S	W	NPS
<i>Ericameria palmeri</i> var. <i>pachylepis</i>	broad-scaled Palmer's goldenbush, Palmer's rabbitbrush	ERIPALP1	ASTERACEAE	P	S	W	NPS
<i>Ericameria pinifolia</i>	pine-bush	ERIPIN	ASTERACEAE	P	S	W	NPS
<i>Galium angustifolium</i>	narrow-leaf bedstraw	GALANG	RUBIACEAE	P	HS	W	NPHS
<i>Gutierrezia sarothrae</i>	San Joaquin matchweed	GUTSAR	ASTERACEAE	P	HS	W	NPHS
<i>Hazardia squarrosa</i>	sawtooth goldenbush	HAZSQU	ASTERACEAE	P	S	W	NPS
<i>Heteromeles arbutifolia</i>	christmas berry, toyon	HETARB	ROSACEAE	P	S	W	NPS
<i>Isomeris arborea</i>	bladderpod	ISOARB	CAPPARACEAE	P	S	W	NPS
<i>Isocoma menziesii</i>	goldenbush	ISOMEN	ASTERACEAE	P	HS	W	NPHS
<i>Juglans californica</i>	California black walnut	JUGCAL	JUGLANDACEAE	P	T	W	NPT
<i>Keckiella antirrhinoides</i>	bush penstemon	KECANT	SCROPHULARIACEAE	P	S	W	NPS
<i>Keckiella cordifolia</i>	climbing bush penstemon	KECCOR	SCROPHULARIACEAE	P	S	W	NPS
<i>Lonicera subspicata</i>	honeysuckle	LONSUB	CAPRIFOLIACEAE	P	VS	W	NPVS
<i>Lotus scoparius</i>	California broom	LOTSCO	FABACEAE	P	HS	W	NPHS
<i>Lupinus albifrons</i>	silver lupine, silver bush lupine	LUPALB	FABACEAE	P	HS	W	NPHS
<i>Malacothamnus fasciculatus</i>	chaparral mallow	MALFAS	MALVACEAE	P	S	W	NPS
<i>Malosma laurina</i>	laurel sumac	MALLAU	ANACARDIACEAE	P	S	W	NPS
<i>Mimulus aurantiacus</i>	bush monkey flower	MIMAU	SCROPHULARIACEAE	P	S	W	NPS
<i>Opuntia littoralis</i>	coastal prickly pear	OPULIT	CACTACEAE	P	S	W	NPS
<i>Opuntia oricola</i>	prickly-pear	OPUORI	CACTACEAE	P	S	W	NPS
<i>Opuntia prolifera</i>	coastal cholla	OPUPRO	CACTACEAE	P	S	W	NPS
<i>Polygala cornuta</i> var. <i>fishiae</i>	Fish's milkwort	POLCORF	POLYGALACEAE	P	S	W	NPS
<i>Prunus ilicifolia</i>	holly-leaf cherry	PRUILI	ROSACEAE	P	S	W	NPS
<i>Quercus agrifolia</i>	coast live oak; encina	QUEAGR	FAGACEAE	P	T	W	NPT

<i>Quercus berberidifolia</i>	scrub oak	QUEBER	FAGACEAE	P	S	W	NPS
<i>Rhamnus crocea</i>	spiny redberry	RHACRO	RHAMNACEAE	P	S	W	NPS
<i>Rhamnus ilicifolia</i>	holly-leaf redberry	RHAILI	RHAMNACEAE	P	S	W	NPS
<i>Rhus integrifolia</i>	lemonadeberry	RHUINT	ANACARDIACEAE	P	S	W	NPS
<i>Ribes sp.</i>	gooseberry	RIBES	GROSSULARIACEAE	P	S	W	NPS
<i>Ribes speciosum</i>	fuchsia-flower gooseberry	RIBSPE	GROSSULARIACEAE	P	S	W	NPS
<i>Salvia apiana</i>	white sage	SALAPI	LAMIACEAE	P	S	W	NPS
<i>Salvia leucophylla</i>	purple sage	SALLEU	LAMIACEAE	P	S	W	NPS
<i>Salvia mellifera</i>	black sage	SALMEL	LAMIACEAE	P	S	W	NPS
<i>Sambucus mexicana</i>	blue/desert elderberry	SAMMEX	CAPRIFOLIACEAE	P	S	W	NPS
<i>Tetradymia comosa</i>	cotton-thorn, hairy horsebrush	TETCOM	ASTERACEAE	P	S	W	NPS
<i>Toxicodendron diversilobum</i>	western poison oak	TOXDIV	ANACARDIACEAE	P	S	W	NPS
<i>Viguiera laciniata</i>	San Diego sunflower	VIGLAC	ASTERACEAE	P	S	W	NPS
<i>Achillea millefolium</i>	yarrow	ACHMIL	ASTERACEAE	P	F	F	NPF
<i>Acourtia microcephala</i>	sacapellote	ACOMIC	ASTERACEAE	P	F	F	NPF
<i>Allium haematochiton</i>	red-skin onion	ALLHAE	LILIACEAE	P	F	F	NPF
<i>Allium praecox</i>	early onion	ALLPRA	LILIACEAE	P	F	F	NPF
<i>Ambrosia acanthicarpa</i>	annual bur- sage/weed	AMBACA	ASTERACEAE	A	F	F	NAF
<i>Ambrosia psilostachya</i>	western ragweed	AMBPSI	ASTERACEAE	P	F	F	NPF
<i>Amsinckia menziesii</i>	ranchers fireweed	AMSMEN	BORAGINACEAE	A	F	F	NAF
<i>Antirrhinum nuttallianum</i>	Nuttall's snapdragon	ANTNUT	SCROPHULARIACEAE	A	F	F	NAF
<i>Aphanes occidentalis</i>	western lady's mantle	APHOCC	ROSACEAE	A	F	F	NAF
<i>Apiastrum angustifolium</i>	mock parsley	APIANG	APIACEAE	A	F	F	NAF
<i>Asclepias fascicularis</i>	narrow leaf milkweed	ASCFAS	ASCLEPIADACEAE	P	F	F	NPF
<i>Astragalus gambelianus</i>	Gambel's locoweed	ASTGAM	FABACEAE	A	F	F	NAF
<i>Astragalus trichopodus</i>	ocean locoweed	ASTTRI	FABACEAE	P	F	F	NPF
<i>Bloomeria crocea</i>	golden stars	BLOCRO	LILIACEAE	P	F	F	NPF
<i>Bowlesia incana</i>	american bowlesia	BOWINC	APIACEAE	A	F	F	NAF

<i>Calandrinia ciliata</i>	red maids	CALCIL	PORTULACACEAE	A	F	F	NAF
<i>Calystegia macrostegia</i>	morning glory	CALMAC	CONVOLVULACEAE	P	VF	VF	NPVF
<i>Calochortus sp.</i>	mariposa-lily	CALOC	LILIACEAE	P	F	F	NPF
<i>Calochortus splendens</i>	splendid mariposa	CALSPL	LILIACEAE	P	F	F	NPF
<i>Calochortus weedii</i> var. <i>weedii</i>	Weed's mariposa lily	CALWEEW	LILIACEAE	P	F	F	NPF
<i>Camissonia bistorta</i>	California/southern sun cup	CAMBIS	ONAGRACEAE	A	F	F	NAF
<i>Camissonia californica</i>	false-mustard	CAMCAL	ONAGRACEAE	A	F	F	NAF
<i>Camissonia hirtella</i>	field sun cup	CAMHIR	ONAGRACEAE	A	F	F	NAF
<i>Camissonia sp.</i>	suncup	CAMIS	ONAGRACEAE	X	F	F	NXF
<i>Camissonia robusta</i>	robust suncup	CAMROB	ONAGRACEAE	A	F	F	NAF
<i>Cardionema ramosissima</i>	tread lightly, beach sand mat	CARRAM	CARYOPHYLLACEAE	P	F	F	NPF
<i>Castilleja affinis</i>	coast paint-brush	CASAFF	SCROPHULARIACEAE	P	F	F	NPF
<i>Castilleja exserta</i>	purple owl's-clover	CASEXS	SCROPHULARIACEAE	A	F	F	NAF
<i>Centaurium venustum</i>	canchalagua	CENVEN	GENTIANACEAE	A	F	F	NAF
<i>Chamaesyce polycarpa</i>	small-seeded spurge, smallseed sandmat	CHAPOL	EUPHORBIACEAE	P	F	F	NPF
<i>Chenopodium californicum</i>	California goosefoot, pigweed	CHECAL	CHENOPODIACEAE	P	F	F	NPF
<i>Chlorogalum parviflorum</i>	small-flower soap-plant; amole	CHLPAR	LILIACEAE	P	F	F	NPF
<i>Chlorogalum pomeridianum</i>	wavy-leaf soap-plant	CHLPOM	LILIACEAE	P	F	F	NPF
<i>Chorizanthe procumbens</i> var. <i>albiflora</i>	Pala spine-flower	CHOPROA	POLYGONACEAE	A	F	F	NAF
<i>Cirsium occidentale</i>	cobwebby thistle, western thistle	CIROCC	ASTERACEAE	A	F	F	NAF
<i>Claytonia perfoliata</i>	miner's lettuce	CLAPER	PORTULACACEAE	A	F	F	NAF
<i>Clarkia purpurea</i>	purple clarkia, winecup clarkia	CLAPUR	ONAGRACEAE	A	F	F	NAF
<i>Crassula connata</i>	pygmy weed	CRACON	CRASSULACEAE	A	F	F	NAF
<i>Cryptantha intermedia</i>	nievitas cryptantha	CRYINT	BORAGINACEAE	A	F	F	NAF

<i>Cryptantha microstachys</i>	tejon cryptantha	CRYMIC2	BORAGINACEAE	A	F	F	NAF
<i>Cryptantha sp.</i>	cryptantha	CRYPT	BORAGINACEAE	A	F	F	NAF
<i>Cucurbita foetidissima</i>	calabazilla	CUCFOE	CUCURBITACEAE	P	VF	VF	NPVF
<i>Cuscuta californica</i>	dodder; witch's hair	CUSCAL	CUSCUTACEAE	A	VF	VF	NAVF
<i>Cuscuta californica var. californica</i>	dodder; witch's hair	CUSCALC	CUSCUTACEAE	A	VF	VF	NAVF
<i>Cuscuta sp.</i>		CUSCU	CUSCUTACEAE	A	VF	VF	NAVF
<i>Datura wrightii</i>	western jimsonweed, sacred thorn-apple	DATWRI	SOLANACEAE	P	F	F	NPF
<i>Daucus pusillus</i>	rattlesnake weed	DAUPUS	APIACEAE	A	F	F	NAF
<i>Delphinium parryi</i>	Parry's larkspur	DELPAR	RANUNCULACEAE	P	F	F	NPF
<i>Dichelostemma capitatum</i>	blue dicks	DICCAP	LILIACEAE	P	F	F	NPF
<i>Dichondra occidentalis</i>	western dichondra; western ponyfoot	DICOCC	CONVOLVULACEAE	P	F	F	NPF
<i>Dodecatheon clevelandii</i>	Padre's shooting star	DODCLE	PRIMULACEAE	P	F	F	NPF
<i>Dudleya lanceolata</i>	coastal dudleya	DUDLAN	CRASSULACEAE	P	F	F	NPF
<i>Dudleya pulverulenta</i>	chalk-lettuce	DUDPUL	CRASSULACEAE	P	F	F	NPF
<i>Emmenanthe penduliflora</i>	whispering bells	EMMPEN	HYDROPHYLLACEAE	A	F	F	NAF
<i>Epilobium canum</i>	California-fuchsia	EPICAN	ONAGRACEAE	P	F	F	NPF
<i>Eremocarpus setigerus</i>	doveweed	ERESET	EUPHORBIACEAE	A	F	F	NAF
<i>Erigeron foliosus</i>	leafy daisy	ERIFOL	ASTERACEAE	P	F	F	NPF
<i>Eriastrum sapphirinum</i>	wooly-star	ERISAP	POLEMONIACEAE	A	F	F	NAF
<i>Eschscholzia californica</i>	California poppy	ESCCAL	PAPAVERACEAE	A	F	F	NAF
<i>Eucrypta chrysanthemifolia</i>	spotted hideseed, common eucrypta	EUCCHR	HYDROPHYLLACEAE	A	F	F	NAF
<i>Filago arizonica</i>	Arizona herba impia, Arizona cottonrose	FILARI	ASTERACEAE	A	F	F	NAF
<i>Filago californica</i>	California filago	FILCAL	ASTERACEAE	A	F	F	NAF
<i>Galium nuttallii</i>	san diego bedstraw	GALNUT	RUBIACEAE	P	VF	F	NPVF
<i>Gilia angelensis</i>	grassland gilia	GILANG	POLEMONIACEAE	A	F	F	NAF
<i>Githopsis diffusa ssp. filicaulis</i>	San Gabriel bluecup, mission canyon bluecup	GITDIFF	CAMPANULACEAE	A	F	F	NAF

<i>Gnaphalium bicolor</i>	bicolor cudweed	GNABIC	ASTERACEAE	P	F	F	NPF
<i>Gnaphalium californicum</i>	California everlasting	GNACAL	ASTERACEAE	A	F	F	NAF
<i>Gnaphalium canescens</i> ssp. <i>beneolens</i>	fragrant everlasting	GNACANB	ASTERACEAE	P	F	F	NPF
<i>Gnaphalium palustre</i>	lowland cudweed	GNAPAL	ASTERACEAE	A	F	F	NAF
<i>Gnaphalium stramineum</i>	cotton-batting plant	GNASTR	ASTERACEAE	A	F	F	NAF
<i>Grindelia camporum</i> var. <i>bracteosum</i>	rayless gumplant	GRICAMB	ASTERACEAE	P	F	F	NPF
<i>Guillenia lasiophylla</i>	California mustard	GUILAS	BRASSICACEAE	A	F	F	NAF
<i>Harpagonella palmeri</i>	Palmer's grappling-hook	HARPAL	BORAGINACEAE	A	F	F	NAF
<i>Hemizonia fasciculata</i>	fascicled tarweed	HEMFAS	ASTERACEAE	A	F	F	NAF
<i>Hemizonia</i> sp.		HEMIZ	ASTERACEAE	A	F	F	NAF
<i>Hemizonia paniculata</i>	San Diego tarweed	HEMPAN	ASTERACEAE	A	F	F	NAF
<i>Hesperocnide tenella</i>	western nettle	HESTEN	URTICACEAE	A	F	F	NAF
<i>Heterotheca grandiflora</i>	telegraph weed	HETGRA	ASTERACEAE	A	F	F	NAF
<i>Jepsonia parryi</i>	coast jepsonia	JEPPAR	SAXIFRAGACEAE	P	F	F	NPF
<i>Lasthenia californica</i>	common goldfields	LASCAL	ASTERACEAE	A	F	F	NAF
<i>Lastarriaea coriacea</i>	lastarriaea	LASCOR	POLYGONACEAE	A	F	F	NAF
<i>Lepidium nitidum</i>	shining peppergrass	LEPNIT	BRASSICACEAE	A	F	F	NAF
<i>Lessingia filaginifolia</i>	California-aster; cudweed-aster	LESFIL	ASTERACEAE	P	F	F	NPF
<i>Linaria canadensis</i>	blue toadflax	LINCAN	SCROPHULARIACEAE	A	F	F	NAF
<i>Linanthus dianthiflorus</i>	farinose ground pink	LINDIA	POLEMONIACEAE	A	F	F	NAF
<i>Lotus hamatus</i>	grab lotus	LOTHAM	FABACEAE	A	F	F	NAF
<i>Lotus purshianus</i>	Spanish clover	LOTPUR	FABACEAE	A	F	F	NAF
<i>Lotus salsuginosus</i> var. <i>salsuginosus</i>	alkali lotus	LOTSALS	FABACEAE	A	F	F	NAF
<i>Lotus strigosus</i>	strigose lotus	LOTSTR	FABACEAE	A	F	F	NAF
<i>Lotus wrangelianus</i>	calf lotus	LOTWRA	FABACEAE	A	F	F	NAF
<i>Lupinus bicolor</i>	miniature lupine	LUPBIC	FABACEAE	A	F	F	NAF

<i>Lupinus concinnus</i>	bajada lupine	LUPCON	FABACEAE	A	F	F	NAF
<i>Lupinus hirsutissimus</i>	stinging lupine	LUPHIR	FABACEAE	A	F	F	NAF
<i>Lupinus microcarpus</i> var. <i>microcarpus</i>	valley lupine, chick lupine	LUPMICM	FABACEAE	A	F	F	NAF
<i>Lupinus succulentus</i>	arroyo lupine	LUPSUC	FABACEAE	A	F	F	NAF
<i>Lupinus truncatus</i>	collar lupine	LUPTRU	FABACEAE	A	F	F	NAF
<i>Malacothrix saxatilis</i>	cliff desert dandelion, cliff-aster	MALSAX	ASTERACEAE	P	F	F	NPF
<i>Marah macrocarpus</i>	cucamonga manroot, chilicothe	MARMAC	CUCURBITACEAE	P	VF	F	NPVF
<i>Micropus californicus</i> var. <i>californicus</i>	Q-tips, slender cottonweed	MICCALC	ASTERACEAE	A	F	F	NAF
<i>Mirabilis californica</i>	wishbone bush	MIRCAL	NYCTAGINACEAE	P	F	F	NPF
<i>Navarretia hamata</i>	skunkweed	NAVHAM	POLEMONIACEAE	A	F	F	NAF
<i>Nemacladus ramosissimus</i>	smallflower threadplant, nuttall's nemacladus	NEMRAM	CAMPANULACEAE	A	F	F	NAF
<i>Osmadenia tenella</i>	osmadenia	OSMTEN	ASTERACEAE	A	F	F	NAF
<i>Oxalis albicans</i>	California wood sorrel	OXAALB	OXALIDACEAE	P	F	F	NPF
<i>Paeonia californica</i>	California peony	PAECAL	PAEONIACEAE	P	F	F	NPF
<i>Parietaria hespera</i>	western pellitory	PARHES	URTICACEAE	A	F	F	NAF
<i>Parietaria hespera</i> var. <i>hespera</i>	rillita pellitory, pellitory	PARHESH	URTICACEAE	A	F	F	NAF
<i>Pectocarya linearis</i> ssp. <i>ferocula</i>	slender pectocarya	PECLINF	BORAGINACEAE	A	F	F	NAF
<i>Pellaea andromedaefolia</i>	coffee fern	PELAND	PTERIDACEAE	P	F	F	NPF
<i>Pellaea mucronata</i>	bird's foot cliff-brake	PELMUC	PTERIDACEAE	P	F	F	NPF
<i>Pentagramma triangularis</i>	goldenback fern, silverback fern	PENTRI	PTERIDACEAE	P	F	F	NPF
<i>Phacelia cicutaria</i>	caterpillar phacelia	PHACIC	HYDROPHYLLACEAE	A	F	F	NAF
<i>Phacelia distans</i>	wild-heliotrope	PHADIS	HYDROPHYLLACEAE	A	F	F	NAF
<i>Phacelia parryi</i>	Parry's phacelia	PHAPAR	HYDROPHYLLACEAE	A	F	F	NAF
<i>Phacelia</i>	branching phacelia	PHARAM	HYDROPHYLLACEAE	P	F	F	NPF

<i>ramosissima</i>							
<i>Pholistoma auritum</i>	fiesta flower	PHO AUR	HYDROPHYLLACEAE	A	F	F	NAF
<i>Pholistoma racemosum</i>	San Diego fiesta flower	PHORAC	HYDROPHYLLACEAE	A	VF	F	NAVF
<i>Plagiobothrys collinus</i>	popcornflower	PLACOL	BORAGINACEAE	A	F	F	NAF
<i>Plantago erecta</i>	dot-seed plantain	PLAERE	PLANTAGINACEAE	A	F	F	NAF
<i>Plagiobothrys nothofulvus</i>	rusty popcornflower	PLANOT	BORAGINACEAE	A	F	F	NAF
<i>Potentilla glandulosa</i>	cinquefoil	POTGLA	ROSACEAE	P	F	F	NPF
<i>Psilocarphus tenellus</i> var. <i>tenellus</i>	slender woolly-heads	PSITENT	ASTERACEAE	A	F	F	NAF
<i>Pterostegia drymarioides</i>	granny's hairnet	PTEDRY	POLYGONACEAE	A	F	F	NAF
<i>Rafinesquia californica</i>	California chicory	RAFCAL	ASTERACEAE	A	F	F	NAF
<i>Ranunculus californicus</i>	buttercup	RANCAL	RANUNCULACEAE	P	F	F	NPF
<i>Salvia columbariae</i>	chia	SALCOL	LAMIACEAE	A	F	F	NAF
<i>Sanicula arguta</i>	sharp-tooth sanicle	SANARG	APIACEAE	P	F	F	NPF
<i>Sanicula crassicaulis</i>	pacific sanicle	SANCRA	APIACEAE	P	F	F	NPF
<i>Sarcostemma cynanchoides</i>	milkvine	SARCYN	ASCLEPIADACEAE	P	VF	F	NPVF
<i>Scrophularia californica</i>	California bee plant; figwort	SCRCAL	SCROPHULARIACEAE	P	F	F	NPF
<i>Selaginella</i> sp.	mossfern	SELAG	SELAGINELLACEAE	P	F	F	NPF
<i>Selaginella bigelovii</i>	bigelow's mossfern	SELBIG	SELAGINELLACEAE	P	F	F	NPF
<i>Selaginella cinerascens</i>	mesa spike-moss/mossfern	SELCIN	SELAGINELLACEAE	P	F	F	NPF
<i>Sidalcea malvaeflora</i>	checker-bloom	SIDMAL	MALVACEAE	P	F	F	NPF
<i>Silene anthirrina</i>	snapdragon catchfly	SILANT	CARYOPHYLLACEAE	A	F	F	NAF
<i>Sisyrinchium bellum</i>	blue-eyed-grass	SISBEL	IRIDACEAE	P	F	F	NPF
<i>Solidago californica</i>	California goldenrod	SOLCAL	ASTERACEAE	P	F	F	NPF
<i>Solanum douglasii</i>	Douglas' nightshade	SOLDOU	SOLANACEAE	P	F	F	NPF
<i>Solanum parishii</i>	Parish's nightshade	SOLPAR	SOLANACEAE	P	F	F	NPF
<i>Solanum umbelliferum</i>	blue witch	SOLUMB	SOLANACEAE	P	F	F	NPF
<i>Solanum xanti</i>	purple nightshade,	SOLXAN	SOLANACEAE	P	F	F	NPF

	chaparral nightshade						
<i>Stachys ajugoides</i>	hedgenettle	STAAJU	LAMIACEAE	P	F	F	NPF
<i>Stephanomeria diegensis</i>	San Diego wreath plant	STEDIE	ASTERACEAE	A	F	F	NAF
<i>Stephanomeria exigua</i>	small wreath-plant	STEEXI	ASTERACEAE	A	F	F	NAF
<i>Stellaria nitens</i>	shining chickweed	STENIT	CARYOPHYLLACEAE	A	F	F	NAF
<i>Stephanomeria sp.</i>	stephanomeria	STEPH	ASTERACEAE	A	F	F	NAF
<i>Stephanomeria virgata</i>	virgate wreath plant	STEVIR	ASTERACEAE	A	F	F	NAF
<i>Stylocline gnaphalioides</i>	everlasting nest-straw	STYGNA	ASTERACEAE	A	F	F	NAF
<i>Triodanis biflora</i>	venus looking-glass	TRIBIF	CAMPANULACEAE	A	F	F	NAF
<i>Trifolium ciliolatum</i>	tree clover	TRICIL	FABACEAE	A	F	F	NAF
<i>Trifolium depauperatum var. truncatum</i>	dwarf sack clover	TRIDEPT	FABACEAE	A	F	F	NAF
<i>Trifolium sp.</i>		TRIFO	FABACEAE	A	F	F	NAF
<i>Trifolium gracilentum var. gracilentum</i>	pin point clover	TRIGRAG	FABACEAE	A	F	F	NAF
<i>Trichostema lanceolatum</i>	vinegar weed	TRILAN	LAMIACEAE	A	F	F	NAF
<i>Trifolium microcephalum</i>	maiden clover	TRIMIC	FABACEAE	A	F	F	NAF
<i>Trifolium wildenovii</i>	valley clover	TRIWIL	FABACEAE	A	F	F	NAF
<i>Uropappus lindleyi</i>	silver puffs	UROLIN	ASTERACEAE	A	F	F	NAF
<i>Verbena lasiostachys</i>	western vervain	VERLAS	VERBENACEAE	P	F	F	NPF
<i>Vicia americana var. americana</i>	american vetch	VICAMEA	FABACEAE	P	VF	F	NPVF
<i>Vicia hassei</i>	slender vetch	VICHAS	FABACEAE	A	VF	F	NAVF
<i>Vicia ludoviciana var. ludoviciana</i>	deerpea vetch	VICLUDL	FABACEAE	A	VF	F	NAVF
<i>Viola pedunculata</i>	johnny jump-up	VIOPED	VIOLACEAE	P	F	F	NPF
<i>Achnatherum coronata</i>	giant stipa	ACHCOR	POACEAE	P	G	G	NPG
<i>Achnatherum diegoense</i>	San Diego County needlegrass	ACHDIE	POACEAE	P	G	G	NPG
<i>Agrostis exarata</i>	spike red-top	AGREXA	POACEAE	P	G	G	NPG
<i>Agrostis sp.</i>	bent-grass	AGROS	POACEAE	P	G	G	NPG

<i>Agrostis pallens</i>	leafy-bent	AGRPAL	POACEAE	P	G	G	NPG
<i>Aristida adscensionis</i>	six-weeks three-awn	ARIADS	POACEAE	A	G	G	NAG
<i>Aristida purpurea</i>	purple three awn	ARIPUR	POACEAE	P	G	G	NPG
<i>Bothriochloa barbinodis</i>	cane bluestem; plumed beardgrass	BOTBAR	POACEAE	P	G	G	NPG
<i>Bromus carinatus</i>	California brome	BROCAR	POACEAE	P	G	G	NPG
<i>Distichlis spicata</i>	salt grass	DISSPI	POACEAE	P	G	G	NPG
<i>Elymus glaucus</i> ssp. <i>glaucus</i>	blue wild rye	ELYGLAG	POACEAE	P	G	G	NPG
<i>Hordeum depressum</i>	low barley	HORDEP	POACEAE	A	G	G	NAG
<i>Hordeum intercedens</i>	little barley	HORINT	POACEAE	A	G	G	NAG
<i>Juncus bufonius</i>	toad rush	JUNBUF	JUNCACEAE	A	G	G	NAG
<i>Juncus sp.</i>	rush	JUNCU	JUNCACEAE	P	G	G	NPG
<i>Juncus patens</i>	spreading rush	JUNPAT	JUNCACEAE	P	G	G	NPG
<i>Leymus condensatus</i>	giant wild rye	LEYCON	POACEAE	P	G	G	NPG
<i>Melica imperfecta</i>	coast range melic	MELIMP	POACEAE	P	G	G	NPG
<i>Muhlenbergia microsperma</i>	littleseed muhly	MUHMIC	POACEAE	A	G	G	NAG
<i>Nassella lepida</i>	foothill needlegrass/stipa	NASLEP	POACEAE	P	G	G	NPG
<i>Nassella pulchra</i>	purple needlegrass/stipa	NASPUL	POACEAE	P	G	G	NPG
<i>Nassella sp.</i>	needlegrass, stipa	NASSE	POACEAE	P	G	G	NPG
<i>Poa secunda</i>	malpais bluegrass	POASEC	POACEAE	P	G	G	NPG
<i>Vulpia octoflora</i>	six-weeks fescue, slender fescue	VULOCT	POACEAE	A	G	G	NAG

Ants

Species level data.

We generated no metrics from these data.

Presence-Absence Data

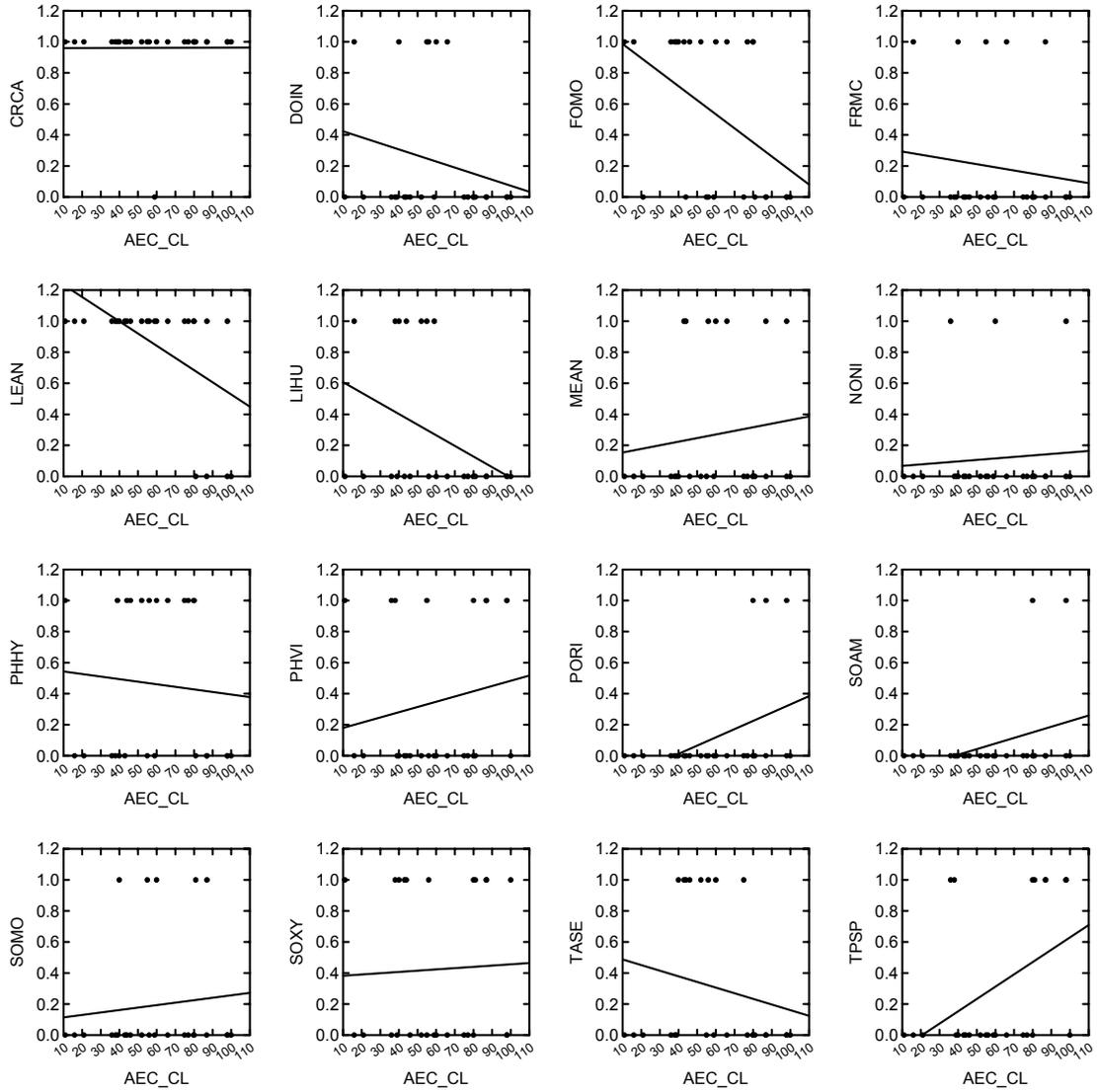


Figure 1. Presence-Absence by AEC for Ants. We screened only the 16 of 38 ant types identified to species.

Relative Abundance Data

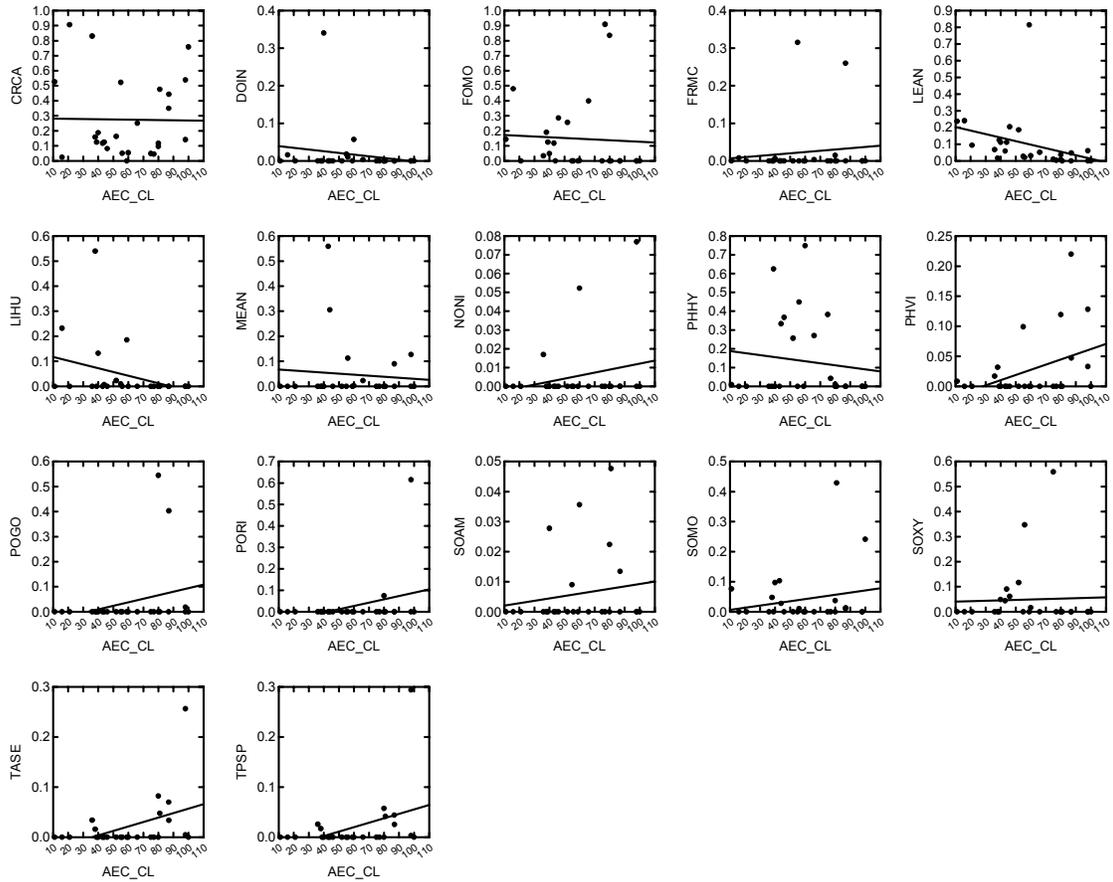


Figure 2. Relative abundance by AEC for Ants. We screened only the 16 of 38 ant types identified to species.

Genera Specific

We created 2 metrics from these data.

Presence-Absence Data.

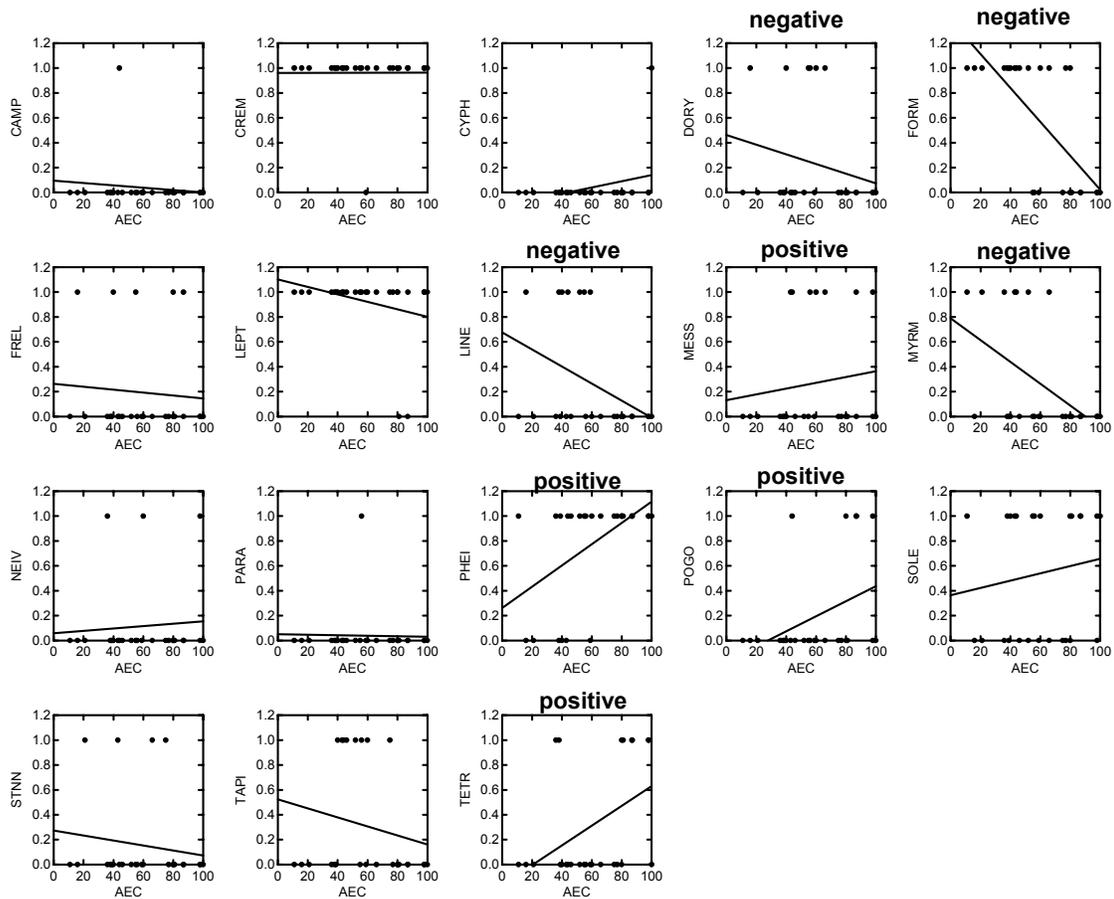


Figure 3. Presence-Absence by AEC for 18 Ant Genera.

Negative responses.

- DORY: Threshold response at ~65% AEC.
- FORM: Threshold response at ~80% AEC.
- LINE: Threshold response at ~60% AEC.
- MYRM: Threshold response at ~65% AEC.

Level of support based on existing knowledge: Speculative. It is possible soil disturbance from past agricultural can explain the absence of DORY and MYRM, but this was speculation from Dr. David Holway (UCSD). Argentine Ants (LINE) likely dropped out as our more disturbed sites are dry and away from mesic refuges (stream beds). We have no explanation for FORM.

Other possible genera. STNN and TAPI showed unimodal responses, not appearing at both high and low AEC.

Positive responses.

- MESS: Threshold response at ~40% AEC.

POGO: Threshold response at ~40% AEC.

TETR: Threshold response at ~35% AEC.

PHEI: Increase in proportional occupancy with increasing AEC.

Level of support based on existing knowledge: Positive responses may be caused by the granivorous diets of some of the ants in these genres. If exotic cover is correlated positively with seed availability, a likely scenario, then some seed-eating ants may well be more common in areas with exotic grasses.

Other possible species. SOLE increased in proportional occupancy with increasing AEC. However, the relationship was weak and SOLE was absent from sites with both low and high AEC.

Relative Abundance Data.

We generated no metrics from these data.

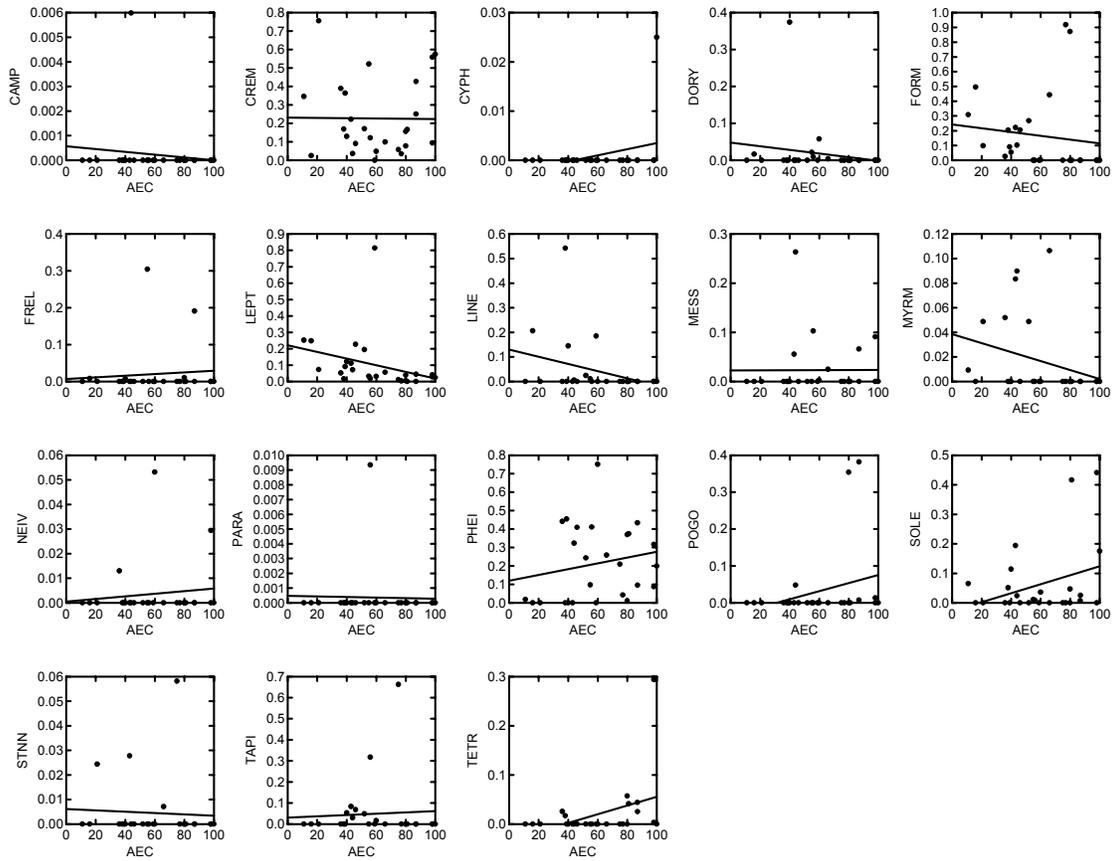
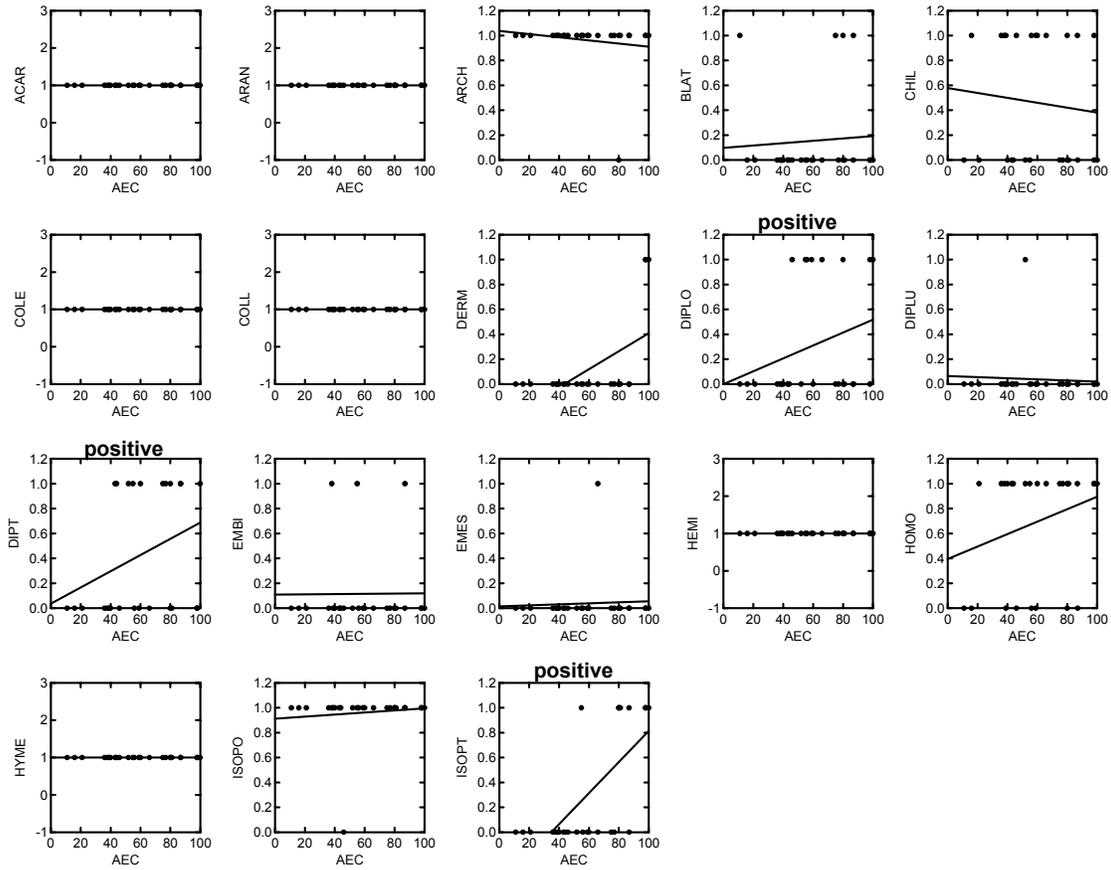


Figure 4. Relative abundance by AEC for Ant Genera.

Arthropods

Presence-Absence Data

1 metric created from these data: Proportion of tolerant species.



(Additional orders on the following page)

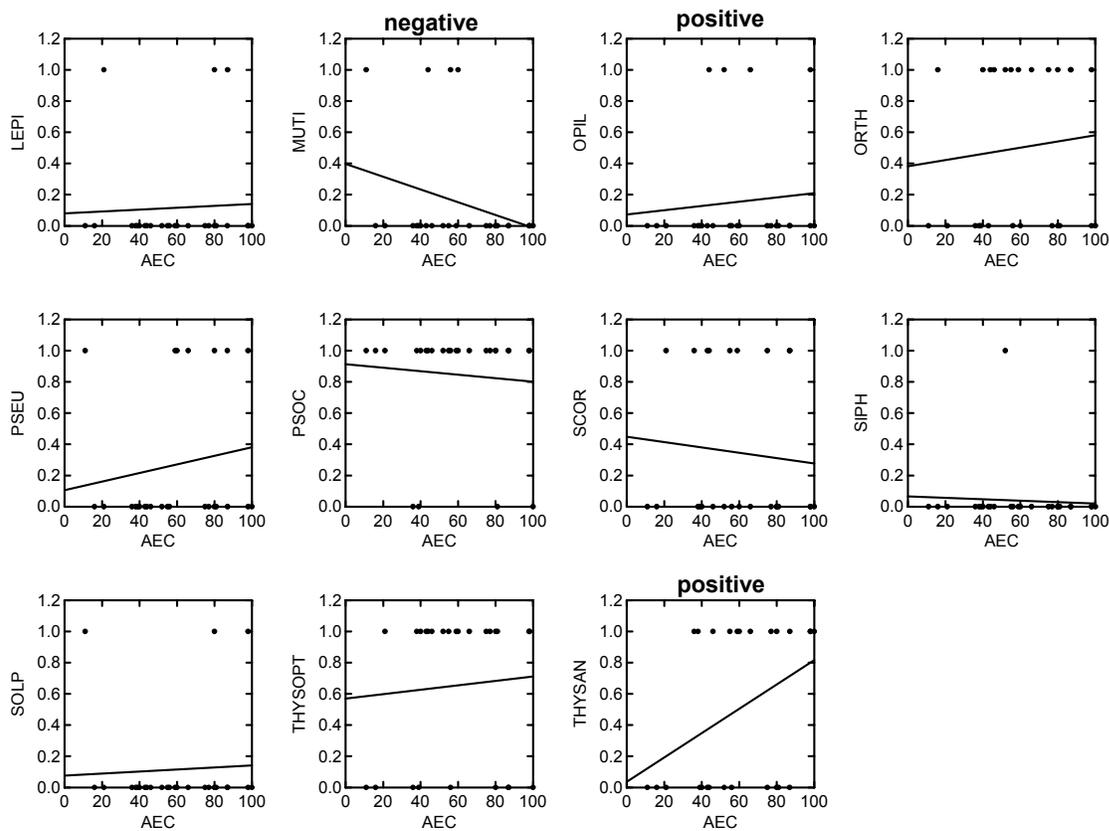


Figure 5. Presence-Absence by AEC for Arthropod Orders.

Negative responses.

MUTI: Threshold response at ~0.65 AEC.

Level of support based on existing knowledge: Basically none.

Other possible species. CHIL had a slight decline in the proportion of sites occupied with increasing AEC, the pattern was weak and in the year 2 dataset, CHIL was only found on sites with AEC above ~55% AEC. Surprisingly, the scorpions (SCOR) did not show declines with increasing AEC despite studies indicating they might (Appendix A).

Positive responses.

DIPLO: Threshold response at ~40% AEC.

DIPT: Threshold response at ~40% AEC.

ISOPT: Threshold response at ~50% AEC.

OPIL: Threshold response at ~40% AEC.

THYSAN: Threshold response at ~35% AEC.

Level of support based on existing knowledge: Basically unknown.

Other possible species. EMBI data were too sparse for confident prediction. Both ORTH and PSEU were found predominantly at sites with higher AEC. However, we also

observed sites with absences across the gradient. For example, PSEU was absent at more sites than present at high AEC.

Exotic species.

No metrics created from these data.

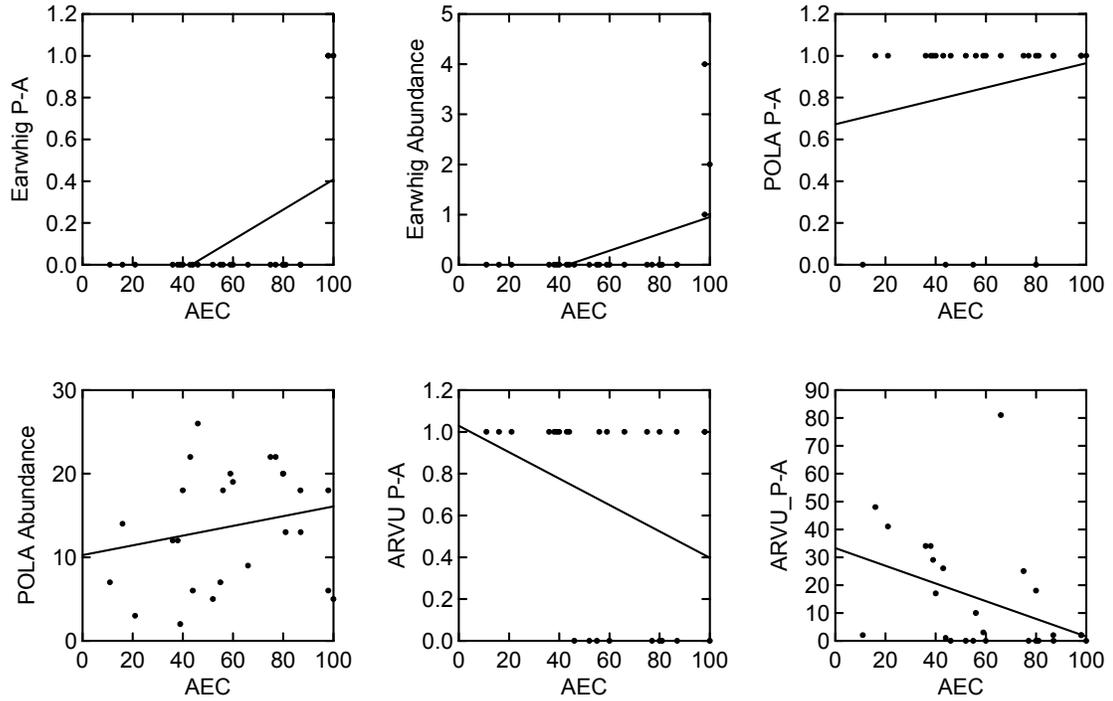


Figure 6. Presence Absence (P-A) and Abundance of 3 exotic species versus Absolute Exotic Cover for the “BOTH” data set.

Birds

Presence-Absence Data Year 1.

We generated 2 metrics from these data: We present year 1, year 2, and both year data, but only label species on the both year scatterplots.

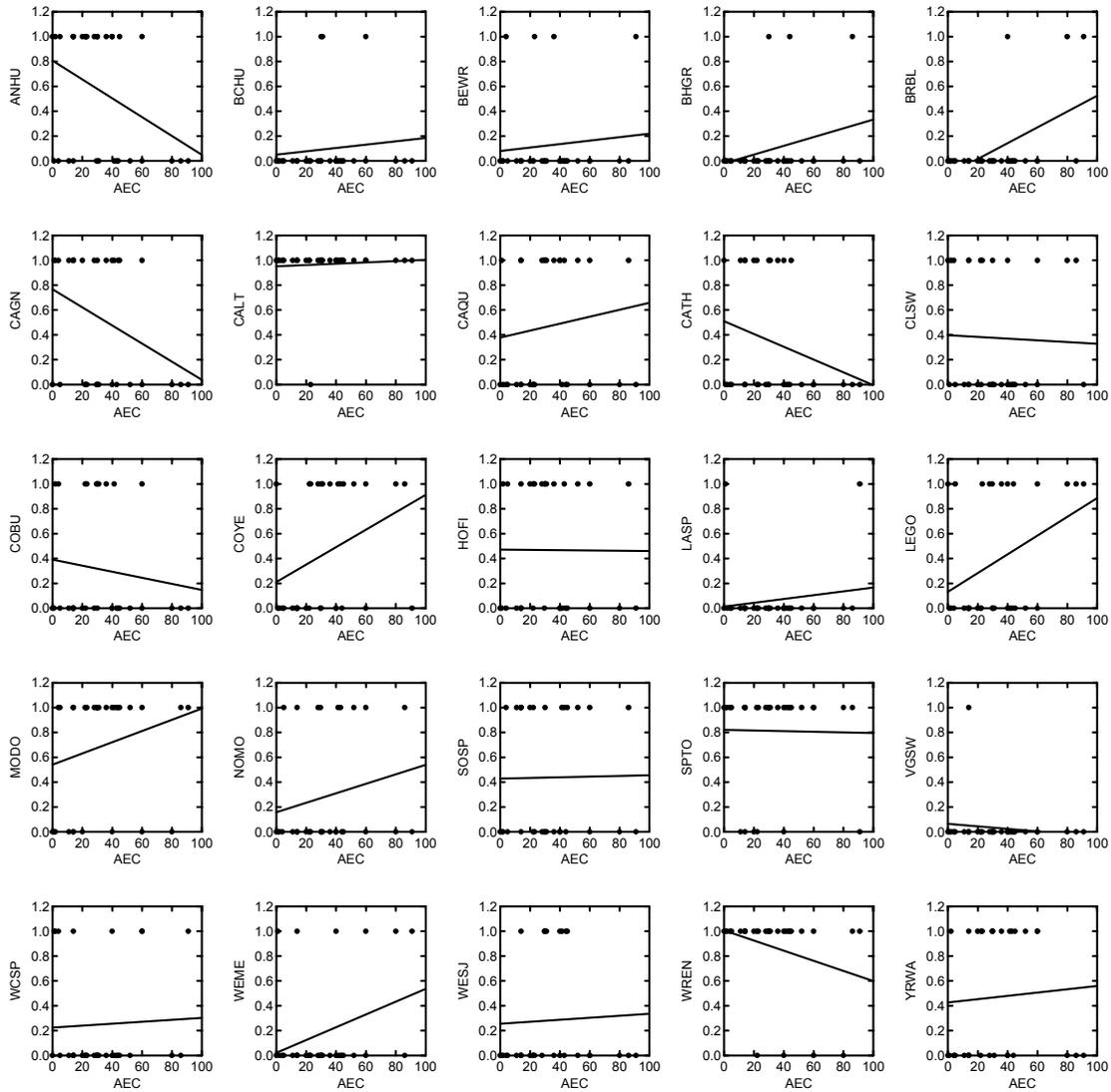


Figure 7. Presence-Absence versus AEC for Birds YEAR 1 Data.

Presence-Absence Data Year 2

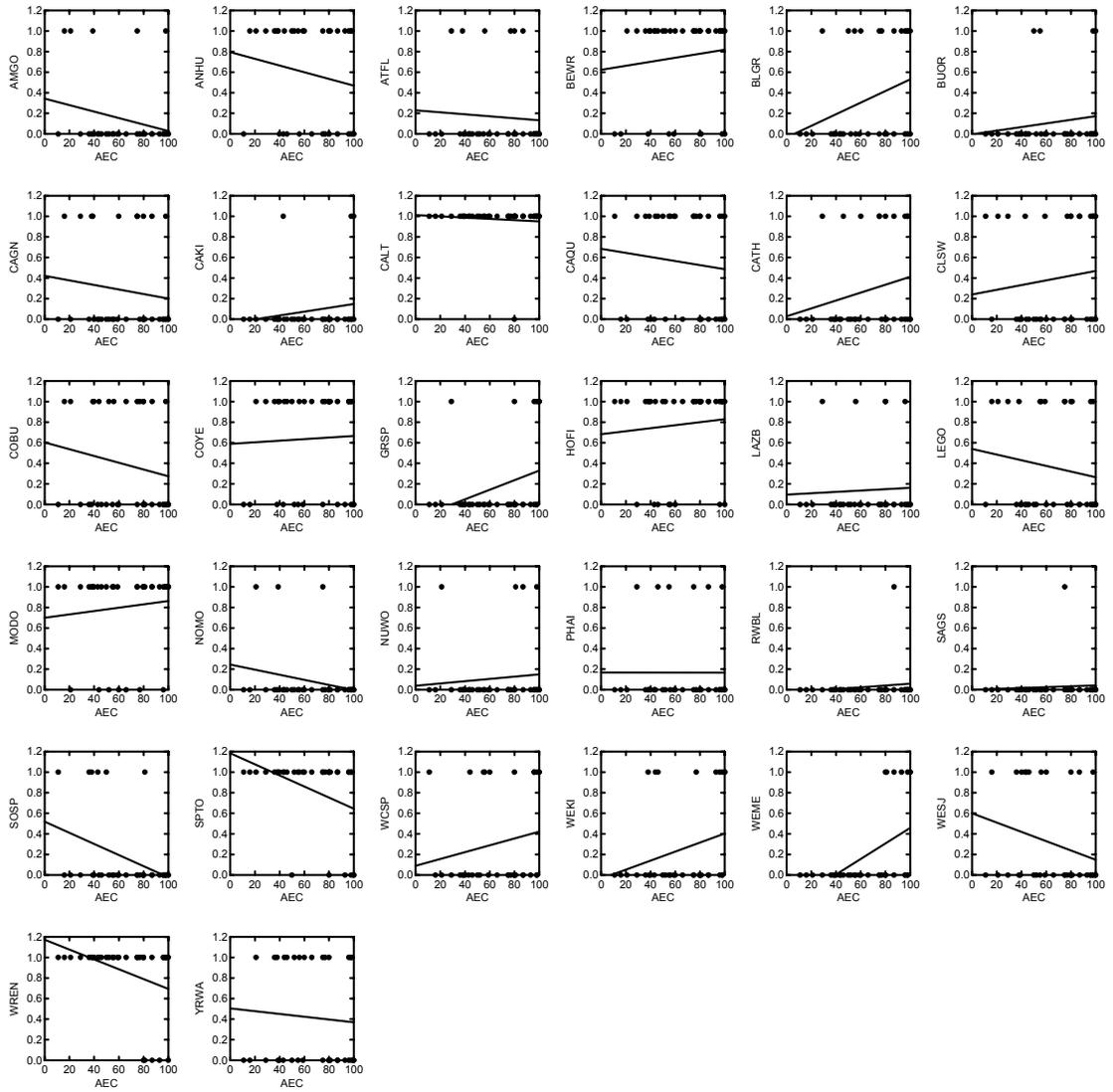


Figure 8. Presence-Absence versus AEC for Birds YEAR 2 Data.

Presence-Absence Data BOTH

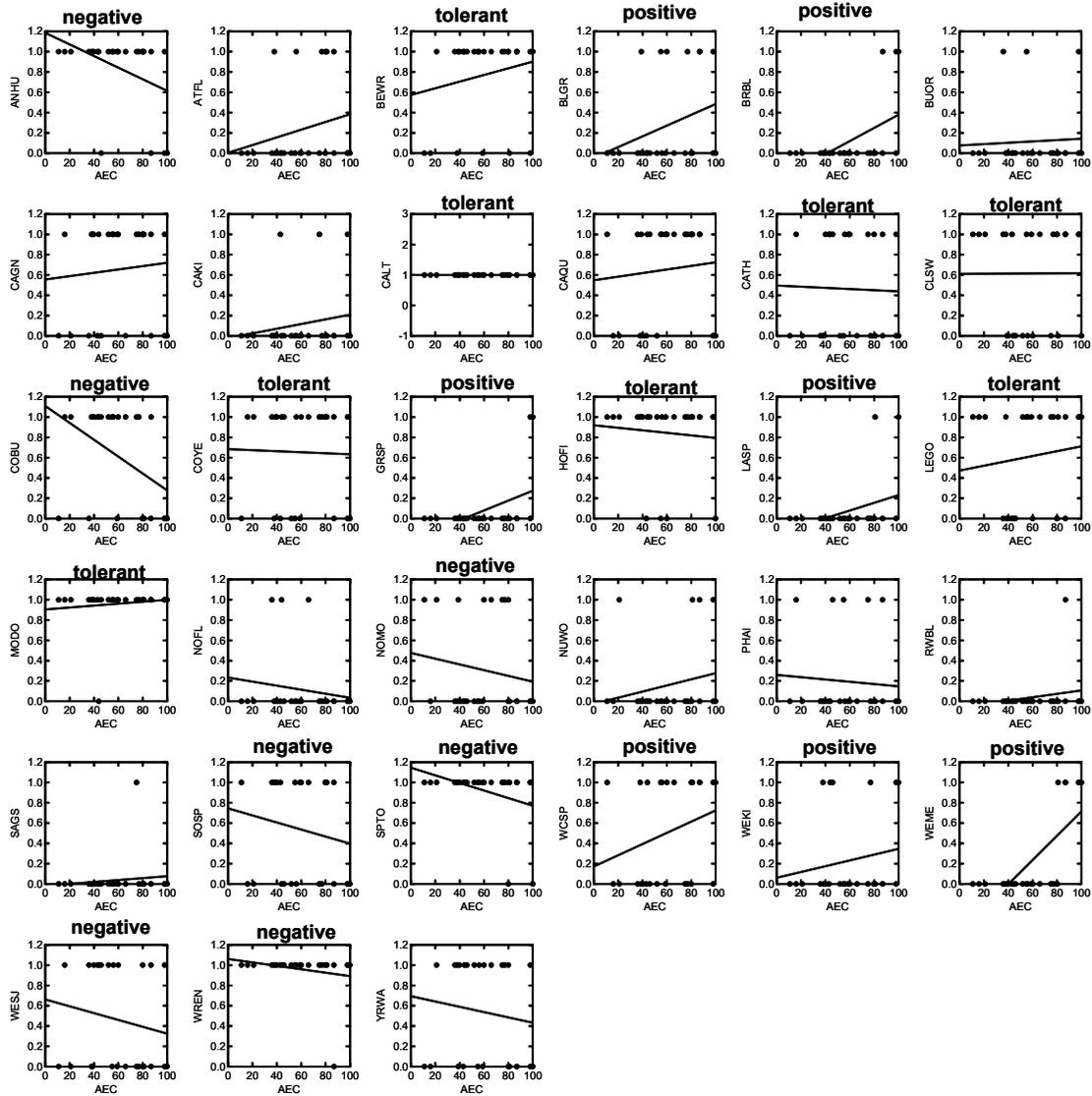


Figure 9 Presence-Absence versus AEC for Birds YEAR 2 Data.

Negative responses.

- ANHU. Threshold response in year 1, general decline in occupancy with increasing AEC.
- COBU. Threshold response in year 1, general decline in occupancy with increasing AEC.
- NOMO. General threshold response at ~75% AEC.
- SOSP. General threshold response at ~80% and general decline in occupancy with increasing AEC.
- SPTO. Decline in occupancy at high levels of AEC, particular in year-long data.

WESJ. General decline in occupancy with increasing AEC.

WREN. Decline in occupancy at high levels of AEC, particular in year-long data.

Level of support based on existing knowledge: The declines in occupancy by these species are expected given their prevalence in shrub-dominated habitats.

Other possible species. NOFL was simply too rare, and PHAI rarely occurs in CSS.

Positive responses.

BLGR. General increase in occupancy with increasing AEC. Some threshold signal.

BRBL. Threshold response at ~80% AEC.

GRSP. Threshold response at ~85% AEC.

LASP. Threshold response at ~75% AEC.

WCSP. General increase in occupancy with increasing AEC. Threshold response at ~40% AEC.

WEKI. General increase in occupancy with increasing AEC. Threshold response at ~35% AEC.

WEME. Threshold response at ~75% AEC.

Level of support based on existing knowledge: Support for the majority of these species is high. Many are known grassland birds.

Relative Abundance Year 1.

We generated 2 metrics from this variable. We present year 1, year 2, and both year data, but only label species on the both year scatterplots.

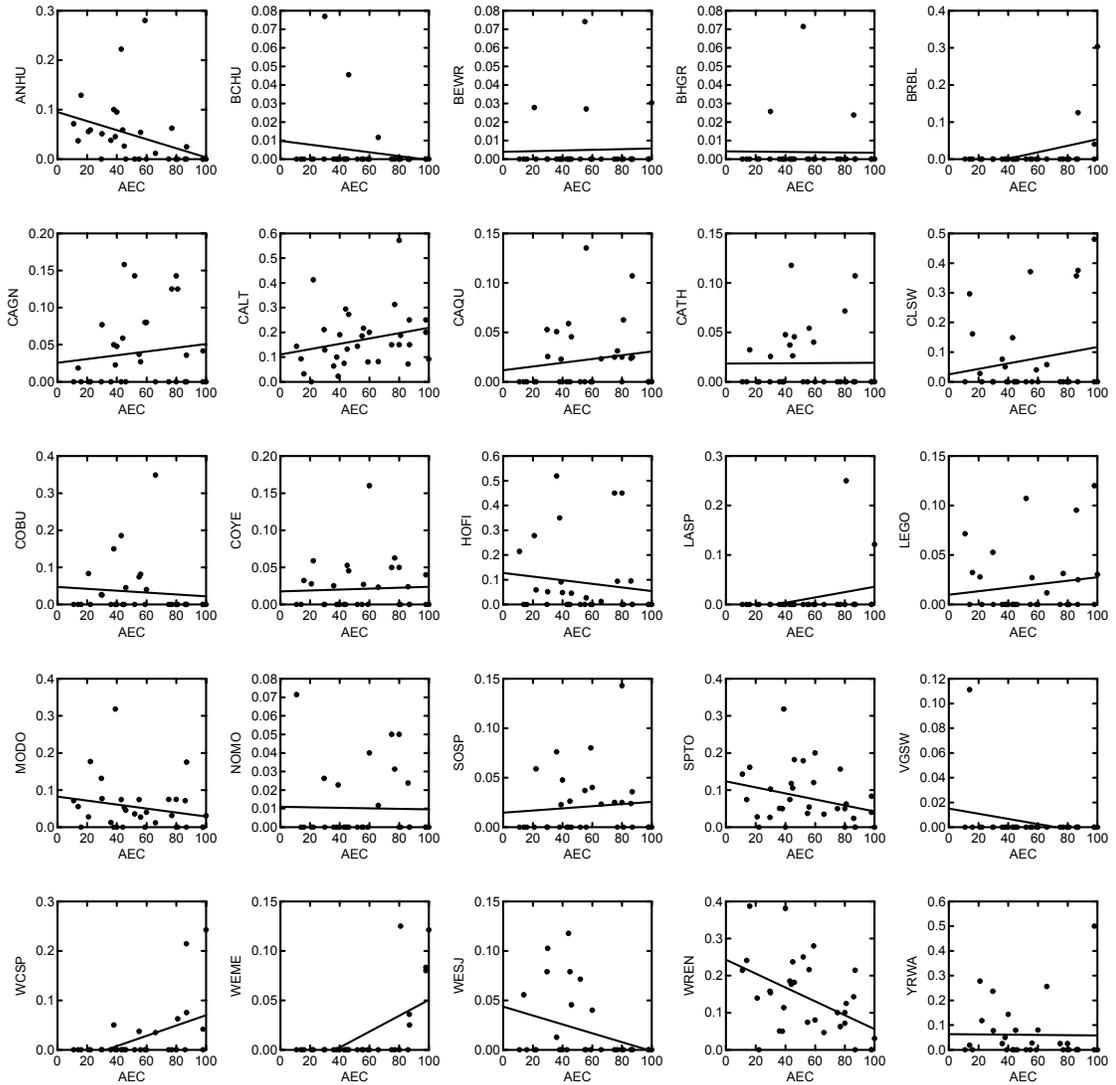


Figure 10. Relative Abundance versus AEC for Birds YEAR 1 Data.

Relative Abundance Year 2.

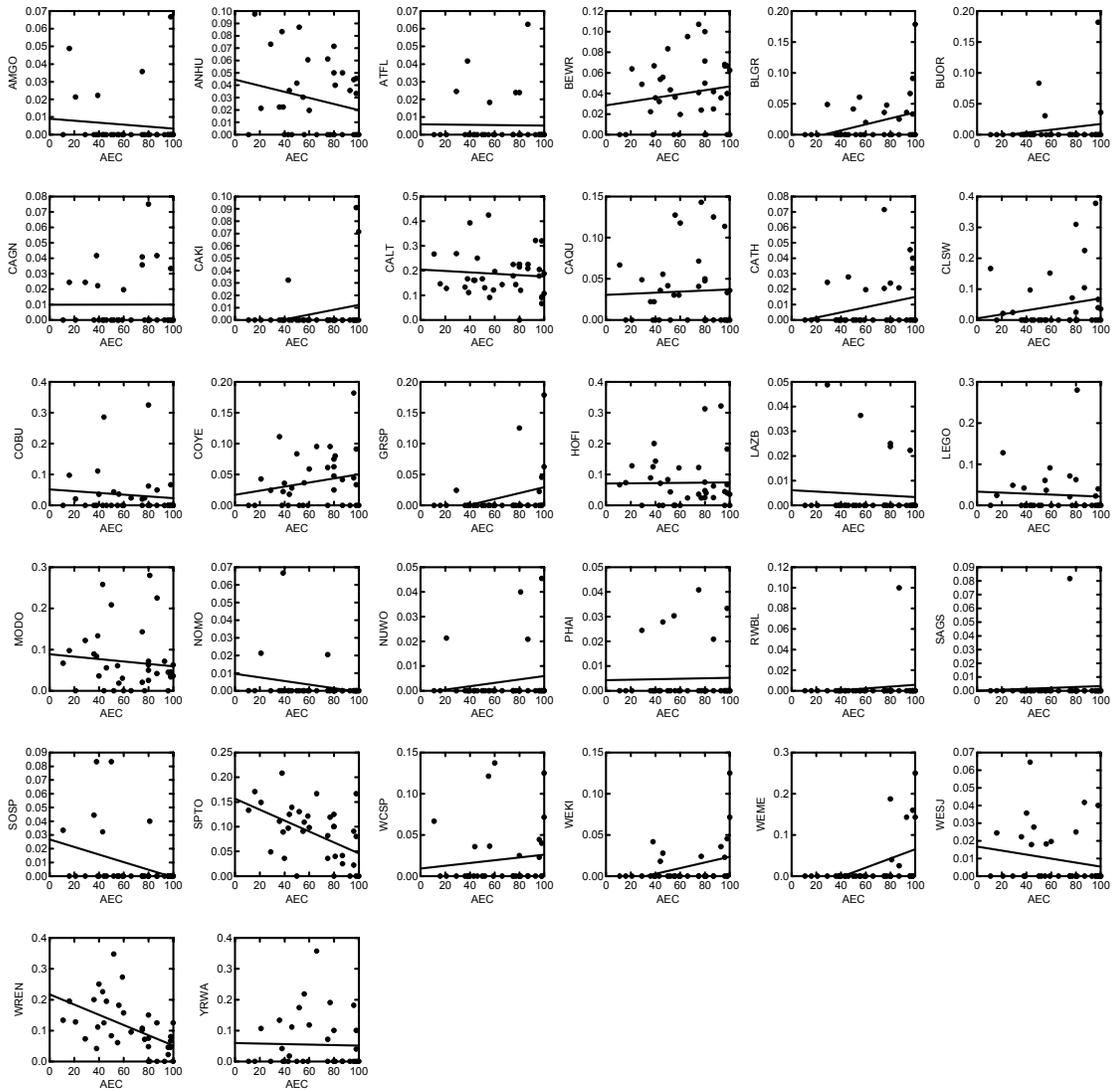


Figure 11. . Relative Abundance versus AEC for Birds YEAR 2 Data.

Relative Abundance Both.

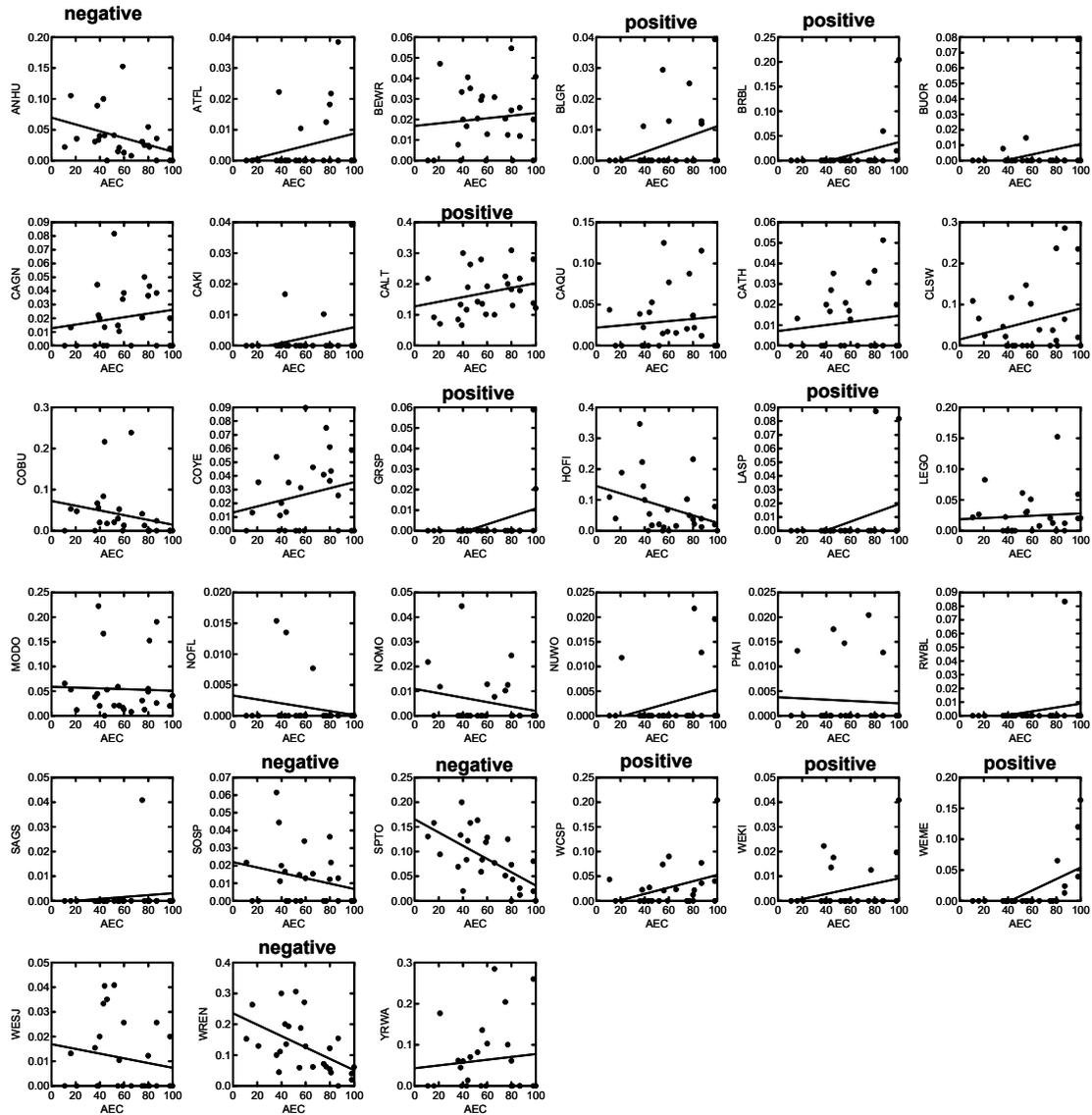


Figure 12. Relative Abundance versus AEC for Birds BOTH Data.

Negative responses.

ANHU. Decline in rel. abundance and maximum rel. abundance with increasing AEC.

SOSP. Decline in rel. abundance with increasing AEC.

SPTO. Decline in rel. abundance with increasing AEC.

WREN. Decline in rel. abundance with increasing AEC.

Level of support based on existing knowledge: The declines in relative abundance by these species was expected given their prevalence in shrub dominated habitats.

Other possible species. HOFI negative response in the BOTH dataset is not found in other datasets. NOFL is simply too rare while NOMO patterns are not consistent across datasets.

Positive responses.

- BLGR. Increase in rel. abundance with increasing AEC.
- BRBL. Threshold response at ~80% AEC.
- CALT. Increase in rel. abundance with increasing AEC.
- GRSP. Threshold response at ~85% AEC.
- LASP. Threshold response at ~75% AEC.
- WCSP. Increase in rel. abundance with increasing AEC.
- WEKI. Increase in rel. abundance with increasing AEC.
- WEME. Threshold response at ~75% AEC.

Level of support based on existing knowledge: Support for the majority of these species is high. Many are known grassland birds.

Other possible species.

ATFL. The overall response is unimodal, with an increase then decrease in relative abundance with increasing AEC.

Herpetofauna

Presence/Absence Data

2 metrics created from this variable.

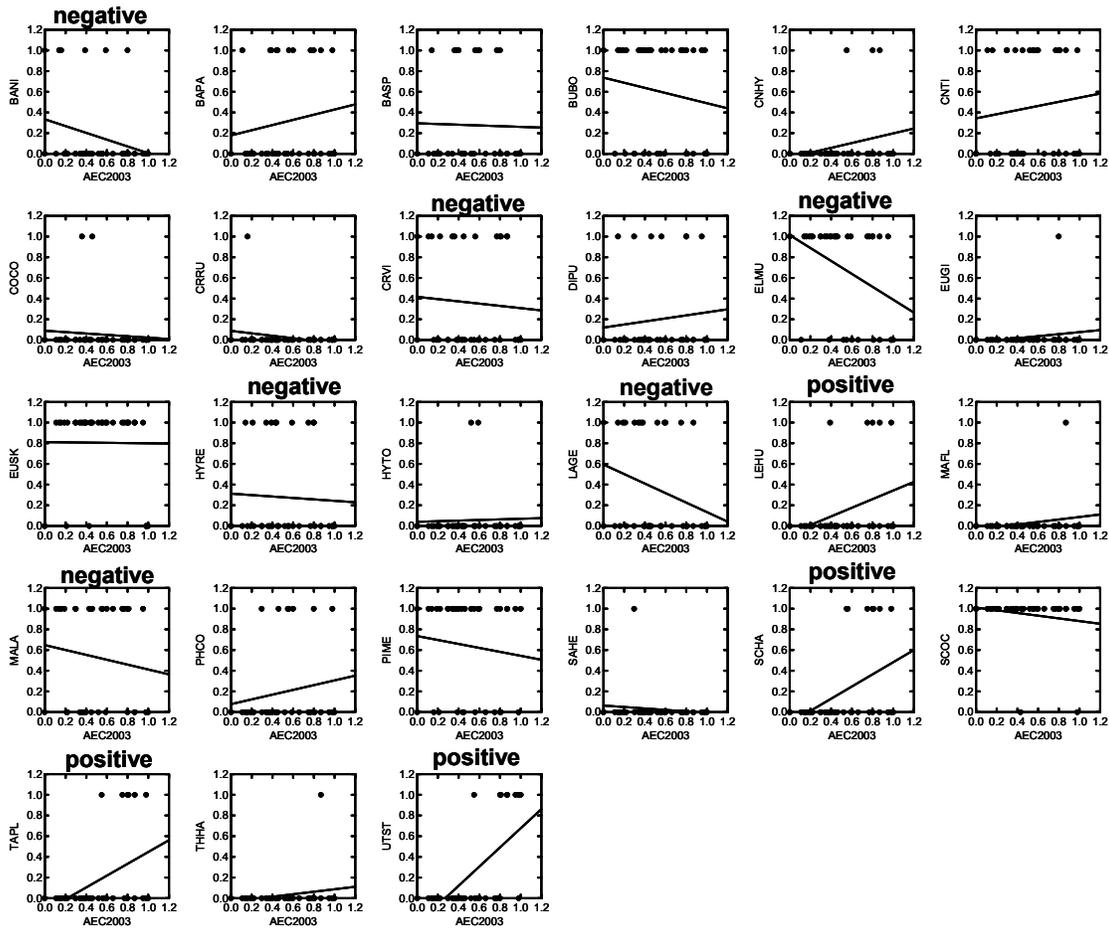


Figure 13. Presence-Absence versus AEC for Herpetofauna. Herp data collected from 1998 to 2003 on most sites.

Negative responses.

BANI: Negative threshold response disappearing from plots after ~0.80 AEC.

CRVI: Negative threshold response disappearing from plots after ~0.85 AEC

ELMU: Proportional change in presence with AEC. No threshold in Presence/Absence, but much higher proportions of sites present at low AEC than high.

HYRE: Negative threshold response disappearing from plots after ~0.85 AEC.

LAGE: Negative threshold response disappearing from plots after ~0.85 AEC and weak proportional response with lower proportion of sites occupied with increasing AEC.

MALA: Proportional change in presence with AEC.

Level of support based on existing knowledge: *Masticophis lateralis* is known to prefer CSS habitat over grasslands based on historic work and radiotracking data by a PhD student of Dr. Diffendorfer and Dr. Fisher (USGS). Thus, its response across the gradient is expected and its inclusion in a metric well supported. Support for the other species is not as strong. BANI, HYRE, ELMU, and LAGE, are considered associated with more mesic environments and more dense cover. In coastal southern California mesic environments tend to contain more leaf litter cover and perhaps more shrubs, though oak-woodlands (habitat we did not include in our plots) are typical associated with more mesic conditions. Perhaps these species do not select more open, drier conditions found at sites with higher amounts of exotic cover but detailed studies of the species habitat preferences are lacking. CRVI is a sit and wait, ambush predator who specializes on small mammals. Perhaps CRVI avoids open grasslands to escape predation by raptors or because they lack shaded areas for thermoregulation.

Positive responses.

- LEHU: Positive threshold response occurring after ~0.30 AEC.
- SCHA: Positive threshold response occurring after ~0.45 AEC.
- TAPL: Positive threshold response occurring after ~0.45 AEC.
- UTST: Positive threshold response occurring after ~0.45 AEC.

Level of support based on existing knowledge: LEHU, SCHA and UTST are associated with open habitats or habitats with loose soils so their absence from sites with dense shrub cover (sites with low AEC) is expected. TAPL is found in grasslands, but has also been reported to occur in chaparral and oak woodland. Thus, the response we observed for TAPL is not fully expected based on the literature.

Other possible species. BASP refers to individuals not separated to species, only to genus and was not used. BUBO and BAPA seem to show a negative response and could possibly be used. We excluded BUBO because, as a frog, BUBO is associated with water and perhaps does not respond the disturbance gradient we measured.

BAPA and CNTI were ‘tough calls’ as their patterns, though positive, are similar to some of the negative responding species we included (i.e. LAGE). In the case of positive responding species, we already had 4 species with strong signals, so we did not need species with weak or questionable signals to create a positive response metric. Furthermore, for both BAPA and CNTI absences were prevalent across all AEC values, as were presences so the proportional change in occupancy across the gradient was weaker than for other species we included. For example, when comparing BAPA to LAGE, for LAGE we found ~6 out of 10 sites occupied from 0-0.2 AEC, and 1 out of 8 occupied for 0.8-1.0 AEC; a 60% occupancy rate declining to a ~12% within increasing AEC. For BAPA, the relationships is not as strong; ~1 out 10 for 0-0.2AEC and 3 out of 10 for 0.8-1.0 AEC and change in occupancy from only 10% to 30%. We considered keeping BAPA, as it showed a strong threshold response if one assumes the single presence at ~0.15 AEC is a rare event. However, given 4 species with much stronger signals, we did not feel we needed to include these weaker cases.

Relative Abundance Data.

We generated 1 metric from this variable.

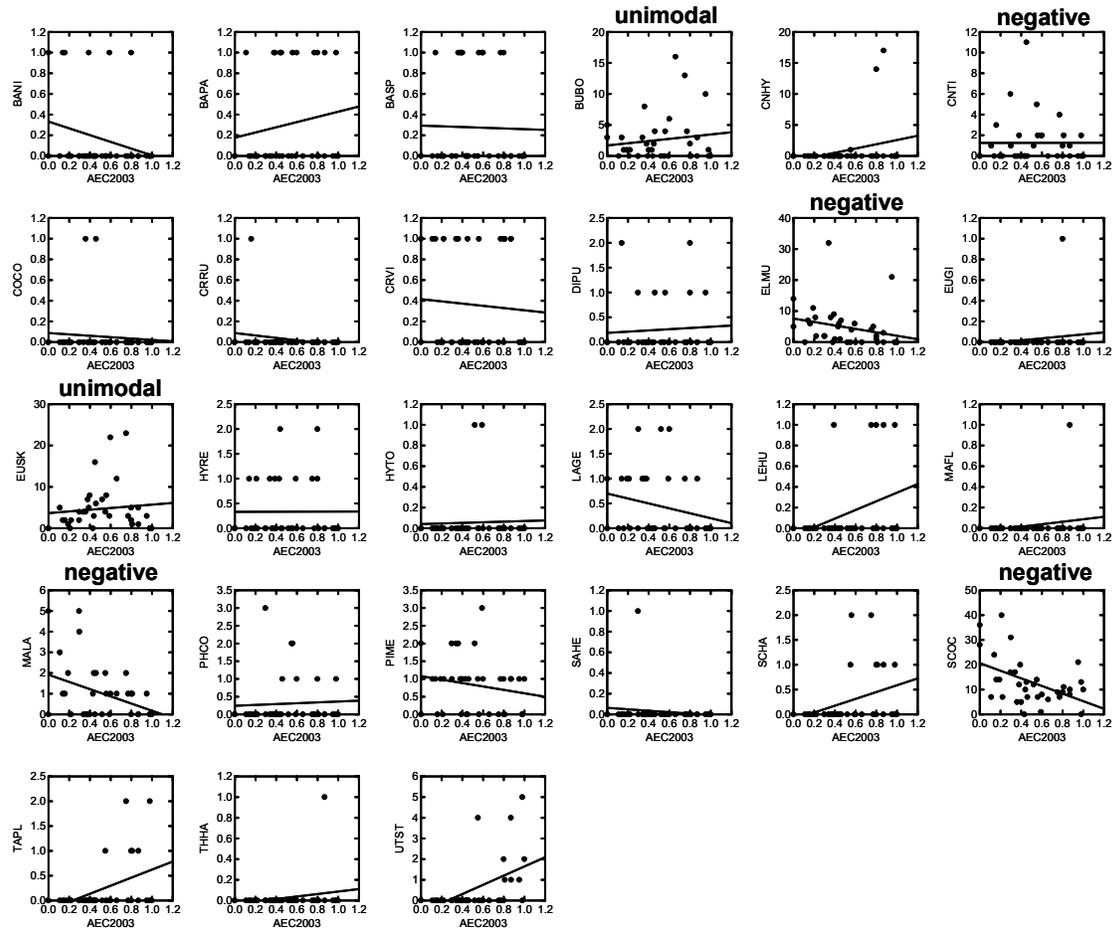


Figure 14. Presence-Absence versus AEC for Herpetofauna. Herp data collected from 1998 to 2003 on most sites.

Negative responses.

CNTI: Negative ceiling effect where maximum abundances decline with increasing AEC.

ELMU: Negative ceiling effect where maximum abundances decline with increasing AEC. Two plots with high abundances exist, in a moderate to highly disturbed site.

MALA Negative ceiling effect where maximum abundances decline with increasing AEC.

SCOC. General decline in abundance within increasing AEC.

Level of support based on existing knowledge: *Masticophis lateralis* is known to prefer CSS habitat over grasslands based on historic work and more recent radiotracking. Thus, its response across the gradient is expected and it's inclusion in a metric well supported. Support for the other species is not as strong. BANI, HYRE, ELMU, and LAGE, are

considered associated with more mesic environments and more dense cover. In coastal southern California mesic environments tend to contain more leaf litter cover and perhaps more shrubs, though oak-woodlands (habitat we did not include in our plots) are typical associated with more mesic conditions. Perhaps these species do not select more open, drier conditions found at sites with higher amounts of exotic cover but detailed studies of the species habitat preferences are lacking. CRVI is a sit and wait, ambush predator who specializes on small mammals. Perhaps CRVI avoids open grasslands to escape predation by raptors or because they lack shaded areas for thermoregulation.

Other possible species. UTST showed a strong positive response. However, this response was mainly one of presence and absence, as the number of individuals varied widely once the species began appearing on plots. In addition, this was the only species showing a positive response in abundance to AEC. We used this species in the metric based on presence/absence instead.

Mammals

Proportion adult.

No metric created from this variable.

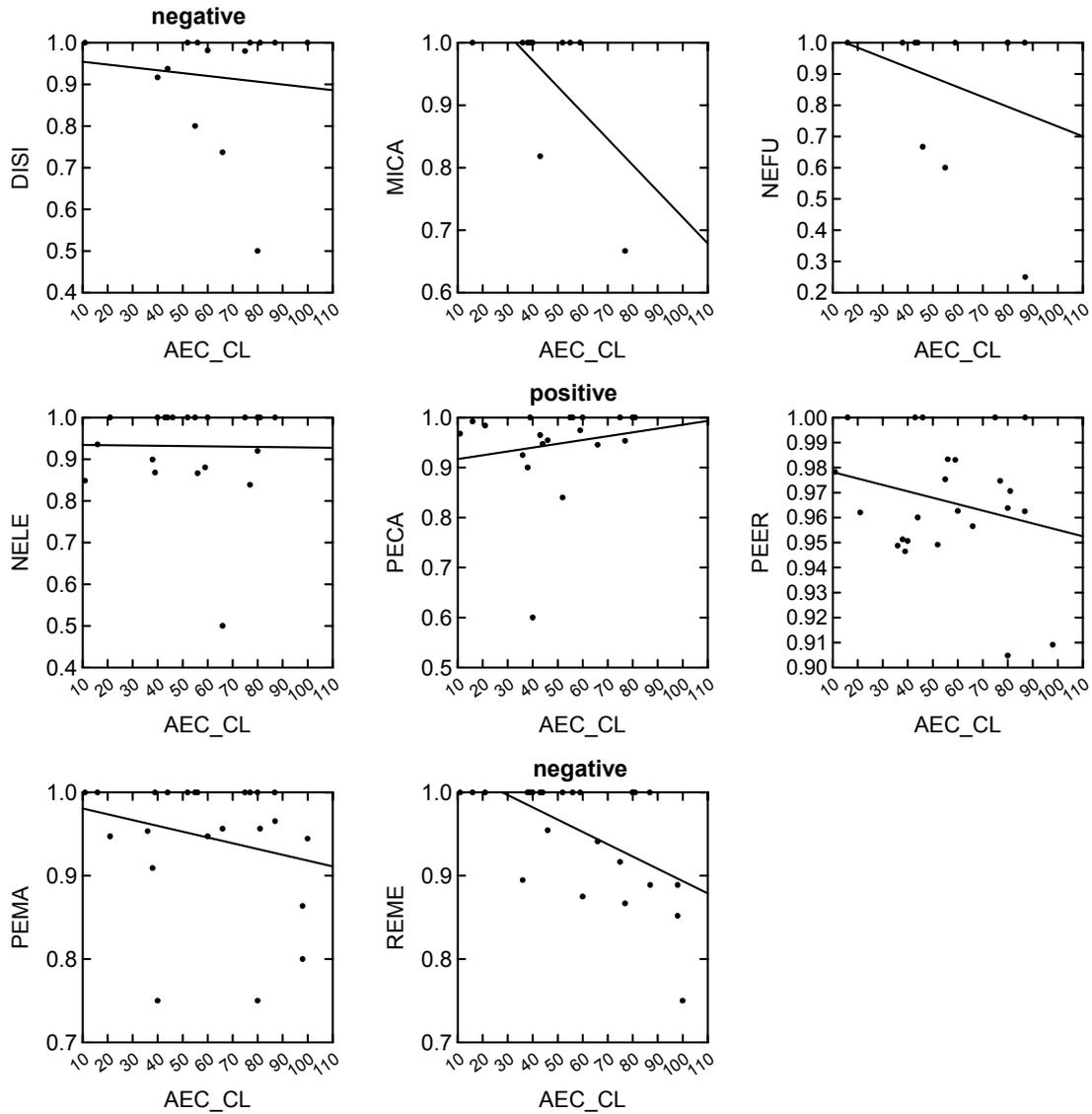


Figure 15. Proportion of adults versus AEC.

Female Body Weight.

We generated no metrics from this variable.

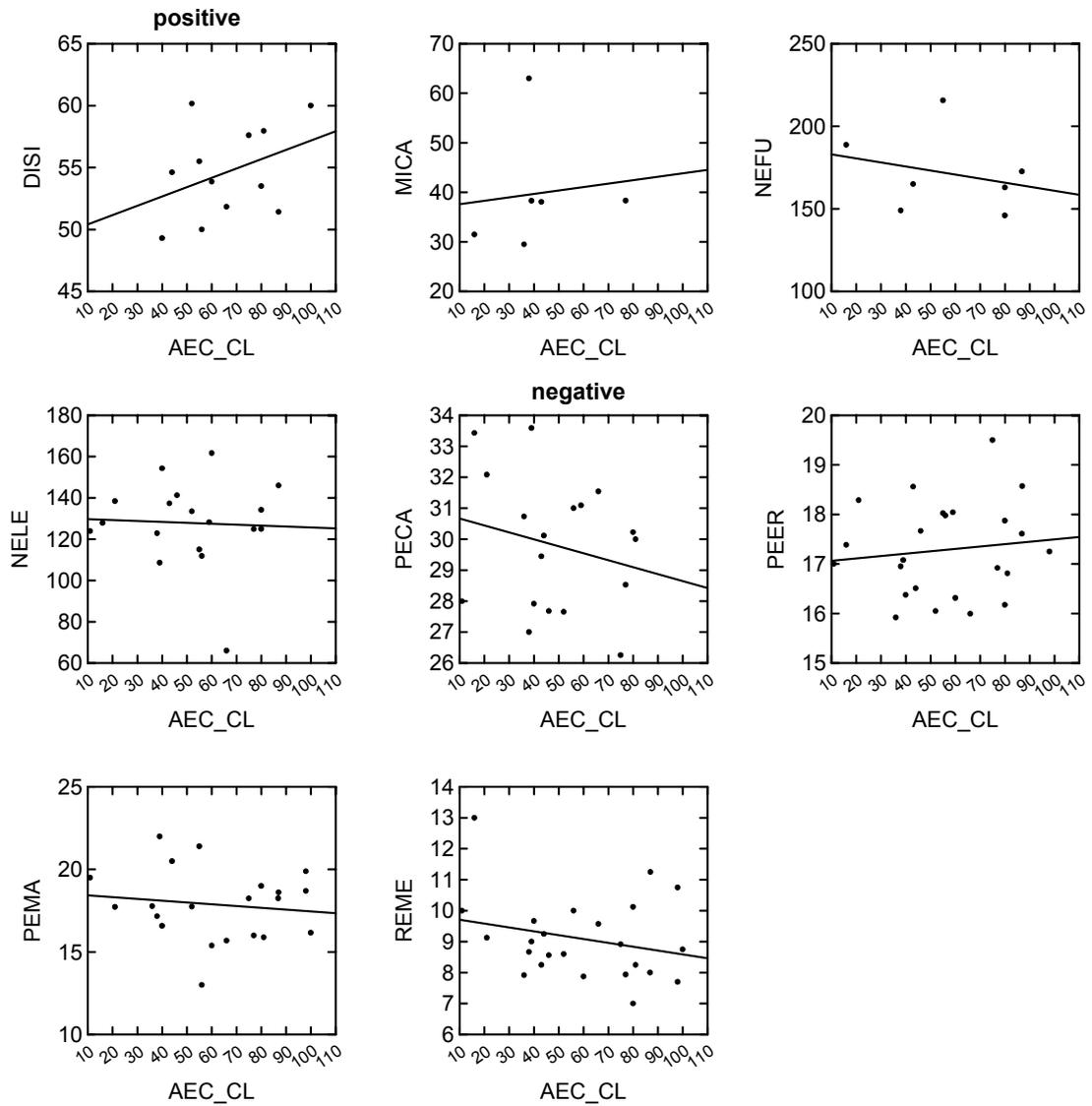


Figure 16. Female body weight versus AEC. Body weight was averaged across all adults captured on a plot.

Male Body Weights

We generated no metrics from this variable.

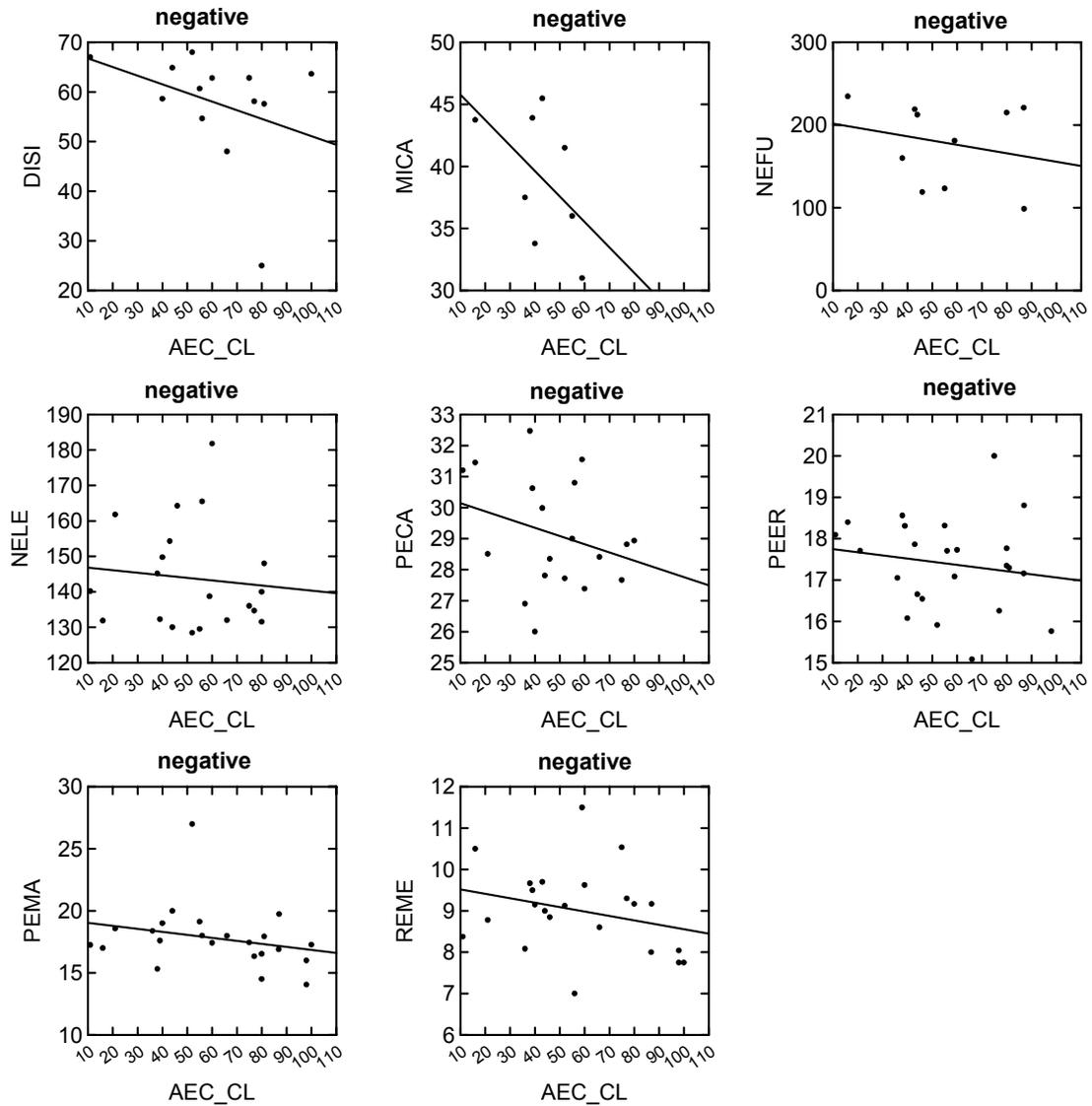


Figure 17. Male body weight versus AEC. Body weight was averaged across all adults captured on a plot.

Male Reproduction.

We generated no metrics from this variable.

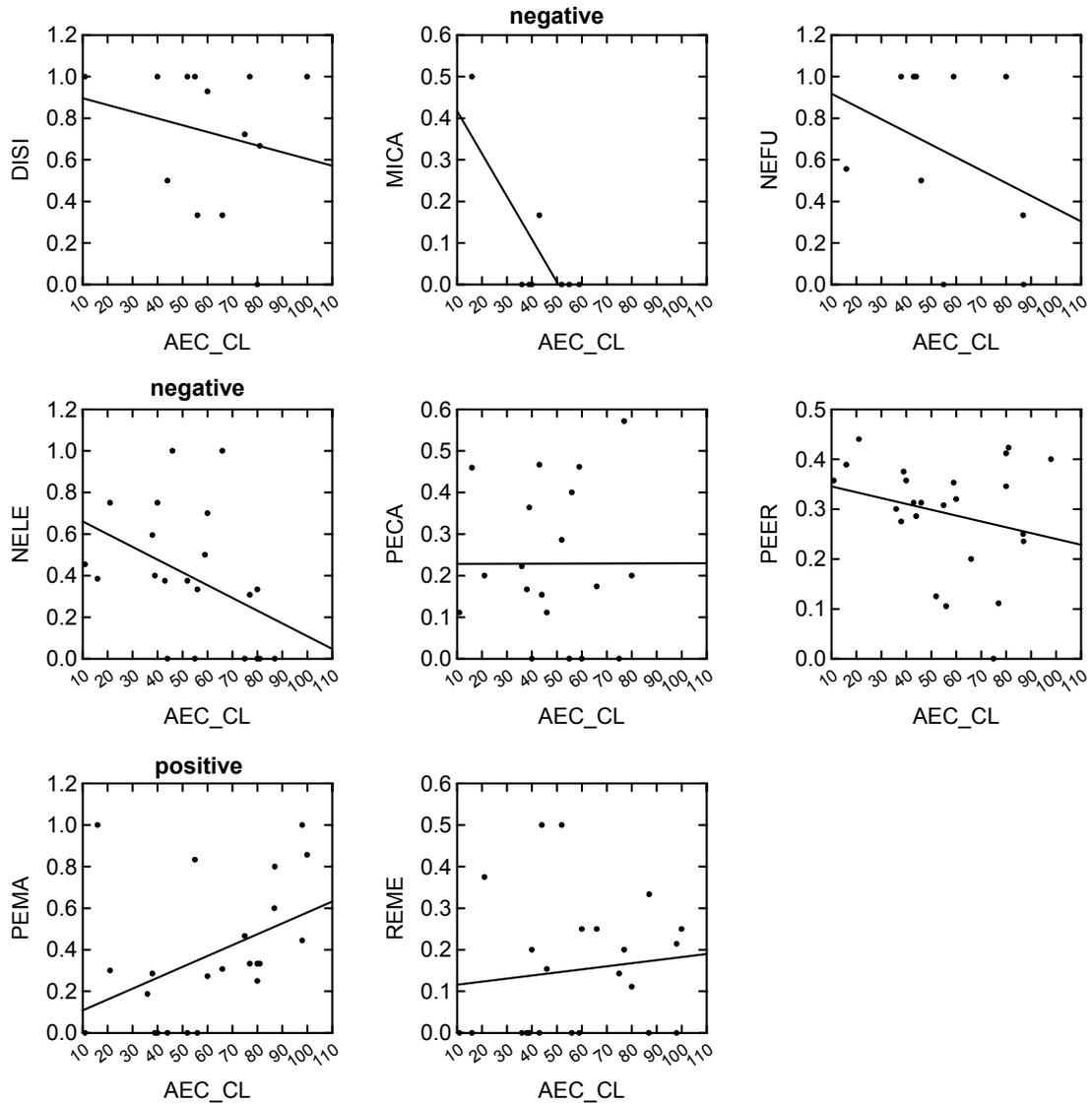


Figure 18. Male Reproduction versus AEC. Proportion reproductive was the number of adults with external signs of reproduction divided by the total number of adults on a plot.

Female Reproduction.

We generated no metrics from this variable.

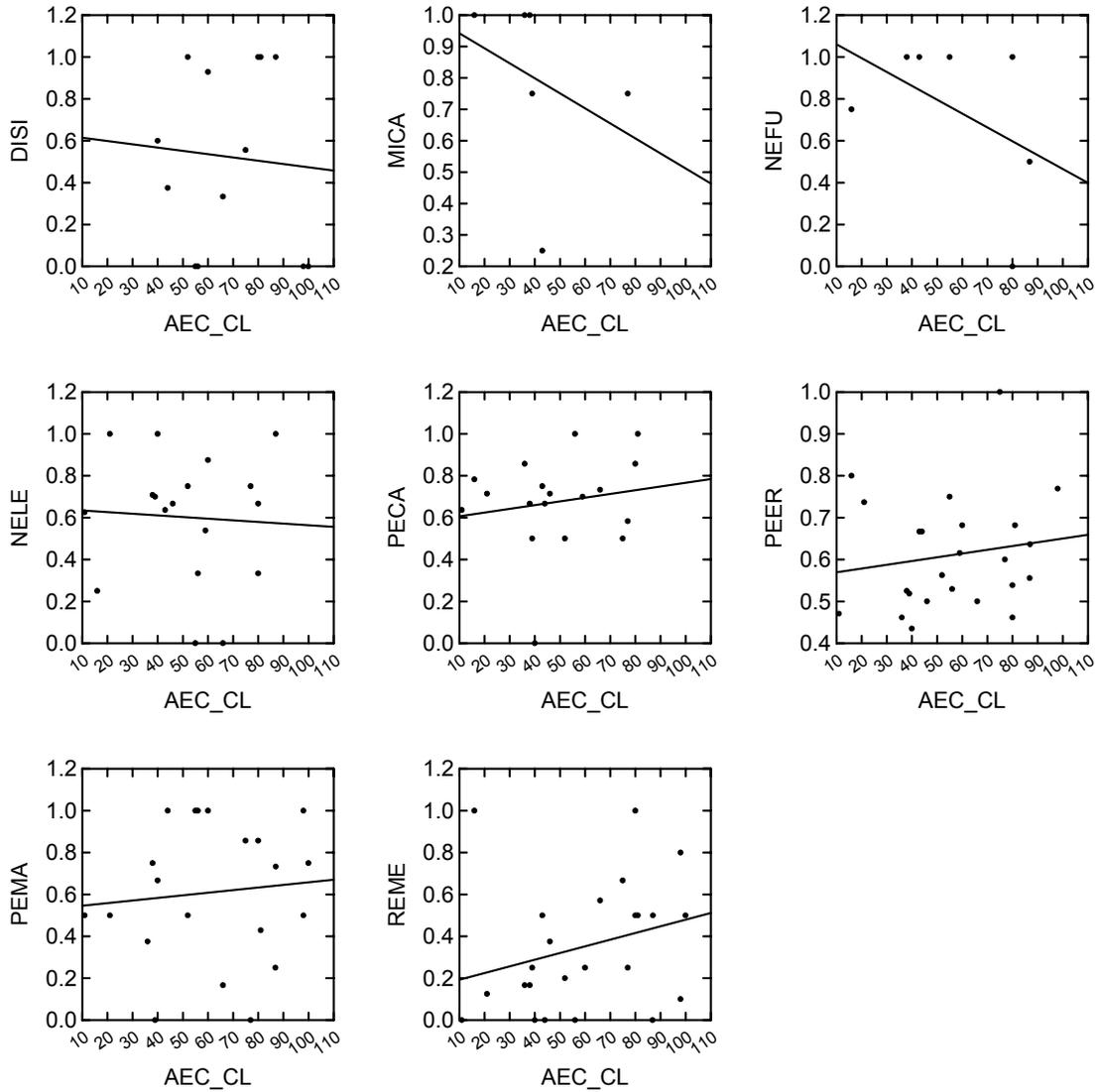


Figure 19. Female Reproduction versus AEC. Proportion reproductive was the number of adults with external signs of reproduction divided by the total number of adults on a plot.

Mammal presence absence.

We generated 2 metrics from these data.

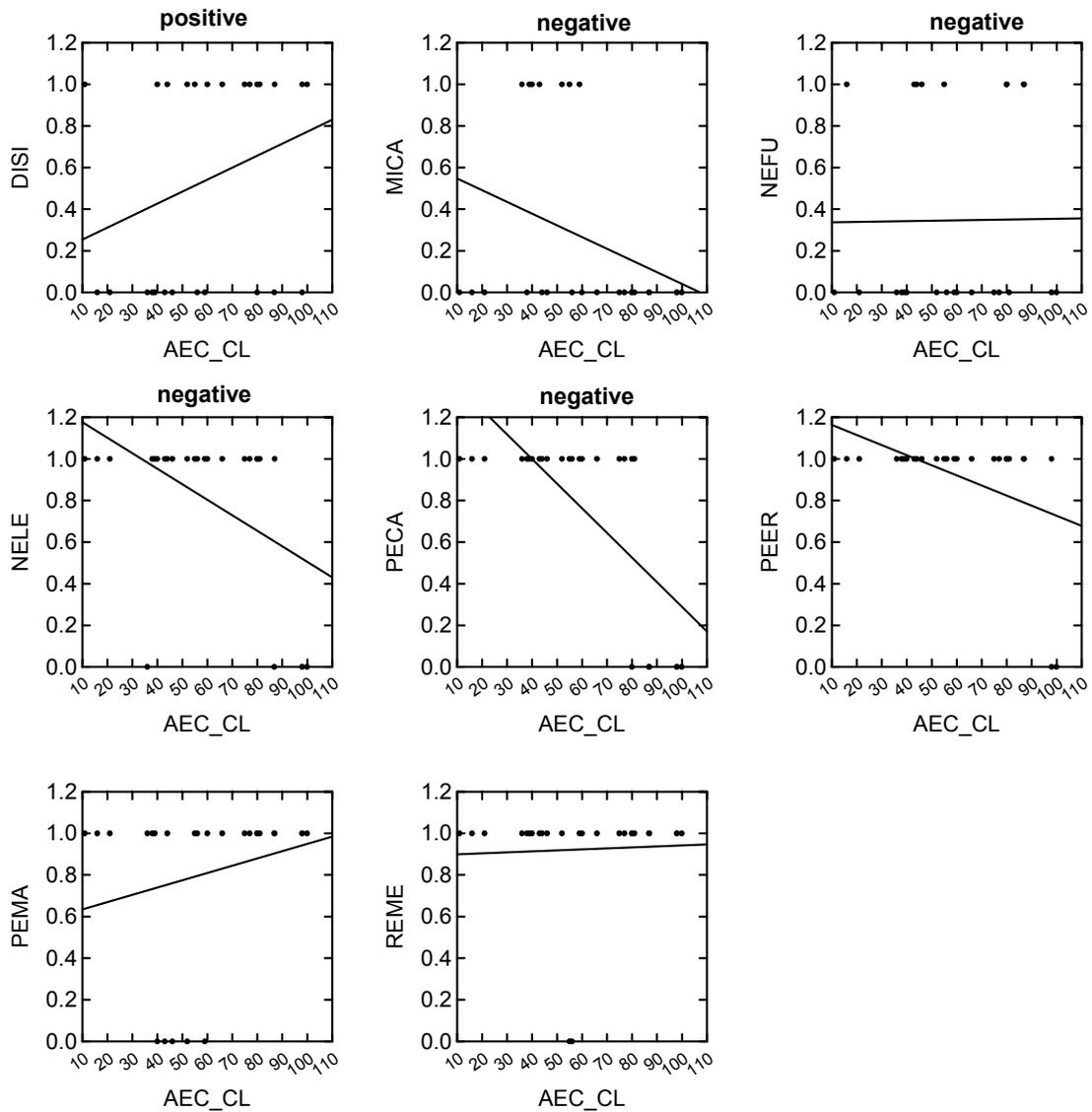


Figure 20. Presence-Absence by AEC for small mammals.

Negative responses.

MICA: Threshold effect at ~0.65.

NELE: Strong threshold effect at ~0.85 AEC. Almost no absences at lower AEC values.

PECA: Strong threshold effect at 0.80 AEC.

Level of support based on existing knowledge: All three of these species are reported to favor shrub-dominated CSS. Thus, their absence from sites with more disturbance fits well with previous research.

Other possible species. NEFU showed a possible threshold effect with no presence after ~0.85 AEC. However, NEFU was absent from many sites with lower AEC values.

Positive or Tolerant responses.

DISI: A threshold effect at ~0.45. However, DISI was captured on a site with almost 0 AEC.

PEER, PEMA, REME. Found at all levels of AEC.

Level of support based on existing knowledge: DISI, being a kangaroo rat, favors more open habitats and avoids closed canopy, dense, CSS. PEER, PEMA, and REME are considered habitat generalist relative to the other species, so their presence on almost all plots is not surprising.

Mammal Relative Abundance.

We generated 1 metric from these data.

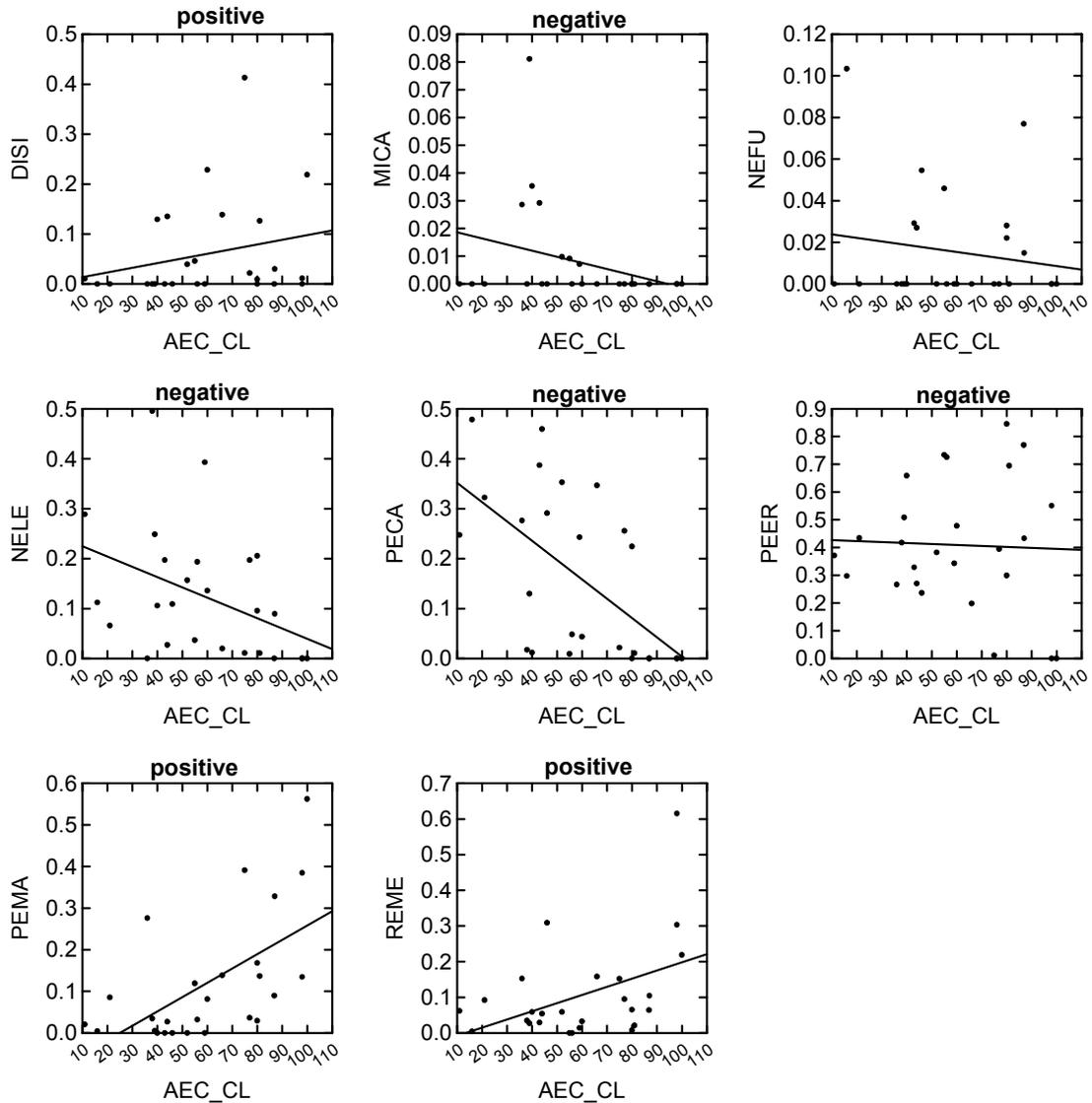


Figure 21. Relative abundance of small mammals versus AEC.

Negative responses.

MICA: Ceiling effect of declining maximum values with increasing AEC.

NELE: General decline and a ceiling effect.

PECA: General decline

PEER: General, but highly variable decline.

Level of support based on existing knowledge: MICA, NELE, and PECA are reported to favor shrub-dominated CSS. Thus, the declines in relative abundance from sites with

more disturbance fits well with previous research. For PEER, research to date, does not fully address the issue of changes in abundance across habitat types.

Other possible species. NEFU showed a possible negative response but it was driven by an extremely abundance of NEFU at one site with low AEC and their absence from sites at high levels of AEC.

Positive responses.

DISI: A ceiling effect with increasing maximum values at greater AEC.

PEMA: General but highly variable. Slight ceiling effect.

REME: General but highly variable. Stronger ceiling effect than PEMA.

Level of support based on existing knowledge: All species are known to prefer or tolerate more open habitats, so their abundances, particularly relative to other species in the community, should increase with disturbance.

Vegetation

Summary metrics.

We generated 2 metrics from these data.

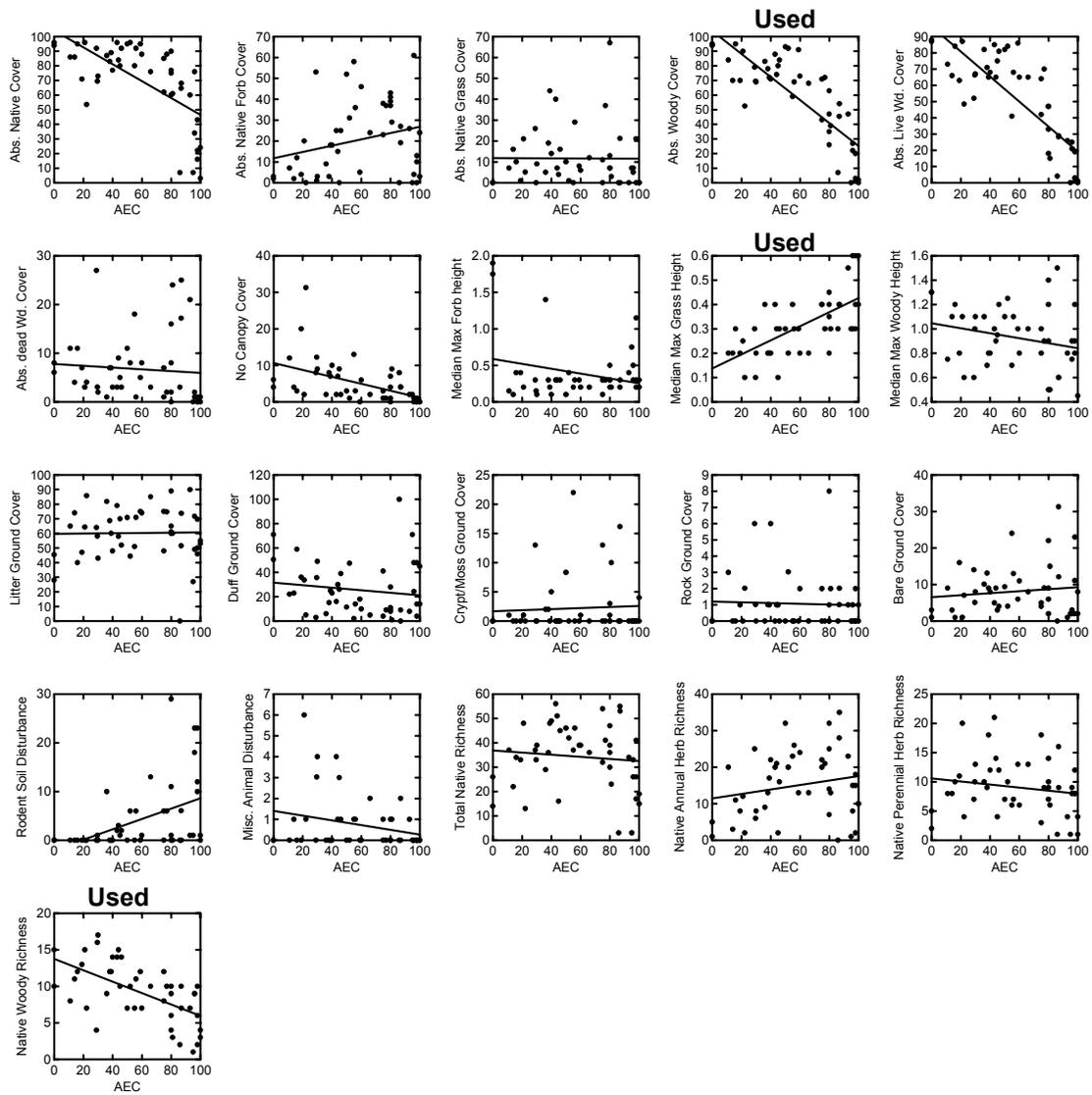


Figure 22. Summary vegetation variables vs. AEC. See Table immediately below for definitions of each variable. “Used” refers to variables used as IBI metrics.

Table 3. Name and description of summary vegetation variables presented in Fig. 22.

<i>Variable Name</i>	<i>Description</i>
Absolute N cover	Percent absolute native cover = Total number of sample points with at least one native hit / Total number of sample points*100; ignores growth form and life history.
Abs Native forb cover	Percent absolute native forb cover = Total number of sample points with at least one native forb hit / Total number of sample points*100; ignores life history.
Abs Native grass cover	Percent absolute native grass cover = Total number of sample points with at least one native grass hit / Total number of sample points*100; ignores life history.
Abs woody cover	Percent absolute woody cover. Total number of sample points with at least one woody hit / Total number of sample points*100. "Woody" hits include shrubs, halfshrubs, woody vines, and trees of both native and exotic origin.
Abs live Wd cover	Percent absolute live woody cover. Total number of sample points with at least one live woody hit / Total number of sample points*100
Abs dead Wd cover	Percent absolute dead only woody cover. Total number of sample points with at least one dead woody hit AND no live woody hits / Total number of sample points*100
No canopy cover	Percent of points with no canopy cover, or "bare ground" = Total number of sample points with no canopy hits / Total number of sample points*100
Median Max Forb Height	Median height of the tallest forb (regardless of origin) across sample points where forbs are present.
Median Max Grass Height	Median height of the tallest grass (regardless of origin) across sample points where grasses are present.
Med Max Woody Height	Median height of the tallest woody plant species (regardless of origin) across sample points where woody plants are present. "Woody" plants include shrubs, halfshrubs, woody vines, and trees.
Litter ground cover	Percent of ground surface covered by plant litter from any source (shrub, grass, etc.) = Total number of sample points with litter ground cover / Total number of sample points*100.
Duff ground cover	Percent of ground surface covered by plant duff (i.e. litter depth >2cm) from any source (shrub, grass, etc.) = Total number of sample points with duff ground cover / Total number of sample points*100
Crypt/moss ground cover	Percent of ground surface covered by cryptogamic crust or moss = Total number of sample points with cryptogam or moss ground cover / Total number of sample points*100
Rock ground cover	Percent of ground surface covered by exposed rock = Total number of sample points with duff ground cover / Total number of sample points*100. Value ignores any plant canopy cover above the ground surface.
Bare ground cover	Percent exposed soil, i.e. ground surface NOT covered by plant material, cryptogams, rock, etc. = Total number of sample points with exposed soil / Total number of sample points*100. Value ignores any plant canopy cover above the ground surface.
Rodent Soil Disturbance	Percent of soil surface disturbed by small mammal (often gopher) digging or burrowing, aka "rodent disturbance", = Total number of sample points with rodent disturbance / Total number of sample points*100. Attempts are made to only record disturbance occurring within the current growing season.
Misc Animal Disturbance	Percent of sample points disturbed by animal activity other than small mammal digging (often, but not limited to, trails) = Total number of sample points with animal disturbance / Total number of sample points*100. Attempts are made to only record disturbance occurring within the current growing season.
Total Native Richness	Total native plant species richness across the entire plot or array, ignoring life history and growth form.
Native annual	Total native annual herb species richness across the entire plot or array. Herbs include

herb richness	both grasses and forbs.
Native perennial herb richness	Total native perennial herb species richness across the entire plot or array.
Native woody richness	Total native woody species richness across the entire plot or array.

Negative responses.

Absolute Woody Cover: General decline with increasing AEC.

Native Woody Richness: General decline with increasing AEC.

Level of support based on existing knowledge: High. Many studies show a loss of shrubs with increasing levels of disturbance.

Other possible variables. Abs. Native Cover, and Abs. Live Wd. Cover showed negative trend but measured the same phenomena as Absolute Woody Cover. To avoid correlated metrics, we only include one of these three variables in the IBI.

Positive responses.

Median Max. Grass Height.

Level of support based on existing knowledge: Moderate. Grasses grow to shorter heights under shaded shrub canopies.

Other possible variables. Rodent Soil Disturbance showed and increase with increasing AEC. However, many sites with high AEC had minimal disturbance.

Presence-Absence Data.

We generated one metric from these data.

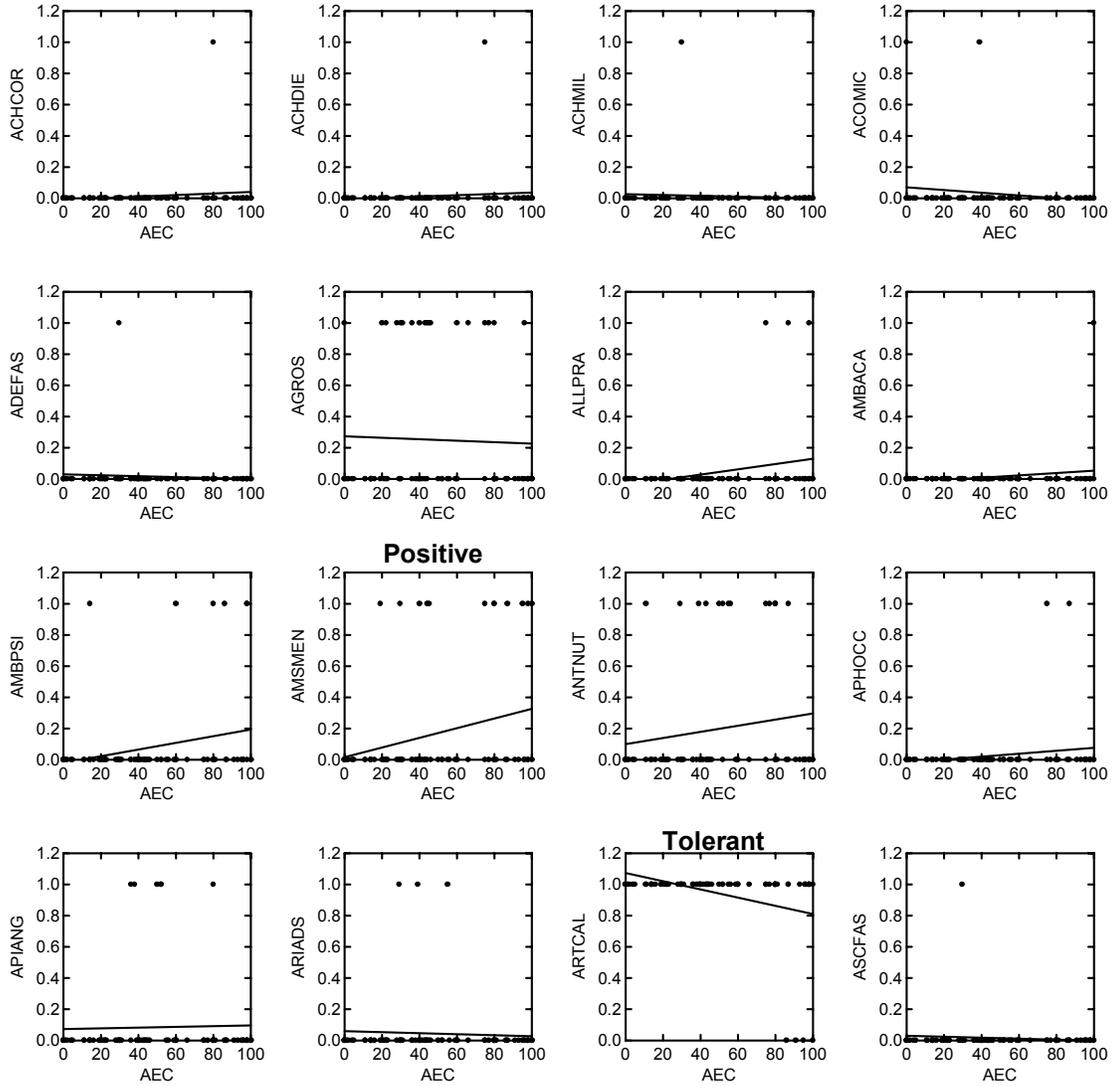
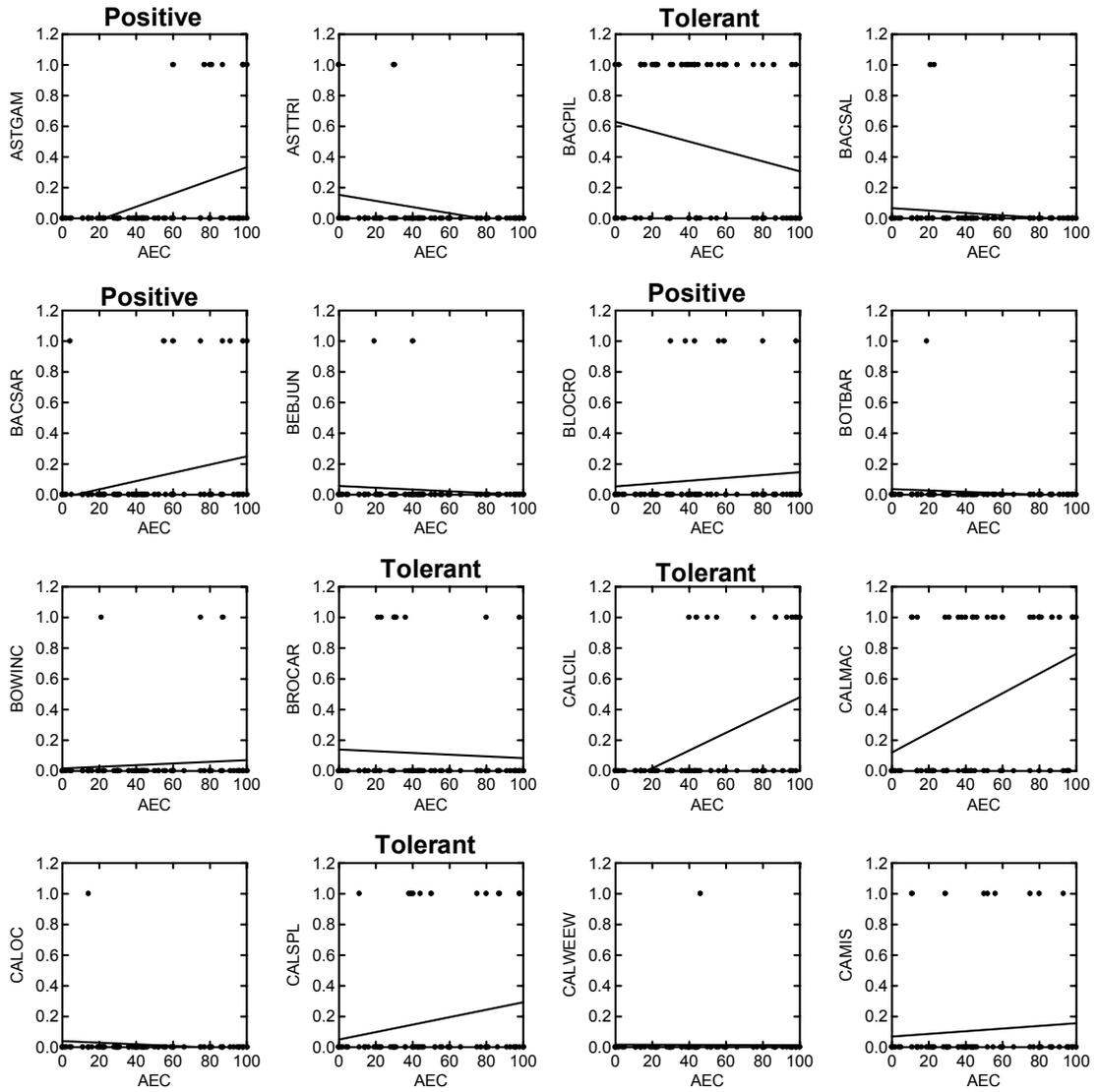
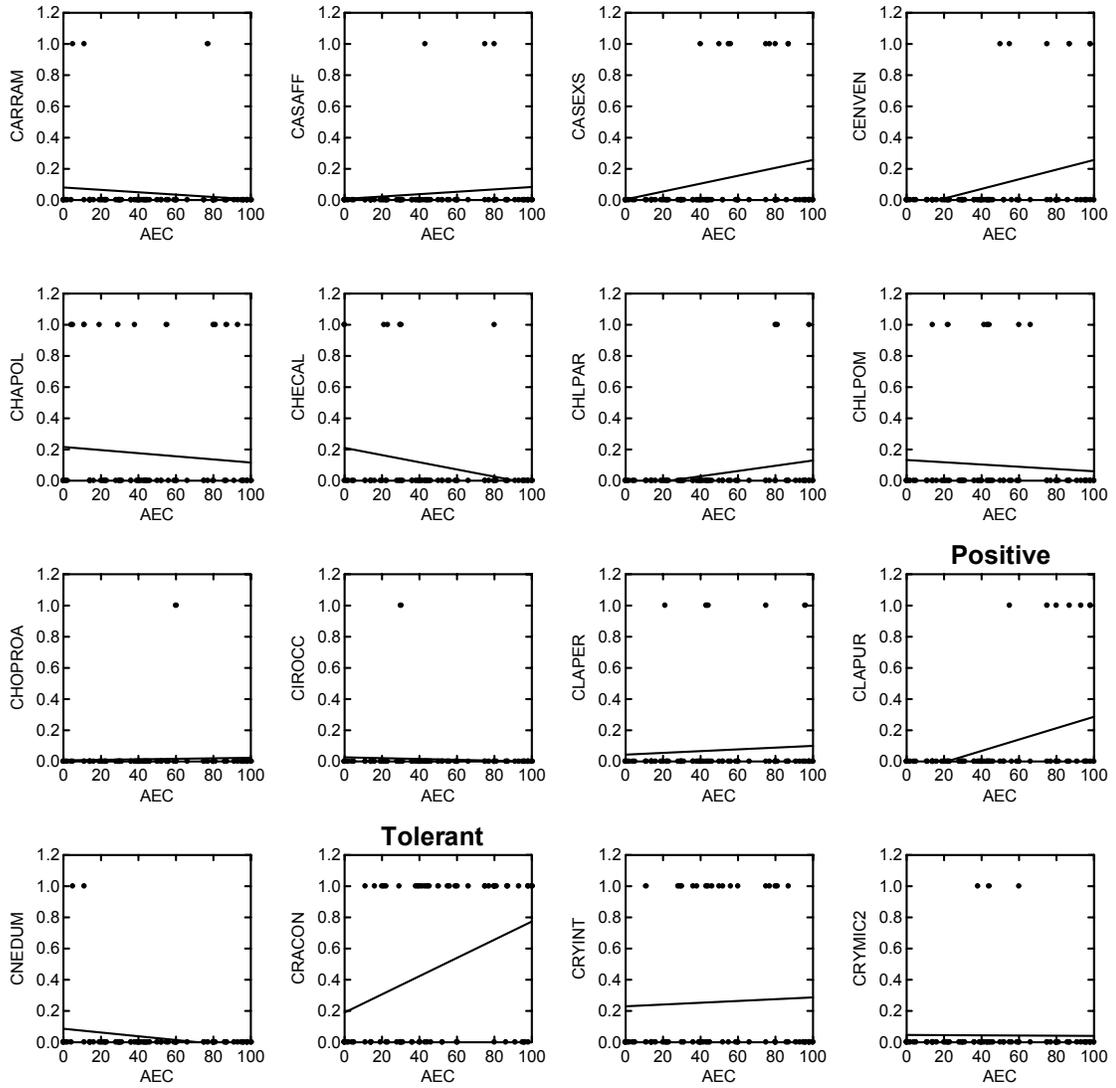


Figure 23. Presence-Absence vs Absolute Exotic cover for native plant species. Data come from all study plots. AEC values from 2003 used in all cases except when only 2002 data was available.

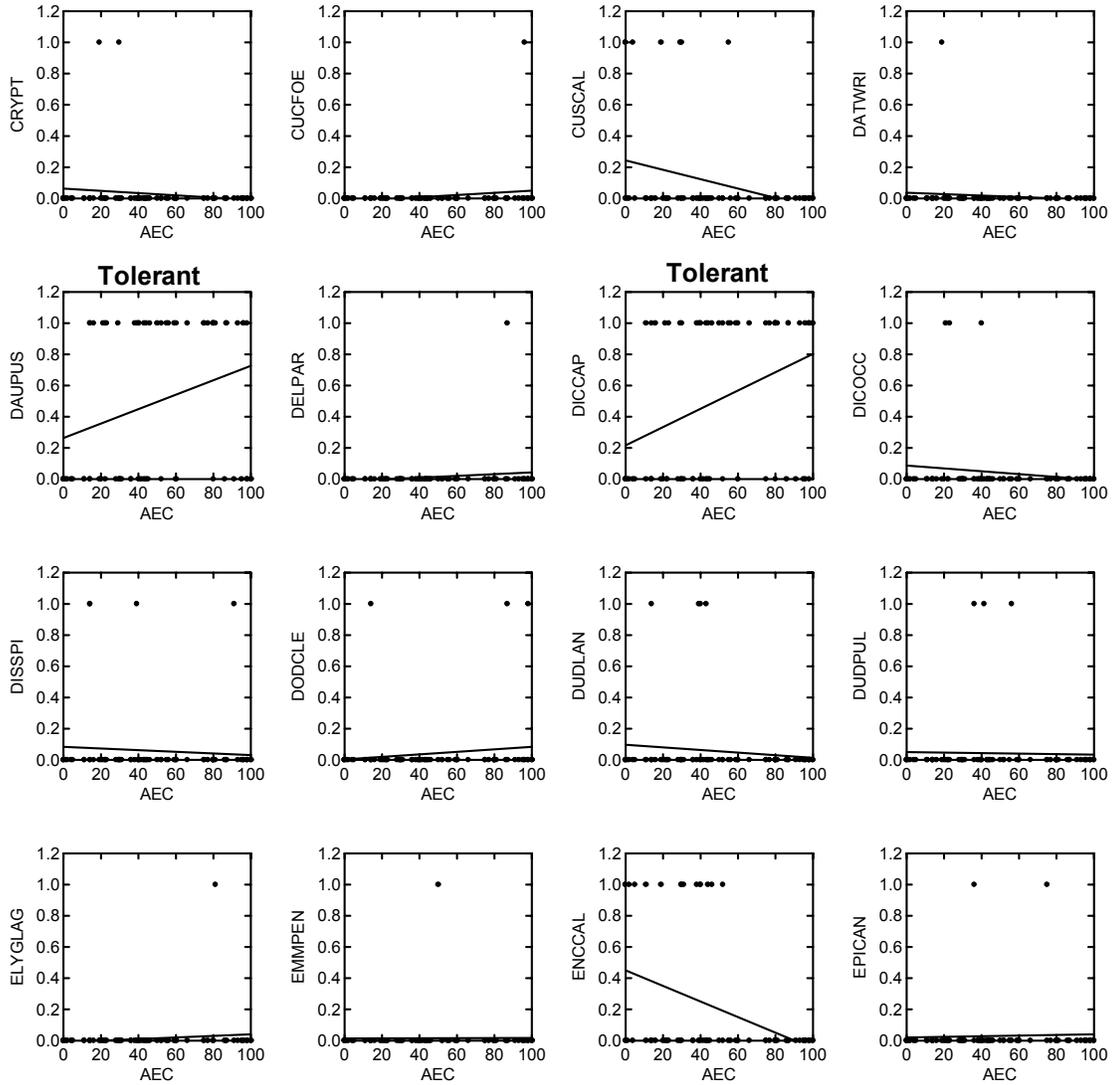
Presence-Absence Data continued



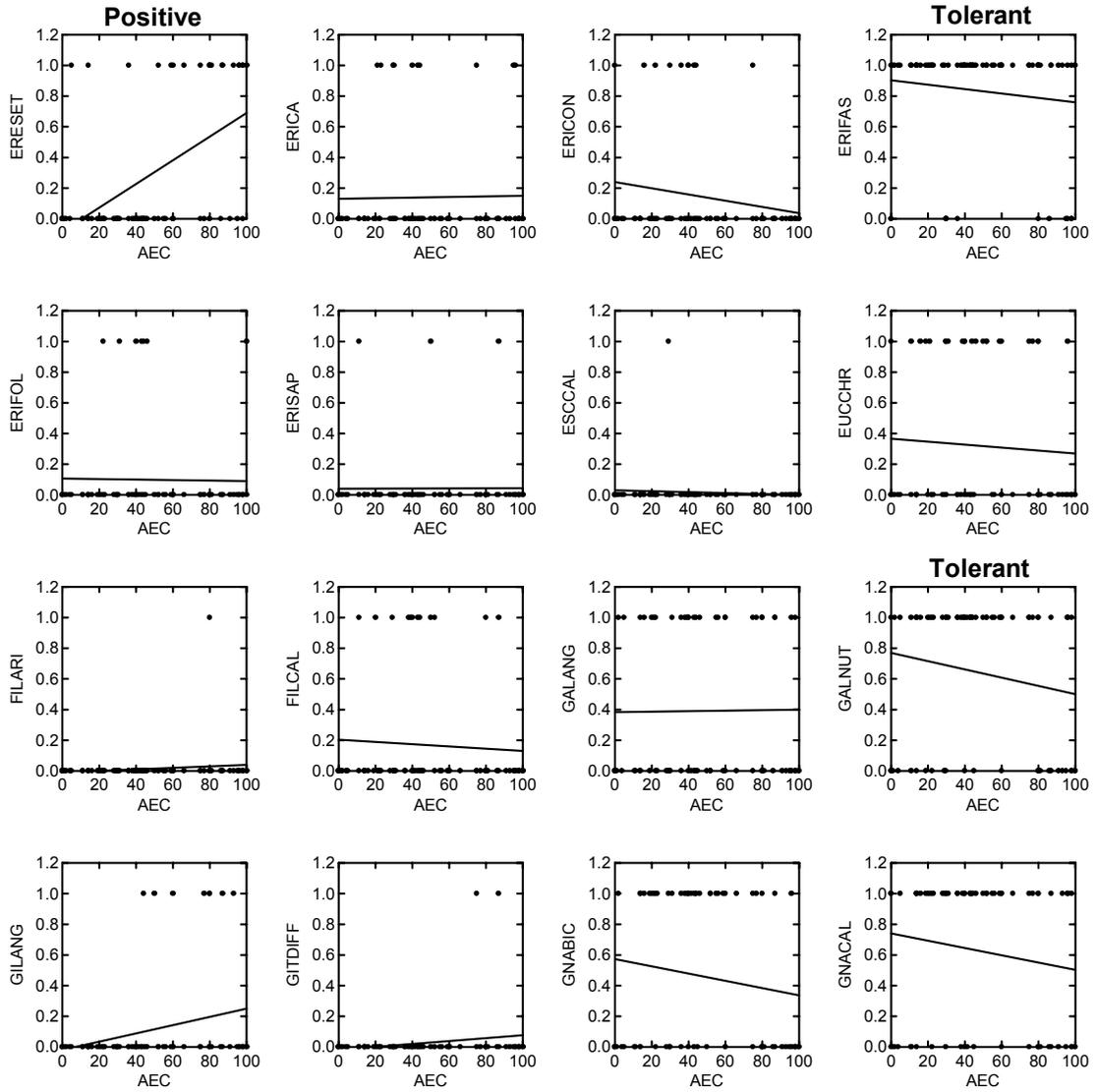
Presence-Absence Data Continued



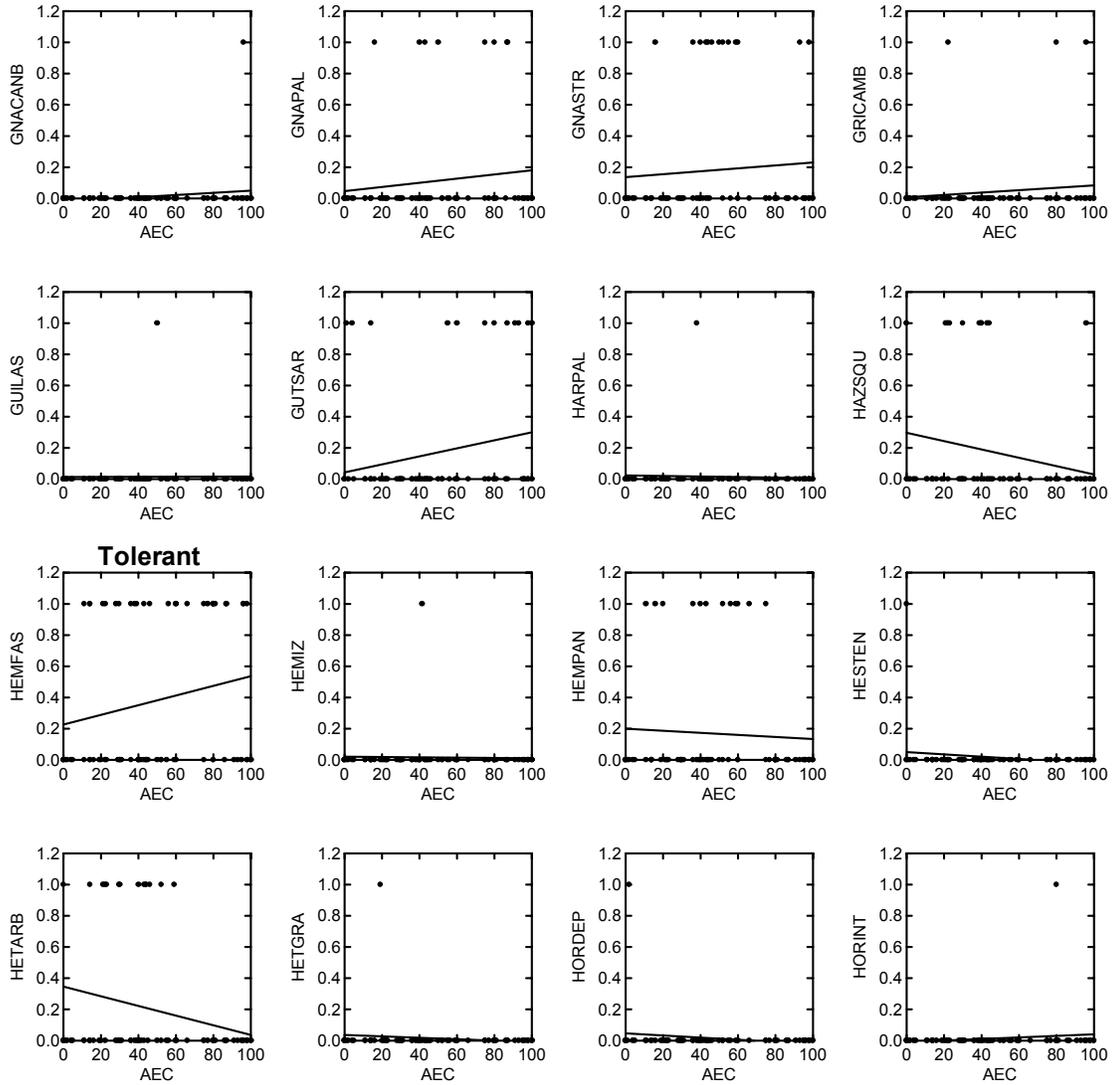
Presence-Absence Data Continued



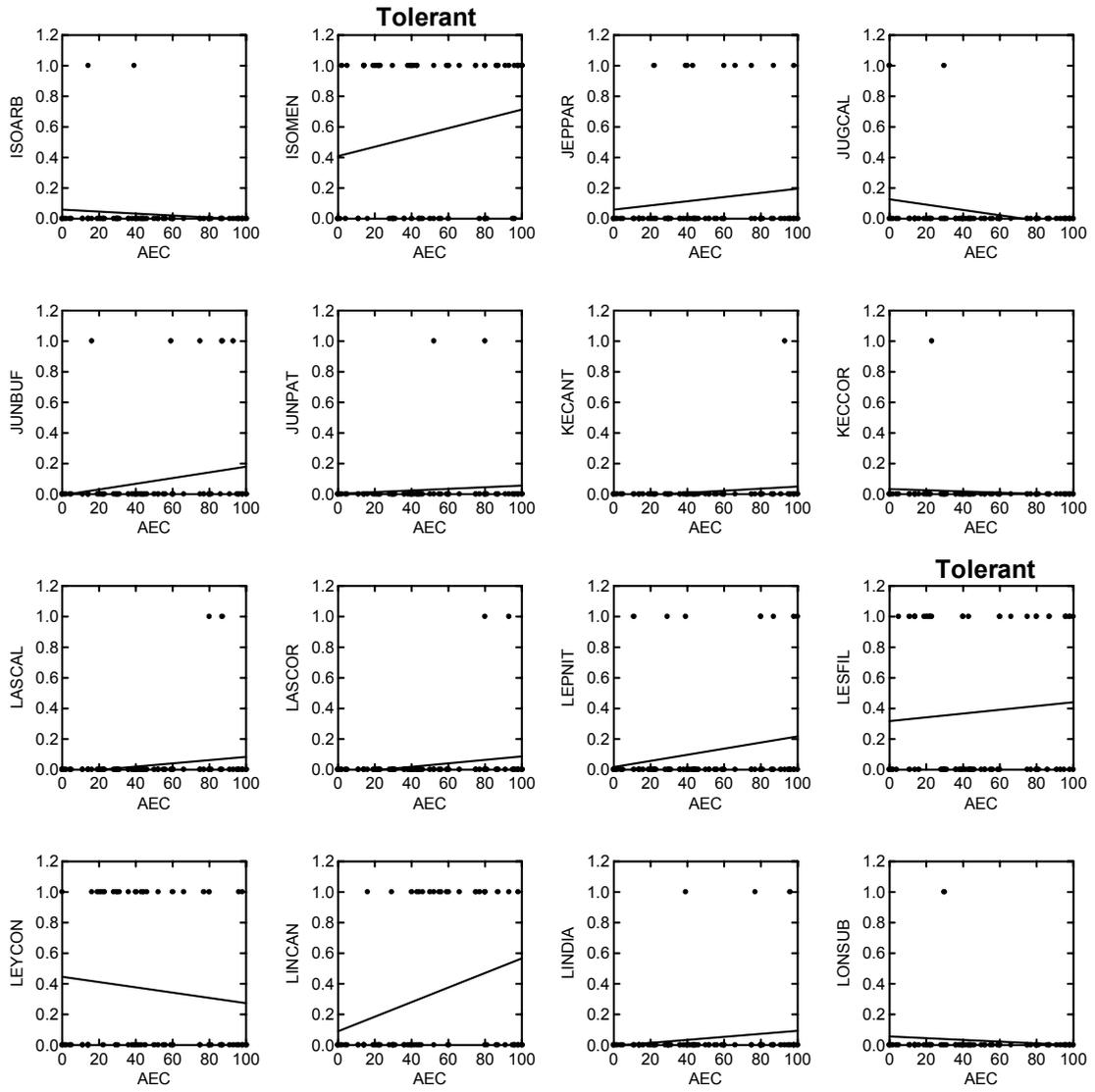
Presence-Absence Data Continued



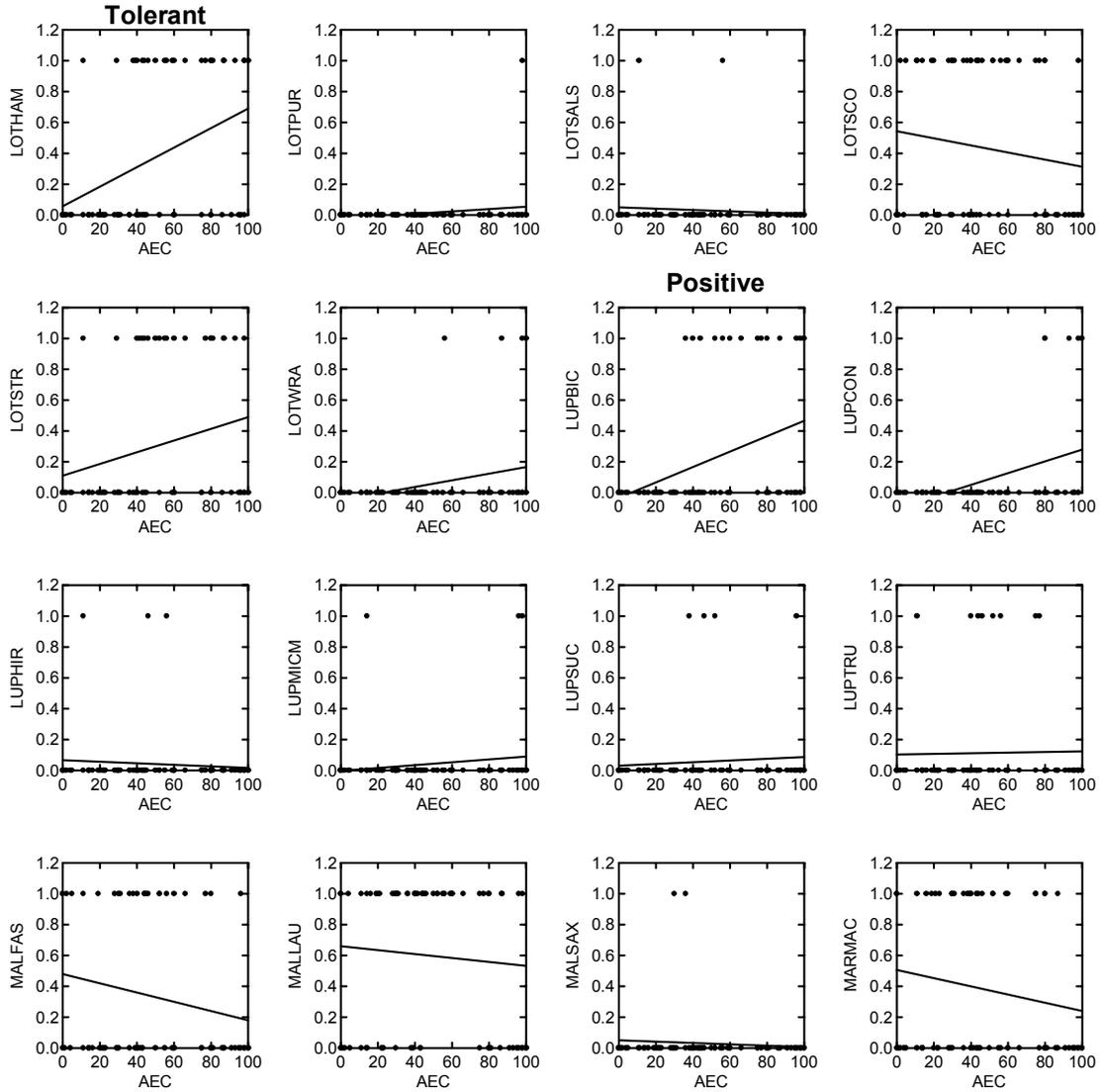
Presence-Absence Data Continued



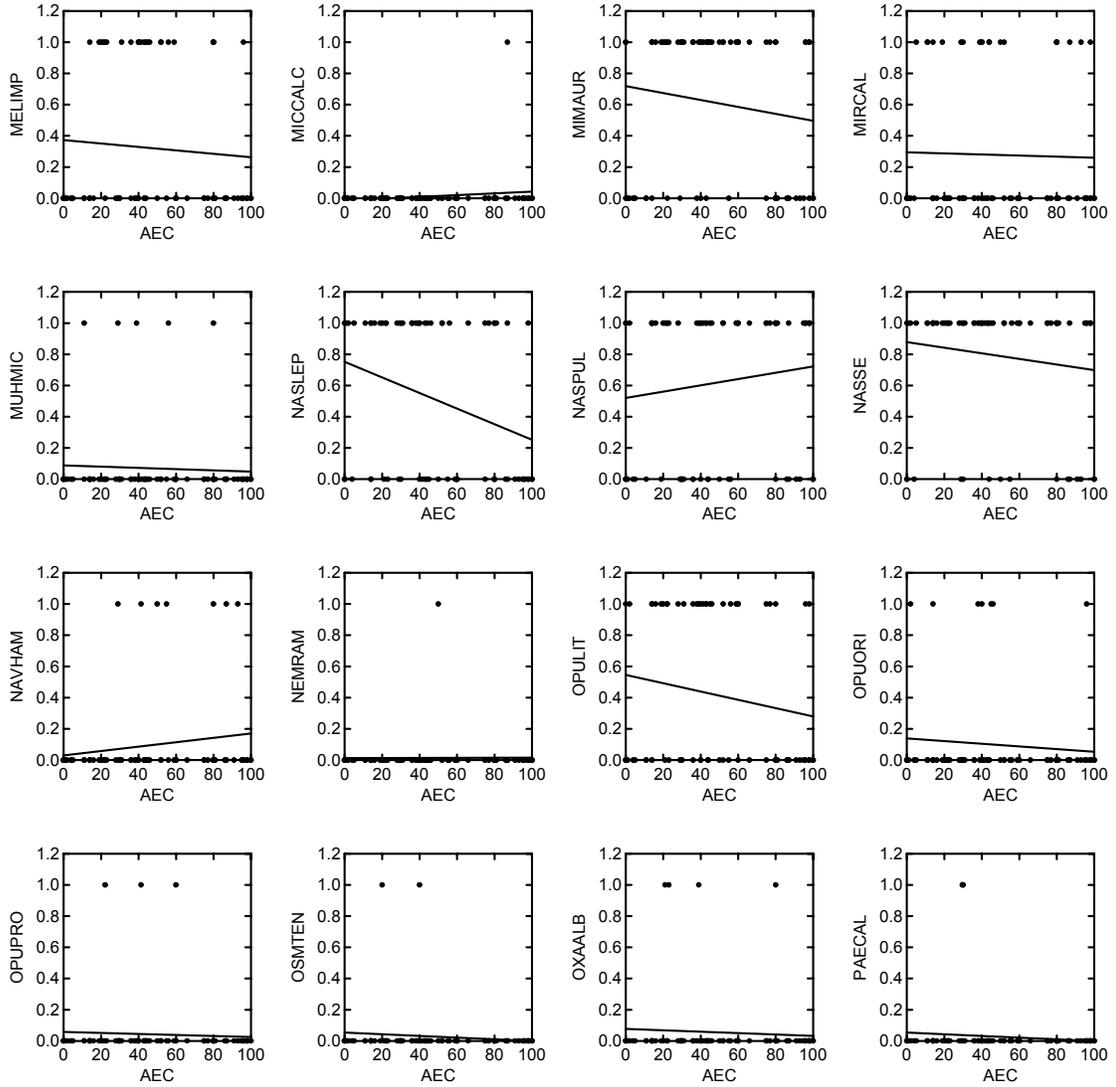
Presence-Absence Data Continued



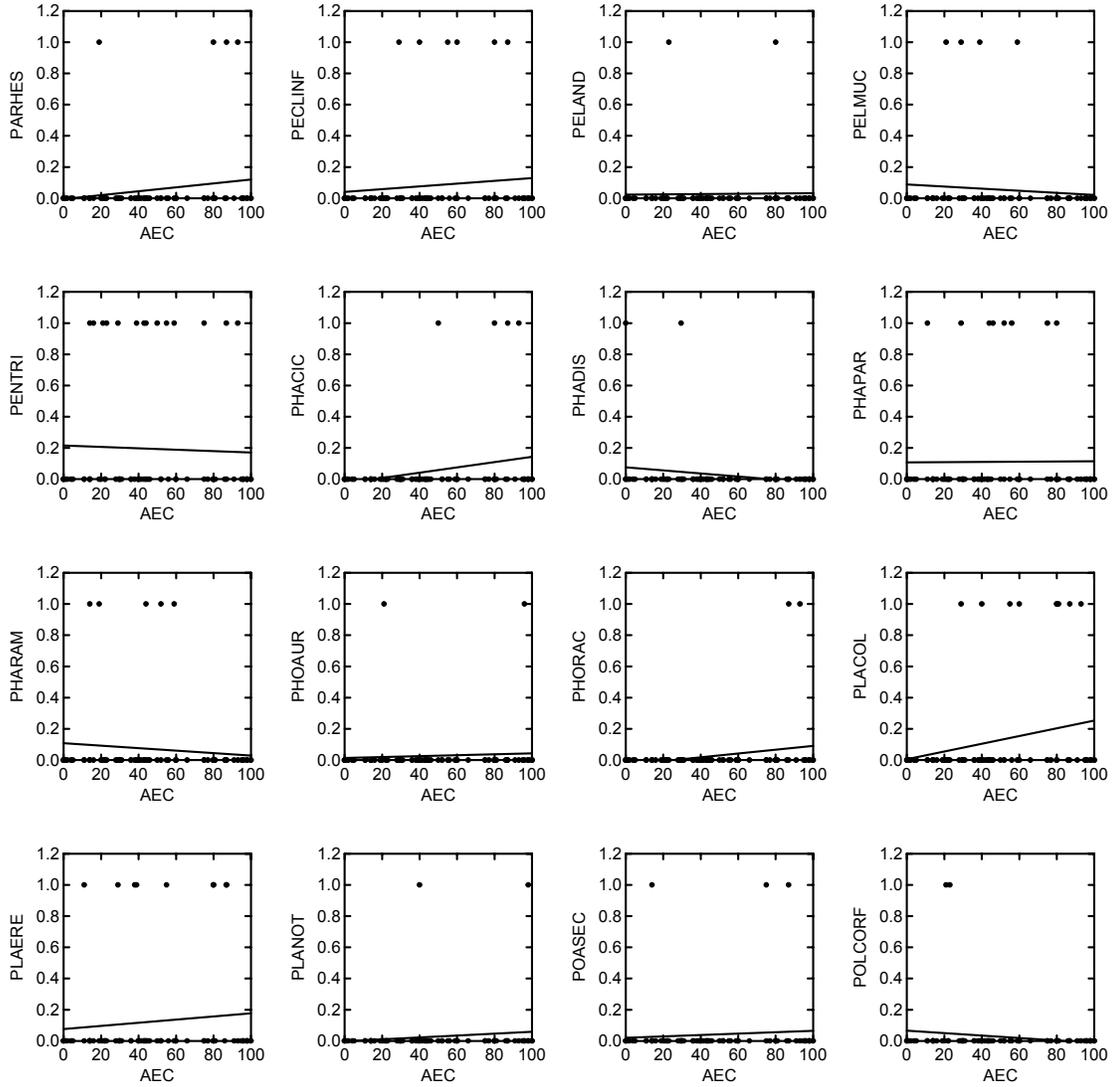
Presence-Absence Data Continued



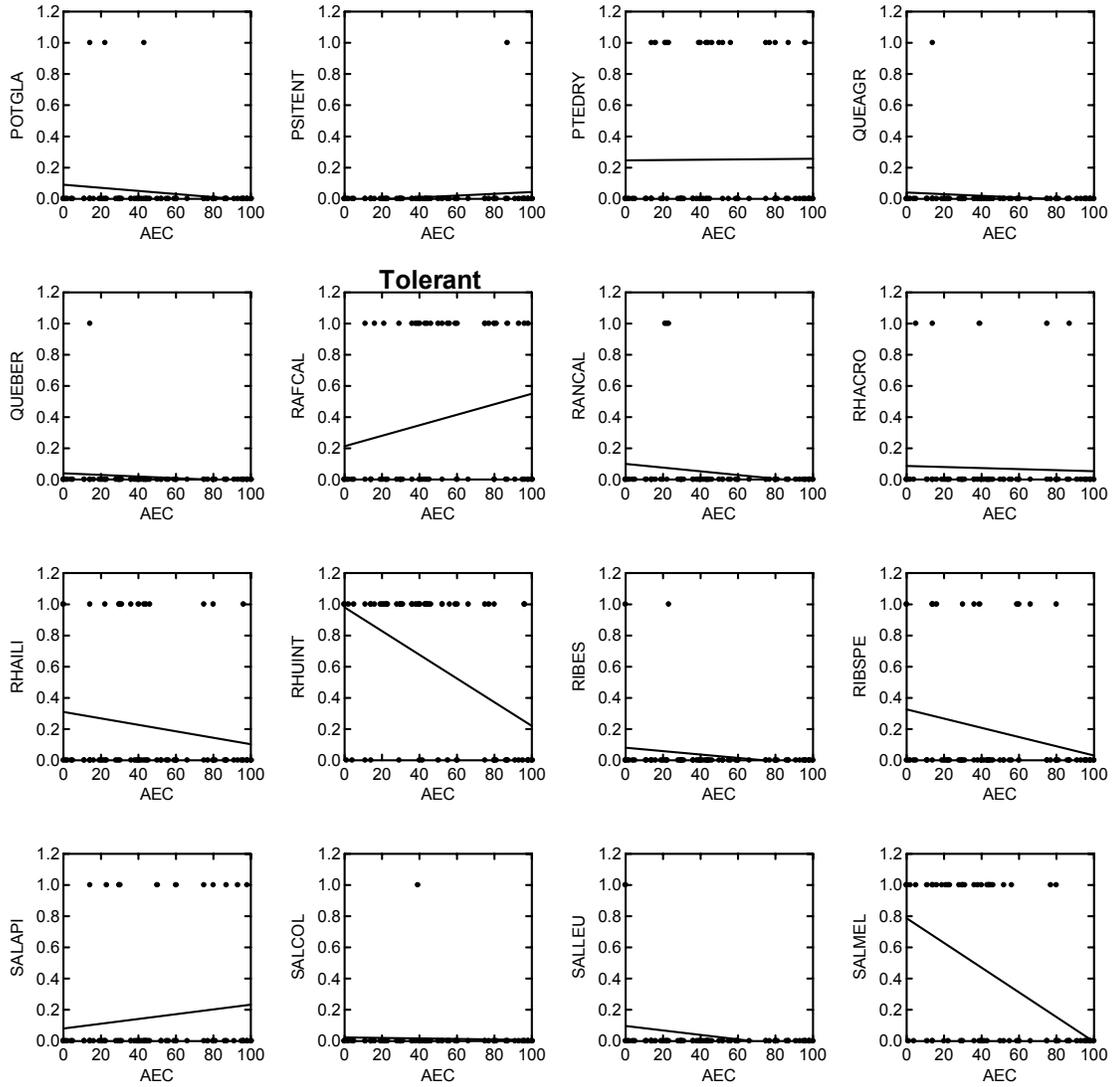
Presence-Absence Data Continued



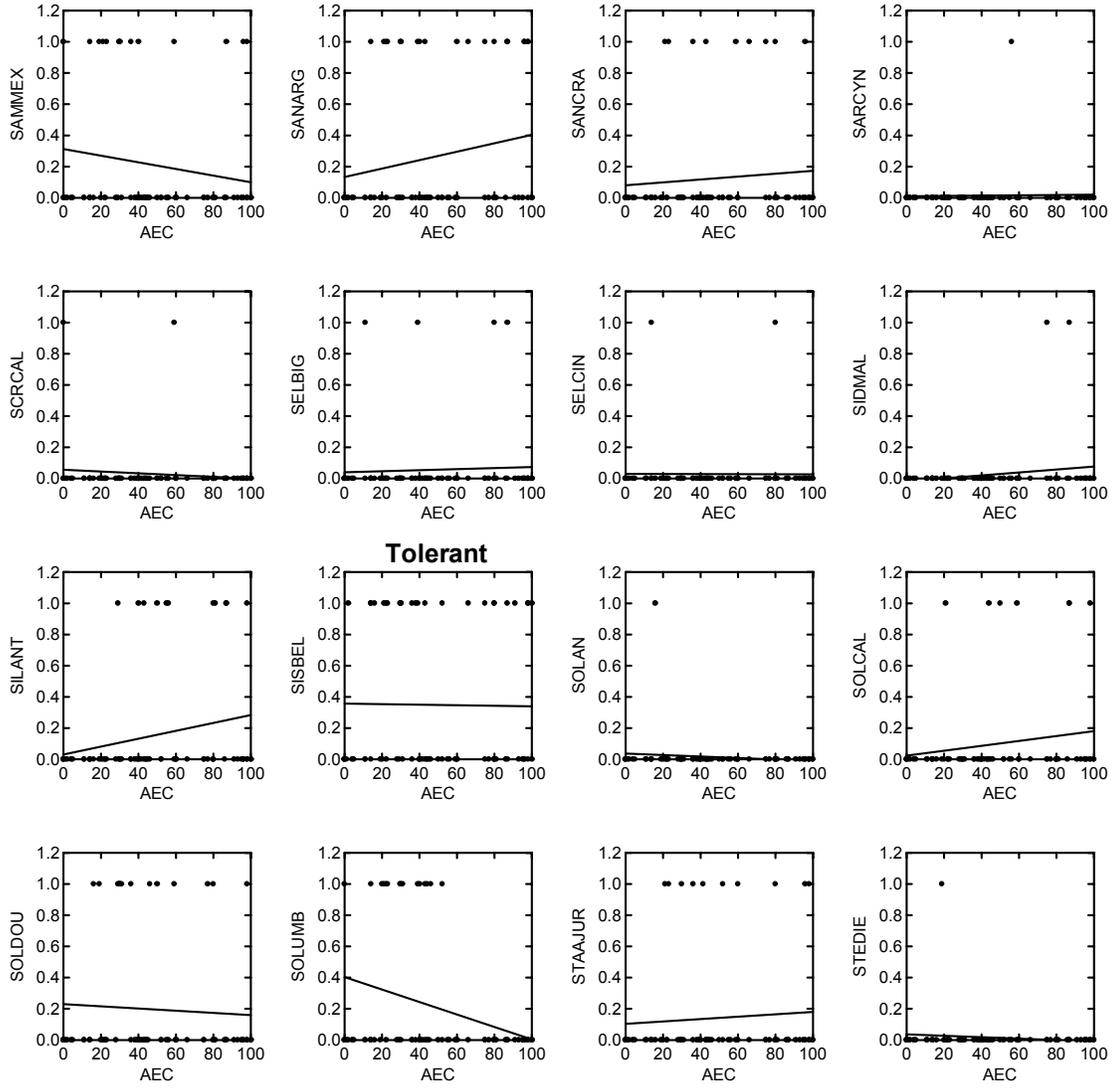
Presence-Absence Data Continued



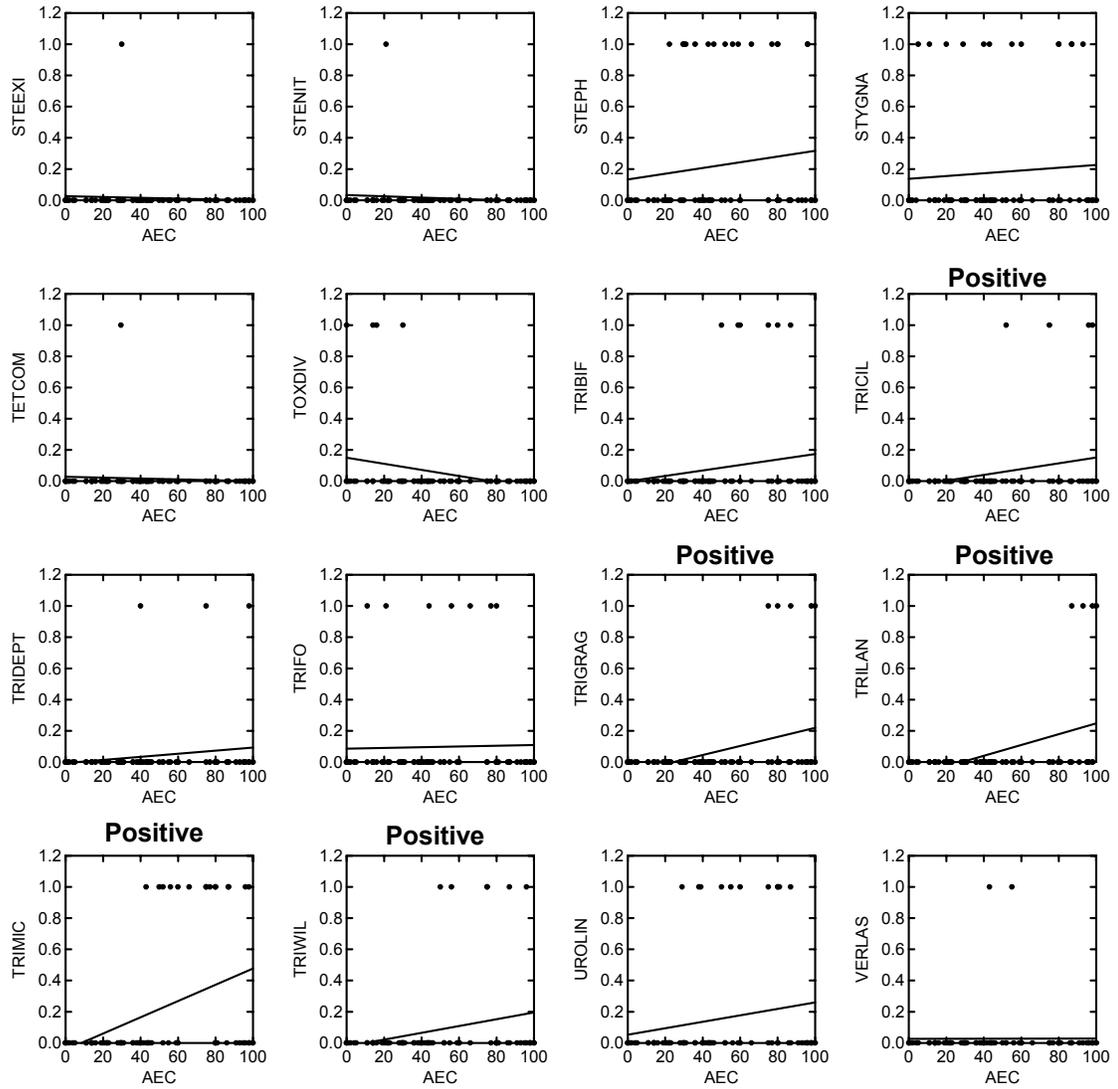
Presence-Absence Data Continued



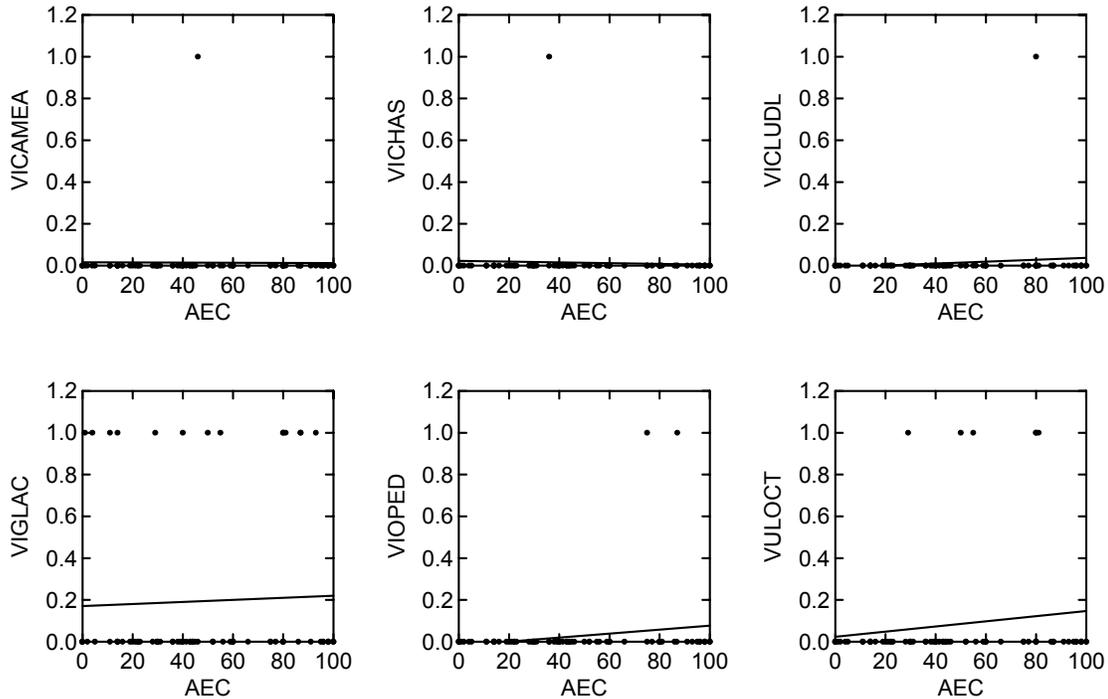
Presence-Absence Data Continued



Presence-Absence Data Continued



Presence-Absence Data Continued



Negative responses.

A number of species showed negative responses. However, the responses were not consistent across years and/or reserves. Furthermore, when we combined the remaining species into a single variable, it did not adequately separate levels of disturbance. Thus, no metric was created using negatively responding species and no species are labeled.

Positive responses. See Table 4.

Table 4. Candidate species for the Proportion Tolerant metric based on presence-absence scatterplots. Species used in the metric had either positive responses or showed tolerance, by being present across a wide range of AEC. Species considered but not included in the metric are also in the table. Toss = species not used. Keep = species included in metric.

Status	Species Code	Reasons.
Toss	ALLPRA	Geophyte, only data is from RJER, inconsistent responses. However, it's common name is "early onion" so if it emerges before the grasses it could have a mechanism for tolerance. Jepson says it likes grassy slopes. Many of our geophytes show up on grassy slopes, so they might all show positive responses, but the real response for most is quadratic.
Keep	AMSMEN	Common in disturbed areas & grasslands. May be officially classified as an agricultural weed. Common name "Rancher's fireweed" seems to indicate a propensity for weediness, and it is toxic to livestock.
Keep	ASTGAM	OK but uncertain. We have seen it in grassier areas, and Jepson indicates it is in open, grassy areas & shrubland. Perhaps a quadratic response, likely ok as a tolerant.
Keep	BACSAR	Shrub. This species does favor disturbed areas, even where invaded.

Keep	BLOCRO	Fine to keep, but little explanation. Common grassland species, so should be tolerant of grassy conditions.
Toss	CENVEN	Only at RJER. Might be OK as this annual may come up a later than others like hemfas, but maybe not as late season. Frequency data is sparse/uninformative. It may well be a positive responder, but our data does not produce much confidence.
Keep	CLAPUR	May have a quadratic response but remain at high AEC longer than some other species. Its absence at low AEC levels could be a fluke. Perhaps it should occur in intact shrublands as habitat preferences are flexible (i.e. shrublands and grasslands). Also true of Cenven, Lupbic, Lupcon, and the Trifoliums below.
Keep	EREST	Matches field experience and the frequency data
TOSS	GUTSAR	Only at RJER and frequency data indicates a possible negative response.
TOSS	JEPPAR	Found on some plots with relatively low AEC, but it is most common at RJER. Maybe just "tolerant". Frequency data is sparse, but could indicate a negative quadratic trend.
Keep	LUPBIC	Another lupine and one common after fire. Quite common in disturbed areas and grasslands.
TOSS	SALAPI	Mostly at RJER; frequency data, though sparse, suggests possible negative quadratic trend. Likely a fluke. A common CSS dominant (white sage).
Keep	TRICIL	Sparse Data. Questionable.
Keep	TRIGRAG	Mostly at RJER; frequency data suggests possible negative response, however this may be one of those legumes that benefits from disturbance (as opposed to invasion)
Keep	TRILAN	Only at RJER, but at most invaded sites; frequency data suggest possible quadratic response to invasion, but anecdotally this species does seem to be most abundant in disturbed/invaded areas. Why only at RJER? Not a range issue, but it is defensible as a positive responder.
Keep	TRIMIC	More tolerant than positive. Not found on sites with <40% AEC. Perhaps intolerant to shrubs? We have seen it growing underneath shrubs.
Keep	TRIWIL	OK, though seems more tolerant, like TRIMIC. We have seen this on plots with low AEC, but only outside shrub canopies.
Keep	ARTCAL	Freq response quadratic, but lit supports a tolerant P/A response
Keep	BACPIL	Freq response negative, but lit supports a tolerant P/A response
Keep	BROCAR	Freq response weak; sparse data
Keep	CALCIL	Freq response positive
Toss	CALMAC	Freq response quadratic, w/ very low freqs toward upper end
Keep	CALSPL	Freq response positive
Toss	CHAPOL	Freq response negative; mostly at RJER
Keep	CRACON	Freq response quadratic
Toss	CRYINT	Freq response negative; drops out after 86% AEC
Keep	DAUPUS	Freq response quadratic
Toss	DICCAP	Freq response flat
Keep	ERIFAS	Freq response flat
Toss	EUCCHRC	Freq response negative
Toss	GALANG	Freq response mostly negative, but present at very high AEC; more reasons to expect sensitivity?
Keep	GALNUT	Freq response negative or quadratic
Toss	GNABIC	Freq response mostly negative, but present at very high AEC
Toss	GNACAL	Freq response mostly negative, but present at very high AEC
Keep	HEMFAS	Freq response positive or quadratic
Toss	ISOMEN	Freq response flat
Toss	LEPNIT	Freq response negative; mostly at RJER
Keep	LESFIL	Freq response flat

Toss	LEYCON	Freq response negative
Toss	LINCAN	Freq response quadratic, w/ fairly low freqs toward upper end
Keep	LOTHAM	Freq response positive
Toss	LOTSTR	Freq response quadratic, w/ fairly low freqs toward upper end
Toss	MALLAU	Freq response quadratic, w/ very low freqs toward upper end
Toss	MIMAU	Freq response negative
Toss	MIRCAL	Highest invasion sites where occurs are RJER, and there it occurs in rocky, uninvaded pockets
Toss	NASLEP	Freq response negative across sites and at individual sites; lit doesn't support tolerance
Toss	NASPUL	Freq response inconsistent across sites; lit doesn't support tolerance
Toss	PLACOL	Mostly at RJER, otherwise drops out after 60 AEC
Toss	PTEDRY	Freq response mostly negative, but present at very high AEC
Keep	RAFCAL	Freq response quadratic
Toss	RHAILI	Freq data sparse
Toss	RHUINT	Freq response negative
Toss	SAMMEX	Sparse P/A data; no freq data
Toss	SANARG	Freq response quadratic, w/ fairly low freqs toward upper end
Toss	SANCRA	Freq response negative
Toss	SILANT	Freq response negative; mostly at RJER
Keep	SISBEL	Freq response flat
Toss	SOLCAL	sparse data
Toss	SOLDOU	sparse data
Toss	STAAJUR	sparsish data; only at OCCR
Toss	STYGNA	Freq response negative; mostly at RJER

Summary Tables

Table 5. IBI Scores for each plot using Year 1, Year 2, and Both datasets. Sample timing refers to the years the sample was collected. Year 1 = 2002, Year 2 = 2003. AEC is the Absolute Exotic Cover for each plot using 2003 data or 2002 data when 2003 was not available.

Reserve	USGS Site	Plot	Sample timing			IBI Score			AEC
			Year1	Year2	Both	Year1	Year2	Both	
OCCR	AWC	1	1	1	1	68	68	72	21
OCCR	AWC	2	1	1	1	62	54	60	39
OCCR	AWC	3	1	0	0	52			86
OCCR	AWC	4	0	1	0		41		98
OCCR	AWC	13	1	0	0	60			14
OCCR	AWC	14	1	1	1	70	70	72	16
OCCR	AWC	15	1	1	1	66	60	64	59
OCCR	AWC	16	1	1	1	72	68	68	43
OCCR	AWC	17	1	1	1	58	62	64	40
CHSP	Chino	7	1	0	0				95
CHSP	Chino	8	1	0	0				0
CHSP	Chino	10	1	0	0				0
CHSP	Chino	11	1	0	0				19
CHSP	Chino	17	1	0	0	52			29.592
CHSP	Chino	18	1	0	0	66			30
RJER	RJER	4	0	1	0		22		100
RJER	RJER	5	1	1	1	22	26	24	100
RJER	RJER	6	1	1	1	22	24	22	98
RJER	RJER	7	1	1	1	34	44	46	55
RJER	RJER	8	0	1	0		50		29
RJER	RJER	9	1	1	1	26	38	30	87
RJER	RJER	10	0	1	0		38		93
RJER	RJER	11	1	1	1	28	30	28	81
RJER	RJER	12	0	1	0		56		50
RJER	RJER	13	0	1	0		40		75
RJER	RJER	14	0	1	0		34		80
RJER	RJER	15	1	1	1	24	26	22	98
RJER	RJER	18	1	1	1	26	38	34	86.869
RJER	RJER	19	1	1	1	38	34	40	80
OCCR	SJHW 1	14	1	1	1	54	58	58	46
OCCR	SJHW 1	16	1	1	1	42	44	46	56
OCCR	SJHW 1	17	1	1	1	50	48	48	60
OCCR	SJHW 1	18	1	0	0	64			45
OCCR	SJHW 1	19	1	1	1	52	56	50	77
OCCR	SJHW 1	20	1	1	1	52	56	54	80
OCCR	SJHW 1	21	1	1	1	36	40	38	75
OCCR	SJHW 1	22	0	1	0		34		96
OCCR	SJHW2	4	0	1	0		48		80
OCCR	SJHW 2	5	1	1	1	60	60	66	52

OCCR	SJHW 2	6	1	1	1	60	62	62	44
OCCR	SJHW 2	7	1	1	1	58	60	62	11
OCCR	SJHW 2	8	0	1	0		38		96
OCCR	SJHW 2	9	1	1	1	46	42	46	66
OCCR	SJHW 2	10	1	1	1	60	60	58	36
OCCR	SJHW 2	11	1	1	1	58	56	58	38
OCCR	SJHW 2	12	1	0	0	46			22.222

Table 6. Summary scores for Ant based IBI metrics for each plot and dataset. Value is the calculated raw quantity of the variable. Score is the 1,3, or 5 category based on the value. APIG = Ant Proportion of Intolerant Genera. APTG = Ant Proportion of Tolerant Genera.

Ants

Reserve	USGS Site	Plot	APIG						APTG					
			Year1		Year2		Both		Year1		Year2		Both	
			Value	Score										
OCCR	AWC	1	0.50	5	0.33	5	0.40	5	0.25	3	0	5	0.00	5
OCCR	AWC	2	0.33	5	0.00	1	0.25	3	0.33	3	0.5	1	0.25	3
OCCR	AWC	3	0.33	5					0.33	3				
OCCR	AWC	4			0.50	5					0.00	5		
OCCR	AWC	13	0.33	5					0.33	3				
OCCR	AWC	14	0.50	5	0.50	5	0.50	5	0.00	5	0.00	5	0.00	5
OCCR	AWC	15	0.50	5	0.00	1	0.50	5	0.00	5	0.00	5	0.00	5
OCCR	AWC	16	0.33	5	0.25	3	0.25	3	0.17	5	0.25	3	0.13	5
OCCR	AWC	17	0.50	5	0.33	5	0.38	5	0.17	5	0.00	5	0.00	5
CHSP	Chino	7												
CHSP	Chino	8												
CHSP	Chino	10												
CHSP	Chino	11												
CHSP	Chino	17	0.00	1					0.00	5				
CHSP	Chino	18	0.17	3					0.00	5				
RJER	RJER	4			0.00	1					0.50	1		
RJER	RJER	5	0.00	1	0.00	1	0.00	1	0.00	5	0.00	5	0.20	3
RJER	RJER	6	0.00	1	0.00	1	0.00	1	0.33	3	0.50	1	0.40	1
RJER	RJER	7	0.20	3	0.25	3	0.29	3	0.20	3	0.25	3	0.14	5
RJER	RJER	8			0.00	1					0.25	3		
RJER	RJER	9	0.00	1	0.00	1	0.00	1	0.00	5	0.60	1	0.50	1
RJER	RJER	10			0.00	1					0.67	1		
RJER	RJER	11	0.00	1	0.00	1	0.00	1	0.00	5	0.50	1	0.50	1
RJER	RJER	12			0.20	3					0.20	3		
RJER	RJER	13			0.29	3					0.14	5		
RJER	RJER	14			0.00	1					0.25	3		
RJER	RJER	15	0.00	1	0.00	1	0.00	1	0.17	5	0.50	1	0.57	1
RJER	RJER	18	0.00	1	0.00	1	0.00	1	0.40	1	0.50	1	0.57	1
RJER	RJER	19	0.00	1	0.00	1	0.00	1	0.20	3	0.50	1	0.43	1
OCCR	SJHW 1	14	0.25	3	0.33	5	0.20	3	0.25	3	0.33	3	0.20	3

OCCR	SJHW 1	16	0.20	3	0.00	1	0.13	3	0.20	3	0.25	3	0.25	3
OCCR	SJHW 1	17	0.17	3	0.14	3	0.22	3	0.17	5	0.29	3	0.22	3
OCCR	SJHW 1	18	0.00	1					0.00	5				
OCCR	SJHW 1	19	0.33	5	0.33	5	0.25	3	0.33	3	0.33	3	0.25	3
OCCR	SJHW 1	20	0.33	5	0.33	5	0.25	3	0.33	3	0.33	3	0.25	3
OCCR	SJHW 1	21	0.00	1	0.00	1	0.00	1	0.00	5	0.25	3	0.20	3
OCCR	SJHW 1	22			0.20	3					0.20	3		
OCCR	SJHW2	4			0.50	5					0.50	1		
OCCR	SJHW 2	5	0.50	5	0.00	1	0.43	5	0.17	5	0.00	5	0.14	5
OCCR	SJHW 2	6	0.25	3	0.30	3	0.27	3	0.25	3	0.33	3	0.27	3
OCCR	SJHW 2	7	0.25	3	0.33	5	0.33	5	0.25	3	0.17	5	0.17	5
OCCR	SJHW 2	8			0.00	1					0.00	5		
OCCR	SJHW 2	9	0.50	5	0.00	1	0.38	5	0.17	5	0.50	1	0.25	3
OCCR	SJHW 2	10	0.33	5	0.17	3	0.29	3	0.00	5	0.33	3	0.29	3
OCCR	SJHW 2	11	0.67	5	0.20	3	0.33	5	0.33	3	0.40	1	0.17	5
OCCR	SJHW 2	12	0.67	5					0.33	3				

ANTS

Site	Plot	APIG						APTG					
		Year1		Year2		Both		Year1		Year2		Both	
		Value	Score	Value	Score	Value	Score	Value	Score	Value	Score	Value	Score
AWC	1	0.5	5	0.33	5	0.40	5	0.25	3	0	5	0.00	5
AWC	2	0.33333	5	0	1	0.25	3	0.33333	3	0.5	1	0.25	3
AWC	3	0.33333	5					0.33333	3				
AWC	4			0.50	5					0.00	5		
AWC	13	0.33333	5					0.33333	3				
AWC	14	0.5	5	0.50	5	0.50	5	0	5	0.00	5	0.00	5
AWC	15	0.5	5	0.00	1	0.50	5	0	5	0.00	5	0.00	5
AWC	16	0.33333	5	0.25	3	0.25	3	0.16667	5	0.25	3	0.13	5
AWC	17	0.5	5	0.33	5	0.38	5	0.16667	5	0.00	5	0.00	5
Chino	7												
Chino	8												
Chino	10												
Chino	11												
Chino	17	0.00	1					0.00	5				
Chino	18	0.17	3					0.00	5				
RJER	4			0.00	1					0.50	1		
RJER	5	0	1	0.00	1	0.00	1	0	5	0.00	5	0.20	3
RJER	6	0	1	0.00	1	0.00	1	0.33333	3	0.50	1	0.40	1
RJER	7	0.2	3	0.25	3	0.29	3	0.2	3	0.25	3	0.14	5
RJER	8			0.00	1					0.25	3		
RJER	9	0	1	0.00	1	0.00	1	0	5	0.60	1	0.50	1
RJER	10			0.00	1					0.67	1		
RJER	11	0	1	0.00	1	0.00	1	0	5	0.50	1	0.50	1
RJER	12			0.20	3					0.20	3		
RJER	13			0.29	3					0.14	5		

RJER	14			0.00	1					0.25	3		
RJER	15	0	1	0.00	1	0.00	1	0.16667	5	0.50	1	0.57	1
RJER	18	0	1	0.00	1	0.00	1	0.4	1	0.50	1	0.57	1
RJER	19	0	1	0.00	1	0.00	1	0.2	3	0.50	1	0.43	1
SJHW 1	14	0.25	3	0.33	5	0.20	3	0.25	3	0.33	3	0.20	3
SJHW 1	16	0.2	3	0.00	1	0.13	3	0.2	3	0.25	3	0.25	3
SJHW 1	17	0.16667	3	0.14	3	0.22	3	0.16667	5	0.29	3	0.22	3
SJHW 1	18	0	1					0	5				
SJHW 1	19	0.33333	5	0.33	5	0.25	3	0.33333	3	0.33	3	0.25	3
SJHW 1	20	0.33333	5	0.33	5	0.25	3	0.33333	3	0.33	3	0.25	3
SJHW 1	21	0	1	0.00	1	0.00	1	0	5	0.25	3	0.20	3
SJHW 1	22			0.20	3					0.20	3		
SJHW2	4			0.50	5					0.50	1		
SJHW 2	5	0.5	5	0.00	1	0.43	5	0.16667	5	0.00	5	0.14	5
SJHW 2	6	0.25	3	0.30	3	0.27	3	0.25	3	0.33	3	0.27	3
SJHW 2	7	0.25	3	0.33	5	0.33	5	0.25	3	0.17	5	0.17	5
SJHW 2	8			0.00	1					0.00	5		
SJHW 2	9	0.5	5	0.00	1	0.38	5	0.16667	5	0.50	1	0.25	3
SJHW 2	10	0.33333	5	0.17	3	0.29	3	0	5	0.33	3	0.29	3
SJHW 2	11	0.66667	5	0.20	3	0.33	5	0.33333	3	0.40	1	0.17	5
SJHW 2	12	0.66667	5					0.33333	3				

Table 7. Summary scores for the Arthropod IBI metric for each plot and dataset. Value is the calculated raw quantity of the variable. Score is the 1,3, or 5 category based on the value. APTO = Arthropod Proportion of Tolerant Orders.

			Arthropods					
			APTO					
			Year1		Year2		Both	
Reserve	USGS Site	Plot	Value	Score	Value	Score	Value	Score
OCCR	AWC	1	0.00	5	0	5	0.00	5
OCCR	AWC	2	0.33	1	0	5	0.00	5
OCCR	AWC	3	0.17	1				
OCCR	AWC	4			0.09	3		
OCCR	AWC	13	0.33	1				
OCCR	AWC	14	0.00	5	0.00	5	0.00	5
OCCR	AWC	15	0.00	5	0.11	3	0.13	3
OCCR	AWC	16	0.00	5	0.00	5	0.08	3
OCCR	AWC	17	0.00	5	0.00	5	0.00	5
CHSP	Chino	7						
CHSP	Chino	8						
CHSP	Chino	10						
CHSP	Chino	11						
CHSP	Chino	17	1.00	1				

CHSP	Chino	18	0.33	1				
RJER	RJER	4			0.31	1		
RJER	RJER	5	0.50	1	0.25	1	0.29	1
RJER	RJER	6	0.67	1	0.27	1	0.19	1
RJER	RJER	7	0.20	1	0.27	1	0.22	1
RJER	RJER	8			0.09	3		
RJER	RJER	9	0.60	1	0.09	3	0.13	3
RJER	RJER	10			0.08	3		
RJER	RJER	11	0.50	1	0.14	3	0.09	3
RJER	RJER	12			0.00	5		
RJER	RJER	13			0.13	3		
RJER	RJER	14			0.08	3		
RJER	RJER	15	0.67	1	0.20	1	0.17	1
RJER	RJER	18	0.80	1	0.10	3	0.13	3
RJER	RJER	19	0.60	1	0.20	1	0.15	3
OCCR	SJHW 1	14	0.25	1	0.14	3	0.15	3
OCCR	SJHW 1	16	0.40	1	0.10	3	0.08	3
OCCR	SJHW 1	17	0.17	1	0.10	3	0.13	3
OCCR	SJHW 1	18	0.00	5				
OCCR	SJHW 1	19	0.33	1	0.11	3	0.15	3
OCCR	SJHW 1	20	0.33	1	0.00	5	0.18	1
OCCR	SJHW 1	21	0.50	1	0.08	3	0.07	3
OCCR	SJHW 1	22			0.09	3		
OCCR	SJHW2	4			0.00	5		
OCCR	SJHW 2	5	0.17	1	0.22	1	0.13	3
OCCR	SJHW 2	6	0.38	1	0.09	3	0.13	3
OCCR	SJHW 2	7	0.25	1	0.00	5	0.00	5
OCCR	SJHW 2	8			0.00	5		
OCCR	SJHW 2	9	0.17	1	0.08	3	0.17	1
OCCR	SJHW 2	10	0.33	1	0.11	3	0.08	3
OCCR	SJHW 2	11	0.00	5	0.00	5	0.07	3
OCCR	SJHW 2	12	0	5				

Table 8. Summary scores for the Bird IBI metrics for each plot and dataset. Value (“V”) is the calculated raw quantity of the variable. Score (“S”) is the 1,3, or 5 category based on the value.

		Birds																								
		BPTS						BPIS						BRAT						BRAI						
		Year1		Year2		Both		Year1		Year2		Both		Year1		Year2		Both		Year1		Year2		Both		
Reserve	USGS Site	V	S	V	S	V	S	V	S	V	S	V	S	V	S	V	S	V	S	V	S	V	S	V	S	
OCCR	AWC	1	0.00	5	0.00	5	0.00	5	0.36	3	0.36	3	0.36	5	0.00	5	0.13	3	0.07	5	0.22	3	0.30	3	0.26	3
OCCR	AWC	2	0.00	5	0.00	5	0.07	3	0.50	5	0.38	3	0.43	5	0.02	5	0.11	5	0.08	5	0.50	5	0.22	3	0.36	5
OCCR	AWC	3	0.00	5					0.33	3					0.07	3					0.19	3				
OCCR	AWC	4			0.06	3					0.24	3					0.10	5					0.27	3		
OCCR	AWC	13	0.00	5					0.40	3					0.09	3					0.41	5				
OCCR	AWC	14	0.00	5	0.00	5	0.00	5	0.38	3	0.45	5	0.36	5	0.03	5	0.15	3	0.09	5	0.68	5	0.49	5	0.54	5
OCCR	AWC	15	0.00	5	0.00	5	0.00	5	0.50	5	0.33	3	0.33	3	0.08	3	0.12	3	0.10	3	0.76	5	0.45	5	0.58	5
OCCR	AWC	16	0.00	5	0.00	5	0.00	5	0.50	5	0.44	5	0.50	5	0.07	3	0.16	3	0.12	3	0.48	5	0.42	5	0.43	5
OCCR	AWC	17	0.00	5	0.00	5	0.00	5	0.38	3	0.44	5	0.43	5	0.19	3	0.39	3	0.30	3	0.57	5	0.32	3	0.40	5
CHSP	Chino	7																								
CHSP	Chino	8																								
CHSP	Chino	10																								
CHSP	Chino	11																								
CHSP	Chino	17	0.00	5					0.50	5					0.21	3					0.26	3				
CHSP	Chino	18	0.00	5					0.36	3					0.13	3					0.41	5				
RJER	RJER	4			0.42	5					0.00	1					0.75	1					0.00	1		
RJER	RJER	5	0.44	1	0.50	5	0.55	1	0.11	1	0.13	1	0.09	1	0.88	1	0.75	1	0.84	5	0.03	1	0.13	1	0.06	1
RJER	RJER	6	0.29	1	0.25	5	0.29	1	0.14	1	0.17	1	0.18	1	0.32	3	0.27	3	0.31	5	0.04	1	0.09	1	0.06	1
RJER	RJER	7	0.10	3	0.18	5	0.13	3	0.40	3	0.27	3	0.31	3	0.22	3	0.61	1	0.38	5	0.15	3	0.18	3	0.15	1
RJER	RJER	8			0.12	3					0.48	5					0.34	3					0.20	3		
RJER	RJER	9	0.33	1	0.18	5	0.27	1	0.11	1	0.27	3	0.20	1	0.38	1	0.28	3	0.31	3	0.03	1	0.08	1	0.05	1
RJER	RJER	10			0.38	5					0.13	1					0.54	1					0.04	1		

RJER	RJER	11	0.38	1	0.10	3	0.19	1	0.25	1	0.30	3	0.25	3	0.63	1	0.16	3	0.30	3	0.19	3	0.12	1	0.13	1
RJER	RJER	12			0.09	3					0.27	3					0.21	3					0.21	3		
RJER	RJER	13			0.08	3					0.15	1					0.21	3					0.14	3		
RJER	RJER	14			0.25	5					0.25	3					0.31	3					0.13	1		
RJER	RJER	15	0.33	1	0.15	5	0.14	1	0.17	1	0.23	3	0.21	1	0.38	1	0.52	1	0.44	5	0.08	1	0.20	3	0.14	1
RJER	RJER	18	0.25	1	0.00	5	0.16	1	0.25	1	0.20	3	0.21	1	0.50	1	0.21	3	0.32	5	0.25	3	0.21	3	0.22	3
RJER	RJER	19	0.00	5	0.00	5	0.00	5	0.40	3	0.17	1	0.23	3	0.57	1	0.21	3	0.31	3	0.21	3	0.12	1	0.15	1
OCCR	SJHW 1	14	0.00	5	0.08	3	0.07	3	0.36	3	0.25	3	0.29	3	0.27	3	0.28	3	0.28	3	0.41	5	0.36	3	0.39	5
OCCR	SJHW 1	16	0.00	5	0.07	3	0.06	3	0.29	3	0.29	3	0.29	3	0.22	3	0.13	3	0.16	3	0.03	1	0.31	3	0.30	3
OCCR	SJHW 1	17	0.00	5	0.15	5	0.12	3	0.54	5	0.31	3	0.41	5	0.20	3	0.35	3	0.30	3	0.36	3	0.29	3	0.31	3
OCCR	SJHW 1	18	0.00	5					0.46	5					0.13	3					0.47	5				
OCCR	SJHW 1	19	0.00	5	0.15	5	0.11	3	0.36	3	0.23	3	0.28	3	0.31	3	0.21	3	0.24	3	0.28	3	0.19	3	0.21	3
OCCR	SJHW 1	20	0.00	5	0.07	3	0.06	3	0.40	3	0.29	3	0.38	5	0.15	3	0.25	3	0.20	3	0.18	3	0.33	3	0.24	3
OCCR	SJHW 1	21	0.00	5	0.00	5	0.00	5	0.40	3	0.31	3	0.28	3	0.15	3	0.22	3	0.22	3	0.18	3	0.24	3	0.15	1
OCCR	SJHW 1	22			0.23	5					0.23	3					0.31	3					0.09	1		
OCCR	SJHW 2	4			0.00	5					0.33	3					0.23	3					0.18	3		
OCCR	SJHW 2	5	0.00	5	0.00	5	0.00	5	0.38	3	0.50	5	0.42	5	0.14	3	0.13	3	0.14	3	0.50	5	0.57	5	0.55	5
OCCR	SJHW 2	6	0.00	5	0.15	5	0.13	3	0.50	5	0.38	3	0.31	3	0.29	3	0.21	3	0.23	3	0.47	5	0.30	3	0.34	3
OCCR	SJHW 2	7	0.00	5	0.11	3	0.08	3	0.50	5	0.33	3	0.42	5	0.14	3	0.33	3	0.26	3	0.43	5	0.30	3	0.33	3
OCCR	SJHW 2	8			0.18	5					0.18	3					0.25	3					0.14	1		
OCCR	SJHW 2	9	0.06	3	0.00	5	0.06	3	0.38	3	0.38	3	0.35	5	0.12	3	0.14	3	0.12	3	0.12	3	0.26	3	0.16	3
OCCR	SJHW 2	10	0.00	5	0.00	5	0.00	5	0.42	5	0.42	5	0.33	3	0.06	3	0.13	3	0.08	5	0.23	3	0.40	5	0.28	3
OCCR	SJHW 2	11	0.10	3	0.08	3	0.13	3	0.40	3	0.33	3	0.33	3	0.15	3	0.21	3	0.18	3	0.20	3	0.42	5	0.31	3
OCCR	SJHW 2	12	0.00	5					0.29	3					0.41	1					0.12	3				

Table 9. Summary scores for the Herp IBI metrics for each plot and dataset. Value (“V”) is the calculated raw quantity of the variable. Score (“S”) is the 1,3, or 5 category based on the value. HPIS = Herpetofauna Proportion of Intolerant Species. HPTS = Herpetofauna Proportion of Tolerant Species, HRAI = Herpetofauna Relative Abundance of Intolerant Species. The yellow values were interpolated scores because these three plots did not contain herpetofauna sampling arrays.

		Herpetofauna																		
		HPIS						HPTS						HRAI						
Reserve	USGS Site	Year1		Year2		Both		Year1		Year2		Both		Year1		Year2		Both		
		V	S	V	S	V	S	V	S	V	S	V	S	V	S	V	S	V	S	
OCCR	AWC	1	0.75	5	0.75	5	0.75	5	0.00	5	0.00	5	0.00	5	0.96	5	0.96	5	0.96	5
OCCR	AWC	2	0.38	3	0.38	3	0.38	3	0.13	3	0.13	3	0.13	3	0.74	3	0.74	3	0.74	3
OCCR	AWC	3	0.50	5				0.00	5					0.80	5					
OCCR	AWC	4			0.14	1				0.37	1					0.41	3			
OCCR	AWC	13	0.50	5				0.00	5					0.71	3					
OCCR	AWC	14	0.38	3	0.38	3	0.38	3	0.00	5	0.00	5	0.00	5	0.83	5	0.83	5	0.83	5
OCCR	AWC	15	0.40	3	0.40	3	0.40	3	0.00	5	0.00	5	0.00	5	0.45	3	0.45	3	0.45	3
OCCR	AWC	16	0.60	5	0.60	5	0.60	5	0.00	5	0.00	5	0.00	5	0.67	3	0.67	3	0.67	3
OCCR	AWC	17	0.17	1	0.17	1	0.17	1	0.00	5	0.00	5	0.00	5	0.35	3	0.35	3	0.35	3
CHSP	Chino	7																		
CHSP	Chino	8																		
CHSP	Chino	10																		
CHSP	Chino	11																		
CHSP	Chino	17	0.22	1				0.00	5					0.77	3					
CHSP	Chino	18	0.43	3				0.00	5					0.84	5					
RJER	RJER	4			0.13	1				0.38	1					0.41	3			
RJER	RJER	5	0.00	1	0.00	1	0.00	1	0.33	1	0.33	1	0.33	1	0.77	3	0.77	3	0.77	3
RJER	RJER	6	0.00	1	0.00	1	0.00	1	0.44	1	0.44	1	0.44	1	0.56	3	0.56	3	0.56	3
RJER	RJER	7	0.10	1	0.10	1	0.10	1	0.30	1	0.30	1	0.30	1	0.60	3	0.60	3	0.60	3
RJER	RJER	8			0.38	3				-0.04	5					0.61	3			
RJER	RJER	9	0.17	1	0.17	1	0.17	1	0.33	1	0.33	1	0.33	1	0.26	1	0.26	1	0.26	1

Table 10. Summary scores for the Mammal IBI metrics for each plot and dataset. Value (“V”) is the calculated raw quantity of the variable. Score (“S”) is the 1,3, or 5 category based on the value. HPIS = Herpetofauna Proportion of Intolerant Species. MNSR = Mammal Native Species Richness, MPTS = Mammal Proportion of Tolerant Species, MRAI = Mammal Relative Abundance of Intolerant Species.

		Mammals																							
		MNSR						MPTS						MRAI											
		Year1		Year2		Both		Year1		Year2		Both		Year1		Year2		Both							
Reserve	USGS Site	V	S	V	S	V	S	V	S	V	S	V	S	V	S	V	S	V	S						
OCCR	AWC	5	3	5	3	5	3	0.60	5	0.60	5	0.60	5	0.27	3	0.44	3	0.39	3						
OCCR	AWC	5	3	6	3	6	3	0.40	5	0.50	5	0.50	5	0.43	3	0.48	5	0.46	5						
OCCR	AWC	3	1	1				0.00	5					1.00	5										
OCCR	AWC			0	1											0.00	1								
OCCR	AWC	7	5					0.43	5					0.82	5										
OCCR	AWC	5	3	5	3	6	3	0.40	5	0.40	5	0.50	5	0.57	5	0.60	5	0.59	5						
OCCR	AWC	5	3	3	5	3	3	0.40	5	0.33	5	0.40	5	0.62	5	0.68	5	0.64	5						
OCCR	AWC	6	3	6	3	6	3	0.33	5	0.33	5	0.33	5	0.56	5	0.64	5	0.61	5						
OCCR	AWC	5	3	6	3	6	3	0.60	5	0.50	5	0.50	5	0.09	1	0.24	3	0.15	3						
CHSP	Chino																								
CHSP	Chino																								
CHSP	Chino																								
CHSP	Chino																								
CHSP	Chino	5	3					0.80	1					0.22	3										
CHSP	Chino	7	5					0.57	5					0.67	5										
RJER	RJER			1	1					1.00	1					0.00	1								
RJER	RJER	3	1	2	1	3	1	1.00	1	1.00	1	1.00	1	0.00	1	0.00	1	0.00	1						
RJER	RJER	2	1	2	1	2	1	1.00	1	1.00	1	1.00	1	0.00	1	0.00	1	0.00	1						
RJER	RJER	3	1	7	5	7	5	1.00	1	0.43	5	0.43	5	0.00	1	0.09	1	0.06	1						
RJER	RJER			3	1					0.33	5					0.03	1								
RJER	RJER	4	1	6	3	6	3	0.75	1	0.67	3	0.67	3	0.03	1	0.12	3	0.09	1						
RJER	RJER			5	3					0.40	5					0.17	3								

RJER	RJER	11	5	3	6	3	6	3	0,8	1	0,67	3	0,02	1	0,03	1	0,02	1
RJER	RJER	12			7	5					0,43	5			0,27	3		
RJER	RJER	13			4	1					0,75	1			0,00	1		
RJER	RJER	14			3	1					1,00	1			0,00	1		
RJER	RJER	15	4	1	3	1	4	1	1,00	1	1,00	1	0,00	1	0,00	1	0,00	1
RJER	RJER	18	4	1	4	1	4	1	0,75	1	0,75	1	0,00	1	0,00	1	0,00	1
RJER	RJER	19	5	3	4	1	5	3	0,60	5	0,50	5	0,09	1	0,10	1	0,10	1
OCCR	SJHW 1	14	4	1	5	3	5	3	0,50	5	0,40	5	0,33	3	0,43	3	0,40	3
OCCR	SJHW 1	16	4	1	3	1	4	1	0,50	5	0,33	5	0,09	1	0,32	3	0,24	3
OCCR	SJHW 1	17	6	3	6	3	6	3	0,67	3	0,67	3	0,12	3	0,23	3	0,18	3
OCCR	SJHW 1	18	7	5					0,57	5			0,15	3				
OCCR	SJHW 1	19	6	3	6	3	6	3	0,67	3	0,67	3	0,41	3	0,49	5	0,45	3
OCCR	SJHW 1	20	6	3	6	3	7	5	0,67	3	0,50	5	0,42	3	0,38	3	0,43	3
OCCR	SJHW 1	21	4	1	6	3	6	3	0,75	1	0,67	3	0,03	1	0,05	1	0,03	1
OCCR	SJHW 1	22			2	1					1,00	1			0,00	1		
OCCR	SJHW 2	4			6	3					0,50	5			0,12	3		
OCCR	SJHW 2	5	6	3	5	3	6	3	0,50	5	0,60	5	0,43	3	0,63	5	0,52	5
OCCR	SJHW 2	6	6	3	7	5	7	5	0,67	3	0,57	5	0,54	5	0,45	3	0,49	5
OCCR	SJHW 2	7	4	1	6	3	6	3	0,50	5	0,67	3	0,46	5	0,58	5	0,54	5
OCCR	SJHW 2	8			3	1	1				0,67	3			0,08	1		
OCCR	SJHW 2	9	6	3	6	3	6	3	0,67	3	0,67	3	0,36	3	0,37	3	0,37	3
OCCR	SJHW 2	10	5	3	5	3	5	3	0,60	5	0,60	5	0,30	3	0,32	3	0,30	3
OCCR	SJHW 2	11	5	3	5	3	5	3	0,60	5	0,60	5	0,46	5	0,54	5	0,51	5
OCCR	SJHW 2	12	3	1					1,00	1			0,00	1				

Table 11. Summary scores for the Plant IBI metrics for each plot and dataset. Value (“V”) is the calculated raw quantity of the variable. Score (“S”) is the 1,3, or 5 category based on the value. AWC = Absolute Woody Cover. TNWR = Total Native Woody Richness, PPTS = Plant Proportion of Tolerant Species.

		Plants																							
		AWC												TNWR						PPTS					
Reserve	USGS Site	Year1		Year2		Both		Year1		Year2		Both		Year1		Year2		Both		Year1		Year2		Both	
		V	S	V	S	V	S	V	S	V	S	V	S	V	S	V	S	V	S	V	S	V	S	V	S
OCCR	AWC	90.0	5	90.0	5	90.00	5	16	5	15	5	15	5	15	5	0.23	3	0.23	3	0.23	3	0.23	3	0.23	3
OCCR	AWC	79.0	5	72.0	3	72.00	3	15	5	12	3	12	3	12	3	0.29	3	0.23	3	0.23	3	0.23	3	0.23	3
OCCR	AWC	7.0	1					2	1							0.67	1								
OCCR	AWC			1.0	1	0.47	1			6	3	6	3	6	3			0.29	3						
OCCR	AWC	70.0	3					11	3							0.36	1								
OCCR	AWC	95.0	5	95.0	5	95.00	5	11	3	12	3	12	3	12	3	0.24	3	0.24	3	0.24	3	0.24	3	0.24	3
OCCR	AWC	88.0	5	91.0	5	91.00	5	10	3	12	3	12	3	12	3	0.38	1	0.28	3	0.28	3	0.28	3	0.28	3
OCCR	AWC	77.0	5	88.0	5	88.00	5	13	5	14	5	14	5	14	5	0.24	3	0.25	3	0.25	3	0.25	3	0.25	3
OCCR	AWC	68.0	3	71.0	3	71.00	3	10	3	14	5	14	5	14	5	0.24	3	0.20	3	0.20	3	0.20	3	0.20	3
CHSP	Chino	0.0	1					1	1							0.33	3								
CHSP	Chino	93.9	5					15	5							0.04	5								
CHSP	Chino	95.0	5					10	3							0.07	5								
CHSP	Chino	70.0	3					13	5							0.15	5								
CHSP	Chino	69.4	3					16	5							0.15	5								
CHSP	Chino	69.0	3					17	5							0.13	5								
RJER	RJER			0.0	1					3	1							0.60	1						
RJER	RJER	1.0	1	2.0	1	2.00	1	4	1	4	1	4	1	4	1	0.57	1	0.63	1	0.63	1	0.63	1	0.63	1
RJER	RJER	2.0	1	3.0	1	3.00	1	2	1	2	1	2	1	2	1	0.50	1	0.50	1	0.50	1	0.50	1	0.50	1
RJER	RJER	64.0	3	59.0	3	59.00	3	6	3	7	3	7	3	7	3	0.38	1	0.24	3	0.24	3	0.24	3	0.24	3
RJER	RJER			79.0	5					4	1							0.16	5						
RJER	RJER	50.0	3	54.0	3	54.00	3	7	3	7	3	7	3	7	3	0.31	3	0.30	3	0.30	3	0.30	3	0.30	3
RJER	RJER			47.0	3					7	3							0.29	3						

RJER	RJER	11	49.0	3	39.0	1	39.00	1	4	3	1	3	1	0.40	1	0.39	1	0.39	1
RJER	RJER	12			93.0	5				7	3					0.22	3		
RJER	RJER	13			71.0	3				12	3					0.24	3		
RJER	RJER	14			26.0	1					4	1				0.28	3		
RJER	RJER	15	17.0	1	20.0	1	20.00	1	10	3	10	3	10	3	3	0.39	1	0.39	1
RJER	RJER	18	44.0	3	45.5	3	45.45	3	7	3	10	3	10	3	1	0.27	3	0.27	3
RJER	RJER	19	68.0	3	63.0	3	63.00	3	5	1	6	3	6	3	5	0.15	5	0.15	5
OCCR	SJHW 1	14	71.0	3	84.0	5	84.00	5	13	5	14	5	14	5	5	0.18	5	0.18	5
OCCR	SJHW 1	16	59.0	3	69.0	3	69.00	3	11	3	11	3	11	3	3	0.28	3	0.28	3
OCCR	SJHW 1	17	66.7	3	73.0	3	73.00	3	8	3	7	3	7	3	3	0.33	3	0.33	3
OCCR	SJHW 1	18	80.0	5					10	3				0.25	3				
OCCR	SJHW 1	19	58.0	3	72.0	3	72.00	3	9	3	10	3	10	3	3	0.27	3	0.27	3
OCCR	SJHW 1	20	44.0	3	47.0	3	47.00	3	10	3	10	3	10	3	3	0.28	3	0.28	3
OCCR	SJHW 1	21	41.0	3	43.0	3	43.00	3	7	3	8	3	8	3	1	0.50	1	0.50	1
OCCR	SJHW 1	22			27.0	1					9	3	7	3		0.35	1		
OCCR	SJHW 2	4			35.0	1					9	3	14	5		0.40	1		
OCCR	SJHW 2	5	80.0	5	92.0	5	92.00	5	9	3	10	3	10	3	5	0.19	5	0.19	5
OCCR	SJHW 2	6	69.0	3	74.0	3	74.00	3	15	5	15	5	15	5	3	0.18	5	0.18	5
OCCR	SJHW 2	7	89.0	5	84.0	5	84.00	5	11	3	8	3	8	3	3	0.22	3	0.22	3
OCCR	SJHW 2	8			22.0	1					9	3	10	3		0.36	1		
OCCR	SJHW 2	9	65.0	3	68.0	3	68.00	3	12	3	10	3	10	3	1	0.39	1	0.39	1
OCCR	SJHW 2	10	77.0	5	83.0	5	83.00	5	7	3	9	3	9	3	3	0.24	3	0.24	3
OCCR	SJHW 2	11	80.0	5	78.0	5	78.00	5	12	3	12	3	12	3	3	0.33	3	0.33	3
OCCR	SJHW 2	12	52.5	3			90.00		7	3				0.46	1				