Coachella Valley Multiple Species Habitat Conservation Plan Monitoring Program

Final Report

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CENTER FOR CONSERVATION BIOLOGY

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County of Riverside

September 2005

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CHAPTER 1

CCB MONITORING PROGRAM

Background

Ranked as one of six regions of conservation priority (Chaplin et al. 2000; Noss et al., 2001), the Coachella Valley is comprised of a diverse mixture of desert and mountain habitats. It is located in the central portion of Riverside County, the westernmost edge of the Colorado Desert. Riverside County has been experiencing a rapid population growth over the past few decades. With an already substantial population taxing available natural resources, it was recognized that inevitably the needs of this burgeoning population would be incompatible with conservation of sensitive species and natural habitats. To address these potential conflicts, the California State Legislature passed the Natural Community Conservation Planning Act of 1991. The Act aimed to protect regional natural diversity while simultaneously allowing for development and economic growth throughout the State. It differs from the Federal and State Endangered Species Acts in that it is a community based approach to conservation rather than species-centric, thus recognizing the ecological milieu that must be protected in order to protect the species. A direct result of the Act is the formulation of Multiple Species Habitat Conservation Plans (MSHCP) of which Riverside County has one approved (Western Riverside County Multiple Species Habitat Conservation Plan) and several in development, one of them being the Coachella Valley Multiple Species Habitat Conservation Plan (CV MSHCP). The expectation in these plans is that biological and economic interests will come together and forge a management plan that will satisfy both present and future needs of all parties involved.

Key to the successful implementation of a MSHCP is the development of an efficient and effective means of monitoring sensitive species and habitats through time, which will facilitate the detection of threats as quickly as possible so that appropriate management actions may be taken. Such monitoring programs depend on an in-depth biological knowledge of the involved communities and ecosystems, as well as the application of a rigorous scientific approach in the conceptualization, development, and implementation of the monitoring efforts (NCEAS, 1999; Noss et al., 2001; Science Review Panel, 2003)

The CV MSHCP monitoring program is a collaborative effort between professional land managers, wildlife agencies, and researchers from the Center for Conservation Biology (CCB) at the University of California Riverside. The CV MSHCP is spearheaded by the Coachella Valley Association of Governments (CVAG). In 2002, CVAG established a Working Group as a forum for agencies with land management responsibilities to provide input and receive research results. Aside from CVAG and CCB associates, the Working

Group consists of representatives from the California Department of Fish and Game, Agua Caliente Band of Cahuilla Indians, Cabazon Band of Cahuilla Indians, Center for Natural Lands Management, Coachella Valley Mountains Conservancy, Coachella Valley Water District, Palm Springs Desert Museum, Terra Nova, Body Deep Canyon Desert Research Center, U.S. Department of Agriculture, U.S. Fish and Wildlife Service, U.S. Bureau of Land Management, U.S. Forest Service, and U.S. National Park Service.

Coachella Valley Multiple Species Habitat Conservation Plan

The Coachella Valley Multiple Species Habitat Conservation Plan (CV MSHCP) area covers 1.1 million acres, and seeks to protect approximately 747,600 acres of natural desert and mountain natural communities in an effort to conserve 27 plant and animal species (Appendix 1). The Plan Area includes unincorporated Riverside County land east of the crest of the San Jacinto Mountains and the jurisdictional areas of the cities of Palm Springs, Desert Hot Springs, Cathedral City, Rancho Mirage, Palm Desert, Indian Wells, La Quinta, Indio, Coachella, and Thousand Palms.

The CV MSHCP Monitoring Framework

In 2002, the CCB initiated a multi-year collaboration with CVAG to design and develop a monitoring plan framework. It is important to note that these monitoring framework activities were complemented and supported by concurrent CCB activities underway in western Riverside County as part of the *Inland Ecosystems of California: Resource Assessment* Project (Allen et al, 2005). The objective for both of these monitoring plans was to create multiple species monitoring approaches that focused on community dynamics while supporting conservation goals and addressed threats to species, communities, and natural processes. As such, we developed a three-phase implementation program (Table 1.1): Initial Phase – Inventories, Middle Phase – Ecological Relationships, and Final Phase – Long-term Monitoring Methodologies (Allen et al, 2005; CDFG-UC collaboration, 2003).

Table 1.1 Development of the monitoring program in Coachella Valley.

Initial Phase. (2002-2004) Species inventories and linking species to environmental variables:

1. Gathering existing data records from museums, literature, and field notes for species within the planning area.
2. Conduct field surveys to verify existing species records, particularly those for plants.
3. Gather extant environmental GIS layers for the MSHCP area.
4. Create/update/interpret vegetation map for planning area.
5. Create niche models for species using verified species records and extant GIS layers.
6. Survey predicted species locations based on niche models to find new populations.

Middle Phase. (2004-2006) Understand ecological relationships and develop a community approach:

1. Transects collecting multiple species information, community information.
2. Trends – changing/magnitudes of changes – response of species
3. Determine what to measure in monitoring.

Final Phase. (2007-2008) Establish efficient methods for monitoring:

1. Use information obtained from Initial and Middle Phases and implement across conservation area.
2. Community transects and trend detection for management.

The Initial Phase has a spatial focus. The intent is to identify what species are present and their locations, and what are the correlates of distribution. This information is then used to create the conceptual model of the community. The CCB successfully completed Steps 1, 2, 3, and initiated steps 5 and 6 of the Initial Phase (Chapters 3 and 4). These tasks include the development of baseline information and models for targeted monitored species, a means to assess habitat quality and changing habitat quality over time, and modeling sensitive of target species to threats such as exotic vegetation, off-road vehicle impacts, sand stabilization, and edge effects/fragmentation.

The purpose of the Middle Phase is to answer questions at a spatial and temporal scale, and identify important ecological relationships (Chapters 3, 4 and 5). We successfully initiated Steps 1 and 2 of the Middle Phase in 2004 and 2005, including establishment of additional transect cluster sites to improve sampling of the rainfall/temperature gradient across the Coachella Valley, and the testing of additional sampling methods (e.g., presence and absence and occupancy models). We expect to initiate Steps 1 and 2 of the Final Phase in 2007. These tasks will track the health of populations and communities, and identify current and potential threats. It is to be expected that questions will arise during the Final Phase that will require a revisit to Middle Phase investigations, but the Middle Phase tasks will diminish with time and improved understanding of the critical processes regulating community dynamics.

Conceptual Models

The information obtained about target species and communities is summarized into ecosystem conceptual models that emphasize interspecific relationships and factors that influence and regulate population change. These relationships can be envisioned now as testable hypotheses to guide the next phases of research, generate hypotheses about a species without direct observation. These models are iterative processes; as further information on system biology is added to models, hypotheses will be refined and our understanding of system dynamics will improve. For example, a spatial model such as the niche model (Table 1.2) developed by the CCB provides the best conjecture for potential habitat and species distribution, suggests correlates of species distributions, and monitors or predicts changes in species distributions in response to land-use changes or invasive species introductions

(Tennant, 2003). Chapter 2 entitled, *A Framework for Monitoring Multiple Species Conservation Plans,* is a forthcoming publication in *Journal of Wildlife Management.* It describes how these interactive models link species occurrence and abundance with environmental features and processes.

Essential to the monitoring framework is the understanding of what environmental factors drive the abundance of a species over space and time, especially with respect to isolated, fragmented habitat that lack the buffering effects of connectivity to larger populations, e.g. the Coachella Valley fringe-toed lizard (*Uma inornata*). The population dynamics study of the Coachella Valley fringe-toed lizard (Chapter 3) reinforce the need for developing models that explain natural population variance, so that departures from those models can be flagged and appropriate management actions be warranted.

In Chapter 4, we examine the processes and species occurring at the boundaries generated by suburban habitats that encroach on the natural desert, and impact components of that community.

Species Accounts

The CCB was restricted in defining the sampling frame to only those lands currently in some form of conservation ownership. Therefore, any measures of population occurrences, abundance or demographic responses were limited to those areas. Spatial modeling allows us to extrapolate occurrences beyond our sampling frame, but the validity of those extrapolations remains largely untested. Within that restriction we established sampling locations that were randomly located, but were stratified to adequately capture each of the defined community types, and to answer specific questions about the nature of potential threats to the occurrence and viability of focal species (Figure 1.1).

Chapter 4 describes species accounts found at these sites and reflect monitoring programs at various stages of development; these include Coachella Valley fringe-toed lizard (*Uma inornata*), flat-tailed horned lizard (*Phrynosoma mcallii*), Coachella Valley round-tailed ground squirrel (*Spermophilus tereticaudus chlorus*), Coachella Valley giant sand-treader cricket (*Macrobaenetes valgum*), Coachella Valley Jerusalem cricket, *Stenopelmatus cahuilaensis*), Coachella Valley Milkvetch (*Astragalus lentiginosus* var *coachellae*), LeConte's Thrasher (*Toxostoma lecontei*), riparian birds, and rare plant surveys. Where standardized sampling protocols did not exist, (i.e. for the majority of the species), we developed and evaluated the efficacy of new methods. For each of those species we have taken the next step in employing those protocols for determining the occurrence and relative abundance of those species across appropriate habitats within the plan area. Those with more advanced monitoring programs (i.e., Coachella Valley fringe-toed lizard and Flat-tailed horned lizard) include a conceptual "envirogram" which outlines hypotheses for environmental factors that drive the occurrence and abundance of that species. Our framework objectives include testing theses hypotheses.

Figure 1.1 The CCB Study Area for the CV MSHCP

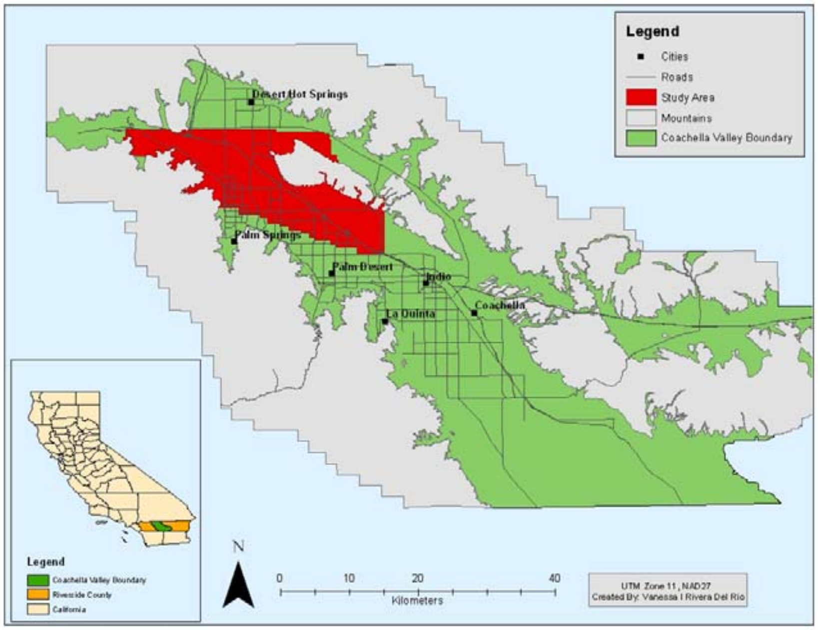
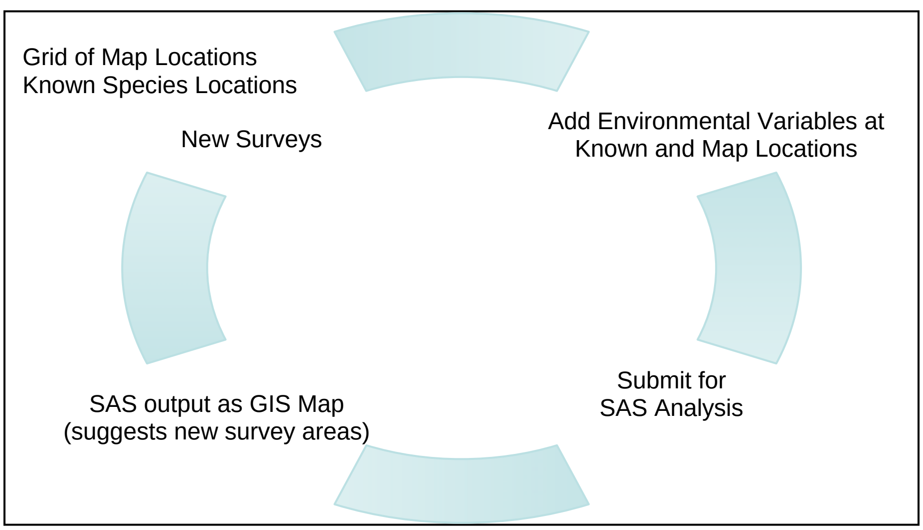


Table 1.2 Overview of niche model process (Tennant, 2003).



For those species where conceptual models are included, we are beginning to quantify driver-response relationships. As driver-response relationships are clarified, those that potentially put species at risk of local extinction are identified for inclusion into adaptive management strategies. The species accounts analyses indicate the need to initiate adaptive management for at least one species. The spatially discrete population decline associated with a habitat boundary impacting flat-tailed horned lizards is one such signal. Off-site management, most readily facilitated by regulatory agencies, focusing on reducing power line perches and anthropogenic nesting sites for predatory birds could mitigate that effect. The broader downward trajectory in the horned lizards' population requires additional research to understand drivers for that decline.

Sampling scale is a critical question in designing monitoring programs. The challenge of a monitoring framework is being able to identify and separate plot-specific idiographic responses from patch scale driver-response relationships from broader landscape scale patterns. The flat-tailed horned lizard example demonstrates how we are meeting that objective.

The explosion of Saharan mustard (*Brassica tournefortii*) in 2005 represents community level stressor that will require management if the mustard densities seen in 2005 persist. Other exotic weeds, such as *Schismus barbatus*, are additional threats that will require adaptive management treatments.

The species accounts include six species associated with the Coachella Valley's aeolian sand habitats. Data were collected for those species at survey sites (transect clusters), indicated by red dots shown on the following map. Mountains framing the valley's aeolian sand communities are shown in brown. The survey sites were confined to lands currently in conservation ownership.

Outreach

It is important to note here that the research undertaken since 2002 has contributed to the compilation of multiple year datasets (Table 1.3), and several publications (Table 1.4), including two doctoral dissertations forthcoming in 2006-2007.

Table 1.3 CCB Datasets for Coachella Valley.

Data Set	Region	Project Period	PI	co-PI	Database type
Boyd Deep Canyon Entomological Dbase	Boyd Deep Canyon Reserve	1984-current	Yanega	Redak	Web Inquiry
Boyd Deep Canyon Transects (Fisher data 1984-1990)	Boyd Deep Canyon Reserve	1984-1987	Fisher	Muth	Access
Boyd Deep Canyon Transects	Boyd Deep Canyon Reserve	1991-2004	Fisher	Muth	Access
CVAG Arthropods & Sand Transect Tracking 2004	Coachella Valley	2004	Allen	Scott, Rotenberry, Redak	Excel

Data Set	Region	Project Period	PI	co-PI	Database type
CVAG Sand Transects 2003, 2004	Coachella Valley	2003-2004	Allen	Scott, Rotenberry, Redak	Excel
CVAG Sand Transects Tracking Sheets 2003	Coachella Valley	2003	Allen	Scott, Rotenberry, Redak	Excel
CVAG 2002-2004 Riparian Bird and Thrasher data	Coachella Valley	2002-2004	Allen	Scott, Rotenberry	Excel
Historical Species Database	Riverside County	2002-2003	MF Allen	T Scott, J Rotenberry	Excel
Rare Plant Database	Riverside County	2003-2005	Allen	Scott, Rotenberry, Allen	Excel
San Jacinto Mammal Trapping 90-92	San Jacinto Wildlife Area	1990-1992	Price	Waser	None
Salton Sea data	Salton Sea	2001	Allen	Boarman	Excel
Species Monitoring DB	Riverside County	2002	Allen	Scott	Excel

Table 1.4. CCB Publications from Coachella Valley Research.

Allen, M.F., Scott, T., Tennant, T., and Wehtje, W (2002) Report to the Coachella Valley Association of Governments I. Assessment of Vegetation Map Boundaries.

Allen, M.F., Tennant, T., and Boarman, B. (2004) Soils of Salton Sea Basin, California. Impacts of Indirect Human Perturbations. UC Riverside Center for Conservation Biology.

Barrows, C.W., Hodges, W., Swartz, M., Allen, M.F, Rotenberry, J.T., Li, B, -L. Chen, X. (2005) A Framework for monitoring multiples species conservation plans. *Journal of Wildlife Management*, Oct 2005 (In Press)

Barrows, C. (2004) Temporal population sinks and sources in a threatened sand dune lizard: *Biological Conservation*, (In review).

Barrows, C. (2000) Tenebrionid Species Richness and Distribution in the Coachella Valley Sand Dunes (*Coleoptera tenebrionidae*): *The Southwestern Naturalist*, 45: 306-312.

Barrows, C. (2004) Indicator Species and Time Series Images Reveal Progress of Dune Habitat Restoration (California). *Ecological Restoration*, 22.

Chen, X., Barrows, C., and Li, B.-L. (2004-Submitted) Ecological profile of habitat characteristics of a sand dune lizard species. *Biological Conservation*.

Chen, X., Barrows, C., and Li, B.-L. (2004-Submitted) Is the Coachella Valley Fringe-toed Lizard (*Uma inornata*) on the edge of extinction at Thousand Palms Preserve in California of USA? *Southwestern Naturalist*

Time and Effort

The elaboration of the monitoring framework for the Coachella Valley Multiple Species Habitat Conservation Plan (CV MSHCP) is a product of a collaborative effort between the Center for Conservation Biology (CCB) and the Coachella Valley Association of Governments, as well as an indirect result of experiences obtained from the CCB's concurrent collaboration with the California Department of Fish and Game's Southern

California Resource Assessment Project (Allen et al., 2005). As such, the time and effort presented here for activities in the Coachella Valley overlap with similar ones undertaken on the conceptual framework and testing conducted on the Western Riverside County Multiple Species Habitat Conservation Plan. A complete description of the overlap between the two MSHCP may be found in the CCB's 2005 report, *Inland Ecosystems of California: Resource Assessment Project*, available for download at http://repositories.cdlib.org/ccb/.

As described in the above-referenced report, the formation of the monitoring framework program was based on research undertaken in the sand and riparian communities in the Coachella Valley, which later served as the foundation for the conceptual development and testing of the community-based monitoring framework described in Chapters 2 and 3. The reasoning lies in the availability of long-term population datasets for a few Covered Species under variable environmental conditions in the Coachella Valley, which facilitated the development and evaluation of the conceptual basis of the monitoring framework. The first series of niche and conceptual models were first developed and evaluated for CV MSHCP Covered Species. Also, the sampling methods later used in western Riverside County were tested first in the Coachella Valley.

Synopsis of CCB activities in Coachella Valley, 2002-2005:

- Riparian bird community surveys (2002, 2003, and 2004) including 5 covered species and 11 sites. Monitoring included point counts for the birds, pitfall trapping for arthropods, vegetation characteristics, and potential threats identification

- Sand dune community surveys (2002, 2003, 2004, and 2005). Includes 25 sampling stations with 5-8, 10 x 100 m belt transects established at each station. Data collection included surveys for Coachella Valley milkvetch, Coachella Valley giant sand-treader crickets, Coachella Valley round-tailed ground squirrels, Coachella Valley fringe-toed lizards, and flat-tailed horned lizards, as well as general reptile, arthropod, and small mammal surveys. Includes surveys in the spring-summer as well as in the fall to track reproductive success for focal species. Vegetation and sand compaction data were collected for each transect each year

- Population dynamics models for flat-tail and fringe-toed lizards to generate spatial and temporal dynamic basis of ecosystem

- Le Conte's thrasher surveys (2002, 2003, 2004, and 2005) development of survey protocol and monitoring on 11 transects

- Rare plant inventories for 5 covered species (2002, 2003, 2004 and 2005)

- Desert tortoise surveys (2004)

- Niche models developed for 4 species

- Framework for monitoring multiple species conservation programs manuscript accepted for publication in the October 2005 Journal for Wildlife Management

Table 1.5. Time and effort for the development and implementation of the CV MHSCP monitoring program.

Task	Fiscal Year 1/May/02 30/Jun/03	1/Jul/03 30/Jun/04	1/Jul/04 31/Apr/05	Total
Project Development and Management				
Faculty Oversight/Review	1,252	1,040	698	3,090
Task Management	1,896	2,000	1,680	5,678
Administrative Support	1,456	1,983	755	4,194
Development/Management Subtotal	***4,604***	***5,023***	***3,133***	***12,962***
Fieldwork				
Rare Plant Surveys	400	800	800	2000
Coachella Valley Sand Dune and Riparian Community Monitoring	4,136	11,629	5,644	21,409
Fieldwork Subtotal	***4,536***	***12,429***	***6,444***	***23,409***
Data Entry/Management				
Arthropod Identification and Data Entry		1,258	725	1,983
Coachella Valley Sand Dune and Riparian Community Data Entry	1,109	1,694	1,309	4,112
Archival Data Entry – Plants and Insects	1,184	891	346	2,421
Data Entry/Management Subtotal	***2,293***	***3,843***	***2,380***	***8,516***
Species Database, GIS, and Niche Modeling				
Faculty Oversight	212	280	150	642
Task Completion	1,989	2,323	2,323	6,635
Species Database/Niche Modeling Subtotal	***2,201***	***2,603***	***2,473***	***7,277***
Data Analysis/Report Preparation				
Faculty Oversight	283	285	73	641
Task Completion	786	2,136	2,166	5,088
Data Analysis/Report Preparation Subtotal	***1069***	***2,421***	***2,239***	***5,729***
Project Total	**14,703**	**26,319**	**16,669**	**57,691**

CHAPTER 2

A FRAMEWORK FOR MONITORING MULTIPLE SPECIES CONSERVATION PLANS

Forthcoming in Journal of Wildlife Management (2005)

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ABSTRACT - The shift from single species conservation initiatives to multiple species conservation plans has not been accompanied by parallel changes in methods to evaluate the success of these efforts, nor to provide managers critical information to employ adaptive management strategies. Layering single species approaches for monitoring multiple species conservation plans is inefficient and may lead to management strategies that have unintended detrimental impacts on target and non-target organisms. Alternative approaches, such as ecosystem monitoring, can also fail to provide adequate protection for listed species and so may not fulfill regulatory requirements. We propose a hybrid approach that employs conceptual and spatial data in an iterative process to create niche models for species and species associations within natural communities. Niche models are composed of testable hypotheses linking species occurrences to environmental parameters over multiple scales. Over the course of an initial data gathering period these hypotheses are evaluated, accepted or rejected, and modified as indicated by new data. Once niche models are corroborated, the focus of monitoring shifts to a greater emphasis on identified anthropogenic and natural environmental drivers of species occurrence and abundance. The focus on environmental drivers supplies managers with direct information as to how, when and where to employ adaptive management strategies when natural variance in those drivers is compromised by anthropogenic stressors. We provide a specific example on the conceptualization, development, and implementation of our hybrid approach from a new Multiple Species Habitat Conservation Plan for the Coachella Valley, in the Colorado Desert of Southern California.

Habitat Conservation Plans (HCPs) were authorized by amendments (Section 10(a)) to the United States federal Endangered Species Act (ESA) in 1982, as a means of compensating for the growing authority of the ESA over private use of land and natural resources (Scott and Sullivan 2000). Section 10(a) allows incidental take (*sensu* Bean et al. 1991) of federally listed species if essential habitat and populations of the species are protected within HCP boundaries. Although the ESA includes language to protect entire ecosystems used by listed species, its implementation has focused on protection of individuals and populations of single species. Unfortunately, the numbers of species at risk of extinction have ballooned beyond most original expectations. For example, in the six county area of southern California there are 102 state or federally listed threatened and endangered species. In response to this proliferation of listed and at-risk species, many HCPs have evolved into plans that include multiple species. Multiple Species Habitat Conservation Plans (MSHCPs) are now the primary implementation tool for regional conservation. They provide ESA Section 10(a) permits for ESA listed and potentially-listed species, perhaps forestalling the need for ESA listing for the latter (Nelson 1999). The assurance of long-term persistence that MSHCP permits require is contingent the development and implementation of management and monitoring protocols that address each species covered under their MSHCP. Today some MSHCPs include more than 140 covered species (e.g. western Riverside County, Dudek & Associates 2003). Unfortunately, the transformation of single-species HCPs into MSHCPs has not been accompanied by a concurrent methodological transformation of modeling or monitoring approaches at required regional scales.

We propose a framework for of uniting single species and ecosystem approaches to address the monitoring needs of multiple species conservation programs. Our approach employs primary data collection to build conceptual and quantitative models, habitat condition assessments, and species-specific surveys that will allow linking species population trajectories with community or ecosystem processes and conditions. Our objectives are to provide compliance within an ESA regulatory framework and to provide a context for data to be evaluated. Management and monitoring will incorporate hypotheses testing to enhance rigor, effectiveness, and utility for conservation of target species and natural communities. This approach is an iterative process of collecting data, developing a model/hypothesis, and evaluating the model/hypothesis statistically by partitioning large-scale models into discrete units that break down complexity. We illustrate the development of this approach with an example using an MSHCP under development in the Coachella Valley of Southern California.

Ultimate goals of biological monitoring programs include evaluating the efficacy of the design and configuration of protected areas, which guide future conservation planning efforts, and providing relevant data to feed into an adaptive management regime (Holling 1978, Walters 1986). This latter goal is the mechanism by which threats to protected species and their habitats are identified and the success of management efforts evaluated. It has the most direct impact on the species and habitats being monitored. There is a range of interpretations as to what adaptive management entails. Here we use a definition in which adaptive management is a science based, data driven process aimed at meeting the need for continuous learning to respond to uncertainty associated with dynamic ecosystems (Busch and Trexler 2003). An overarching goal of adaptive management is to maintain optimally functioning ecosystems, with all their components (Noss and Cooperrider 1994). To meet that goal we need to begin to understand the natural dynamics that characterize populations, communities, and the resources on which they depend (Landres et al. 1999). Hypotheses as to process and community level linkages that characterize sustainable populations, as well as proximate and ultimate stressors that may compromise those processes, and populations need to be identified. When monitoring efforts determine those stressors are evident, management by experiment is employed to test various means postulated to reduce or eliminate the stressor's impact. These management experiments are coupled with focused monitoring to evaluate the experiment's success (Morrison et al. 2001). Once an effective management strategy is identified, it can then be applied as needed. Many monitoring programs fail to provide the necessary input to meet one or more of the criteria for implementing adaptive management identified by others and as we have outlined it here. Our proposed monitoring framework attempts to fill this information gap.

Shortcomings of Current Monitoring Approaches

Under an HCP rubric, single-species monitoring can be deceptively simple: (1) establish protection goals, (2) track populations, and (3) determine whether goals have been met. Effective single-species approaches may also include monitoring various habitat parameters and assessing whether changes in those parameters drive changes in target species populations. A key step here is the establishment of protection goals as thresholds for initiating remedial management (Yoccoz et al. 2001). Should goals be determined using first year population estimates as a baseline? Should they be based on some long-term regional mean population level or should they be based on a theoretical optimal or maximum? There is no theory on which to base the correct approach. In each case above the protection goals are static or fixed quantities, and natural populations are not static.

An additional problem faced by single species monitoring approaches involves the power of statistical analyses to detect statistically significant population changes in rare species (Funk et al. 2003). Green and Young (1993) provide guidance regarding sampling efforts required to detect a rare species at a given density. The challenge here is developing an *a priori* expected density. Even when population change detection is possible, the effect of stressors on populations of covered species may lag behind the appearance of agents causing that change (e.g., Knick and Rotenberry 2000).

Shortcomings of single-species monitoring approaches quickly become more apparent when applied to MSHCPs. If monitoring plans for many single species are simply layered on top of each other, complex, community-level patterns and processes may be lost (Fischer et al. 2004). When several species covered under an MSHCP are distributed along an environmental gradient, natural dynamics or anthropogenic effects can benefit one or more covered species at the expense of others. Determining management priorities under this realistic scenario is hardly trivial and may require choosing which covered species to manage for and which to manage against under a species-focused monitoring and management paradigm (Vogel and Hicks 2000, Roemer and Wayne 2003). Nevertheless, MSHCPs do require species specific monitoring, particularly federally listed threatened or endangered species. In these cases community and ecosystem parameters are often ignored.

Alternatives to species-focused monitoring include: (1) ecosystem monitoring (Busch and Trexler 2003), (2) threats monitoring (Salafsky and Margoulis 1999), (3) indices of biotic integrity (IBI) (Trebitz et al. 2003), and (4) ecological integrity monitoring (Salwasser 1991, Parrish et al. 2003). Each approach focuses on processes, and determining natural variances of those processes (Landres et al. 1999). To a greater or lesser extent these approaches assume that within some acceptable range of environmental conditions, species will maintain viable populations. There are shortcomings to an ecosystem monitoring approach as well. One misuse of this approach is the assumption of constant relationships between species, natural communities, and larger scale population drivers and processes (Landres et al. 1999). However, these relationships may change with short term and long term climatic fluctuations, population density, and the presence of new stressors.

Ecosystem approaches to monitoring have been slow to be embraced in MSHCP planning as a result of regulatory limitations. The ESA is explicitly focused on each species and its habitat, not communities, large scale processes, or threats. A strict ecosystem approach fails to provide species specific data, and so assumptions that species will maintain sustainable populations are often untested. Trebitz et al. (2003) evaluated the utility of an IBI approach for stream fish assemblages, and found that method sufficient for detecting large departures from baselines, but not for identifying more gradual changes that could lead to large departures in the future. Without empirical linkages between focal species, environmental features, and processes, the risk of local extinction may compound over time. Even when linkages between ecosystem processes are made, rare species are often excluded due to low encounter rates (Queheillalt et al. 2002), and it is rare species that conservation plans are often designed to protect. Ecosystem monitoring programs by themselves fail to meet the species nexus required in the ESA and so conservation strategies such as MSHCPs that strive to meet those regulatory requirements still require species-centered monitoring.

Our proposed hybrid monitoring framework consists of multiple steps or phases, each providing an information foundation upon which to build on subsequent phases. The phases are layers of information that provide an increasing understanding of how targeted species fit into larger communities, how they are affected by ecosystem processes, and how perturbations to those processes impact population trajectories.

PHASE 1: FRAMEWORK CONSTRUCTION

Compiling Existing Data

Collecting and organizing information known about a landscape of interest is an important initial step when designing a multiple species monitoring program. Data about a region and its organisms can be gathered from a wide variety of sources at different scales, ranging from museum records and field notes to geographic information system (GIS) databases. Data are available for practically any location on earth, but their quality, scale, and value vary from collection to collection and site to site. Types of data that are important include, but are not limited to, species occurrence and community distribution data, multi-spectral satellite imagery, artificial night sky brightness imagery, weather data (e.g. precipitation, temperature, and growing season), aerial photos, historic photos, land-use maps, and land ownership data. While collecting initial information, the source, scale, precision, and accuracy of each datum (i.e. metadata) must be obtained. Large-scale data, such as satellite imagery, are important because they provide landscape-level context beyond local-level attributes. Strengths and weaknesses in existing knowledge of a given community or system are found during data mining activities. Existing data can be used in initial model building exercises, which often identify critical information gaps. The first phase implementation of a monitoring plan may simply focus on collecting missing information.

Conceptual Models

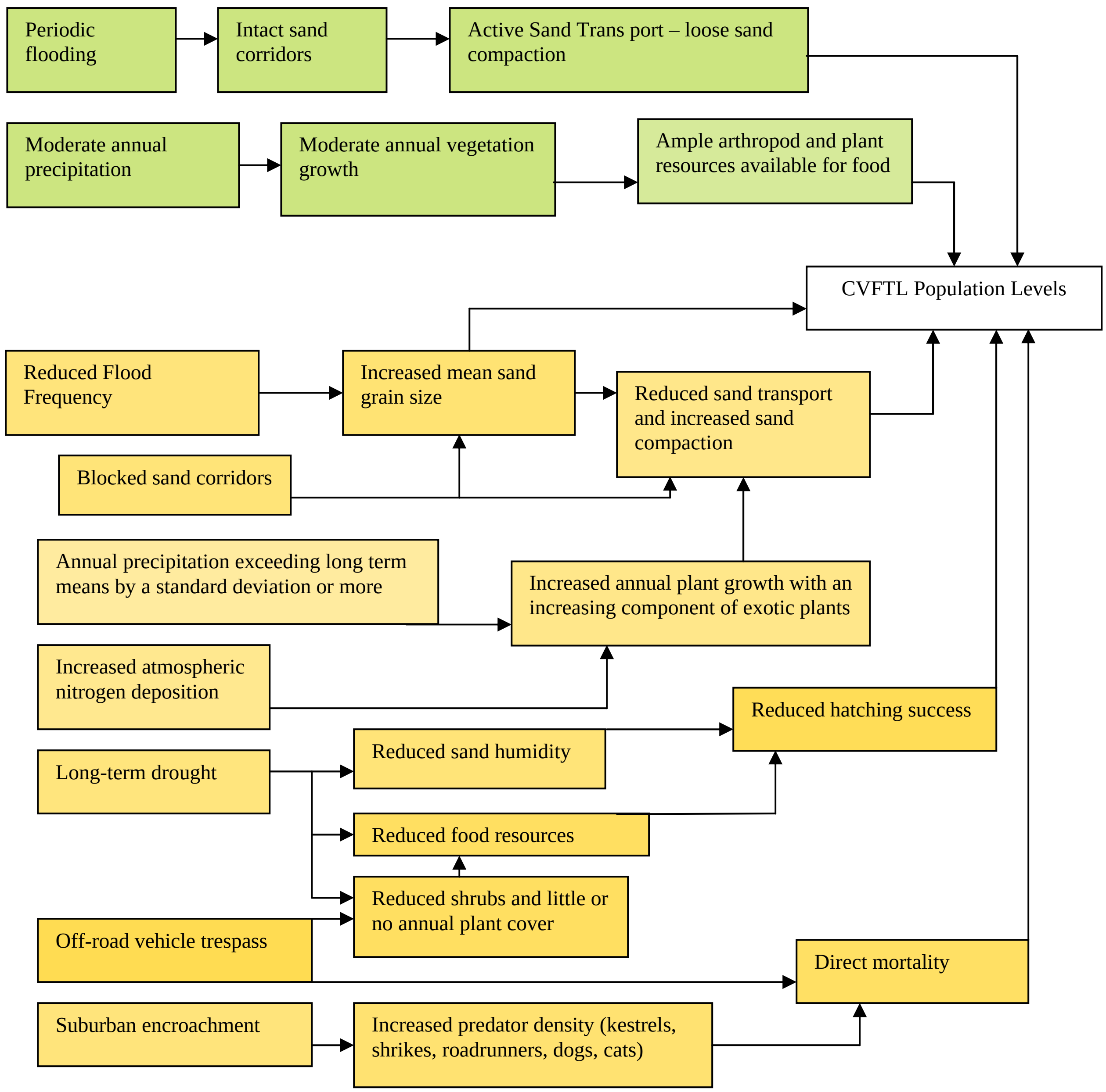
After amassing available data for species and communities of interest, conceptual interaction models can be developed, linking species occurrence and abundance with environmental features and processes. In each case the species are the focus for understanding relevant community and ecosystem properties (MacMahon et al. 1981). Preferably those properties have been empirically determined, or can be framed as testable hypotheses. Factors simultaneously affecting multiple focal species can then be overlain creating a multiple species conceptual model. Conceptual models are essential tools that facilitate forward progress in linking species of concern with relevant ecological parameters. These also form a basis for identifying adaptive management actions related to environmental perturbations (Woodward et al. 1999).

We begin the development of conceptual models with the Malthusian "first principle" of population ecology: populations increase exponentially in an unlimited environment (Turchin 2003). The drivers of population change, whether exponential, declining to extinction or something in between, are the relative contributions of birth (B) and immigration (I) versus deaths (D) and emigration (E) (i.e. a BIDE model; Cohen 1969, 1971). It is the influence of environmental factors on these population parameters that form the structure of the conceptual models we propose here. Developing conceptual models allows population change to be analyzed within a theoretical as well as empirical framework, giving context to monitoring data. In so doing, model-based monitoring programs are heuristic and so avoid the ecological banality that has otherwise characterized population monitoring (Krebs 1991).

Ecological Process and Community Relationship Models.--These models take many forms, such as narrative descriptions, energy-nutrient flow diagrams, and box and arrow designs. An example using envirograms (Andrewartha and Birch 1984) for the Coachella Valley fringe-toed lizard (*Uma inornata*) is shown in Figure 2.1. Envirograms make a dichotomy between those factors that drive or limit population growth. They separate proximate and ultimate drivers of population change, thereby creating a linkage to adaptive management options if a need to change the trajectory of the population is identified. Management can then address both the symptom of a stressor, but when possible, its causes as well. When applied to community-level relationships, the linkages become complex but are nonetheless recognizable for providing testable community-level context for focal species. Figure 2.2 is essentially a series of envirograms for seven aeolian sand community focal species that have been compressed into one diagram. The advantage of this type of diagram is that it identifies where focal species are likely to respond in a parallel or orthogonal manner to changes in particular resource levels. The directions of these responses represent hypotheses that can then be tested. Once corroborated, the responses depicted in a community-level diagram can inform managers as to when a management action aimed at enhancing the population of one species may have a negative effect on another species.

Conceptual models are modified as new data indicate alternative or additional hypotheses. These models also help identify important information needs and help establish a research agenda to fill knowledge gaps. For example, the effect of edges and fragmentation on fringe-toed lizards (Fig.2.1) or any focal aeolian sand species (Fig. 2.2) can be surmised based on theory or anecdotal observation; however, empirical data rarely have been analyzed in a

Figure 2.1 Envirogram for the Coachella Valley fringe-toed lizard**.** Ultimate drivers of stressors are depicted to the left of the diagram whereas more proximate impacts on population levels are shown to the right of the diagram. Drivers of positive population growth are in the top half of the diagram and stressors that result in negative population growth are in the lower half.



manner capable of assessing the strength of that effect with respect to species, edge type or fragment size. Adaptive management efforts to mitigate edge and fragmentation effects would likely be costly and so determining the degree and means by which stressors might impact focal species is a critical first step.

The adaptive management process, along with specific sampling designs, provide an opportunity to evaluate and quantify the strength of identified linkages between biotic and abiotic ecosystem features and processes that may be used as surrogates for single species population monitoring. By focusing on processes and stressors, management actions can be triggered to address anthropogenic perturbations before population changes become statistically or biologically significant. Thus problems with effects lagging behind causes inherent in single species monitoring approaches may be avoided.

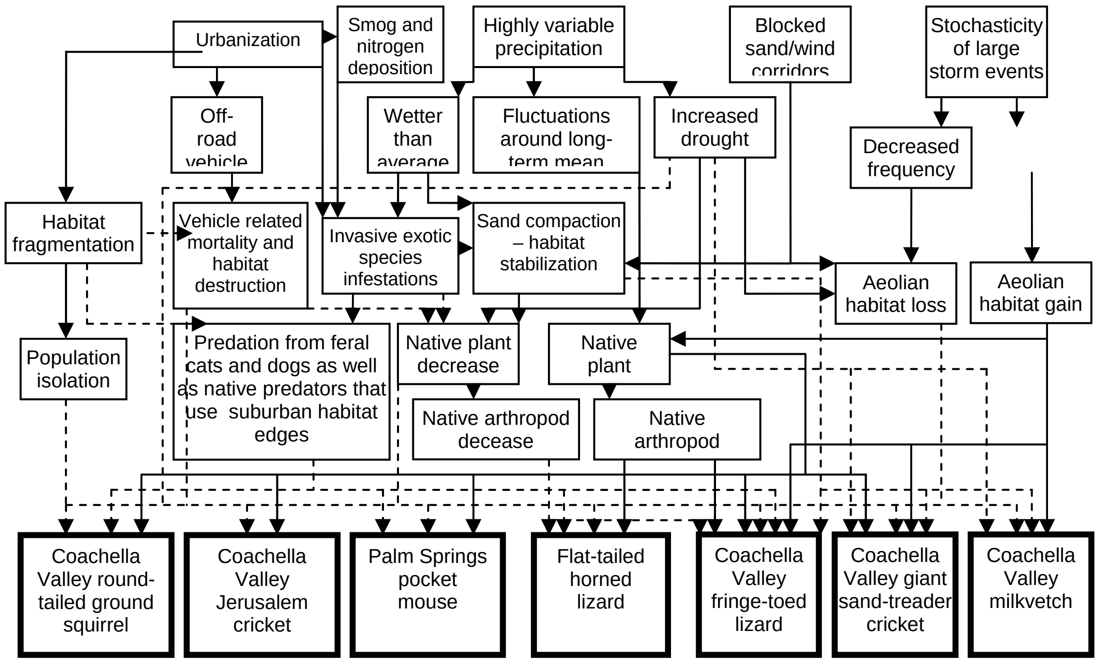
Species data are generally collected at a point, transect, or plot. If monitoring is limited to this structure, then environmental drivers of population change may be missed. Larger scale natural processes that affect population dynamics, such as flooding, erosion, aeolian sand movement (Barrows 1996), fire frequency (Minnich 2001), exotic species invasions (Stylinski and Allen 1999, Atkinson 1989, Seabloom et al. 2003), anthropogenic nitrogen deposition (Fenn et al. 2003), habitat connectivity, patch size and age distribution (Saunders et al. 1991, Martin and McComb 2003), and predator guild structure (Crooks and Soulé 1999) constitute a short list of environmental parameters that drive population trajectories, but are rarely adequately quantified when sampling is limited to a few species-focused points or plots. These drivers often originate far from occupied habitat (i.e., sand inputs to aeolian systems; Barrows 1996), and by the time their impacts are evident at the species level they may be impossible to mitigate. Both conceptual and spatial models should identify hypotheses as towhat processes are relevant to the species and community in question, and should guide appropriate sampling to quantify the effect of these larger scale processes.

Creating linkages between species occurrence and abundance with environmental parameters will be scale dependent. An *a priori* understanding of the scale at which changes in natural processes and habitat metrics begin to impact communities and their component species is often elusive (Fischer et al. 2004). Constructing conceptual models often provides direction as to proper scaling (Figures 2.1 and 2.2), but sometimes iterative sampling may be required to identify which scale provides the best predictive model. We use a sampling framework that allows analyses at multiple scales. In this structure sampling points, transects, or plots are analyzed as independent data, or clustered to be analyzed by natural community type, by protected area, and by region.

The structure of the populations being sampled is another important aspect of sampling. Many populations are in fact subsets of larger interconnected populations, i.e. metapopulations (Wilson 1992, Hanski and Gilpin 1997). Metapopulation theory has been embraced by many conservation biologists as a framework for protecting and managing fragmented habitats even if the populations don't always precisely fit the classic metapopulation structure (Wiens 1996). The fragmentation that often precipitates the development of habitat conservation plans can isolate sub-populations when they lack the ability to immigrate across anthropogenic landscape elements. Fragmented populations, whose viability naturally depends on connectivity to other larger populations through immigration and emigration, may require more intensive monitoring and management if they

Figure 2.2 Aeolian sand community conceptual model. Ultimate drivers are distributed along the top tier boxes, with more proximate drivers in the middle tiers. Representative aeolian sand community species included here are those for which coverage is being sought under the Coachella

Valley MSHCP. Arrows with solid lines leading from proximate drivers to species boxes are those facilitating positive population growth. Arrows with dashed lines indicate factors causing negative growth. Scientific names for species are in the text.



are to persist. Metapopulations may be monitored by determining presence or absence throughout a matrix of habitat patches. Analytic tools are available to evaluate population viability of metapopulations using occupancy data (Noon and McKelvey 1996). The degree to which a metapopulation model is applicable to a given species should be addressed in the conceptual modeling stage of the monitoring framework.

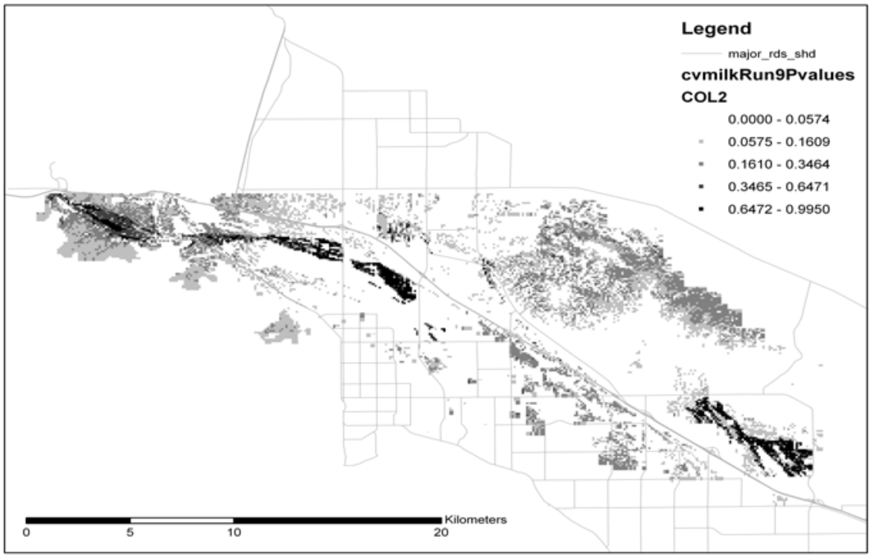
Spatial Models

Spatially-explicit conceptual models generate hypotheses about environmental correlates that dictate species occupancy within a habitat matrix. Such hypotheses can then be applied in a predictive manner to a landscape over the entire region of interest. Geographical information system (GIS) analysis tools are essential components to this process. The dependent variable for most models will be a GIS-layer containing spatial coordinates of each target species observation, such as point data extracted from historical field notes, regional natural heritage program databases, and current monitoring data. Independent variables may include spatial configurations of soils or vegetation types within a specified distance of the species locality (e.g., vegetation types, interdigitation of different vegetation types, amount of edge or ecotone between different vegetation types), distance measures (e.g., distance from a locality to the nearest attribute *Z*, where *Z* might be a road, an urban boundary, a particular vegetation type), or any other spatially explicit attribute suggested by the conceptual model. Interpretation of high-resolution satellite images yields a wealth of attributes that are important indicators of species occurrence at local scales which can be extrapolated and tested at larger scales. In addition to using variables positively correlated with the presence of a species, we also suggest using negatively correlated variables, especially those that are related to previously identified threats to the target species or community. Variables that quantify or capture variation associated with attributes that reserve managers can manipulate will have the highest potential for making these models applicable to adaptive management strategies. There are numerous methods available to create spatially-explicit models; Scott et al. (2002) contains extensive discussions of the pros and cons of many of them. Because many of the sources of species distributional information, especially historical and/or museum records, consist of presence-only data, we suggest Mahalanobis D^2 (Clark et al. 1993, Knick and Dyer 1997) and its partitions (Rotenberry et al. 2002), or Genetic Algorithm for Rule-set Prediction (GARP) (Stockwell and Peters 1999, Raxworthy et al. 2004). These techniques are robust to potential bias in presence-only data, and are little affected by the inclusion of irrelevant independent variables in the modeling process (Stockwell and Peterson 2002, Rotenberry et al. 2002).

The initial product of spatially-explicit conceptual modeling is a map consisting of points or polygons within a study area quantitatively identified by their degree of similarity to those known to be or to have been occupied by a species. An example is a spatial model for the Coachella Valley milkvetch (*Astragalus lentiginosus* var. *coachellae*), an aeolian sand species which commonly co-occurs with Coachella Valley fringe-toed lizards (Figure 2.3).

The map thus represents an index of "habitat suitability" or "potential occupancy" of each point or polygon for a given target species. As mentioned above, these models are intentionally iterative. The initial model can be used as a baseline for comparing against data gathered in subsequent years. With each iteration, new data will refine the accuracy of the model, providing greater confidence that the model represents those locations that are or could be occupied by the covered species. The variables that are inserted into the model are those that are hypothesized to influence the occurrence of a given species. The relative importance of those variables for defining the occurrence or potential occurrence of a species can be determined statistically or by the iterative adding and subtracting of variables in the modeling process. Assessing the influence of variables in a spatially-explicit model provides insights as to which environmental features should be the focus of monitoring efforts. The model also defines a sampling frame for corroborating model accuracy and for determining species occupancy within modeled areas. By viewing habitat and occupancy at a landscape scale, patterns will likely emerge that will improve our understanding of the niches occupied by the target species as well as identifying patterns in the occurrence and effect of potential threats.

Figure 2.3 A spatial model for the Coachella Valley milkvetch (*Astragalus lentiginosus* var *coachellae*) created using the Mahalanobis D^2 statistic. Variables included elevation, reflectance values from multi-spectral satellite imagery, soil types, and plant community variables. Darker colors indicate high similarity to a multivariate mean value from known localities and thus are predictive of potentially suitable milkvetch habitat.



PHASE 2: FRAMEWORK EVALUATION

For the Coachella Valley MSHCP, and arguably for any similar conservation plan, we have proposed at least a five year period when conceptual and spatial models are evaluated and refined. The iterative character of these models correctly implies that this process is never complete, even when empirical data corroborate hypothesized relationships. Nevertheless, there should be a finite, initial period when the strength of modeled relationships is evaluated. Following this initial period, these models will require periodic checks to insure that the postulated drivers of species occurrence and abundance are still operative. This iterative evaluation of hypotheses is integral in our adaptive management approach, and provides a continuing linkage between species occurrences and larger scale parameters. The initial evaluation period should extend through a temporal sequence of environmental variance typical of the target landscape. Drought, flood, and fire cycles are examples of temporal variance that needs to be incorporated into this initial evaluation period. The occurrence of events such as floods and fires may be on the order of decades to centuries. In these instances space-for-time substitutions can be incorporated (Drury and Nisbet 1973, Strayer et al. 1986, Pickett 1989), where a matrix of temporal dynamics is sampled spatially in lieu of waiting for those processes to occur on any given parcel. Inferences can be made from associations in sites with differing time intervals since disturbance and then tested with the reoccurrence of those processes in the future.

Each hypothesized relationship set forth in the conceptual models should be evaluated to determine its ability to predict the occurrence of a species or species association. This can be accomplished by designing sampling strategies that explicitly test these relationships. If habitat fragmentation and edge effects are suspected to be limiting to a species' population size, then sampling should occur with respect to those edges and at varying distances from edges. If qualitative or quantitative differences in habitat conditions are discerned, then a sampling strategy should exploit this knowledge, making sure an adequate number of samples are randomly located across the gradient of occupied habitats to estimate a species' use of and discrimination of differences along that gradient. A purely random sampling array can, by chance, under-sample or over-sample particular habitats. Our approach attempts to maximize information gain about how species distributions respond to local and landscape patterns. In many cases, rather than implement an exhaustive effort focusing on estimating total population size, we suggest an analytical focus on the dynamic response of species and natural communities to temporal and spatial habitat parameters. These data feed naturally into adaptive management strategies, and so directly support efforts to conserve those species and communities.

In a traditional monitoring scheme, a decision to initiate adaptive management action within a single species monitoring paradigm occurs if $N_i < N_0$ (N_i is population abundance in year i, N_0 is a baseline and/or threshold population abundance). There is often a focus on quantifying N_i to the highest resolution and over the largest sampling frame practical in order to identify statistical departures from N_0. For many reasons this approach is inadequate. The primary problems are that assessing N_i relative to N_0 over a relatively short sampling period fails to address the naturally dynamic character of populations and no hypotheses are being evaluated. Needless, expensive, and potentially harmful management actions could be undertaken in order to fix a population that is not broken.

An Example: The Coachella Valley Aeolian Sand Community

The shortcomings of a traditional monitoring approach described above can be illustrated from the following real situation. Approved in 1986, a single species HCP for the Coachella Valley fringe-toed lizard (Bean et al. 1991) preceded the development of the Coachella Valley MSHCP, now nearing completion (Coachella Valley Association of Governments, 2004). Under the original plan the lizards have been monitored annually.

From 1988 through 1990 there was a steady decline in the lizard population (Figure 2.4A). The U. S. Fish and Wildlife Service interpreted this decline as an indication that the HCP was flawed, and began discussions toward revoking the 10(a) permit. However, in 1991 annual rainfall returned to near average levels after two years of drought; the lizard

Figures 2.4A-C. The relative abundance of Coachella Valley fringe-toed lizards through time frames relevant to HCP monitoring and evaluation. When viewed as time fragments rather than the full time sequence, different conclusions may result. The first six years of monitoring data are shown in Figure 2.4A. The most recent 13 years of data are shown in Figure 2.4B. The full time sequence (along with precipitation levels) is shown in Figure 2.4C. A one-year lag time between the precipitation stimulus and the effect on the adult lizard population size is demonstrated. Index of lizard abundance is a measure of relative abundance, using mean number of lizards seen per six repeated visits on a transect, per year.

Figure 2.4A

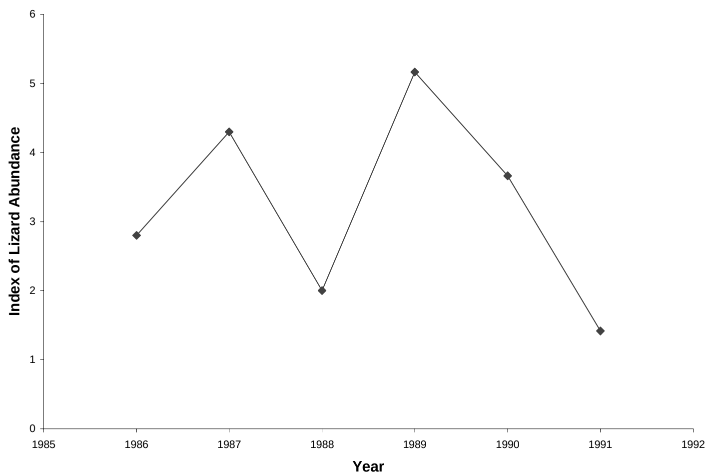


Figure 2.4B

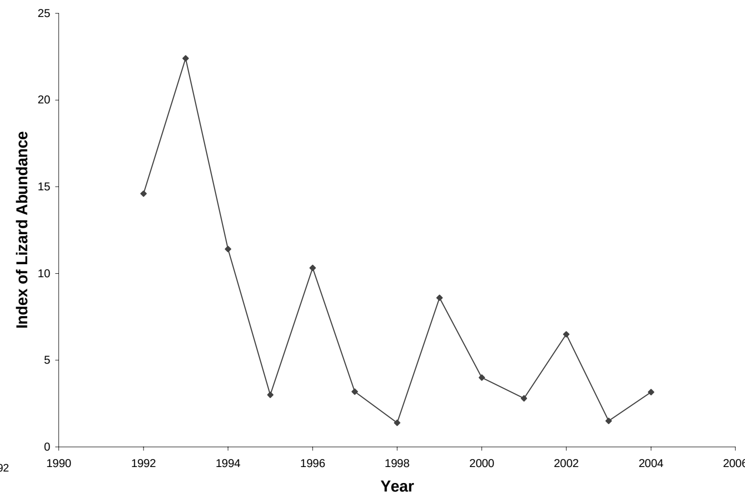
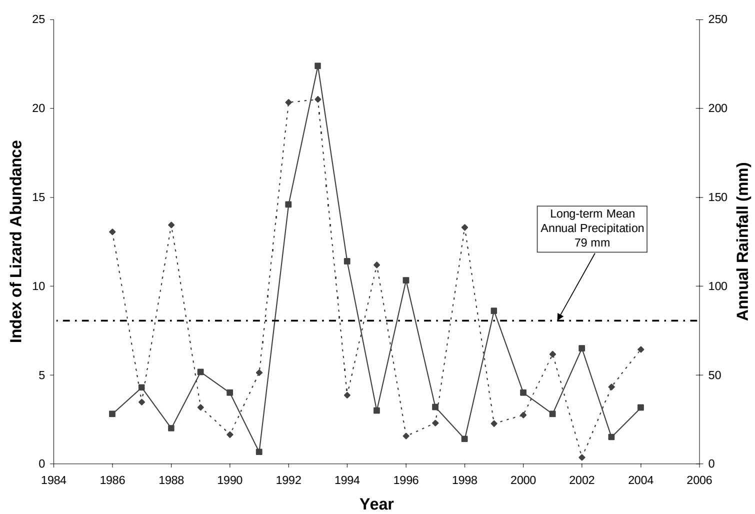


Figure 2.4C



population responded with high levels of reproductive success, and in 1992 their population had reached the highest levels recorded. In retrospect, despite low lizard numbers in 1989 and 1990, the population was not at risk. The lizard population decline from 1992 to 2004 was even more dramatic (Figure 2.4B). If monitoring had been initiated in 1992 so that there was no perspective gained from having observed the population levels of the late 1980s, it would have been easy to assume that this population was on a trajectory toward extinction. However, if the data are viewed within a context of primary population drivers identified in a conceptual model such as the envirogram in Figure 2.1, in this case annual rainfall, even relatively long term population declines may be viewed as part of a natural dynamic, rather than a symptom of a species at risk of extinction (Figure 2.4C). Focusing on short-term $N_i < N_0$ determinations in isolation, no matter how precise, will not provide guidance to managers as to what, when, or if remedial actions are warranted.

Our proposal is to develop conceptual, spatial, and numeric models that describe key drivers of population change in a species, in order to provide a species-centric nexus to natural community and landscape metrics. Such an approach might be depicted as $N_i = f(a, b, c,\ldots)$ where each parameter (a, b, c,…) represents environmental factors that impact the abundance of species N in year i. Conceptual models were used to define the parameters and identify the linkage between individual species and habitat and/or landscape metrics. Spatial models quantify the parameter's importance in defining the occurrence of a species. Each of these models constitutes a series of testable hypotheses regarding the linkage between a species and its habitat. Collectively these models constitute a niche model which defines the boundaries, limits and drivers of population growth for a species. Each habitat parameter includes year to year variance, which in turn may impact population dynamics. When that variance is modified by anthropogenic influences to the extent that the species occurrence and abundance are negatively impacted, then a threshold for initiating management action has been reached. The focus here is on tracking environmental parameters and detecting changes in population trajectories and community dynamics. A high resolution quantification of N_i is less important within this monitoring framework.

Population change may result from multiple causal factors. Here again conceptual models help identify potential multiple population drivers and stressors. The strength of any one driver may change and be dependent on the co-occurrence of other drivers in space and time. Returning to the fringe-toed lizard envirogram (Figure 2.1), this approach results in a model where: Lizard Abundance = f(annual rainfall, food abundance, sand compaction, perennial vegetation cover, habitat fragmentation, time since most recent sand deposition event). Each environmental variable initially represents hypothesized relationships that span multiple spatial and temporal scales, and that can be evaluated and quantified, creating a niche model. Assuming the linkage between environmental parameters and lizard numbers can be established, an effective monitoring program would focus on anthropogenic constraints to the natural dynamics of those parameters (Landres et al. 1999). These data then feed directly into adaptive management strategies. Since those same parameters impact other species, the monitoring program becomes increasingly community based rather than focused on single species. Future measures of N_i would be designed to insure that the relationships between the species and independent variables remains coupled, not simply whether N_i increases or decreases at specific points in time.

The fringe-toed lizard is a component of an aeolian sand community, including many species that are restricted to the Coachella Valley. Members of this community that are also covered under the Coachella Valley MSHCP include Coachella Valley milkvetch, flat-tailed horned lizard, (*Phrynosoma mcallii*), Coachella Valley giant sand-treader cricket (*Macrobaenetes valgum*), Coachella Valley Jerusalem cricket (*Stenopelmatus cahuilaensis*), Palm Springs round-tailed ground squirrel (*Spermophilus tereticaudus chlorus*), and Palm Springs pocket mouse (*Perognathus longimembris bangsi*). Conceptual models for each of these species have overlapping parameters (Figure 2.2), but the effect of each driver differs between species (Figs. 3.5A and 3.5B). Each of these species is evaluated as part of a more comprehensive community monitoring approach that also includes general arthropod surveys (prey base for the lizards and indicators of community species richness), annual and perennial vegetation surveys (including non-native plants), and an analysis of sand compaction. The result is a database that spans trophic relationships, tracks potential stressors, allows analysis of inter-specific patterns in abundance and distribution, and evaluates the effect of the various drivers on the abundance of those covered species.

In 2003 annual precipitation was approximately 40% below the long-term mean (Western Regional Climate Center, Indio reporting station). Milkvetch and sand-treader crickets were nearly undetectable in the driest, eastern portions of their range, but were more common in the wetter, western portions. In 2004 annual precipitation was 5% below the long-term mean. Eastern populations of both species had a 70-90% increase in their numbers; however, westernmost populations had a 90% decrease. Paralleling milkvetch and sand-treader cricket population decreases was an increase in invasive exotic annual plants on western sites.

By contrast, Jerusalem crickets occurred in their greatest abundance on those same western sites. Several important implications can be drawn from these data:

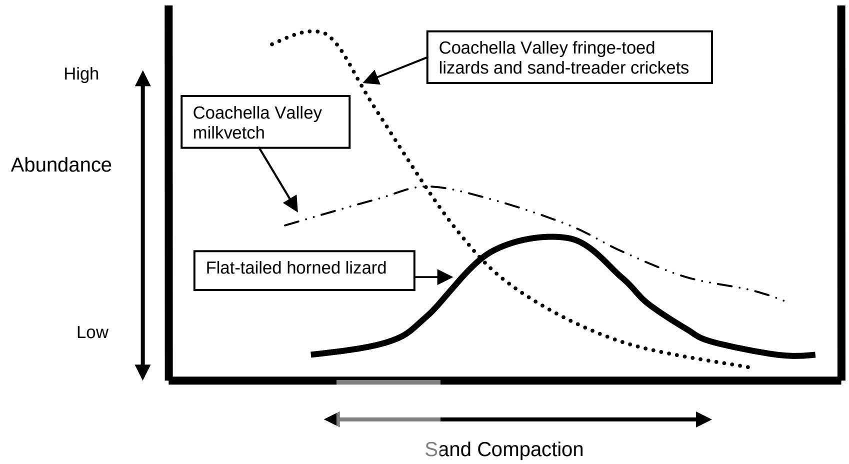
1. The data are consistent with the conceptual niche model (Figure 2.2) and provide a start for quantifying that model.

2. Although data indicate the sand-treader cricket and milkvetch were nearly absent in eastern portions of their range in 2003, these data were not viewed as an indication of species at risk, but as consistent with a natural response to drought as predicted in the conceptual model.

3. A precise measure of N_i was less important than a measure of the direction of change in the populations

Figures 2.5 A-B Distribution of aeolian sand community members along environmental gradients. Figure 2.5A depicts the relationship of four of the species to sand compaction, indicating non-linear but predictable patterns of occurrence. Figure 2.5B depicts the relationship of four of the species to increasing levels of exotic annual vegetation.

Figure 2.5A



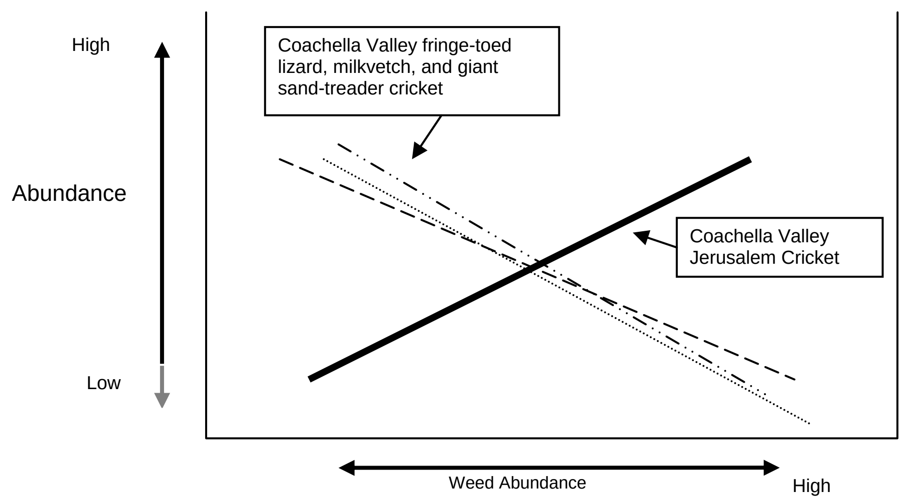


Figure 2.5B

4. By collecting data on environmental variables that are hypothesized drivers of a population trajectory, managers now have targets as to where to direct adaptive management experiments (i.e., reduce exotic plant cover on the westernmost sites to protect milkvetch, sand-treader cricket, and fringe-toed lizard populations).

5. The abundance of Jerusalem crickets on the same sites where population declines were measured for other covered species emphasizes a need to implement exotic vegetation control measures that are benign to the Jerusalem cricket population.

Without this kind of community approach a zealous effort to remove exotic vegetation could have resulted in the undesirable effect of damaging the core Jerusalem cricket population.

MANAGEMENT IMPLICATIONS

Single-species approaches are inefficient for monitoring and evaluating the success of multiple species habitat conservation plans. Managers and scientists have been calling for a paradigm shift away from single-species monitoring and toward ecosystem-based approaches for over a decade (Salwasser 1991, Salafsky and Margoulis 1999, Busch and Trexler 2003, Parrish et al. 2003). Our hybrid approach attempts to bridge the gap between these paradigms and regulatory implications of the ESA. In making this bridge, flaws inherent in the single-species approach can be remedied while still allowing links to those species. By looking at habitats, processes, landscapes, and threats to the integrity of these components, we can more readily incorporate theoretical frameworks into hypothesis development and subsequent sampling designs. Questions such as the importance of connectivity, habitat patch dynamics, and population demographic synchrony (e.g. Chen et al. unpublished) can be identified early in the implementation of a conservation plan, and hopefully answered while there are still opportunities to engage in adaptive management based solutions.

Using this framework, the number of species covered under MSHCPs becomes less daunting. Rather than measuring N_i precisely, surveys can incorporate species occupancy, relative species abundance, and community characterizations into a larger scale sampling framework that focuses on population drivers and stressors identified in conceptual models. Resources that are required for rigorously exhaustive sampling designs to obtain statistically valid estimates of N_i, can be directed to evaluating specific relationships with hypothesized population drivers. While rigorous sampling designs are still required (Green 1984), the questions are not limited to how many of a given species are present in a given year. As species-population-driver linkages are confirmed, monitoring can shift increasingly to those drivers. As the nature of population drivers are more completely understood, thresholds for management actions can be based on the natural, temporal, and spatial variation in those drivers, and can then identify where natural variance is negatively influenced by anthropogenic factors.

This hybrid approach provides biologists and managers an ever-increasing knowledge base on the importance and character of processes that drive population change at multiple scales. Importantly, it also provides a means of evaluating when those processes become dysfunctional and feeds seamlessly into an adaptive management strategy by providing direct guidance as to what components of a habitat warrant remedial management. Time and resources saved by making this direct link can then be focused on actions that remove stressors and improve the prognosis for survival for species and natural communities.

ACKNOWLEDGMENTS

Funding for the development of this monitoring framework was provided by the Coachella Valley Association of Governments, the County of Riverside, and the California Department of Fish and Game. We particularly thank James Sullivan, Yvonne Moore and Eric Loft. Figure preparation was aided by Tracy Tennant and helpful editorial suggestions were provided by Katie Barrows. Comments by Michael Morrison substantially improved this manuscript.

CHAPTER 3

POPULATION DYNAMICS OF A THREATENED SAND DUNE LIZARD

CAMERON W. BARROWS

ABSTRACT--Understanding how and why the abundance of a species changes in space and time is an essential component to effective endangered species conservation. Key to this understanding is being able to distinguish natural population dynamics from a downward trajectory of a species at risk of extinction. For many species in arid environments, rainfall drives population changes. This is the case for Coachella Valley fringe-toed lizards (*Uma inornata*), a species listed as threatened under the U. S. Endangered Species Act. At low rainfall levels the lizards exhibit negative population growth until annual precipitation exceeds 40-50 mm. Fluctuation in the lizards' population growth is also correlated with changes in their diet. A regression model using rainfall and diet to explain lizard population dynamics resulted in a R^2 of 0.956, $p < 0.0001$. As drought is common in their arid environment, it is not unusual for this lizard species to endure consecutive years of population declines. Fringe-toed lizard population counts during extended droughts often approach zero, yet the populations quickly rebound during periods of near average rainfall. If counts approaching zero are not reliable thresholds for when remedial management actions are warranted, then monitoring based management decisions need to use more heuristic criteria. Departures from the rainfall-diet-population growth model may provide the signal needed for management actions.

Natural population fluctuations are common in species (Pechmann et al., 1991; Blaustein et al., 1994) and are generally no cause for alarm. However not all declines are natural (Gibbons et al., 2000). Distinguishing between natural population dynamics versus a downward trajectory of a population at risk of extinction becomes a critical challenge for insuring the conservation of endangered species (Pechmann et al., 1991). Barrows et al. (2005) proposed a conceptual framework for addressing this key problem. Central to that framework is an understanding of what environmental factors drive the abundance of a species over space and time. This understanding is particularly acute in isolated, fragmented habitats that lack the buffering effects of connectivity to larger populations.

The following is an analysis of twenty years of data on population drivers for isolated populations of a species listed as threatened under the U. S. Endangered Species Act, the Coachella Valley fringe-toed lizard (*Uma inornata*). Historically, desert sand dunes systems, the sole habitat for this species (Stebbins, 1944; Norris, 1958; Barrows, 1997), stretched across much of the floor of the Coachella Valley in the Colorado Desert of southern California, providing nearly continuous habitat for a diverse community of aeolian sand-adapted species. Over the past three decades, increases in human population and suburban development have resulted in a 95% loss of this habitat (Barrows, 1996) and severe fragmentation of the remaining viable habitat. Implementation of a regional conservation initiative aimed at finding an adequate balance between species protection and continued economic development, termed a habitat conservation plan (HCP), was begun in 1986. The HCP requires that monitoring occur in order to assess the "status of the fringe-toed lizard populations". Population estimates alone rarely provide sufficient information to identify thresholds for when or how an adaptive management regime (Holling, 1978; Walters, 1986) should be employed for a species that is the focus of a conservation effort (Barrows et al., 2005). The objective was to identify one such threshold using an analysis of the population dynamics of *U. inornata* and correlations between, and annual rainfall and food resources to generate a predictive model to track and forecast natural population oscillations. Departures in the natural population fluctuations predicted by such a model could signal a need for remedial management actions.

In arid ecosystems, highly variable and unpredictable precipitation often becomes the driver of biological processes (Noy-Meir, 1973). Support for this axiom can be found across a broad range of taxa and regions (Mayhew, 1965, 1966; Pianka, 1970; Ballinger, 1977; Whitford and Creusere, 1977; Seely and Louw, 1980; Dunham, 1981; Abts, 1987; Robinson, 1990; Brown and Ernest, 2002; Castañeda-Gaytán et al., 2003; Germano and Williams, 2005). Climatic effects may be particularly acute in extremely arid deserts as variation in annual precipitation increases with decreases in mean annual rainfall (Noy-Meir, 1973; Bell, 1979; MacMahon, 1979).

Maximizing reproductive success during periods of high resource abundance may be critical for sustaining population viability through extended droughts. For desert lizards, higher reproductive success often correlates with increased rainfall (Robinson, 1990). This may be due to increased food availability from annual plant growth and phytophagous insects (Pianka, 1970; Ballinger, 1977; Ballinger and Ballinger, 1979; Seely and Louw, 1980; Dunham, 1981; Robinson, 1987 and 1990). This pattern should have the strongest correlation with longer-lived species, which can forego breeding except when environmental conditions may result in higher reproductive success (Williams, 1966; Tinkle, 1969).

Methods

Study Area

Data were collected on three separate active sand dunes within the Thousand Palms Preserve in the Coachella Valley, Riverside County, California. Historically, the valley soils were overlain with extensive sand dunes arising from flood outwash events from the San Bernardino Mountains to the northwest, San Jacinto Mountains to the southwest, and the Little San Bernardino and Indio Hills to the north. The Coachella Valley is classified as an extremely arid (Noy-Meir, 1973) shrub desert with a mean annual rainfall of 79 to 125 mm (Western Regional Climate Center, 2005).

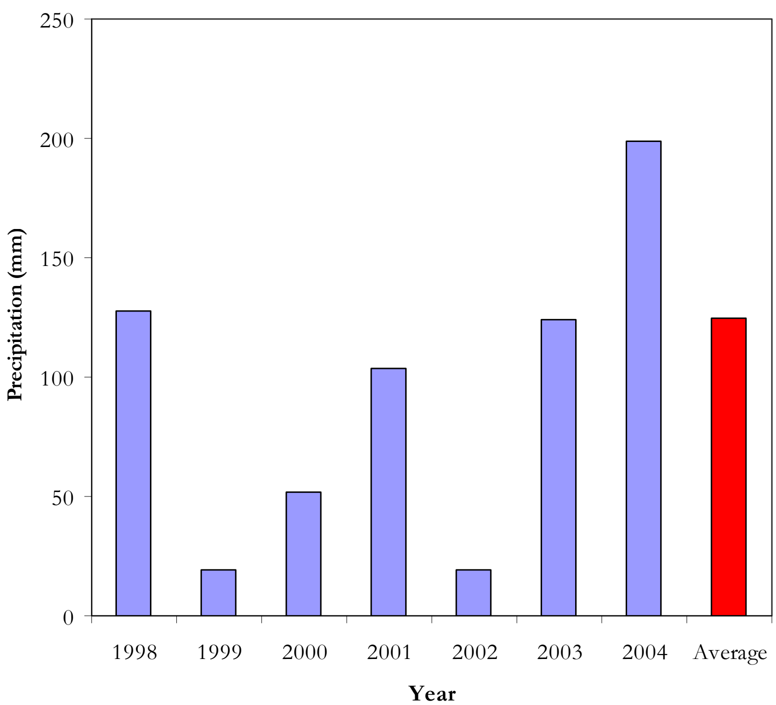


Figure 3.1. Annual precipitations from 1998 to 2004, compared with 62 year mean (Western Regional Climate Center, www.wrcc.dri.edu).

The lowest rainfall year occurred in 2002, with just 4-7 mm recorded across the valley floor. Temperatures show similar extremes ranging from a low approaching 0° C in the winter to highs exceeding 45° C commonly recorded during July and August.

The Thousand Palms Preserve (TPP) is in the central Coachella Valley, and has a 60-year average precipitation of 79 mm. Aeolian sand at TPP is finer than in dune habitats further west and forms active sand fields and dunes that are migrating over a fine silt substrate. Two plots (TPP1 and TPP2) were located approximately 2 km apart (33° 47' N, 116° 20' W) on separate active dunes, each > 50 ha, with sparse perennial vegetation dominated by creosote bush (*Larrea tridentata*) and saltbush species (*Atriplex canescens* and *A. polycarpa*). While these dunes were separated in terms of their physical location, there was likely biological connectivity between them. The inter-dune habitat consisted of an aeolian sand hummock habitat where fringe-toed lizards also occurred, although in much lower densities. Data were collected from 1986 to 2005 (TPP1) and from 1990 to 2005 (TPP2). A third plot (TPP3) was established on a physically and biologically isolated dune (33° 51' N, 116° 19' W) approximately 1 ha in size, 6 km north of TPP1 and TPP2. The habitat on this site differs from TPP1 and TPP2 in that it is adjacent to a natural desert fan palm (*Washingtonia filifera*) oasis and includes phreatophyte vegetation such as honey mesquite (*Prosopis glandulosa* var. *torreyana*) and arrowweed (*Pluchea sericea*). Data were collected at TPP3 from 1996 to 2000.

Survey Protocols

The TPP1 and TPP2 plots are 1000 m x 10 m belt transects. The small size of the available habitat at TPP3 did not allow the use of identical survey protocols. Area of surveyed habitat was the same on all plots. The number of repeated surveys required per year was determined using a Power Analysis with a standardized effect size of 3.3 and standard deviation of 1.5 (15 year mean, $\alpha = 0.05$, $\beta = 0.80$). Survey data are comprised of counts of *U. inornata* observed within the transects. Transects were surveyed at least six times per year in a spring (May/June) census.

In 1990 an autumnal census period (September/October) was added to the TPP1 and TPP2 plots to include young of the year, which emerge between July and September (Mayhew, 1966). TPP1 and TPP2 transects were surveyed an additional six times per year in September and October. On the TPP3 plot, a complete census was conducted eight to ten times during the same autumnal census period as at the TPP1 and TPP2 sites. Plant density and species richness were measured by counting all perennial plant species on the plots. Rainfall totals for the rain year (July through June) were recorded from a rain gauge on the TPP preserve and at the nearby Indio Fire Station.

Data Analyses

The observed mean annual rate of lizard population increase ($\bar{r}$) was calculated using $\bar{r} = \ln(N_{i+1}/ N_i)$ where N_i is the mean count of lizards observed during spring surveys in year i.

I analyzed fecal pellets (scat) to determine diet composition at all study locations. Scat were collected monthly (March through June) on and adjacent to the three plots from March 1996 through October 2002. This noninvasive method (Pietruszka et al., 1986, and references therein) was used because of the protected status of the lizard and the unknown consequences of repeated captures and disruption on normal lizard activities. Scat from adult *Uma* is distinguishable from scat of sympatric species by shape and size. Any scat of uncertain origin was discarded. Plant content in scat consisted of seeds, leaf fibers, and flower parts. Seed numbers could be identified in the scat, but most plant material could not be quantified with confidence as to the number of leaves or flowers consumed. Therefore, plant frequency was quantified as present or absent in each scat. For statistical analyses invertebrate frequencies were quantified based on the proportion of the lizards' diet (not including plants) for taxonomic groups (Formicidae, Coleoptera, Orthoptera, etc.). Diet analyses are confined to those data collected during the spring (March through June) season, the period when the lizards should be preparing for reproduction. Prey proportions were transformed (arcsine transformation) before inclusion in statistical analyses that required normal data distributions.

Multivariate models including rainfall and diet variables were limited as to the number of independent variables that could be included due to the number of years when both diet samples and rainfall were recorded. Combining TPP1 and TPP2 yielded an N=14. In order to create a model that allowed statistical inference, the number of variables needed to be limited so that the variable to observation ratio was ≤ 1:7 (Tabachnick and Fidell, 2001); models were therefore limited to two independent variables. Statistical analyses were performed using Systat 10.0 (Wilkinson, 1990). A threshold of $\alpha = 0.05$ for statistical significance is used throughout this paper.

Results

With rare exceptions, annual rainfall dynamics coincided with fluctuations in measures of population growth, $\bar{r}$ for each site considered in this analysis (Figure 3.1). While the direction of the correlation between $\bar{r}$ and annual rainfall was consistent, the amplitude of $\bar{r}$ with regard to rainfall was less unpredictable, and so the correlation between these parameters was relatively low, but nevertheless significant for the two sites with the longest data record. R^2 values for the three sites were 0.349, $p = 0.012$ for TPP1; 0.354, $p = 0.032$ for TPP2; and 0.279, $p = 0.471$ for TPP3. When annual rainfall exceeded 40 – 50 mm there was a shift from a negative to positive $\bar{r}$ at all four sites (Figure 3.2). Although there is limited data for corroboration, it appears that at annual rainfall amounts > 200 mm there may be a shift back to negative $\bar{r}$. TPP1 and TPP2 differed from TPP3 in vegetation and food resources available to the lizards. The TPP1 and TPP2 plots were similar with a sparse perennial shrub density of 134 plants/ha and 70 plants/ha, respectively. Just three shrub species were present on these plots; creosote bush comprised 13% and 63%, and saltbush constituted 87% and 37%, of perennial shrubs at the two sites. The juxtaposition of plot TPP3 with a natural palm oasis and upwelling groundwater along the San Andreas earthquake fault resulted in a much more mesic environment, with a perennial shrub density of 331 plants/ha. Five shrub species comprised the perennial vegetation community, with creosote bush at 2%, saltbush at 73%, arrowweed at 17%, alkali goldenbush (*Isocoma acradenia*) at 5%, and honey mesquite at 3% of the total composition. At all TPP plots, long lived annuals or biennials such as bugseed (*Dicoria canescens)* and coldenia (*Tiquilia plicata)* were present only in the wettest years.

Figure 3.1 Time series depicting the relationship between annual rainfall and population growth ($\bar{r}$) for the July through June rain year. The solid black line with solid squares at the data nodes represents annual rainfall; dashed line with open squares at the data nodes represents TPP1; open triangles represent TPP2; "x" represent TPP3.

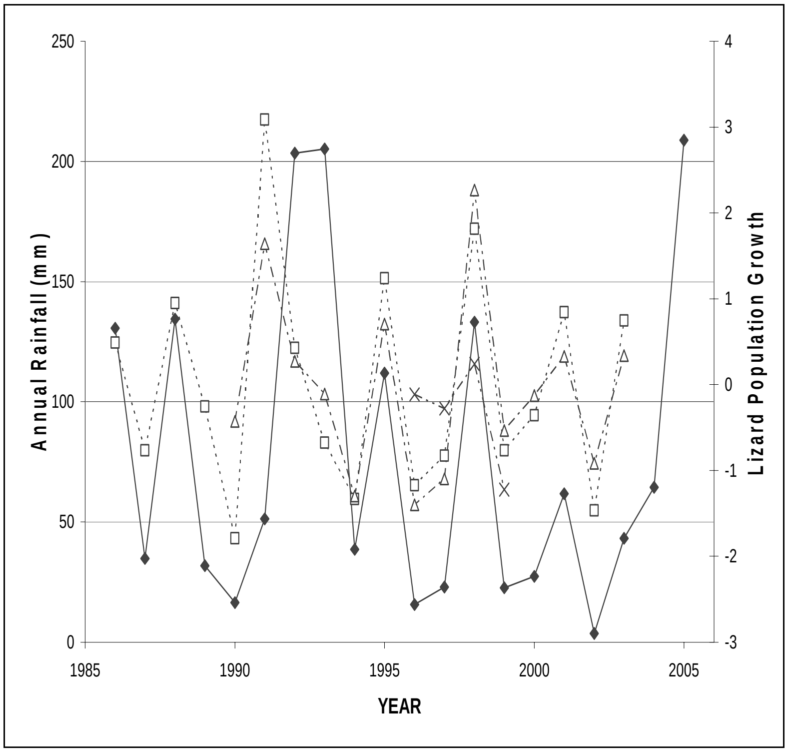
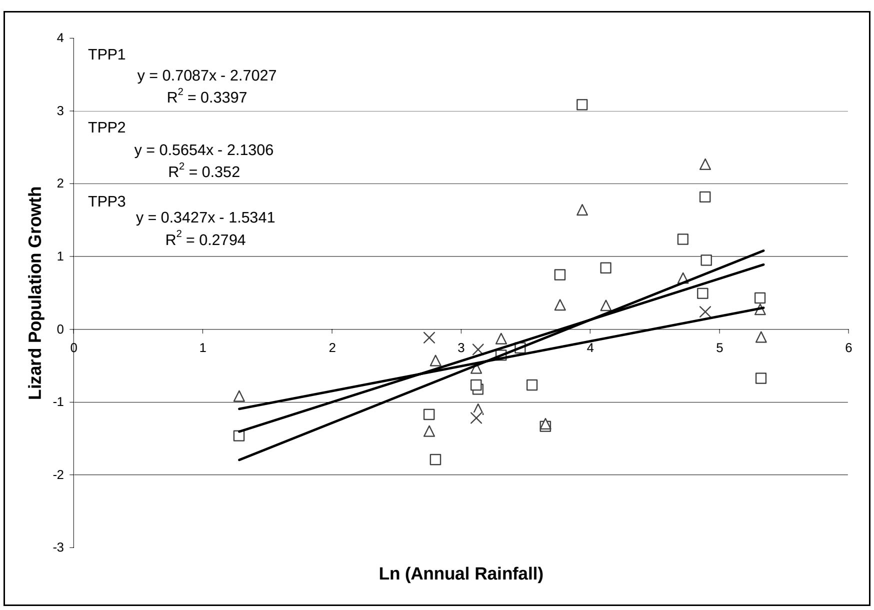
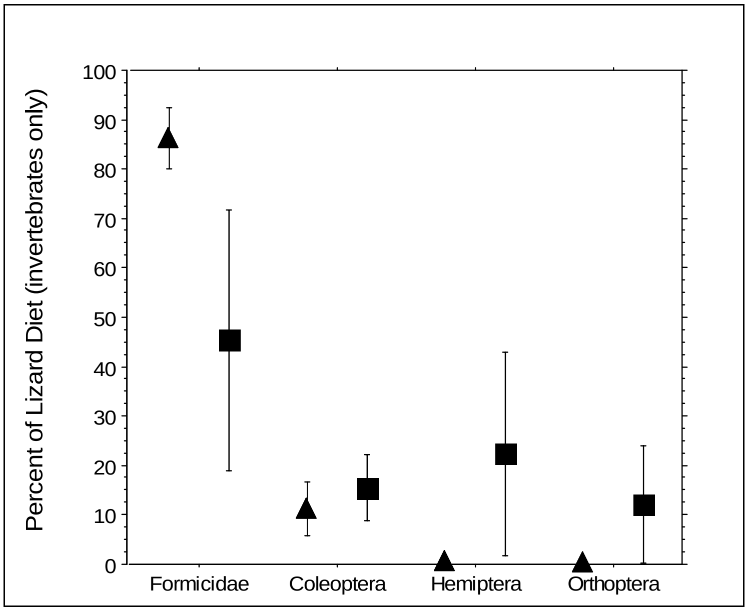


Figure 3.2 Relationship of natural log transformed annual rainfall versus population growth ($\bar{r}$) at the TPP sites.



A total of 701 *U. inornata* scat were collected at the TPP1 and TPP2 plots over a seven-year span (scat contained 7,758 arthropod prey items); 374 scat were collected on the TPP3 plot over five years (scat contained 2,676 arthropod prey items). Frequencies of plants, ants, and other invertebrates consumed varied between years of positive and negative population growth (Figure 3.3). Population growth ($\bar{r}$) was positive for only two of the seven years that data was collected.

Figure 3.3 The proportion of four diet categories at the combined TPP1 and TPP2 sites showing the difference in fringe-toed lizard diets between years with positive (squares) and negative (triangles) population growth ($\bar{r}$).



Diet shifts between positive and negative $\bar{r}$ years were seen on the TPP1 and TPP2 plots. The proportion of lizard scats with plants and with harvester ants (*Pogonomyrmex* sp.) differed ($U_{0.05\,(1)\,2,\,5} = 10$, $p < 0.05$) with more plants and fewer ants ingested during years of positive $\bar{r}$ than years of negative $\bar{r}$. No statistically significant differences in the proportion of plant to invertebrate prey were detected on the TPP3 site. However, data collection on TPP3 was curtailed two years earlier than expected because the population became extirpated. Pairwise Pearson's correlation values for four diet variables, rainfall and population growth at the TPP1 and TPP2 plots are shown in Table 3.1.

Table 3.1. Pearson's correlation coefficients relating Coachella Valley fringe-toed lizard population growth ($\bar{r}$) to diet variables and precipitation at 4 sites.

Variables	TPP1	TPP2	TPP3
Annual Rainfall	0.933	0.806	0.529
Ants	-0.742	-0.848	-0.723
Beetles	-0.057	-0.025	0.412
Invertebrates (- Ants)	0.624	0.806	0.686
Plants	0.516	0.310	-0.217

Due to overall similarity between patterns of response to rainfall and diet variables, TPP1 and TPP2 data were combined to create a multivariate model to explain the variance in population growth at these sites. Annual rainfall and the proportion of harvester ants in the lizards' diet were the variables that had the most consistent (between plot) and strongest correlation with the lizards' population growth (from Table 1). The resultant multi-variable, linear regression model yielded an $R^2 = 0.956$, $p < 0.0001$.

Discussion

The relationship between annual rainfall and population growth for Coachella Valley fringe-toed lizards is consistent with that for other desert species (Mayhew, 1965; Pianka, 1970; Ballinger, 1977; Whitford and Creusere, 1977; Seely and Louw, 1980; Dunham, 1981; Abts, 1987; Robinson, 1990; Brown and Ernest, 2002; Castañeda-Gaytán et al., 2003). Even small fluctuations in annual rainfall corresponded to parallel demographic changes for fringe-toed lizards. Although statistically significant positive linear correlations between rainfall and population growth were measured, this relationship may not extend to high rainfall years. At the highest rainfall levels recorded the lizards' population growth declined. While a fit with a linear model was statistically significant, the relationship between rainfall and the lizards' population growth may not be truly linear; rather there may be a rainfall threshold at about 40-50 mm, above which sufficient resources are present to enable positive population growth.

During dry years the lizards' diets were dominated by harvester ants and the lizards often appeared extremely thin. I examined differences in the lizards' diet with respect to their population growth. If rainfall patterns produced enhanced resource abundance and availability the result should be reflected in the lizards' diet. For the two sites with the sparsest perennial vegetation, diet differences between years of high positive and negative population growth were significant. Combining precipitation and the proportion of ants in the lizards' diet into a multivariate model explained over 95% of the variance in population growth over a seven year period. During years of higher rainfall the lizards ate relatively more annual plants and larger, phytophagous invertebrates, primarily species of Hemiptera and Orthoptera. Reproduction during these years increased. At a finer scale, Durtsche (1992) described a similar shift in foraging strategies by female *U. inornata* during the breeding period in spring. Such a foraging strategy would increase fat deposition rates and increase the lizards' reproductive success (Robinson, 1990). The increased abundance of annual plants and arthropods in wetter years provides lizards with a greater array of food choices. Previous studies analyzing fringe-toed lizard diets have suggested that the lizard diets reflect available resources (Durtsche, 1992, 1995; Gadsden and Palacios-Orona, 1997).

The shift to negative population growth when annual rainfall exceeded 200 mm appears inconsistent with the hypothesis that higher rainfall leads to increased annual plants, increased arthropods, and then increased lizard reproduction. Several hypotheses could explain this response to high precipitation levels. Andrews and Wright (1994) determined experimentally that an extended period of above average rainfall resulted in a population decline due to reduced egg viability in a tropical lizard. Extremely moist conditions in the egg chamber can cause increased fungal and microorganism infections (Tracy, 1980), or reduced gas exchange (Packard and Packard, 1984). Additionally, the assumption that the relationship between rainfall and food resource abundance is linear at the higher rainfall levels may not be correct. High rainfall (approximately 210 mm) in 2005 did not result in increased abundance in harvester ants or beetles on the sand dune habitat (Barrows, unpubl. data)

Coachella Valley fringe-toed lizards were observed exhibiting negative population growth 10 out of the 20 years of my study. The population dynamics described here are consistent with a non-equilibrium paradigm (Picket et al., 1992), where population levels reflect stochastic changes in resource availability. As drought is common in arid environments, it is not unusual for this species to endure consecutive years of population declines. During extended droughts mean population counts often approached zero, yet the populations quickly rebounded during periods of near average rainfall. From the standpoint of developing a meaningful monitoring program, counts approaching zero, or consecutive years of negative population growth are thus not reliable thresholds for when remedial management actions are warranted to maintain the continued viability of this population. Monitoring based management decisions need to use more heuristic criteria.

Maintaining viable populations of endangered species in highly variable environments presents a unique challenge. Such populations can be stressed by fragmentation (Saunders et al., 1991, Martin and McComb, 2003, Chen et al., 2005), compromised processes maintaining suitable habitat (Barrows, 1996), invasions of exotic species (Atkinson, 1989), and changes in predator population composition and density (Crooks and Soule, 1999), among other factors. Addressing these stressors or their effects through remedial management may be critical to the continued existence of endangered species. Identifying when the species' populations are being negatively impacted by the stressors thus should become a focus of monitoring activities. Separating natural population dynamics from anthropogenic stressor effects can be difficult without a conceptual and quantitative framework from which to evaluate population changes. Departures from the rainfall-diet-population growth model may provide the signal needed for management actions. The influence of potential population stressors, such as population isolation or the invasion of exotic vegetation, may be interpreted as to the magnitude of departure from the regression model. Without this conceptual context, with empirical support, managers charged with insuring that lizard populations persist would have difficulty discerning when management actions are warranted. Developing models to explain natural population variance so that departures from those models can be identified should be a critical conservation goal for any species at risk of extinction.

The Thousand Palms Preserve where my research was focused is a remnant of a once extensive sand dune ecosystem (Barrows, 1996). The fragmented nature of the remaining protected habitat begs questions as to the long-term viability of these populations. Chen et al. (2005) modeled the long-term viability of *U. inornata* on the same plots I have described in this analysis. They determined that on isolated habitats < 100-200 ha in size, long-term viability of this species was doubtful. Certainly the extinction of the TPP3 population, occurring on an isolated sand dune just 1 ha in size is consistent with that prediction. The TPP1 and TPP2 plots occurred on sand dunes that were each less than 60 ha, but occurred in a mosaic of sand dunes that totaled about 250 ha. These dunes were imbedded in a matrix of inter-dune, sand hummock habitat where fringe-toed lizards also occurred, albeit in low densities. The lizards could easily travel between the higher quality sand dune habitats. The total occupied habitat was closer to 750 ha, and so the prospects for long-term persistence for the TPP1 and TPP2 plots may be more secure than a focus on only the individual habitat patches where the plots were located would indicate. Nevertheless, on-going monitoring of these sites, using the dynamic context I have provided here, will provide early detection and thus an opportunity for remedial response if habitat fragmentation becomes a population stressor here.

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CHAPTER 4

COACHELLA VALLEY SPECIES ACCOUNTS

4.1 COACHELLA VALLEY FRINGE-TOED LIZARD, *UMA INORNATA*

Cameron W. Barrows

Coachella Valley fringe-toed lizards (CVFTL) are endemic to the Coachella Valley. They are restricted to aeolian sand habitats which once encompassed roughly 260 km^2 (100 mi^2) of the valley floor. Today no more than 5% of that habitat remains viable, with intact ecosystem processes. CVFTL populations have been monitored in two separate efforts since 1985 and 1986 at a limited number of locations (one and four sites, respectively). Since 2002, UCR's Center for Conservation Biology has engaged in a broader approach, both in aerial extent and in the species being surveyed. This monitoring effort includes all of the aeolian sand habitats under conservation ownership as its sampling frame. In 2002 surveys included 12 sites (clusters) with a total of 80, 10 m x 100 m belt transects. In 2003 the sampling included 18 sites with a total of 116 transects; in 2004 there were 22 clusters including 134 transects, and in 2005 there were 17 clusters totaling 106 transects. The changes in sampling effort corresponded to different questions being asked each year. Cluster sites were stratified between four subdivisions of the Coachella Valley aeolian sand community: active sand dunes, sand hummocks, ephemeral sand fields, and mesquite dunes. CVFTL occurred in each of these subdivisions each year, although their relative abundance varied in each habitat type from year to year. Our primary questions with regard to CVFTL are:

1. Does the sampling protocol provide a reliable, repeatable indication of the lizards' range, occupancy and relative abundance?

2. What are the habitat affinities for this species?

3. What are potential stressors? For example, do habitat edges have an impact on the lizards' distribution and abundance?

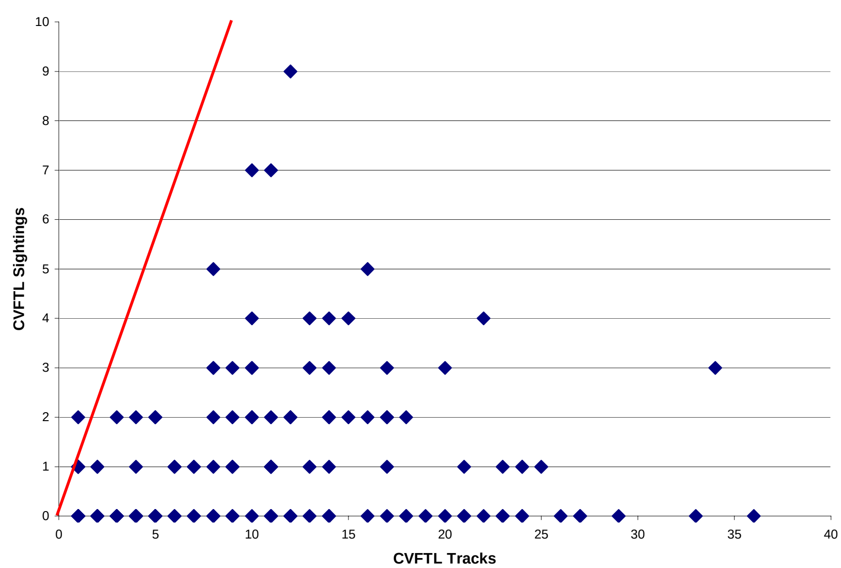
<u>Survey Protocols and Detection Issues</u>

Methods available to survey CVTL include mark-recapture, distance sampling, transects using the number of sightings as the primary metric, and transects using diagnostic tracks as the primary survey metric. A mark-recapture survey occurred concurrently through a long-term research effort by staff at the UCR-Deep Canyon Desert Research Station. The effort required to employ that method limited them to one 2.25 ha plot. As our objective was to survey across all the remaining habitats available to CVFTL, we rejected mark-recapture as a potential method. Distance sampling requires being able to sight all lizards along the transect center line, which is a criterion that would not likely be met (see below), and requires large sample sizes (>50) which could also be difficult to meet and still have enough replicated samples for statistical analyses.

The method we chose included the establishment of 10 m x 100 m belt transects as the primary sampling unit. Each transect was re-surveyed up to six times in order to determine sampling variance and means. Surveyors slowly walked each transect looking for lizards as well as their diagnostic tracks. When multiple tracks were located they could often be separated to different individuals by differences in size, or would be followed until the tracks diverged as separate individuals. When the number of individuals could not be determined, then a conservative minimum number of lizards were recorded for that transect. Transects were clustered in a 2.5 ha area so that between-transect differences in habitat were minimized. Data analysis could then be scaled up from transect, cluster, habitat type, to general Coachella Valley summaries.

We compared the relative detection rates of sightings versus tracks as a method for population estimates. We did not estimate true population size; rather we estimated relative abundance between clusters, habitats, and between years. Figure 4.1.1 plots the number of CVFTL sightings and trackways (tracks left by a single lizard) per cluster per survey. The red diagonal line indicates when the number of sightings equaled the number or trackways. Points to the right of that line represent surveys where trackway counts > sightings.

Figure 4.1.1 CVFTL sightings and trackways per cluster per survey.



One concern of using sighting data was the large number of false negatives (surveys in which no lizards were sighted on a cluster, even though their diagnostic tracks were present) that would be recorded. In 2004 and 2005, on approximately 65% of the surveys in which tracks were present, no lizards were seen. The analysis below (Table 4.1.1) shows what proportion of the total lizards recorded on a cluster during a survey would have been counted if sightings were the only metric used.

Table 4.1.1 Total lizards recorded on a cluster in 2004 and 2005.

Habitat Type	2004	2005
Active Dunes	0.07	0.04
Sand Hummock	0.00	0.00
Ephemeral Sand Field	0.37	0.33
Mesquite Dune	0.13	0.08

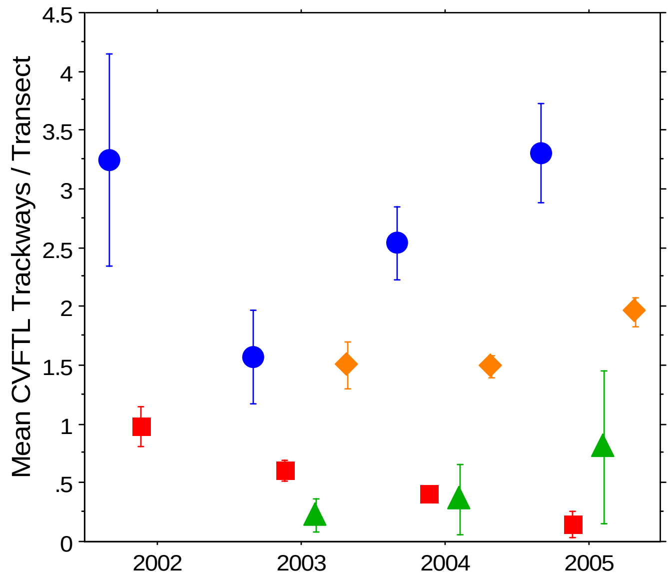
Using tracks avoids many of the detection problems associated with lizard surveys, such as the lizards ducking behind shrubs, burying in the sand, or leaving the belt transect at the approach of the surveyors. CVFTL are vigilant and can be evasive; they usually detect surveyors before being detected themselves. Detecting CVFTL by sightings is dependent on shrub density, annual plant cover, when or if the lizards take evasive action, what sort of behavior the lizards use for evasion, and whether the survey is conducted at the optimal temperatures and times of day. However, if the lizard is or was active on that site it will leave a track that an experienced surveyor can distinguish from any other species, and whether the track was from that day or from a previous day. Differences in cryptic behavior, habitat conditions and temperature are nullified, from the standpoint of detecting the lizards, as long as the substrate includes fine sand.

<u>Relative Abundance by Year and Habitat</u>

Research presented in this report has demonstrated that direction of CVFTL population growth can be predicted from annual precipitation data. Prediction accuracy improves with the inclusion of additional resource condition variables, such as diet. Annual precipitation in 2002 was the lowest on record for the Coachella Valley (3-7 mm). Due to the one year time lag between the lizards' response to the drought, in terms of reproductive success, and when the populations were surveyed CVFTL relative abundance in 2003 would be predicted to be at a low ebb. Population abundance patterns depicted in Figure 4.1.2 support that prediction.

From 2003 through 2005 there was increasing rainfall each year, with precipitation approaching mean levels in 2004 (64 mm) and rainfall in 2005 nearly three times mean levels (210 mm). CVFTL should have positive population growth through those years, and for active dunes, ephemeral sand fields and mesquite dunes that was in fact what was recorded.

Figure 4.1.2 Error bars indicate one standard error. Blue circles represent active dune habitat, red squares indicate sand hummocks, green triangles depict ephemeral sand fields, and orange diamonds represent mesquite dune habitats.



The only habitat in which the CVFTL showed a continued decline was in the sand hummocks. This decline was not confined to CVFTL; flat-tailed horned lizards (*Phrynosoma mcallii)*, shovel-nose snakes (*Chionactis occipitalis)*, sidewinders (*Crotalus cerastes)*, and banded geckos (*Coleonyx variegatus)* all showed similar declines in the sand hummock habitat over the same time period. At the same time there were population increases in all the rodent species in that habitat. Whether there is a causal link between the increased rodent populations and the decline in reptiles was not tested.

For CVFTL, though not for the other reptiles mentioned, the sand hummock habitat may not be viable habitat in most years (Table 4.1.2). CVFTL populations are correlated with non-stabilized sand, low shrub density, low cover from annual plants, sparse cover from the exotic annual grass (*Schismus barbatus)*, and relatively dense cover of another exotic, Russian thistle (*Salsola tragus)*. Of all habitats sampled, sand hummocks have the mean highest sand compaction, and the highest percent cover of annual plants, indicating a poor fit to preferred CVFTL habitat. Sand hummocks may be a “population sink” habitat in most years.

Table 4.1.2 Pearson pairwise correlations for CVFTL habitat associations.

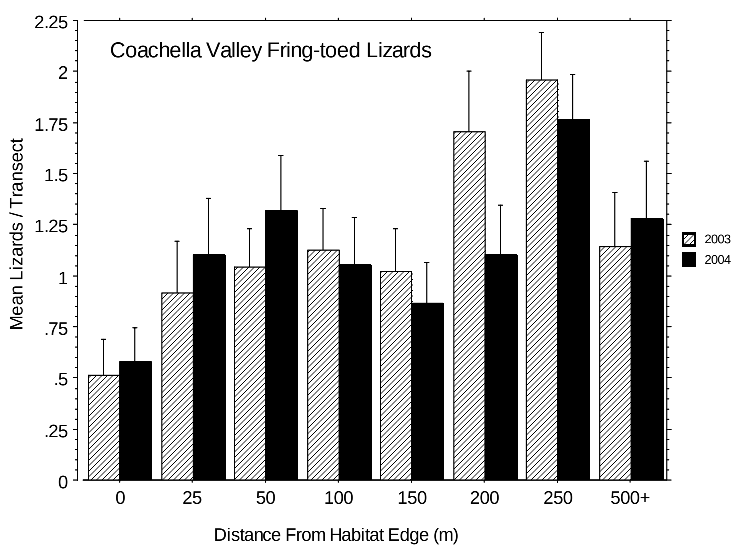
Habitat Variable	2003	2004	2005
Sand Compaction	-0.449	-0.604	-0.539
Shrub Density	-0.167	-0.254	-0.294
% Cover Annual Plants	-0.365	-0.415	-0.333
% Cover *Schismus barbatus*	-0.278	-0.258	-0.346
% Cover *Salsola tragus*	0.151	0.219	0.381

Population Stressors

The current patterns of CVFTL population abundance may be impacted by future fragmentation and encroachment as a result of conversion to suburban landscapes along the conserved habitat boundaries. This hypothesis was tested at the Thousand Palms Preserve where clusters were established along the preserve edge and in core habitats. Nine clusters were established along the preserve boundary and five clusters were located within the core of the preserve.

Based on the data shown in Figure 4.1.3 it appears that there may be an edge effect that extends no farther than 25 m from the preserve fence line. However, a logistic regression using edge distance, sand compaction, and shrub density as independent variables identified more compacted sand along the immediate preserve boundary as the best explanatory variable for the variance.

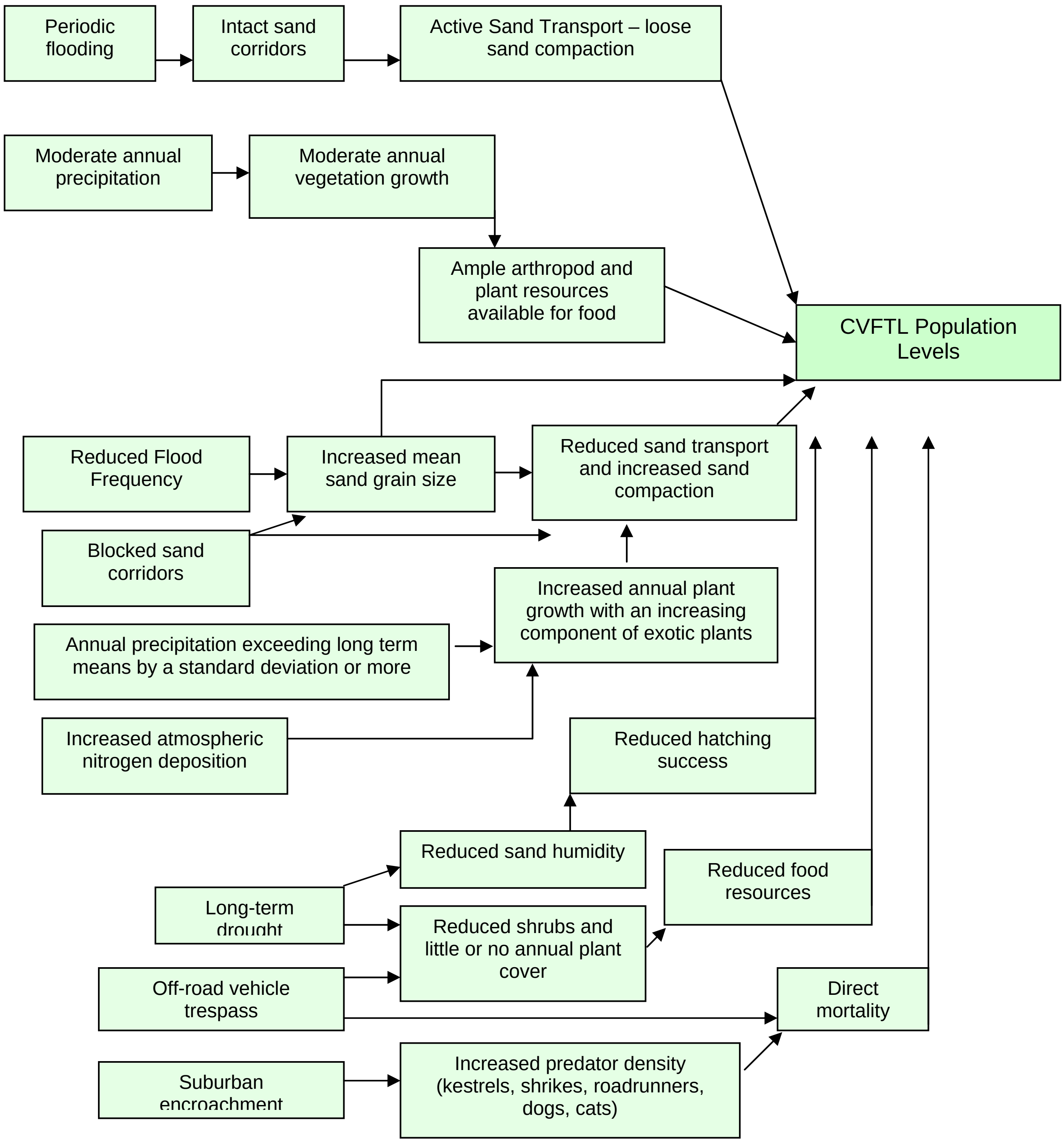
Figure 4.1.3 Relative abundance of CVFTL with respect to the preserve boundary.



Although the data have not yet been fully analyzed, it appears that the invasion of Saharan mustard (*Brassica tournefortii)* in 2005 may be a stressor to CVFTL populations, especially if it becomes a persistent feature on the preserve. The mustard appears to promote higher sand compaction, and as was shown above, CVFTL are sensitive to that habitat variable. There also appears to be impacts to native annual plant reproduction, which in the long term could also impact food resources for the lizards. The survey protocol was able to detect departures from the predicted CVFTL population growth that correlated with the density of mustard on that transect or cluster.

As described in Chapter 3, the CVFTL community envirogram (Figure 4.1.4) illustrates relationships, both known and hypothesized, that describe population responses to ecosystem processes.

Figure 4.1.4 CVFTL Envirogram. Ultimate drivers of stressors are depicted to the left of the diagram whereas more proximate impacts on population levels are shown to the right of the diagram. Drivers of positive population growth are in the top half of the diagram (in green) and stressors that result in negative population growth are in the lower half (in orange).



4.2 FLAT-TAILED HORNED LIZARDS, *PHRYNOSOMA MCALLII*

Cameron W. Barrows

Flat-tailed horned lizards (FTHL) have the smallest range of any horned lizard found in the U.S. They reach their northern and western-most distribution in the Coachella Valley. Historical records indicate that FTHL once occurred throughout much if not all the aeolian sand habitats that once covered the valley floor. FTHL were surveyed each year between 2002 and 2005 on conservation lands within that historical distribution. In 2002 surveys included 12 sites (clusters) with a total of 80 10 m x 100 m belt transects. In 2003 that effort included 18 sites with a total of 116 transects; in 2004 there were 22 clusters including 134 transects, and in 2005 there were 17 clusters totaling 106 transects. Cluster sites were stratified between four subdivisions of the Coachella Valley aeolian sand community: active sand dunes, sand hummocks, ephemeral sand fields, and mesquite dunes. Our primary questions with regard to FTHL were:

1. Whether our sampling protocol provided a reliable, repeatable indication of their range, occupancy and relative abundance,

2. What were the habitat affinities for this species, and

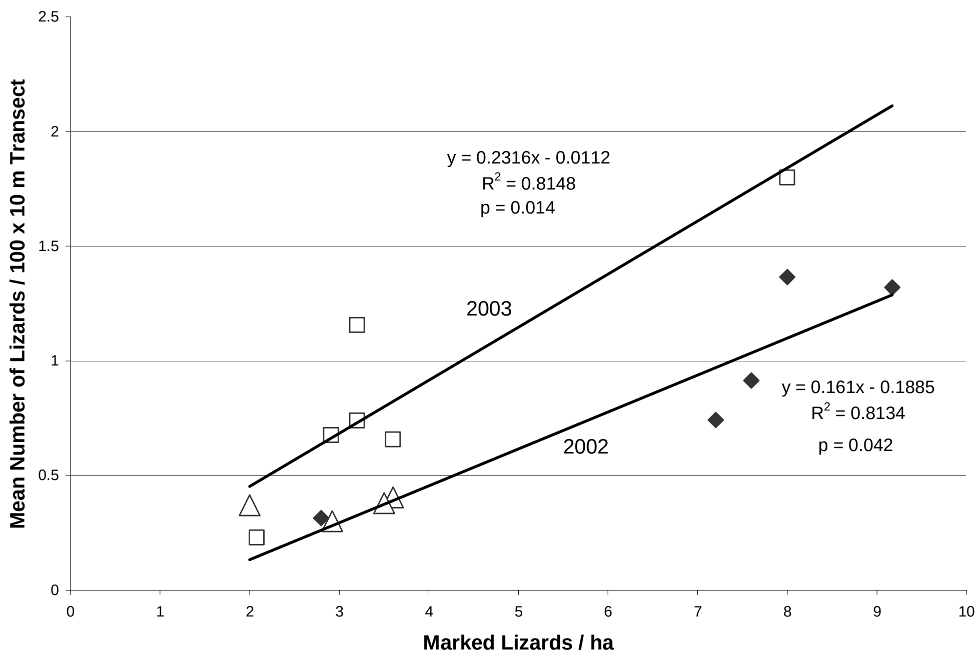
3. Determine whether potential stressors, such as whether habitat edges had an impact on the FTHL distribution and abundance.

Survey Protocols and Detection Issues

Methods to survey FTHL potentially include mark-recapture, distance sampling, counts dependent on sightings and counts dependent on tracking. Distance sampling involves walking line transects and then measuring the precise perpendicular distance of all sightings from that line. It includes the assumption that all individuals on or near the center line will be counted, an assumption that could not necessarily be met with FTHL due to their cryptic coloration and behavior. Distance sampling also requires reasonably large sample sizes (> 50) which is a criterion that could not be met on the plot sizes we chose. Although many FTHL were seen during surveys, their presence was invariably determined by initially tracking the animals first. Mark-recapture sampling, generally believed to be the most accurate but also time intensive method, was employed in 2002 through 2004 on five clusters (Figure 4.2.1) where tracking belt transect techniques were also employed so as to evaluate the efficacy of that method. There was a close agreement between the two methods. The differences between 2003 and the results for 2002 (diamonds) and 2004 (open triangles) may reflect some unexplained change in the lizards' behavior that year, but highlight the temporal limits to the accuracy of detection estimates. Regression lines for 2002 and 2004 data are nearly identical.

The problems associated with detection of such a highly cryptic species such as FTHL appear to be resolved using their diagnostic tracks as the primary metric of their occurrence and abundance. This method is effective only where there are fine sands available. However, even in areas dominated by hard packed silt or sand, the FTHL seemed to seek out the patches of available loose fine sand and so their presence could be reliably detected.

Figure 4.2.1 Comparisons of Mark-recapture sampling undertaken in 2002 - 2004 on five clusters.



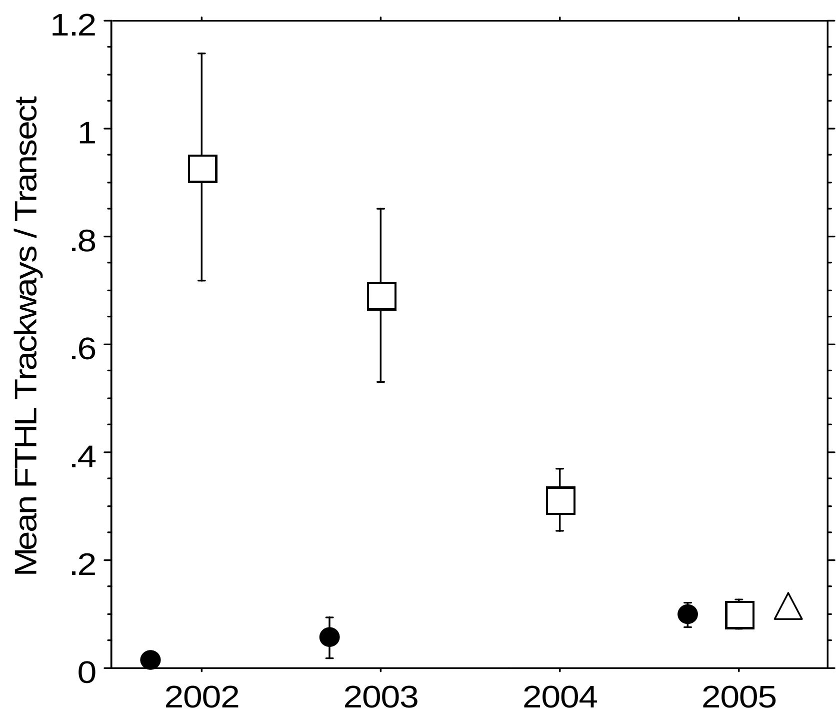
Surveys were repeated on each transect six times between late May and mid July. Surveys were conducted as early in the morning as the lizards became active (usually by 0800 hrs, and when the temperature 1 cm above the sand reached 35° C), and were concluded by 1100 hrs when the high angle of the sun reduced track definition.

<u>Relative Abundance by Year and Habitat</u>

No FTHL were detected outside the Thousand Palms Preserve during any surveys. Further west, on habitats previously occupied by FTHL, there has been an ephemeral loss of much of the aeolian sand, making the substrate more suitable for the desert horned lizard, *P. platyrhinos*. Desert horned lizards were found on all the western plots previously occupied by FTHL. The ephemeral character of the western sites likely resulted in a dynamic distribution for FTHL, however today, due to habitat fragmentation, the opportunity to maintain that dynamic character has been eliminated.

The temporal patterns of abundance by habitat on the Thousand Palms Preserve is depicted in Figure 4.2.2. Closed circles indicate active dune habitat, open squares depict sand hummocks, and triangles represent mesquite dune habitat. Sand hummock habitat is clearly the primary habitat for FTHL within the Coachella Valley, although in 2005 FTHL were observed on a greater variety of aeolian habitats, in greater numbers on non- sand hummock habitats than had previously been observed. In 2002 the observed FTHL abundance on the sand hummock habitat within the Thousand Palms Preserve exceeded that reported for any location elsewhere in the lizards' range. The steady decline in abundance since that year is an opportunity to explore those factors that influence FTHL abundance. FTHL were not alone in this decline. Coachella Valley fringe-toed lizards, *Uma inornata*, Shovel-nose snakes, *Chionactis occipitalis*, sidewinders, *Crotalus cerastes*, and banded geckos, *Coleonyx variegatus,* all showed similar declines.

Figure 4.2.2 Temporal patterns of abundance by habitat on the Thousand Palms Preserve. Error bars indicate one standard error.



In 2002 precipitation was the lowest on record for the Coachella Valley (3-7 mm). Since that year there has been increasing precipitation each year, reaching near average levels in 2004 (64 mm), and reaching nearly three times average levels in 2005 (210 mm). Increasing rainfall should indicate increasing resource availability; however that was not the case here. Figure

4.2.3 depicts abundance of the lizards' ant prey (*Pogonomyrmex, Myrmecosystus*, and *Messor*) within the sand hummock habitat over the years when FTHL were surveyed. The parallel decline in both ants and FTHL indicates these patterns are correlated, especially since the FTHL diet consists almost exclusively of these ants.

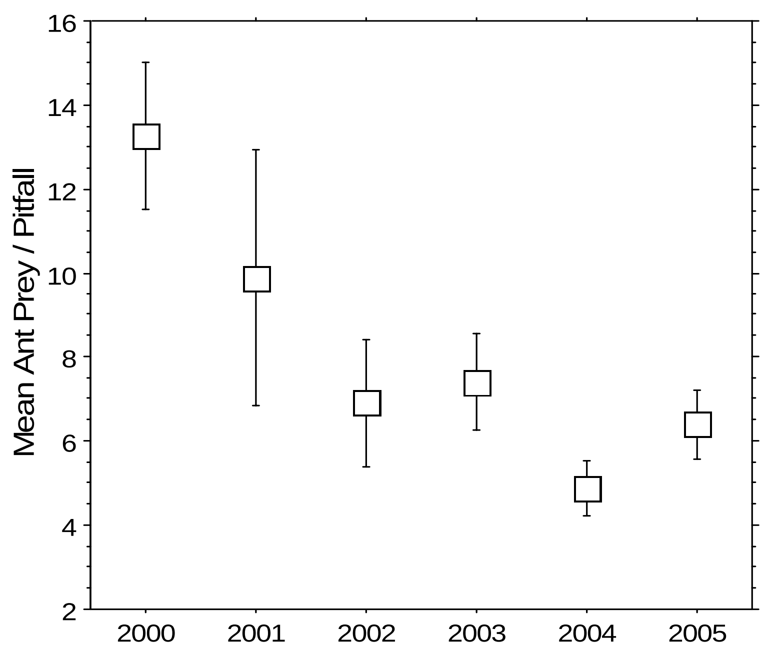
While several hypotheses are available to explain the declining ant population, none were tested. Hypotheses include:

1. Increasing precipitation may foster disease, or parasitism of the ants, or their food caches may be put at risk due to fungal invasions.

2. Increasing precipitation has resulted in an increased rodent population; research elsewhere has convincingly demonstrated a competitive relationship between harvester ants and Heteromyid rodents.

3. In 2004, increased primary productivity was negated by an explosion of sphinx moth caterpillars which consumed most of the annual vegetation prior to seed set. In 2005 native seed production was compromised by the invasion of exotic Saharan mustard, *Brassica tournefortii*. The relationship between increased precipitation and resources available to the ants may not be as tight as was the *a priori* assumption.

Figure 4.2.3 Abundance of the lizards' ant prey (*Pogonomyrmex, Myrmecosystus*, and *Messor*). Error bars indicate one standard error.



The observed decline in other reptile species may have a similar causal link. Both banded geckos and shovel-nosed snakes are commonly, if not primarily, found in the sand hummocks. They depend on arthropods for prey, and overall arthropod abundance has paralleled the patterns observed for the ants. The FTHL decline, without any ecological context, presents an immediate concern for the future viability of this species. Even within an ecological context those concerns are not alleviated. However, knowing that the decline has not been limited to FTHL, but is part of a broader pattern, points to focused research questions that should allow us to discern the longer term implications of those patterns. Without such research, developing a meaningful adaptive management scenario here could be problematic. For instance, removal or thinning of the rodent population may have the desired competitive release effect on the ants, but the overall impacts of such an action would be difficult to predict. Predators currently utilizing the rodents as a food source may shift to reptiles, resulting in further declines in their populations.

<u>Population Stressors</u>

The current patterns of FTHL population abundance may be impacted by future fragmentation and encroachments as a result of conversion to suburban landscapes along the conserved habitat boundaries. This hypothesis was tested at the Thousand Palms Preserve where clusters were established along the preserve edge and in core habitats. Nine clusters were established along the preserve boundary and five clusters were located within the core of the preserve. Results of this analysis are shown in Figure 4.2.4. An edge effect is evident from 100 m up to 150 m from the preserve boundary. ANOVA results for whether the variance in FTHL abundance was related to distance from the preserve edge ranged from $p = 0.003$ to $p = 0.000000009$ over the three years. A logistic regression using edge distance along with habitat metrics as independent variables identified edge distance as the only variable that explained a significant amount of the variance in the abundance of FTHL each year.

Reasons for this response were evaluated and include:

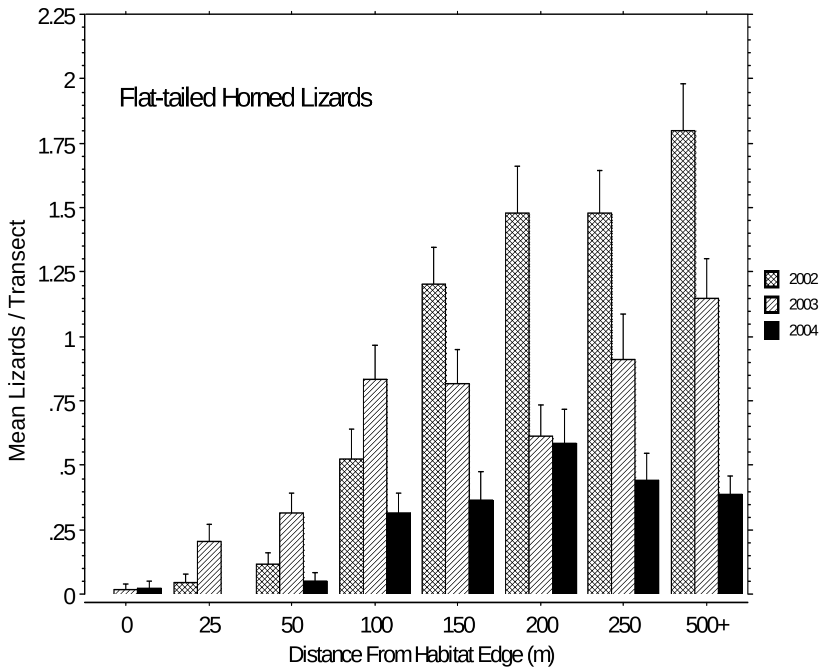
1. Invasions of exotic Argentine ants along the boundary regions.

2. Road mortality along the roads that border the preserve.

3. Enhanced predation pressure along the boundary.

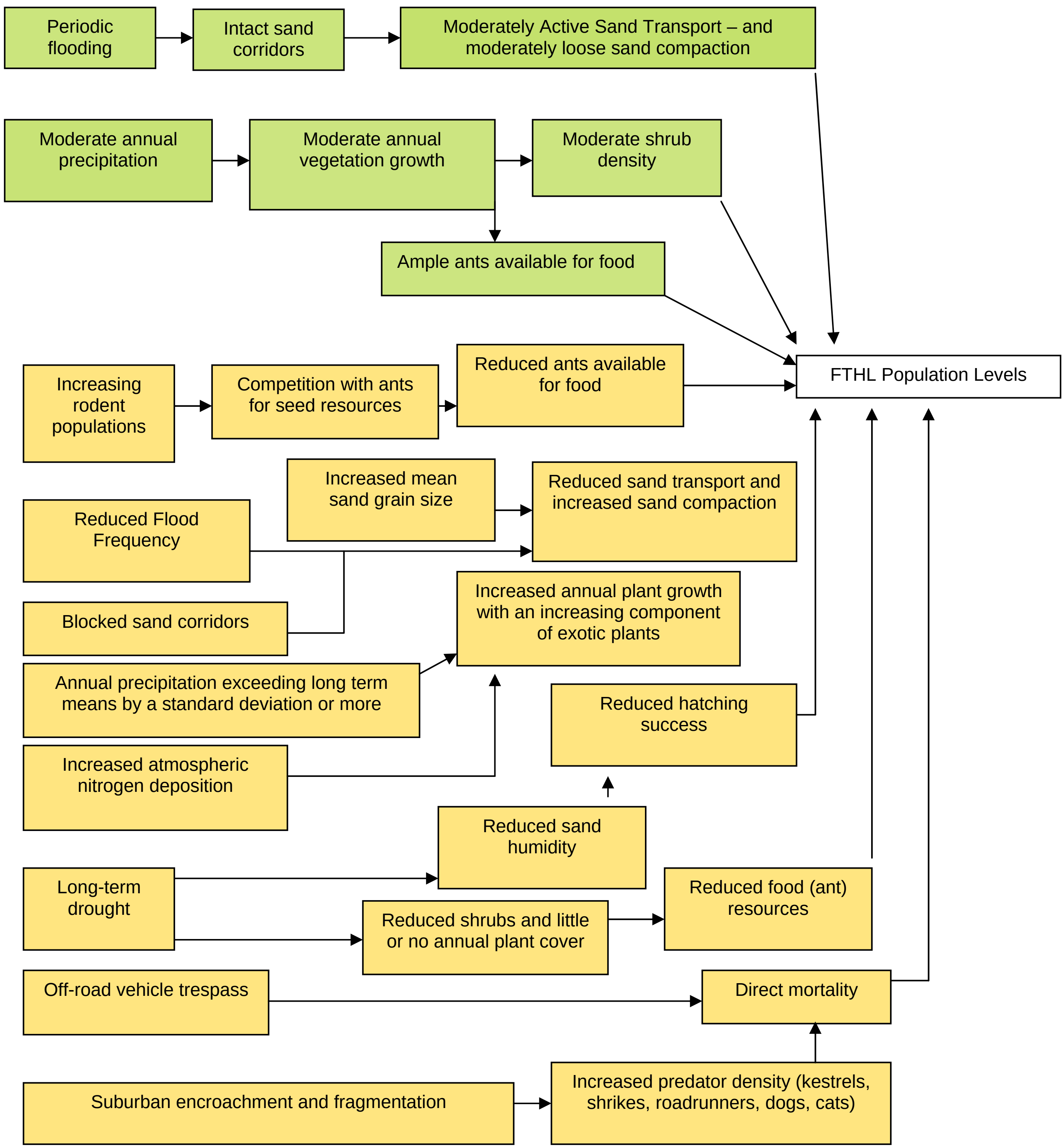
The first hypothesis was rejected since no exotic ants were located, nor was there any reduction in ant abundance along the boundaries. Hypothesis #2 could not be rejected or fully supported as the data were inconclusive. There was a larger edge effect adjacent to the wider road with curbs, but that same edge was more proximate to suburban development than the edge with a narrower, less used road. Hypothesis #3 was supported. American kestrels and loggerhead shrikes nested in suburban areas or in tree rows planted for agriculture and then foraged within the preserve. A power line along the preserve boundary provided perches from which both bird species launched their hunting sorties, resulting in a "dead zone" along the preserve boundary. FTHL was the only species occurring within the boundary region that demonstrated an edge effect.

Figure 4.2.4 The effect of edge processes on the abundance of flat-tailed horned lizards.



The results of this analysis point to an opportunity to initiate an adaptive management strategy. Under the direction of the appropriate agencies, the power line could be re-located or placed underground, thus removing the predators' perches. Additionally, suburban communities could be encouraged to reduce their use of fan palms for landscaping (where the kestrels often nest) and prune those existing palms in March or February to discourage nesting efforts, and to remove potential nest sites.

FTHL Envirogram (green = positive growth, yellow = negative growth)



4.3 COACHELLA VALLEY ROUND-TAILED GROUND SQUIRREL, *SPERMOPHILUS TERETICAUDUS CHLORUS*

Cameron W. Barrows

Round-tailed ground squirrels (RTGS) were surveyed in the years 2003-2005 on plots distributed on conservation lands throughout the remaining aeolian sand community of the Coachella Valley. In 2003 that effort included 18 sites (clusters) with a total of 116 100 m x 10 m belt transects; in 2004 there were 22 clusters including 134 transects, and in 2005 there were 17 clusters totaling 106 transects. The changes in sampling effort corresponded to different questions being asked each year. Cluster sites were randomly stratified between four subdivisions of the Coachella Valley aeolian sand community: active sand dunes, sand hummocks, ephemeral sand fields, and mesquite dunes. Round-tailed ground squirrels occurred in each of these habitat subdivisions each year, although their relative abundance varied in each habitat type and from year to year. Our primary questions with regard to the ground squirrels were:

1. Whether our sampling protocol provided a reliable, repeatable indication of the squirrels' range, occupancy and relative abundance

2. What were the habitat affinities for this species?

3. What are potential stressors? Do habitat edges have an impact on the squirrels' distribution and abundance?

Survey Protocols and Detection Issues

There are at least four potential survey methods that could be potentially employed to count RTGS: mark-recapture live-trapping, sighting counts, vocalization counts, and track detection. Of the four methods, mark-recapture is generally believed to yield the most accurate population estimates, but it is also the most resource expensive in terms of field-person hours / datum. One of our overall objectives here was to evaluate the utility of a community approach to multiple species monitoring. As such, within the resources available to us, a mark-recapture method was not evaluated.

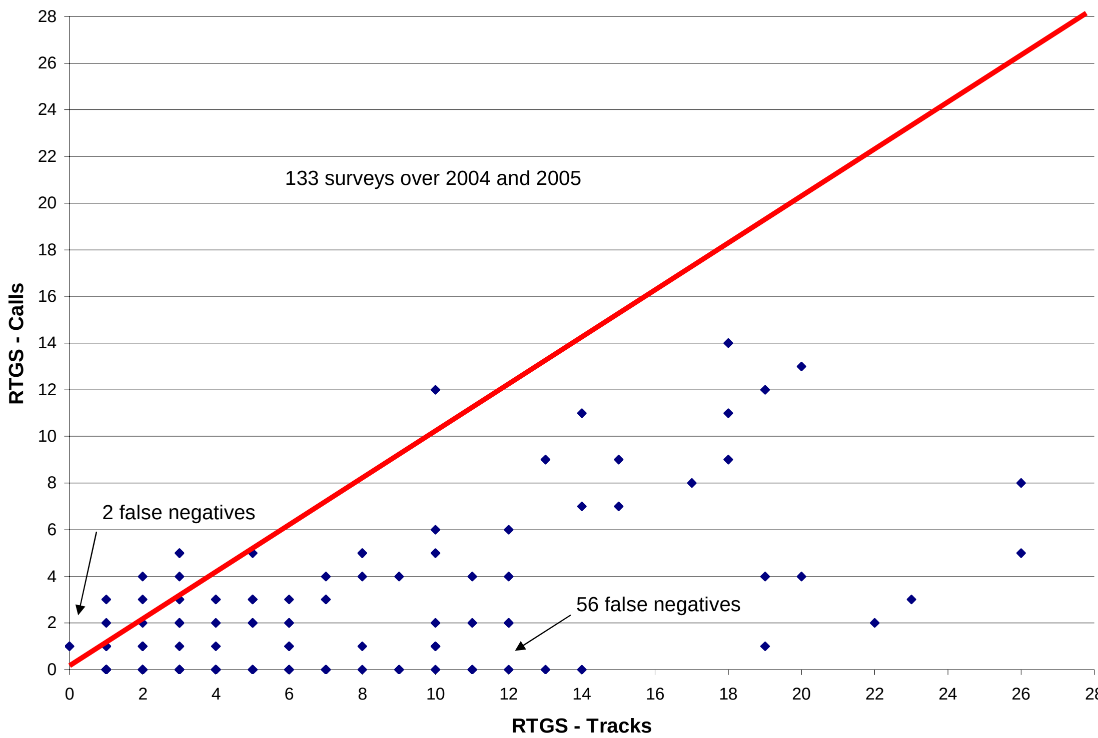
Detection is a critical constraint in any survey protocol. Detection rates can be related to how a species' crypsis varies between habitats, temporally within habitats due to changes in resources, changes in season and activity rates, time of day, reproductive behavior, and density. Lack of detection can also be related to a mismatch between the home range size of the organism and the size of the sample plot. If the home range exceeds the sample plot, then the species has a certain probability of being outside the count area during the survey. Additionally there are detection components related to skills of the surveyors. Without controlling for all of these differences in detection rates, comparing survey data between surveyors, between habitats, etc., can yield spurious conclusions as to the true differences in the abundance or occurrence of a species. Mathematical approaches to resolving this conundrum depend on partitioning data sets to obtain a constant detection probability within each set. Otherwise, the assumption that detection rates are constant across all surveys is rarely rigorously tested, and often likely unmet. The most effective test is to conduct parallel sampling using mark-recapture and an alternative approach simultaneously in time and space. The time and resource constraints indicated above usually eliminate this as an option. RTGS are vigilant and evasive, and so detection issues are critical to address here. However, since we did not employ mark-recapture methodology, we evaluated which alternative method (sighting, calls, and tracks) yields the highest and most consistent detections.

RTGS sightings were exceedingly rare compared to other survey methods, and were not considered a reliable survey metric. Therefore, the following discussion focuses on the relative merits of using calls versus tracking as a survey technique. The basic survey unit was a 10 m x 100 m belt transect. Surveyors walked each transect recording any diagnostic squirrel tracks encountered as well as listening for the sharp pitched, single note warning calls that the squirrels often emit at the approach of danger. Each transect was surveyed six times within a 4-6 week period between late May early July to provide repeated samples and sampling variance measures. Transects were clustered within a 2.5 ha rectangle, termed a "cluster," in order to provide relatively consistent habitat on each transect. Clusters were located in a stratified random manner (stratified by habitat type and condition) in order to provide replications and variances associated with that habitat type.

Figure 4.3.1 summarizes RTGS survey results for 2004 and 2005, comparing total number of calls versus tracks per survey, per cluster. The diagonal red line represents that point when the number of calling RTGS equaled the number of RTGS tracks detected. Points below or to the right of the line indicate more RTGS were detected by tracks than calls, points above the line indicate more RTGS were detected by calls. Other than the obvious greater number of track detections, an important result presented here was the number of false negatives, or surveys in which RTGS were detected with one method but not the other, potentially giving the false impression that the RTGS were absent when in fact they were not. Using calling data alone, surveys would have falsely concluded that RTGS were absent 42% of the time, whereas using tracks alone false negatives occurred 1.5 % of the time.

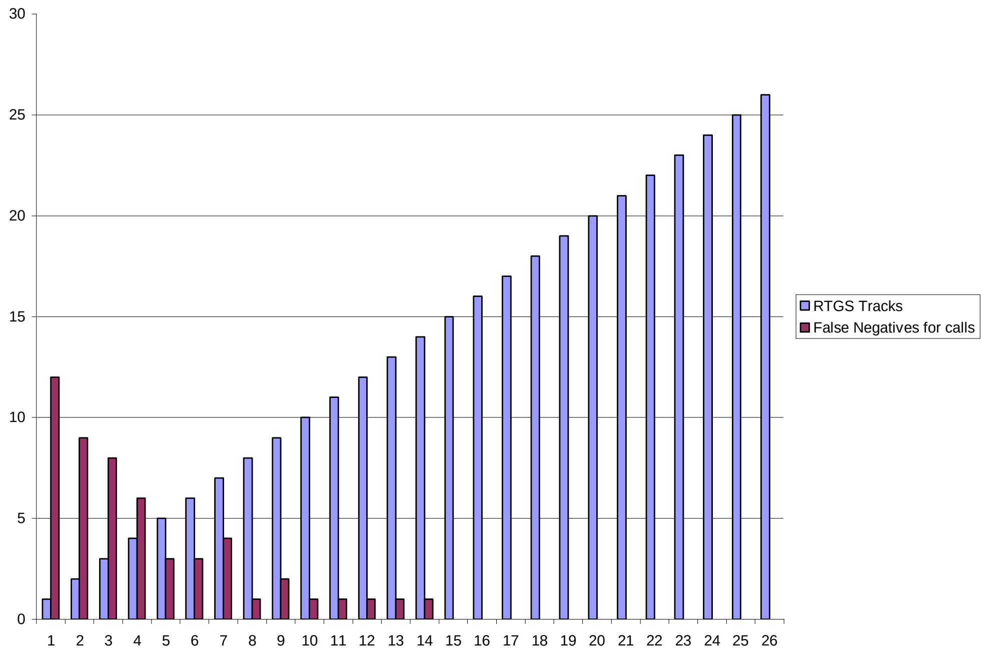
Figure 4.3.2 explores this relationship further. The likelihood of a false negative for the occurrence of RTGS increases at what are apparent low RTGS densities (based on track counts). The "Y" axis is a simple count of tracks per cluster and the number of times there were coincident false negatives from call surveys at those track densities. As RTGS densities decline, call rates decline at an increasing rate. This observed pattern may be interpreted from the standpoint of the function of their warning call. At higher densities the warning call may be used to alert genetically related individuals, and simultaneous calling could serve to confuse a predator. However, at low densities few if any kin would hear the warning, and the call would only serve to focus a predator's attention to the vocalizing squirrel. Using calling as a survey tool, detection probabilities are not constant, and so there will be an under estimate of RTGS occurrence or abundance at those low densities, potentially resulting in unreliable population estimates and spurious habitat correlations.

Figure 4.3.1 Comparison of tracking surveys versus surveys that relied only on counts of the squirrels' alarm calls. The diagonal red line represents surveys where alarm call counts = track counts. Data above that line indicate surveys where alarm calls > tracks, below the line alarm calls < tracks.



Tracking provides higher detection rates than either calling or sighting surveys for RTGS. Whether it represents an accurate population estimate is unknown without mark-recapture data from the same locations. Tracking is unreliable on substrates that do not allow diagnostic track identification; however at all of our sampling locations in aeolian sand habitats that was not a constraint. Given a substrate that leaves identifiable tracks, as long as the RTGS are active they will leave tracks, regardless of absolute densities. The 10 m x 100 m transects are small enough that tracks that are present would not likely be missed by surveyors. The metric resulting from tracking appears to provide a measure of relative abundance for RTGS without biases or errors associated with density (RTGS), vegetation cover, or time of day. As long as season is controlled for in the survey protocols, that would also not be a source of error.

Figure 4.3.2 Number of false negatives, surveys in which squirrels were not recorded giving alarm calls even though tracks were present. The number of false negative surveys (using alarm calls) is reduced with increased squirrel density. Calling rates appear to be related to squirrel density.



Relative Abundance by Year and by Habitat

In Figure 4.3.3, active sand dunes are represented by blue circles, sand hummocks (eastern Coachella Valley) by red squares, ephemeral sand fields (western Coachella Valley) by green triangles, and mesquite dunes by orange diamonds.

Clearly RTGS are found at highest densities in mesquite dunes. However the temporal dynamics of each habitat type may reveal more about the habitat needs of this species. Between 1999 and 2003, the Coachella Valley experienced a severe drought. Rainfall in 2004 was near the annual average and rainfall in 2005 was nearly three times the annual average. From 2003 to 2005 there was a decline in RTGS at each of the mesquite dune sites. During 2004 to 2005 there was an increase in RTGS relative abundance on three out of four sites for both the active dunes and sand hummocks. This increase occurred despite an infestation of Saharan mustard, *Brassica tournefortii*, on the sand hummock sites. There appeared to be a reduced association with mesquite with a coincident increase in association with other habitat features. A pair-wise Pearson Correlation matrix comparing habitat features with RTGS abundance in 2003 through 2005 reveals that same pattern.

Figure 4.3.3 Apparent abundance of RTGS varied by habitat type, and from year to year. Error bars indicate one standard error.

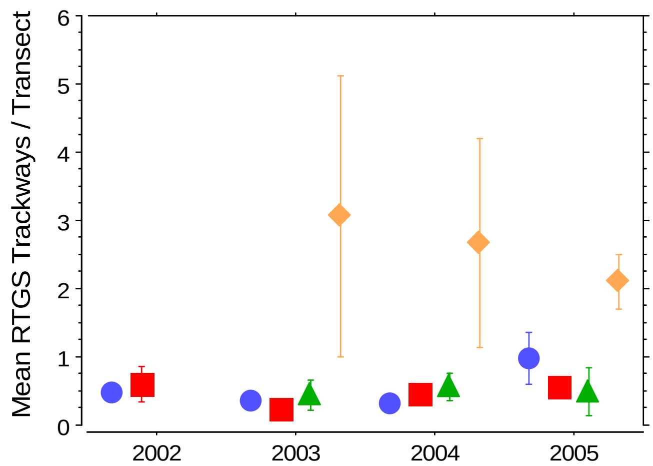


Table 4.3.1 Pearson's correlation values for round-tailed ground squirrel relative abundance and four habitat variables.

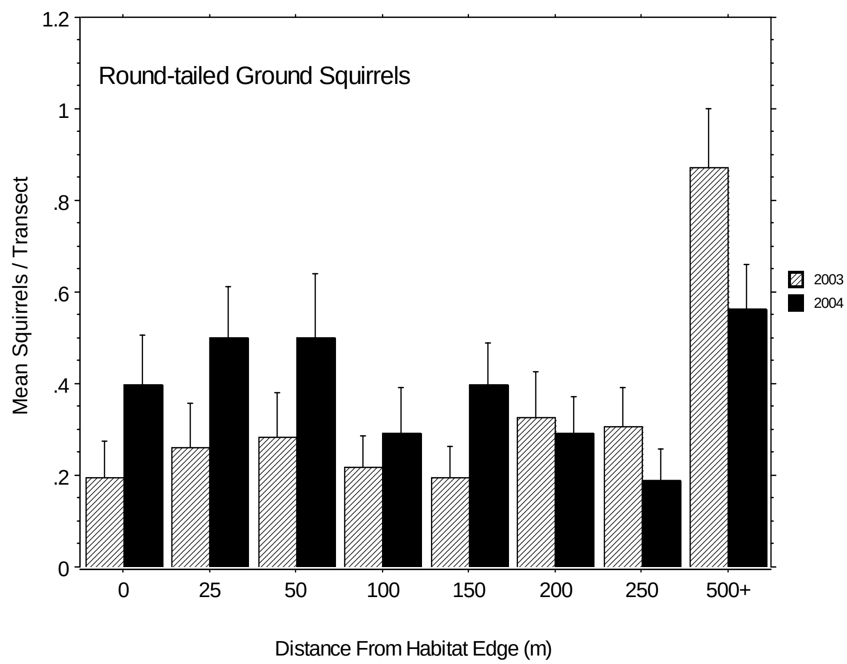
Habitat Variable	2003	2004	2005
Sand Compaction	-0.119	-0.217	-0.539
Creosote Density	0.319	0.299	0.366
Saltbush Density	0.102	0.241	0.338
Mesquite Density	0.787	0.753	0.467

While the mesquite dunes are important habitats for the RTGS, the importance of the other aeolian sand habitats should not be overlooked (Table 4.3.1). Mesquite hummocks occupy just 1/20 or less of the aeolian sand landscape in the Coachella Valley. While the UCR data indicate that the squirrels are 2-10 times more abundant in the mesquite, the squirrels' population size outside the mesquite areas is at least twice that of the population inside the mesquite. The majority of the squirrels are occupying non-mesquite habitat, albeit at lower densities. The fact that squirrels occupied all survey plots indicates a general distribution of the squirrels across a variety of habitat conditions. The lack of synchrony between RTGS populations in mesquite and in other aeolian sand habitats may bode well for their long-term survival. One interpretation of these data may have RTGS using the mesquite as refuges during severe droughts, and then expanding more into other habitats during wetter conditions. The landscape scale patterns of habitat patches within the aeolian habitats of the Coachella Valley appear to provide the RTGS the ability to shift their patterns of abundance as they track available resources.

Population Stressors

The current patterns of RTGS population abundance may be impacted by future fragmentation and encroachment as conversions to suburban landscapes occur along the conserved habitat boundaries. This hypothesis was tested at the Thousand Palms Preserve where clusters were established along the preserve edge and in core habitats (Figure 4.3.4) Nine clusters were established along the preserve boundary and five clusters were located within the core of the preserve.

Figure 4.3.4 Effect of edge processes on the relative abundance of round-tailed ground squirrels.



No evidence of a boundary effect was detected for the RTGS population here. The higher RTGS numbers at 500+ m from the boundary were the result of mesquite habitats being included in one of the core habitat clusters. An ANOVA test of the impact of the preserve boundary for RTGS yielded a $p = 0.8198$ in 2003 and $p = 0.2276$ in 2004.

4.4 COACHELLA VALLEY GIANT SAND-TREADER CRICKET, *MACROBAENETES VALGUM*

Cameron W. Barrows

The Coachella Valley giant sand-treader cricket (CVGSTC) is endemic to the aeolian sand habitats of the Coachella Valley. This is an "annual species"; it completes its life cycle in a 4-6 month period, from mid-winter through mid-spring to early summer. CVGSTC were surveyed in 2004 and 2005. In 2003 the drought conditions reduced this species' active period to less than three months and then only in the western most portions of the Coachella Valley. In 2004 surveys occurred on 22 sites, or clusters, including 134 100 m x 10 m belt transects; and in 2005 there were 17 clusters totaling 106 transects. Cluster sites were stratified between four subdivisions of the Coachella Valley aeolian sand community: active sand dunes, sand hummocks, ephemeral sand fields, and mesquite dunes. CVGSTC occurred in each of these subdivisions each year, although their relative abundance varied in each habitat type and varied from year to year. Our primary questions with regard to the CVGSTC are:

1. Does the sampling protocol provide a reliable, repeatable indication of the CVGSTC range, occupancy and relative abundance

2. What are the habitat affinities for this species.

<u>Survey Protocols and Detection Issues</u>

CVGSTC are nocturnal insects that quickly die if exposed to warm sun and low humidity. Dry, unbaited pitfall trapping works well to sample the presence of this species in areas of high population density, but in areas of lower density, false negatives (no CVGSTC in the pitfalls but their presence was confirmed with other methods) were common.

CVGSTC excavate one or more burrows each night, selecting one to spend the diurnal hours in. Their excavations result in a diagnostic triangular or delta-shaped debris pile outside their burrow that can be 20 cm or more long and 20 cm or more wide at the distal "triangle base." Surveys involved walking each 10 m x 100 m belt transect early to mid morning and counting each aggregation of the diagnostic CVGSTC excavations. It appeared that a single CVGSTC could excavate more than hole per night, and if it had not been especially windy over the past 24 hrs, a cluster of excavations could occur in a roughly 3 m x 3 m area. Attempts to determine how many of these excavations were occupied invariably yielded a single CVGSC per aggregation.

Relative Abundance by Year and by Habitat

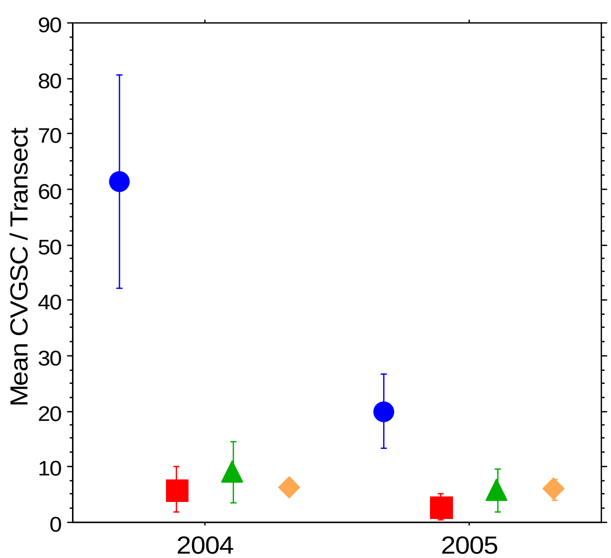
Figure 4.4.1 depicts the relative abundance of CVGSC in 2004 and 2005, by habitat. Blue circles represent active dune sites, red squares indicate sand hummocks, green triangles are ephemeral sand fields, and orange diamonds represent mesquite dunes. Active dunes were clearly the preferred habitat for CVGSC. The apparent population decline between 2004 and 2005 is curious, as CVGSC otherwise seem to only emerge to complete their life cycle during the cool wet season, and 2005 was cooler and wetter than 2004. The declines were measured at virtually every cluster, independent of habitat type, east-west location, or the presence of exotic weed species. Two hypotheses are available to explain this pattern and include:

1. The CVGSC reached its upper tolerance for dealing with water saturated sands. There is no *a priori* reason to predict that a positive relationship between CVGSC abundance and rainfall is linear.

2. The wetter conditions of 2005 temporally shifted the CVGSC' life cycle so that sampling at the same time as sampling was conducted the previous year may have hit different activity levels. The CVGSC were the same size during each years' survey, so this doesn't appear likely, but it is still possible

Additional years of sampling should identify which hypothesis, or an additional hypothesis best explains the observed pattern of CVGSC abundance.

Figure 4.4.1 Giant sand-treader cricket relative abundance separated by year and by community type. Blue circles represent active dunes, red squares represent sand fields/hummocks, green triangles represent ephemeral sand fields, and orange diamonds represent mesquite dunes. Error bars indicate one standard error.



The habitat affinities of CVGSC closely parallel those for the Coachella Valley fringe-toed lizard.

Table 4.4.1 Pearson pairwise correlations for CVGSC and their habitat.

Habitat Variable	2004	2005
Sand Compaction	-0.464	-0.310
Shrub Density	0.018	-0.170
% Cover Annual Plants	-0.119	-0.143
% Cover *Schismus Barbatus*	-0.099	-0.364
% Cover *Salsola tragus*	0.168	0.100

Loose, non-stabilized sand and sparse plant cover characterize CVGSC habitat.

4.5 COACHELLA VALLEY JERUSALEM CRICKET, *STENOPELMATUS CAHUILAENSIS*

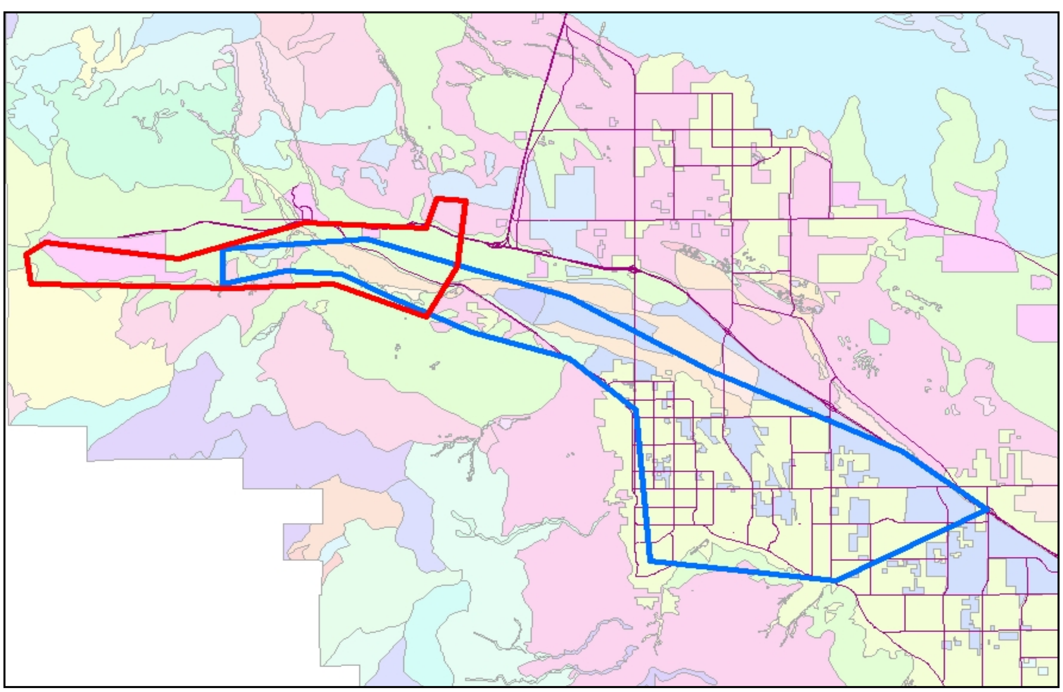
Cameron W. Barrows

Coachella Valley Jerusalem crickets (CVJC) are endemic to the far western portions of the Coachella Valley. Our survey objectives were to define their distribution and to develop an adequate sampling protocol.

Sampling Protocols and Detection Issues

In 2003 efforts were focused on defining the current range of CVJC. Sampling included pitfall trapping [dry, unbaited and baited (oatmeal)], searching under debris, and the placement of 2' x 2' tiles or cover boards. CVJC were located with all three methods. The sampling frame included all historic locations, as well as likely habitat outside those historic known locations. Surveys occurred between January and April, 2003. Based on those surveys a new range for CVJC was defined (Figure 4.5.1). The blue polygon in Figure 4.5.1 encompasses all historic points, whereas the red polygon defines the CVJC range based on our 2003 sampling effort.

Figure 4.5.1 Minimum convex polygons of Jerusalem cricket locations prior to the 2003 CCB surveys (blue), and for the CCB surveys period (2003-2005) (red).



CVJC are difficult to detect. They are nocturnal and fossorial, apparently coming to the sand surface only to forage, and presumably mate, when the soil moisture is high near the soil surface in the winter and early spring. We were able to extend their activity season by augmenting the soil moisture below tiles, supporting the assumption that CVJC are soil moisture limited. Aeolian sands are relatively easy to burrow in and are able to hold soil moisture longer than coarser alluvial sands, which may explain the species apparent association with areas of fine sands.

CVJC sightings were positively associated with waste debris left by humans. To evaluate whether the observed CVJC distribution was artificially skewed toward areas of high debris because there were more places to search for them there, in 2004 we placed arrays of standardized debris consisting of 2' x 2' tiles across our existing transect clusters. Four clusters were selected inside the defined 2003 distribution in the Snow Creek – Windy Points area (18 tiles/cluster for a total of 72 tiles), and three clusters outside the 2003 distribution, but inside the CVJC historic range, near Indian Avenue (a total of 54 tiles). Precipitation in 2003 was below average, whereas the 2004 rainfall was near average; we wanted to determine whether the added rainfall would result in an eastward extension of their observed distribution. In 2005 there was nearly three times average rainfall, and so the tiles were surveyed for CVJC to again see whether there was an eastward extension associated with additional soil moisture.

Occurrence and Relative Abundance by Habitat

In both 2004 and 2005 no CVCJ were located outside the described 2003 distribution. In both years all sightings were on the four Snow Creek – Windy Point clusters. The distribution of sightings by habitat is shown below (Table 4.5.1).

Table 4.5.1 Summary data for Jerusalem cricket surveys in 2004 and 2005.

Habitat Type	2004 (4 visits)	2005 (1 visit)
Active Dune	1	0
Ephemeral Sand Field	10	3

The ephemeral sand fields have a much higher plant density and plant species richness than the active dunes. The active dunes also received a much higher rate of ORV trespass. Determining whether these factors play a roll in the abundances of CVJC will require additional sampling.

The distribution of CVJC described in 2003 appears to be an accurate representation of their current range. No explanations are available for the substantial reduction in their distribution. Perhaps it is an effect of the long-term drought conditions experienced by this region. The "new", westward extension of the CVJC range may not be new at all, as there is no record that those areas were historically searched for this species.

Placing cover boards or tiles appears to be an effective method for determining the occurrence of this species.

4.6 COACHELLA VALLEY MILKVETCH, *ASTRAGALUS LENTIGINOSUS* VAR *COACHELLAE*

Cameron W. Barrows

The Coachella Valley milkvetch (CVMV) is an annual-short lived perennial plant species endemic to the aeolian sand fields of the Coachella Valley. CVMV were surveyed in the years 2003-2005 on plots distributed on conservation lands throughout the remaining aeolian sand community of the Coachella Valley. In 2003 that effort included 18 sites (clusters) with a total of 116 100 m x 10 m belt transects; in 2004 there were 22 clusters including 134 transects, and in 2005 there were 24 clusters totaling 146 transects. Cluster sites were stratified between four subdivisions of the Coachella Valley aeolian sand community: active sand dunes, sand hummocks, ephemeral sand fields, and mesquite dunes. CVMV occurred in each of these subdivisions each year, although their relative abundance varied in each habitat type and varied from year to year. Our primary questions with regard to CVMV were whether our sampling protocol provided a reliable, repeatable indication of the plants' range and relative abundance, what were the habitat affinities for this species, and a determination of potential stressors impacting the distribution and abundance of CVMV.

Survey Protocols

Our basic sampling unit was a 10 m x 100 m belt transect. The belt transects were small enough so that two surveyors walking slowly across the transects could count all CVMV growing there. Five to seven transects were clustered together on a roughly 2.5 ha site, termed a "cluster", with consistent habitat characteristics. Data analyses could be scaled from transects, to clusters, to habitat types, to larger landscape units depending on the "grain" of analyses required. Each transect was surveyed once annually during February.

Relative Abundance by Habitat by Year

CVMV abundance from 2003-2005 is depicted in Figure 4.6.1. Blue circles represent active sand dunes, orange squares indicate sand hummocks, green triangles ephemeral sand fields, and orange diamonds depict data from mesquite dunes.

This summary is dominated by one site, an ephemeral sand field near Windy Point. That site had nearly 200 CVMV per transect, an order of magnitude more than any other measured location. Removing that one cluster from the analysis reveals a much clearer understanding of the abundance and habitat affinities of CVMV across the Coachella Valley (Figure 4.6.2).

Figure 4.6.1 Relative abundance of CVMV 2003-2005, separated by community type. Blue circles represent active dunes, red squares represent sand fields/hummocks, green triangles represent ephemeral sand fields, and orange diamonds represent mesquite dunes. . Error bars indicate one standard error.

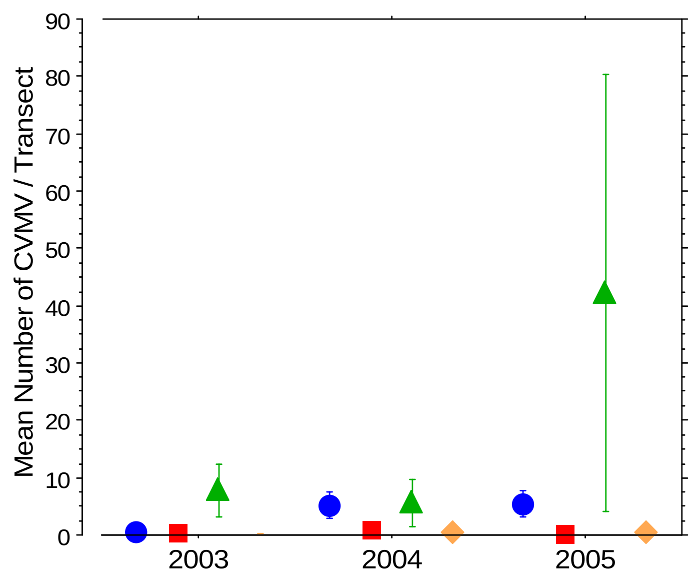
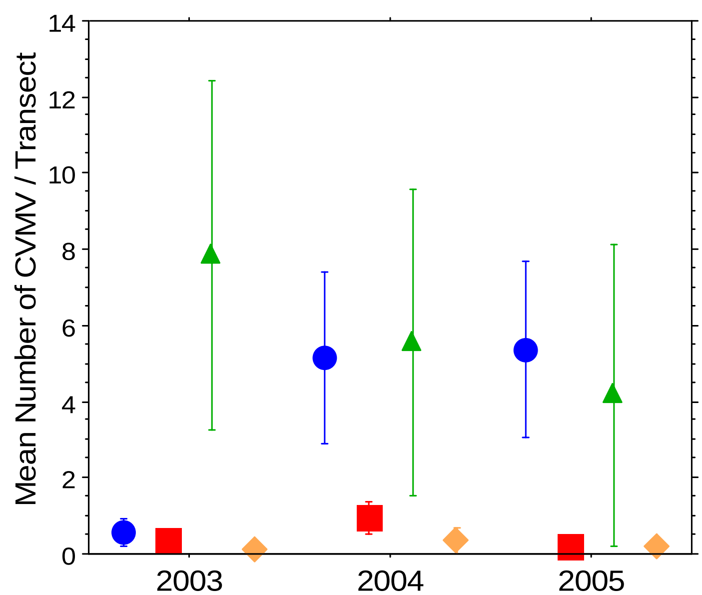


Figure 4.6.2 Relative abundance of CVMV 2003-2005, separated by community type. One transect cluster was removed from the analysis due to its outlier status. Blue circles represent active dunes, red squares represent sand fields/hummocks, green triangles represent ephemeral sand fields, and orange diamonds represent mesquite dunes. Error bars indicate one standard error.



Active sand dunes and ephemeral sand fields are the primary habitat for CVMV. CVMV seeds need to be scarified in order to promote germination, a process most readily accomplished here by wind-born sand abrasion. The ephemeral sand fields and active dunes are habitats with high rates of aeolian sand transport, whereas sand hummocks and mesquite dunes have much lower rates of sand movement, and so are less effective in abrading CVMV seeds.

Temporal changes in CVMV on active sand dunes were significant between 2003 and 2004. Rainfall in 2003 in the center of the Coachella Valley where most of the active sand dune clusters occur was 43 mm, in 2004 it was 64 mm, and in 2005 in was nearly 210 mm. In the western Coachella Valley, where ephemeral sand fields occur, rainfall levels were nearly twice those amounts. It is possible that there is a minimum rainfall threshold for CVMV germination between 43 mm and 64 mm. In the western Coachella Valley that threshold was exceeded each year, whereas in the central valley it was exceeded only in 2004 and 2005.

The apparent, though not statistically significant, decline in the mean CVMV abundance in the ephemeral sand fields from 2003 to 2005 may have multiple causes. At one cluster near Snow Creek, higher rainfall resulted in dense exotic plant invasions. At that site mean CVMV per transect went from 23 plants in 2003 to 1 plant in 2005. Weed densities there have eliminated sand transport activity, and so no seed scarification by sand abrasion occurs. At another cluster east of Indian Avenue, fine aeolian sands were largely blown off the site without replacement (epitomizing the ephemeral character of this habitat). At that site mean CVMV abundance went from 25 plants per transect to 16 plants per transect in 2005. This decline may also reflect reduced seed scarification occurring there due to reduced sand transport. Weed abundance on this site was very low. Aeolian sands returned to this cluster by late spring as a result of winter flooding; CVMV surveys in coming years should provide support for or refute this hypothesis.

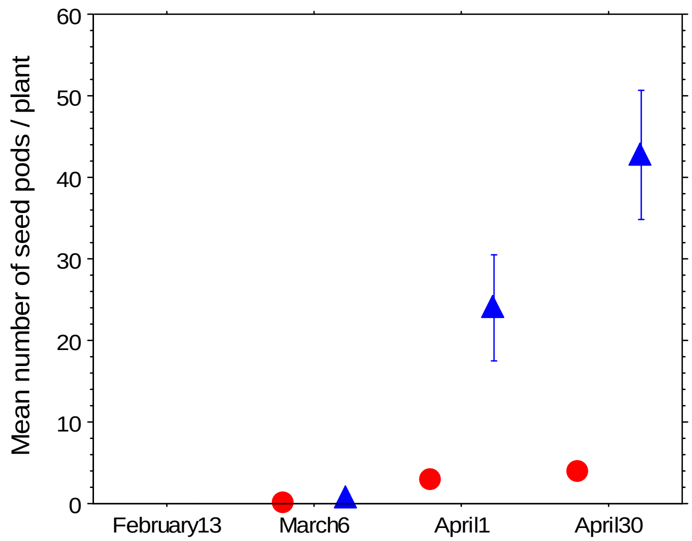
Population Stressors

As mentioned in the previous section, any process that reduces sand transport will likely result in a reduction in CVMV germination. In 2005 there was a severe infestation of Saharan mustard, *Brassica tournefortii*, especially on the sand hummocks of the central Coachella Valley. One potential impact of this infestation is the stabilization of the aeolian sand habitats. Preliminary data support this hypothesis; areas infested by the mustard have higher sand compaction measures than those adjacent weeded sites (ANOVA, $p = 0.004$). In addition, the mustard appears to have a direct impact on CVMV seed production.

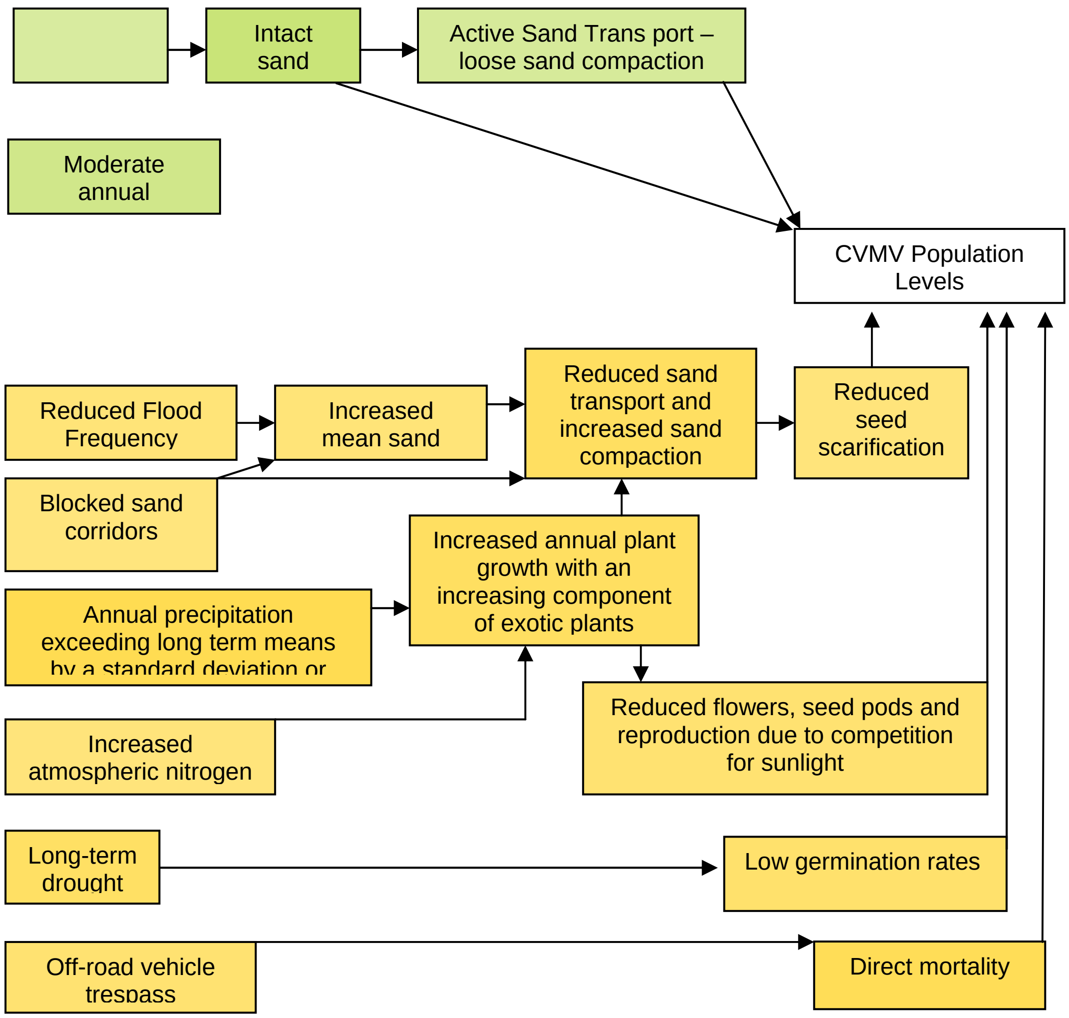
Figure 4.6.3 summarizes the results of a mustard removal experiment associated with a dense stand of both mustard and CVMV. Fruit (seed) production on plants released from mustard competition was substantially higher. Red circles represent the control, or non-weeded plot, whereas blue triangles represent the CVMV pod counts on the weeded plot. If the mustard becomes a persistent component of the aeolian habitats of the Coachella Valley, at the densities seen in 2005, then CVMV will likely be eliminated from sand hummock habitats.

Fortunately, for now the mustard appears less able to invade the active dune and ephemeral sand field habitats.

Figure 4.6.3 Results of a weeding experiment in which seed pods of CVMV were counted on a plot where Saharan mustard was removed (blue diamonds) and the control plot where no weeding occurred (red circles) Error bars indicate one standard error.



CVMV Envirogram (green = positive growth, yellow = negative growth)



4.7 LECONTE'S THRASHER, *TOXOSTOMA LECONTEI*

Darrell Hutchinson

LeConte's Thrasher is an uncommon resident of the deserts of the American Southwest and northwestern Mexico. It inhabits desolate environments, rarely drinks, and feeds on arthropods dug up from the litter. Dependent on desert vegetation, it has lost extensive habitat to development (Sheppard 1996) and is a designated species of concern by California Department of Fish and Game.

In the Mojave Desert, multiple pairs of LeConte's Thrasher occupy adjacent territories and vocalize regularly throughout the breeding season (Barrows pers.com.). Near Maricopa, California, home ranges vary between 3.5-18 ha pair (Sheppard 1970). In the most populated habitats, densities may reach 10 adults per square kilometer (Sheppard 1996). In the Coachella Valley, LeConte's Thrasher appears to be rare and widely dispersed. LeConte's seem to occupy disjunct territories with home ranges as large as 100 ha. Living at low density, it vocalizes less because intensive territorial defense is unnecessary.

Our objective for monitoring LeConte's thrashers was to develop a methodology to enable us to survey for presence of LeConte's Thrasher in the Coachella Valley.

<u>Survey Protocols</u>

Point counts combined with visual surveys are a common method of detecting presence of target bird species. LeConte's Thrasher's plain sandy coloration blends well with the dry vegetation and ground cover. It normally moves by running between shrubs and tends to forage underneath them. In the absence of vocalizations, its presence is difficult to detect.

We employed the use of an audio playback technique to increase detection. The use of audio playbacks is a standard technique whereby observers broadcast vocalizations of a target species into areas of suitable habitat, in order to stimulate a response from occupying individuals. Birds may respond by vocalizing and/or approaching the broadcast location, thus increasing their conspicuousness and detectability.

The CCB acquired an analog field recording of a singing male LeConte's Thrasher near Desert Hot Springs, then digitized and filtered the recording to create a 90 second sound file. The broadcast survey protocol was developed using this recording of a local dialect. By employing a broadcast survey technique, researchers were able to determine the optimum seasonal and daily timing of two survey methods, gather data about local distribution, and compile ecological information to integrate into sand community projects.

Site selection was determined by evaluating historical occurrences, niche model projections, recent sightings, habitat preference and public access. Transects were generally set through the middle of an expansive desert wash away from roads and buildings, although some transects were set along roads with right-of-way access.

Two different survey techniques were evaluated. For the **Broadcast Survey**, one kilometer long transects were established with three broadcast points located 500m apart at the ends and midpoint. At each point, the 90-second song recording was broadcast in opposite directions perpendicular to the transect for a total of three minutes. After each three minute broadcast, a four minute detection period ensued where researchers scanned with binoculars and listened for a vocal response. If no response was detected, then the three minute broadcast and four minute detection period were repeated two more times at ascending playback volumes in order to increase the projection distance without overwhelming nearby birds. If a LeConte's Thrasher was detected anytime during the broadcast or detection periods, playbacks were stopped, data were collected, and the researchers moved to the next point on the transect. If, during data collection, any breeding behaviors were observed, then the survey was terminated. If winds consistently exceeded a 20 km/hr maximum or 15 km/hr average, the survey was considered invalid.

Data collected on LeConte's Thrasher included time and place of detection, time after playback of response, volume of playback, method of initial detection (vocal or visual), distance of initial detection and eventual approach (if any), type and duration of vocalizations, and all other information regarding foraging, vocalizing, nesting, and inter and intra specific behaviors.

To better evaluate the Broadcast Survey, an alternative method of searching for LeConte's Thrasher was developed. The Walking Transect required researchers to walk the one kilometer long transects using only vocal and visual cues to detect LeConte's Thrasher (no broadcast recordings were employed). Data were collected on time, location, distance from transect and method of detection for all LeConte's Thrashers observed. Typically, researchers used the Walking Transect method to reach the endpoint of a transect, then conducted the Broadcast Survey on their return.

<u>Results</u>

During the Broadcast Surveys, playbacks were broadcast at three ascending volumes. Most LeConte's Thrasher detections occurred in response to the loudest volume (11/20 detections), which is expected since a louder projection samples a greater area. However, several detections occurred in response to the lowest volume (6/20 detections). Broadcasting at low volume serves to attract nearby birds without presenting too forceful a rival. A possibly unmated subordinate male was observed singing spontaneously during a Walking Transect survey but not detected later during the Broadcast Survey, suggesting young adults may be cowed by the introduction of simulated rivals.

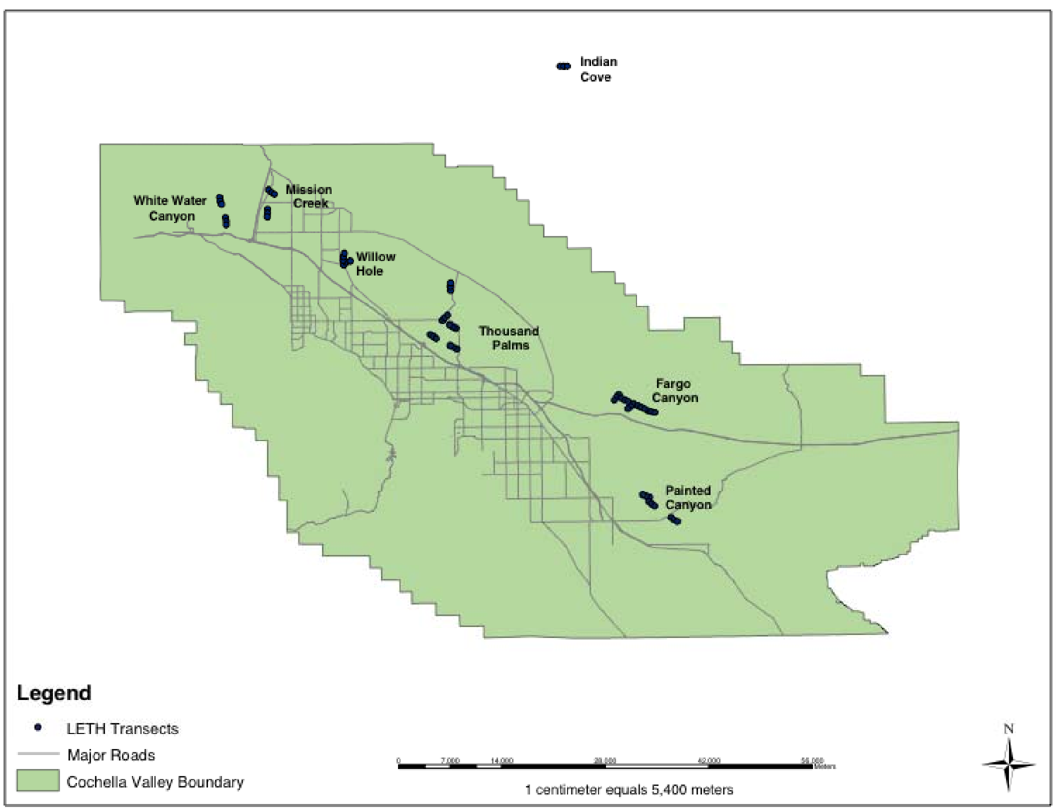
A four minute silent detection period occurs after each playback. The range of response times was –1:00 – 10:00 minutes (mean = 2:07 ± 2.3 minutes, n = 20). A negative indicates birds were detected before the end of the playback (2/20 detections). Most birds were detected within the first two minutes after the end of the playback (12/20 detections) although four were detected during the final two minutes after the end of the playback, indicating a four minute waiting period is appropriate.

The range of distances at which a bird was initially detected was 22 – 275 meters (mean = 120 ±65 m, n = 15). Most birds were first heard and then seen, however five detections occurred where LeConte's Thrasher was heard, positively identified, but not seen.

The total amount of time birds were heard to vocalize ranged between 4 – 300 seconds (mean = 88 ± 90 sec., n = 18). Short vocalizations generally consisted of a single or series of call notes while longer vocalizations contained one or more extended song phrases. Two detections occurred on silent approaches where no vocalizations were heard.

<u>Survey Results</u>

Figure 4.7.1 Location of transects in the Coachella Valley. Indian Cove is located outside the border of Joshua Tree N.P.



During 2004, 20 separate transects were established and surveyed [see Table 4.7.1]. Each transect was surveyed two times. LeConte's Thrasher was detected on four transects - two at Mission Creek, one at Thousand Palms Preserve, and one at Indian Cove; two of which were previously discovered during preliminary investigations. In order to evaluate the effectiveness of the Walking Transect and Broadcast Survey methods, subsequent surveys were conducted only on those four transects known to have LeConte's Thrasher present.

During 2005, eight transects were surveyed: two at Mission Creek, two at Willow Hole, and four at Thousand Palms Preserve [see Table 4.7.2]. Each transect was surveyed twice. LeConte's Thrasher was detected on four transects – two at Mission Creek and two at Thousand Palms Preserve; three of which were previously known from 2004.

Table 4.7.1 Survey Results for 2004.

	# Surveys	# Surveys on transects w/LETH known to occur	# Surveys where LETH detected	Total # Individuals detected
Broadcast survey	46	14	9	15
Walking transect	41	14	5	11

Table 4.7.2 Survey Results for 2005

	# Surveys	# Surveys on transects w/LETH known to occur	# Surveys where LETH detected	Total # Individuals detected
Broadcast survey	17	9	6	13
Walking transect	18	10	0	0

Rates at which LeConte's Thrasher were successfully detected on transects where LeConte's were known to occur are presented in Table 4.7.3. Success rates for both methods in both years are compared. Since some surveys are compromised by wind, high temperatures or unsettled weather, these surveys can be invalidated. Selecting for "Favorable" conditions provides a more accurate measure of how effective each methodology is at detecting LeConte's Thrasher. Including all completed surveys conducted under "Routine" conditions presents a more accurate measure of the sampling effort required.

Table 4.7.3 Success Rates of detection on transects where LeConte's Thrasher is known to occur.

	2004	2005
Under routine sampling conditions (Sampling effort)	Walking Transect 5/14 = 36% Success Rate	Walking Transect 0/10 = 0% Success Rate
	Broadcast Survey 9/14 = 64% Success Rate	Broadcast Survey 6/9 = 67% Success Rate
Under favorable conditions (Effective detector)	Walking Transect 5/12 = 42% Success Rate	Walking Transect 0/8 = 0% Success Rate
	Broadcast Survey 9/12 = 75% Success Rate	Broadcast Survey 6/7 = 68% Success Rate

Discussion

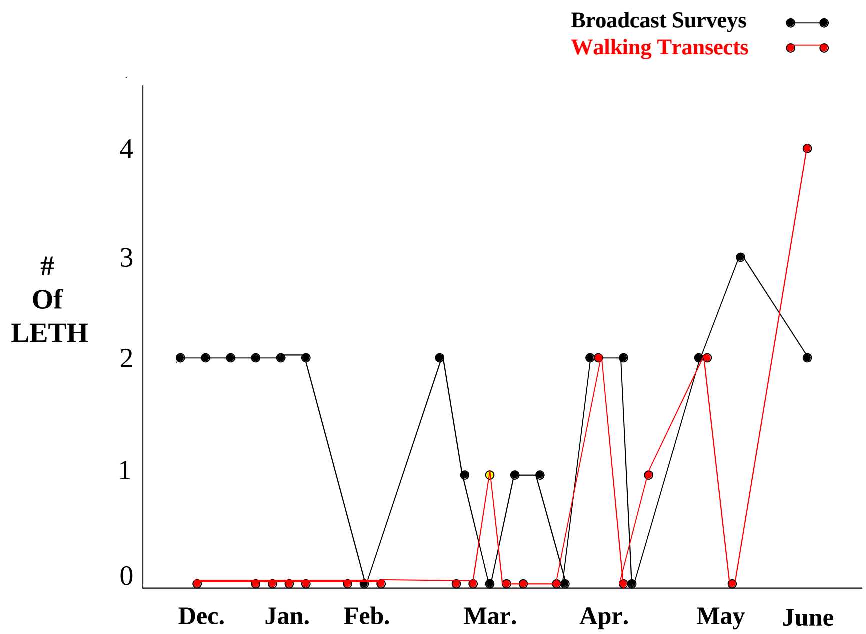
Results indicate that the Broadcast Survey method is more effective at detecting LeConte's Thrasher than the Walking Transect method. One feature of the audio playback is that it prompts LeConte's Thrasher to come up off the ground, where it normally forages, to sit on top of a bush where it can be more easily spotted. The playbacks also stimulate LeConte's Thrasher to respond vocally and approach. Also, by broadcasting across an estimated area of 1000m under favorable conditions, the broadcast survey method effectively increases the sampling area beyond which you could see or hear by just walking through suitable habitat.

Although the Broadcast Survey is most effective, several LeConte's Thrashers were detected during 2004 using the Walking Transect method. The discrepancy between the success rate of the Walking Transect method between 2004 and 2005 can be explained by the seasonal timing of surveys [see Figure 4.7.2].

Walking Transect surveys conducted in April, May and June were much more likely to detect LeConte's Thrasher (4/7 surveys) than surveys conducted earlier in the season (1/12 surveys). By late Spring, LeConte's Thrasher pairs are likely to have fledged one or two broods and be traveling in family groups. The overall increase in numbers and amount of intra specific vocalizing makes them more conspicuous and easier to detect.

The effectiveness of the Broadcast Survey method also varies seasonally. During late December and January (and perhaps earlier), LeConte's Thrashers are engaged in pair bonding behaviors, including territorial defense and nest building. During these months,

Figure 4.7.2 Seasonal Response to Surveys



they appear to be particularly responsive to audio playbacks (6/6 surveys), which simulate the presence of a rival male in their territory. Usually both members of the pair are detected (6/6 surveys) and time vocalizing is extended (mean = 152 ± 0.07 sec., n = 8) as the male sings his primary song. During the nesting period of February, March and April, when adults are occupied with incubation and provisioning, detection rates are more sporadic (6/10 surveys), the detection of only one adult is more likely (3/6 surveys), and time vocalizing is reduced (mean = 45 ± 0.023 sec., n = 7). During the post-breeding season of May and June, Broadcast Surveys remain effective (3/3 surveys) as adults with fledglings are detected, but vocal responses are short (mean = 12 ± 0.01 sec., n = 4) and consist mostly of call notes.

Table 4.7.4 Daily Timing of Surveys.

Time of Day	# of Detections/Surveys	
	Walking Transect	Broadcast Survey
1st Period (7 – 10:00 AM)	3/9	7/9
2nd Period (10 – 12:00 AM)	3/8	5/9
3rd Period (12 – 3:00 PM)	0/3	3/3

Walking Transect Surveys are best conducted in the morning when birds are most active. Broadcast Surveys can be conducted any time of day. Especially during early breeding season

(Dec.–Feb.), LeConte's Thrasher appears to respond to playbacks even during warm afternoons (Table 4.7.4).

Early breeding and post breeding seasons are preferred times to conduct LeConte's Thrasher broadcast surveys. During early breeding, adult pairs are most likely to approach together and vocalize extended song phrases. During post breeding (May-July), family groups are active and more conspicuous although less responsive to playbacks. Conducting surveys during summer also provides information on reproductive success and post breeding dispersal.

Integration into Sand Community Projects

LeConte's Thrasher surveys can be integrated into the Vertebrate Tracking surveys that are conducted on sand transects across the Coachella Valley during late Spring and Fall. With attention to size and pattern, researchers can reliably distinguish LeConte's Thrasher tracks from similar sized resident species, namely Northern Mockingbird, Western Meadowlark, Loggerhead Shrike, Mourning Dove, and Gambel's Quail. Vocalizations and sightings of birds are also recorded during Vertebrate Tracking surveys, which help determine presence of LeConte's Thrasher near sand transects.

During the Vertebrate Tracking surveys of Fall 2003, a single LeConte's Thrasher track was recorded on the Thousand Palms Preserve. The following Spring 2004, a pair of LeConte's Thrashers successfully bred and fledged at least one brood 3 km north of the sand transects on the Preserve. Four and later three sets of LeConte's Thrasher tracks were recorded during that Spring and Fall respectively. By Winter 2004, the presence of a second pair of LeConte's Thrashers was confirmed on the Preserve.

During this same time period at Willow Hole Preserve, no LeConte's Thrasher tracks were recorded during tracking surveys nor were any detected using thrasher surveys. However by Spring 2005, researchers documented LeConte's Thrasher tracks and vocalizations on sand transects and subsequently confirmed the presence of a pair of LeConte's Thrashers using the audio playback technique.

Overlaying sand transects and thrasher transects allows researchers to compare sampling techniques and corroborate results. Vertebrate Tracking surveys can provide evidence of LeConte's Thrasher, while Broadcast Surveys can confirm their presence. Combining survey methods provides ecological information on LeConte's Thrasher distribution, post breeding dispersal and seasonal habitat use.

4.8 RIPARIAN BIRDS

Lori Hargrove

Riparian habitats have been greatly reduced and degraded in southern California, leading to the decline of many riparian bird species. The Coachella Valley Multiple Species Habitat Conservation Plan (MSHCP) has identified five species of riparian birds as targets for conservation on the basis of their rarity and sensitive status (Table 4.8.1). The MSHCP also identified the Brown-headed Cowbird (*Molothrus ater*) as a management concern because it is a known threat to riparian bird species. Brown-headed Cowbirds lay their eggs in nests of other species ("nest parasitism"), and the host parents expend effort rearing cowbird nestlings, typically to the detriment of their own young (Lowther 1993). Threats to the persistence of riparian birds include: loss of riparian habitat (by vegetation clearing or lowered water levels), degradation of riparian habitat (increased invasive exotics, decreased plant diversity, or decreased insect prey base), disturbance (natural or human), nest parasitism, predation, and disease. Kus et al. (2003) reported presence of invasive exotic plants in 94% of riparian habitats surveyed in southern California, most commonly giant reed (*Arundo donax*) and tamarisk or saltcedar (*Tamarix ramosissima*).

Table 4.8.1 Riparian Bird Species Identified as Target Species in the Coachella Valley MSHCP.

Common Name	Scientific Name	Status
Willow Flycatcher/ Southwestern Willow Flycatcher	*Empidonax traillii/* *Empidonax traillii extimus*	State Endangered/ Federally Endangered
Least Bell's Vireo	*Vireo bellii pusillus*	State Endangered/ Federally Endangered
Yellow Warbler	*Dendroica petechia*	State Species of Special Concern
Yellow-breasted Chat	*Icteria virens*	State Species of Special Concern
Summer Tanager	*Piranga rubra*	State Species of Special Concern
Brown-headed Cowbird	*Molothrus ater*	None (potential threat)

As part of developing the monitoring plan, the CCB conducted surveys in the spring-summer of 2002, 2003, and 2004 for six species of birds that occur in riparian habitats within the Plan area (Table 4.8.1), five of which are of conservation concern, and one of which is considered a conservation threat (Brown-headed Cowbird). To determine associations between target riparian bird species and habitat characteristics, the riparian habitat was assessed for vegetation structure and composition, level of human disturbance, and presence of invasive species. There were five key objectives of the riparian bird surveys:

1. Estimate the distribution and abundance of target species within the Plan area,

2. Assess potential threats,

3. Evaluate monitoring survey protocols,

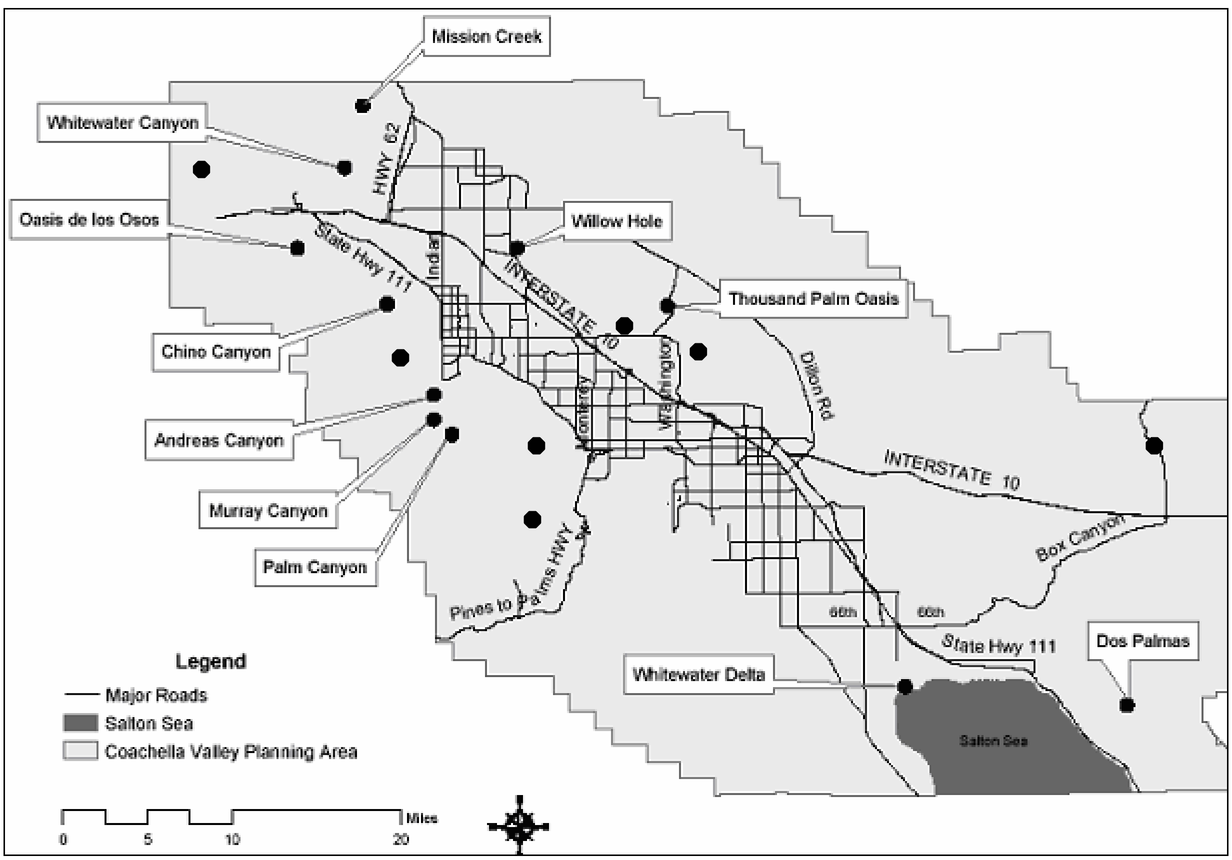
4. Develop predictive habitat models, and

5. Make recommendations for future monitoring and conservation efforts.

Site Selection and Point Establishment

A series of 11 riparian sites and 56 points were initially established and surveyed in 2002. Approximate point count locations were selected in advance by viewing an aerial photo in ArcView (a Geographic Information System program) and selecting points in riparian habitat at roughly 200 m intervals. Riparian habitat was broadly defined to include all riparian

Figure 4.8.1 Locations of CCB Riparian Bird Survey Sites, Spring-Summer 2002-2004.



woodland and shrubs, including mesquite, tamarisk, and palm oases. A reconnaissance visit was conducted to establish actual point count locations, and coordinates were determined using a hand-held GPS unit. Point count locations were positioned linearly in riparian areas that were linear. For patches that could not be penetrated without disturbing the vegetation (e.g., Mission Creek, Chino Canyon), point count locations circumscribed the vegetation. Where bands of riparian vegetation were separated by non-riparian vegetation, distances between points were as great as 700 m (e.g., Dos Palmas, Thousand Palms). In 2003 we expanded the number of sites and points based in part on aerial photos, and in part on riparian bird habitat modeling. Using historical data and results from 2002, a predictive model based on all target species combined was generated to identify additional areas of potentially suitable riparian habitat. Six new sites and 44 new points were established in 2003, and one new site and 16 new points were established in 2004, for a total of 18 sites and 116 points (Figure 4.8.1). The number of points per site varied between one (Willow Hole) and 13 (Whitewater Delta). Sites are under various types of land ownership (Appendix 2.4).

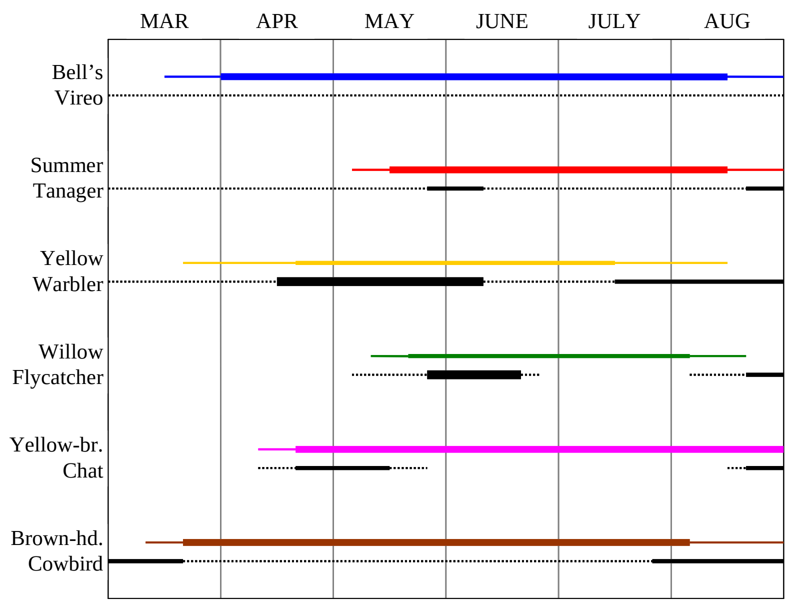
Timing of Surveys

Historical records of breeding and migration (Patten et al. 2003, UCR data, Unitt 1984) were analyzed to determine appropriate survey periods. The breeding populations are often different from migrating populations that breed further north. Among species, while there is some overlap of when the breeding and migrating populations are expected to be present, there is no single ideal time to conduct surveys (Figure 4.8.2). Late May to mid-June has the most amount of overlap, but later surveys are often helpful to distinguish individuals that are potentially breeding from earlier migrating individuals. Thus, to maximize our probability of detection and to ensure sampling of both breeding and migrating populations of all target species, we conducted most surveys between late May and mid-July, with some additional limited sampling earlier in the season, from late April to mid-May. And in 2003 and 2004, most sites received at least two rounds of surveys, the first from late May to mid-June, and the second from late June to mid-July.

Point Count Methods

Point counts were conducted between sunrise and 0900 hours by experienced observers. During point counts, observers stood quietly on or near the location of a point and recorded detections of the target species (identifications by sight or sound) during a 15-minute count period. We did not attempt to identify birds to the sub-species level. Non-target species were also noted before, during, or after the counting period; however, detection of non-target species was a secondary goal because of its potential interference with the detection of target species. We employed a double-observer method to increase and evaluate the probability of detection. For the double-observer method, two observers conducted counts at each point count location on the same day but at slightly different times. The second count was conducted within 30 minutes of the first count. The first observer (Observer 1) went to Point 1 while the second observer (Observer 2) waited at a distance of approximately 200 m or more away from Point 1. When Observer 1 was finished counting at Point 1, he/she radioed to Observer 2 to progress to Point 1, and Observer 1 progressed to Point 2. The two observers progressed through all of the points in this manner, with Observer 2 trailing Observer 1. At the end of the survey, observers compared observations to estimate the total number of individual birds actually present at the site. To further analyze detectability, the 15-minute count period was divided into four intervals, and we recorded whether species were detected in the first (0-3:00), second (3:00-5:00), third (5:00-10:00), or fourth (10:00-15:00) interval. Additionally, we recorded the distance estimated to the nearest meter from the observer/point to each bird detected, and whether initial detection was made visually, or

Figure 4.8.2 Approximate timing of breeding (color) and migrating (black) populations of target species expected in the Coachella Valley. Dotted lines represent a possibility, and solid lines represent the most likely time period for presence, weighted by approximate relative abundance.



by call or song. Detection information and locations of target species relative to the point coordinates were documented on point count data forms (Appendix 2.1).

Incidental observations were also recorded of target species made before or after point counts, during reconnaissance visits, and during vegetation and arthropod sampling. These incidental observations can be used as supplemental data for habitat modeling. However, only controlled data such as those collected by point counts are appropriate for comparisons between sites or years, or for monitoring population trends. While controlled point counts are primarily for estimating abundance, we also recorded incidental behavioral observations including breeding activity.

Additional Data Collected

To document the extent of variation in the vegetation among point count locations and sites, vegetation measurements were collected after conclusion of the bird surveys following a modification of the Point Reyes Bird Observatory point count vegetation assessment protocol (PRBO 2002). In 2003, vegetation measurements were collected at all point count locations. In 2004, vegetation measurements were only collected at new point count locations, and at sites that underwent substantial modification (e.g., due to fire, Oasis de los Osos, or vegetation clearing, Whitewater Delta). Vegetation structure was quantified by identifying layers based on height (e.g., tree layer, shrub layer), and visually estimating total cover, and maximum and minimum heights and trunk diameters of species in each layer. Vegetation composition was quantified by identifying and listing all the most common species present, and visually estimating the relative cover of each species in each height layer. Additional features were noted such as the presence or absence of standing and running water, relative ground cover composition, and number of snags and logs. Also noted were presence of human disturbance (e.g., ORV tracks, trampled vegetation), invasive species, and surrounding landscape characteristics. Vegetation data were recorded on field forms (Appendix 2.2).

Each year, a rapid habitat assessment was conducted at each point count location after each point count to track major changes to the quality or disturbance of the habitat. Presence of water, human activity, other disturbances, and invasive non-native species were graded (Appendix 2.3). In addition, digital pictures of the riparian habitat were taken at each point in each cardinal direction each year.

In 2003 and 2004, arthropod sampling was conducted after the conclusion of point counts to quantify the presence of ground-dwelling arthropods, primarily ants. The qualitative survey of dominant ant species in riparian areas assesses habitat quality, resistance to invasive species, and presence/absence of invasive ant species known to negatively impact reproductive success of the target bird species. The invasive red fire ant (*Solenopsis invicta*) and Argentine ant (*Linepithema humile*) are known to be serious threats to bird populations in other regions. For rapid assessment of the dominant ant species in the community, a single pitfall trap was embedded in the ground near each point. The pitfall trap is a small 12 oz plastic cup placed in the soil such that the lip is flush with the ground. Inside the cup is a plastic funnel directed into a smaller cup containing a solution of non-toxic, biodegradable detergent and salt. The trap is covered with a small board placed on wooden blocks and is collected three days after placement. Traps typically collect a few ants and an occasional beetle.

Habitat Analysis

To test local and landscape habitat variables for their ability to predict riparian bird abundance by site, a correlation analysis was performed. All target bird detections under 100 m were summed and divided by the number of point counts for each point count location

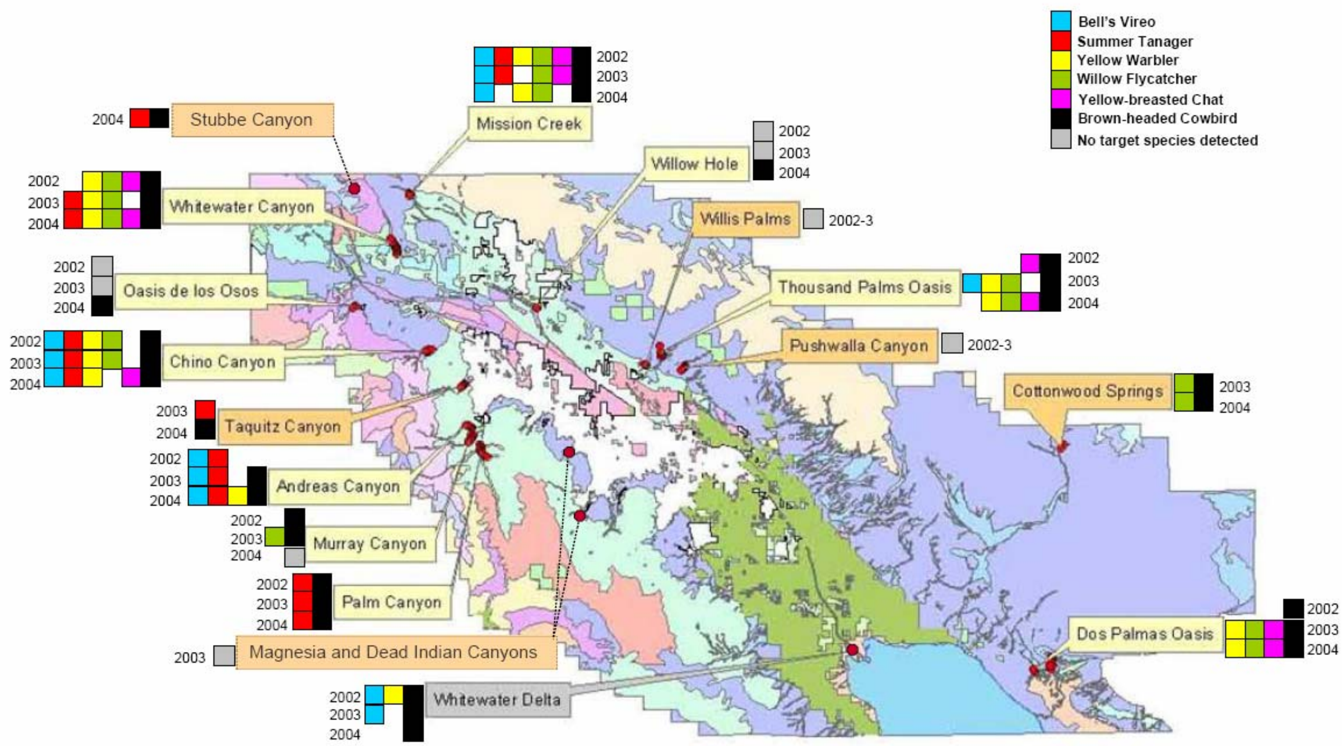
(2003-2004), and summed for each site to generate an abundance estimate by site. Local vegetation variables were estimated at each point count location within a 50 m radius, and then averaged for each site. For this analysis, the variables included native riparian shrub cover “Shrub” (e.g., shrubby willow, cottonwood, sycamore, alder, and *Baccharis*), riparian tree cover excluding palms “Tree,” palm tree cover “Palm,” and non-native invasive shrubs and trees “Invas” (e.g., tamarisk). Local human disturbance was scored from 0-3 in five categories: trash/litter, trampled/cleared vegetation, vehicle presence/tracks, roads/structures, and human presence/tracks. These scores were summed for each point count location, and then averaged for each site. Landscape habitat variables were estimated by land cover habitat types (Holland) using ArcGIS software. A 500 m buffer was drawn around each point count location, and the points were merged into a single buffered area for each site. The total coverage of four land-cover types were used for this analysis: cottonwood-willow riparian woodland “Cott-Willow,” sycamore-alder riparian woodland “Syc-Alder,” palm oasis “Palm,” and “Urban.” All variables were divided by the maximum value for that variable to generate relative scores. Pearson product-moment correlations were generated with SAS 9.1.

Results

A total of 18 riparian sites and 116 points were surveyed between 2002 and 2004. Including all survey rounds and observers, a total of 912 point counts were conducted. Each of the five target riparian bird species, as well as the Brown-headed Cowbird, was observed in the Plan area each year (Figure 4.8.3, Table 4.8.2, Appendix 2.4). There were two sites where all five target species were detected, Chino Canyon and Mission Creek. Four of the five target species were detected at Thousand Palms Oasis and Whitewater Canyon, and three of the five were detected at Andreas Canyon and Dos Palmas Preserve. There were six sites where one or two target species were detected: Cottonwood Springs, Murray Canyon, Palm Canyon, Stubbe Canyon, Tahquitz Canyon, and Whitewater Delta. There were six sites where no target species were detected: Dead Indian Canyon, Magnesia Springs Canyon, Oasis de los Osos, Pushwalla Canyon, Willow Hole, and Willis Palms. Brown-headed Cowbirds were detected at every site where target species were observed. Failure to detect a species at a site does not confirm absence of the species since individuals may go undetected.

Occurrence patterns were fairly consistent over the three years (Figure 4.8.3, Table 4.8.2). For example, at Chino Canyon, Bell’s Vireos, Yellow Warblers, Summer Tanagers, and Brown-headed Cowbirds were found each year. In Palm Canyon, only Summer Tanagers and Brown-headed Cowbirds were detected, but they were present each year. It is unknown to what degree population sizes and occupancy patterns are expected to naturally fluctuate in this system, but major long-term trends can be estimated if these surveys are repeated over future years. Comparing sites surveyed in 2003 and 2004 (and excluding incidental observations during vegetation sampling), there was a slight apparent decrease in number of sites occupied by Bell’s Vireos, five to three. One of the vacated sites, Whitewater Delta, had been cleared of vegetation, and the other site, Thousand Palms Oasis, had only a single

Figure 4.8.3. Summary map of target species occurrence patterns, Coachella Valley, 2002-2004. Observations during point counts and all incidental observations are included.



individual detected in 2003. The number of sites where Yellow Warblers occurred had a slight apparent increase, from four to six, but this is most likely due to slightly earlier surveys in 2004 during the migration peak. The number of sites where Yellow-breasted Chats occurred had a slight apparent increase, from two to four, but the only consistent site with multiple singing males was Dos Palmas Oasis. A new small population of Yellow-breasted Chats was apparently established at Chino Canyon in 2004. There was little or no apparent change in the number of sites where Willow Flycatchers and Summer Tanagers occurred between 2003 and 2004. Brown-headed Cowbirds showed a slight apparent increase in number of sites occupied, from 10 to 12, but only a single individual was detected at the newly occupied sites.

In the areas surveyed where target riparian species were detected, relatively few individuals were observed despite our double-observer method and thorough coverage. While some individuals present in an area may go undetected, males typically sing during the breeding season. Thus, it is expected that low detections reflect relatively low numbers of individuals present. Estimates of the number of individuals present per site were typically only one to three individuals or possible pairs. The maximum numbers of individuals detected in any one

Table 4.8.2. Occurrence Patterns[1] of Target Bird Species and Maximum Number Observed Per Day, Coachella Valley, 2002-2004.

Species	Site	2002	2003		2004		Max #
			Early	Late[3]	Early	Late[3]	
Willow Flycatcher	Chino Canyon	X	O	O	O	O	1
	Cottonwood Springs	--	X	--	X	O	2
	Dos Palmas Oasis	O	X	O	X	O	6
	Mission Creek	X	X	O	X	O	10
	Murray Canyon	O	X	O	O	O	2
	Thousand Palms Oasis	O	X	O	X	O	4
	Whitewater Canyon	X	X	O	X	O	5
Bell's Vireo	**Andreas Canyon[2]**	X	X	X	X	X	4
	Chino Canyon[2]	X	X	X	X	X	7
	Mission Creek[2]	X	X	X	X	O	4
	Thousand Palms Oasis	O	X	O	O	O	1
	Whitewater Delta	X	--	X	--	O	2
Yellow Warbler	Andreas Canyon	O	O	O	X	O	5
	Chino Canyon	X	X	X	X	O	6
	Dos Palmas Oasis	O	X	O	X	O	4
	Mission Creek	X	O	O	X	O	9
	Thousand Palms Oasis	O	X	O	X	O	8
	Whitewater Canyon	O	X	O	X	O	3
	Whitewater Delta	X	--	O	--	O	1
Yellow-breasted Chat	Chino Canyon	O	O	O	X	X	5
	Dos Palmas Oasis	O	X	X	X	X	13
	Mission Creek	X	X	O	O	O	1
	Thousand Palms Oasis	X	O	O	X	O	1
	Whitewater Canyon	X	O	O	X	O	1
Summer Tanager	**Andreas Canyon[2]**	X	X	X	X	X	4
	Chino Canyon[2]	X	X	X	X	X	7
	Mission Creek	X	O	O	O	O	1
	Palm Canyon	X	X	O	O	X	3
	Stubbe Canyon[2]	--	--	--	X	X	6
	Tahquitz Canyon	--	X	O	O	O	1
	Whitewater Canyon[2]	O	X	X	X	X	5
Brown-headed Cowbird	**Andreas Canyon[2]**	O	X	X	X	O	14
	Chino Canyon	X	X	X	X	X	9
	Cottonwood Springs	--	X	--	X	X	4
	Dos Palmas Oasis[2]	X	X	X	X	X	42
	Mission Creek	X	X	X	X	O	3
	Murray Canyon	X	X	O	O	O	3
	Oasis de los Osos	O	O	O	O	X	1
	Palm Canyon	X	X	X	X	X	8
	Stubbe Canyon	--	--	--	X	X	2
	Tahquitz Canyon	--	O	O	X	O	1
	Thousand Palms Oasis	X	X	O	X	O	7
	Whitewater Canyon	X	X	X	X	X	19
	Whitewater Delta	X	--	X	--	X	204
	Willow Hole	O	O	O	O	X	1

[1] X indicates presence, O indicates absence, -- indicates that surveys were not completed; 2002 surveys were conducted between April and July; 2003 and 2004 surveys were conducted in two rounds, one between May and June ("Early") and another between late June and July ("Late")

[2] Sites where breeding was confirmed

[3] Incidental observations during vegetation sampling are not included

day were: ten Willow Flycatchers at Mission Creek, seven Bell's Vireos at Chino Canyon, nine Yellow Warblers at Mission Creek, 13 Yellow-breasted Chats at Dos Palmas Oasis, and seven Summer Tanagers at Chino Canyon. The total number of sites where a target species was detected over the three years was only five for Bell's Vireos and Yellow-breasted Chats, and seven for Willow Flycatchers, Yellow Warblers, and Summer Tanagers. Brown-headed Cowbirds were detected at 14 of the 18 sites, and were abundant at several sites.

Distance sampling was used so that estimated abundance can be adjusted for differences in detection probability, especially by species and observer. This added control will improve the accuracy of any future trend analysis. The double-observer method increased target species detections by 37% over a single observer method, although this varied by species. Bell's Vireos were relatively easily detected, with over half of all detections occurring in the first three minutes of the point count period alone, and the second observer only adding 26% of new detections. On the other hand, Summer Tanagers were less easily detected, probably due to their larger territory sizes and less frequent singing, so the second observer added 50% of new detections. Although additional survey time and observers are always expected to improve detection at any one point, greater coverage area is generally a more effective way to increase total sample size. We surveyed essentially all riparian habitat that was accessible within the Plan area. Probability of occupancy and detection can also vary over the season for different species depending on the timing of migration and the stage of the breeding cycle. To compensate for this variability, most sites received at least two rounds of surveys in 2003 and 2004. This allowed us to detect both early spring migrants and late summer breeders. In late June and July, no spring migrants are expected, so detections are more likely to represent summer residents and possible breeding. However, in late July, probability of detection can be lower for some species such as the Yellow-breasted Chat (Eckerle and Thompson 2001), and a few individuals may begin departing for early fall migration.

All riparian sites are expected to be important to many migratory and resident bird species, regardless of whether or not individuals of the target species are present. For example, no target species were observed at Willow Hole, but numerous other migrating species were observed at this location. In addition, target species utilize many non-breeding riparian sites during migration. For example, Yellow Warblers and Willow Flycatchers were numerous at several sites in May, but absent by late June. The Coachella Valley is an important migratory corridor, and riparian sites offer critical habitat for temporary shelter, foraging, and water. Riparian sites are also expected to be important to many other resident species, whether for breeding, foraging, or as a source of water. A total of 150 bird species was detected at riparian sites in the Plan area from 2002-2004 (Appendix 2.5).

With respect to the breeding status of the target species in the Coachella Valley, the Bell's Vireo and the Summer Tanager were confirmed breeding within the Plan area, and it is very likely that the Yellow-breasted Chat also breeds in the Plan area. Confirmation of breeding was not a primary goal of these surveys, but breeding behavior was incidentally recorded. In 2003, Bell's Vireos were confirmed breeding at Chino Canyon by the detection of a pair with fledglings 15 July. At Andreas Canyon 27 May 2004, a pair of Bell's Vireos was observed with a nest that had been parasitized by Brown-headed Cowbirds, confirming a breeding attempt. At Mission Creek 10 June 2004, two Bell's Vireos were observed making plaintive calls, suggesting breeding activity. The presence of singing males and repeated detections within the season and between years is suggestive of breeding behavior. Singing male Bell's Vireos were detected at the same locations all three years at Andreas Canyon, Chino Canyon, and Mission Creek. This also suggests site fidelity, a common phenomenon in breeding birds. However, with point counts alone, we cannot determine with certainty that these birds are the same individuals that occupied the sites in previous years.

Summer Tanagers were confirmed breeding at one site in 2002 and at three sites in 2004. In 2002, a female Summer Tanager was observed on a nest in Andreas Canyon. In 2004, pairs were observed with nests at Chino Canyon 22 June, and at Whitewater Canyon 7 June, and a female was observed carrying large amounts of insects, presumably either to nestlings or fledglings, at Stubbe Canyon 9 July. Singing male Summer Tanagers have been detected in the same locations at Andreas Canyon and Chino Canyon in 2002, 2003, and 2004, and at least one Summer Tanager has been detected all three years at Palm Canyon and Whitewater Canyon. While Summer Tanagers have declined in the region since the 1960s due to loss of riparian woodland dominated by cottonwood (Patten et al. 2003), our observations suggest a possible recovery trend in the Coachella Valley.

Numerous Yellow-breasted Chats were detected repeatedly at the same locations in 2003 and 2004 at Dos Palmas Preserve, which strongly suggests breeding at this site. In 2004, at least two singing males were detected at Chino Canyon in June and July where none were detected in 2003, suggesting that this may be a new breeding site. Over the three years, a few Yellow-breasted Chats were observed irregularly at Mission Creek, Thousand Palms Oasis, Whitewater Canyon, and Willow Hole early in the season, suggesting that these were migrating individuals.

Willow Flycatchers were detected at least two years in a row during the first round of surveys at Chino Canyon, Cottonwood Springs, Dos Palmas Preserve, Mission Creek, Thousand Palms Oasis, and Whitewater Canyon, and one year at Murray Canyon, but these were all likely migrants. It is not known whether these individuals were *Empidonax traillii extimus*, the subspecies that breeds in southern California (Unitt 1987; Sedgwick 2000) and is a federally-listed endangered subspecies, or whether they were a different subspecies that occurs as migrants in southern California but breed farther north (e.g. *E. t. brewsteri*). Even though multiple singing males were observed at many of these sites, no Willow Flycatchers have been detected during the second rounds of surveys in late June to July. The latest detection was on 19 June 2003 at Cottonwood Springs, still within the normal migration period. Individuals observed in August are presumed to be fall migrants. Over the three years of surveys, no breeding activity has been confirmed for Willow Flycatchers.

Yellow Warblers were observed at least two years at Chino Canyon, Dos Palmas Preserve, Mission Creek, Thousand Palms Oasis, and Whitewater Canyon, and were observed one year at Andreas Canyon and Whitewater Delta. Over the three years of surveys, most detections of Yellow Warblers were during the peak migration period in mid-May, and no breeding activity has been confirmed. A single singing male was observed at Chino Canyon 15 July 2003, suggesting the possibility of breeding at this site, but the individual could also have been an early fall migrant.

The Brown-headed Cowbird was confirmed breeding at two sites but is strongly suspected to be breeding commonly though much of the Plan area. It is suspected to be breeding based on repeated observations of singing males in the vicinity of females away from foraging areas. On 27 May 2004 at Andreas Canyon, a Bell's Vireo nest was incidentally observed containing two vireo eggs and two cowbird eggs. At Dos Palmas Preserve, a Common Yellowthroat (*Geothlypis trichas*) was observed feeding a cowbird fledgling 28 June 2004, and a Bewick's Wren (*Thryomanes bewickii*) was observed feeding a cowbird fledgling 18 August 2003. From 2002-2004, the Brown-headed Cowbird was detected at every site where there were target riparian species. The highest abundance of Brown-headed Cowbirds was recorded at Whitewater Delta, with several large flocks observed. However, this likely represents post-breeding congregation. High rates of nest parasitism by Brown-headed Cowbirds have been reported for all of the target riparian bird species in other regions (Brown 1993; Eckerle and Thompson 2001; Lowther et al. 1999; Robinson 1996; Sedgwick 2000). The Bronzed Cowbird (*Molothrus aeneus*) is another brood parasite that occurs less commonly in the region (Patten et al. 2003), but this species was not observed in the Plan area during these surveys.

The five target bird species are all well-known to be strongly associated with riparian habitat, and this was supported by our preliminary habitat models. To analyze this further, relative bird abundance by site was correlated with a few local and landscape habitat variables expected to be key predictors (Table 4.8.3). Willow Flycatchers were strongly associated with local native riparian shrub cover while Summer Tanagers were strongly associated with local native riparian tree cover. At the landscape scale, Summer Tanagers were associated with cottonwood-willow riparian woodland while Bell's Vireos and Yellow Warblers were more strongly associated with sycamore-alder riparian woodland. Further analysis and habitat modeling is needed, but it is likely that no single riparian type is ideal for all the target species, and instead a range of dominant species and successional stages is needed to support the specialization of each bird species. Although many correlations were not statistically significant, it is interesting that all target species (individually and combined) were negatively associated with palm tree cover, both on the local and landscape scale, and all were negatively associated with urban landscape cover. Local human disturbance and invasive plant species were poor predictors of riparian bird abundance by site, except that Yellow-breasted Chats were positively associated with invasive species. This is not surprising since the largest population of Yellow-breasted Chats was at Dos Palmas Oasis, where tamarisk is very common. Interestingly, Brown-headed Cowbirds were strongly associated with local human disturbance and invasive species, but this was probably skewed by the large numbers of cowbirds at Whitewater Delta, the most disturbed site. Overall, human disturbance was fairly low at most of the riparian sites, but historical reduction of riparian habitat was not analyzed. Invasive exotic plants were common at many of the sites surveyed, with tamarisk being very widespread. The most disturbed site was Whitewater Delta, where the vegetation had been partially cleared in 2003 by tractors, and completely cleared in 2004, eliminating any potential use by riparian bird species in the near future. From the ground-dwelling arthropod sampling at the riparian sites, approximately 40 species of ants were identified, none of which are considered to be invasive exotics or threats to riparian birds.

Table 4.8.3. Correlations of Target Species Abundance with Local and Landscape Habitat Variables by Site.

Target species	Local vegetation cover			Landscape habitat cover			Disturbance		
	Shrub	Tree	Palm	Cott-Willow	Syc-Alder	Palm	Human	Invas.	Urban
Willow Flycatcher	**+++**	+	(-)	(+)	(+)	(-)	(-)	(+)	(-)
Bell's Vireo	(+)	+	(-)	(+)	**+++**	(-)	(+)	(-)	(-)
Yellow Warbler	+	+	(-)	(+)	**++**	(-)	(-)	(-)	(-)
Yellow-breasted Chat	(-)	(-)	(-)	(-)	(-)	(-)	(+)	**++**	(-)
Summer Tanager	(-)	**+++**	(-)	**++**	(+)	(-)	(-)	(-)	(-)
All target riparian birds	(+)	**++**	(-)	+	**++**	(-)	(-)	(+)	(-)
Brown-headed Cowbird	(-)	(-)	(-)	**++**	(-)	(-)	**+++**	**+++**	(-)

( ) indicates no correlation (p>0.10), but the direction of the relationship is indicated (+ or –).

+ indicates a weak positive relationship (0.05<p<0.10), not statistically significant.

++ indicates a positive relationship (0.005<p<0.05), statistically significant.

+++indicates a positive relationship (p<0.005), highly statistically significant.

Summary

During surveys at 18 riparian sites in the Coachella Valley in Spring-Summer 2002-2004, we observed all five target riparian bird species each year: Willow Flycatchers, Bell's Vireos, Yellow Warblers, Yellow-breasted Chats, and Summer Tanagers. Small numbers were detected during surveys, which we believe reflects the presence of relatively few individuals at the sites surveyed, but occupancy patterns were fairly consistent over the three years. Bell's Vireo and Summer Tanager were confirmed breeding in the Plan area, and are suspected to be breeding at several sites. Yellow-breasted Chats are strongly suspected to be breeding in the Plan area, most likely at Dos Palmas Preserve, and possibly Chino Canyon. Willow Flycatchers and Yellow Warblers were observed, but all were likely migrants.

The primary threat to riparian birds is loss of riparian habitat. During this survey period, riparian vegetation was completely removed at Whitewater Delta, rendering it unsuitable for riparian birds. Additional potential threats include habitat degradation, disturbance, nest parasitism, predation, and disease. We observed Brown-headed Cowbirds at every site where target riparian bird species were present. Brown-headed Cowbirds were very abundant at several sites, and breeding was confirmed in the Plan area. Thus, nest parasitism appears to be a significant threat, and it is unknown what effect it may have already had on local riparian bird populations. A single female can potentially parasitize up to 40 nests per season (Lowther 1993), and nest parasitism is known to a common problem for these target species in some areas that have been studied. However, host birds can develop defensive mechanisms against cowbirds, so the presence of cowbirds alone does not necessarily translate into a significant reduction in reproductive success. Native riparian shrubs and trees were positively correlated with riparian bird abundance by site, except for the Yellow-breasted Chat which was most common at Dos Palmas Preserve where tamarisk is abundant. No invasive ants were detected, and direct human disturbance was fairly low at most sites. It is unknown to what degree invasive plants, human disturbance, nest parasitism, predators, disease, or other factors may be affecting reproductive success of riparian birds in this system.

To develop a monitoring protocol and adaptive management process, several factors need to be considered. Controlled point counts with distance sampling, double-observers, and at least two rounds of surveys are recommended for accurate abundance estimates to monitor long-term population trends. Periodically, within any one season, as many riparian sites as possible should be surveyed following these established protocols, including sites with marginal riparian habitat. These protocols are also highly standardized so that results can potentially be compared to other regions and studies. The extent and quality of riparian habitat should be monitored, and vegetation data should be collected periodically to test and refine predictive models. Potential threats such as human disturbance, invasive species, nest parasitism, predation, and disease should be further assessed, and intensive nest monitoring surveys are needed to study their effects on reproductive success. Due to the rarity and migratory behavior of these target species, controlled experiments are unlikely to be practical, and the primary focus may need to be on the conservation and improvement of suitable riparian habitat. Riparian systems are necessarily dynamic due to periodic flood scouring, so many sites with extensive habitat are needed to support a range of successional stages and larger bird population sizes.

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4.9 RARE PLANT SURVEYS

Robert D. Cox

In support of the objectives of the monitoring framework developed by UCR's Center for Conservation Biology, we attempted to identify all historically known locations of the plant species covered under the plan, and determine whether locations occupied by the target species in the past continue to be occupied now. The five covered plant species are known to have relatively restricted distributions within the plan area, and may be nearly absent outside the plan area. Within these limited areas, it is important to know not only the extent of the former distribution of these plants, but to be able to compare former and current distribution patterns.

Database Creation

In order to determine the former distribution of each species, we attempted to locate historic records for the five target species covered by the plan. In 2002, a master database of historic records was created by querying various herbaria and museums, and combining these with records on file in the lab of Dr. Thomas Scott at UCR. This database of observations then required considerable effort to create a useable database of plant locations.

The first step in the process of identifying historic locations for the covered plants was to translate the mostly text descriptions of plant locations provided by the historic records into geographic coordinates that could be used in mapping and visiting the locations on the ground. This step, referred to as "geo-referencing", was completed by examining print and digital maps of the areas described by the text of each record, locating the area described, and extracting a geographic coordinate from these maps. Alternatively, if the historic record itself included a set of geographic coordinates, these were used, rather than producing new coordinates. Finally, the geographic coordinates, were placed on a digital map to help determine historic distribution, and to allow us to organize and prioritize site visits. Once the locations for each species were mapped, those on private property were eliminated by comparison to a map displaying public property within the plan area. This eliminated more than 50% of the records in the database. Another 25% of the records pointed to duplicate locations. After private and duplicate locations were eliminated, 43 records were left in the database. These 43 records pointed to locations which UCR researchers attempted to visit during the study period. Table 1 displays the breakdown of record types for each category.

Table 4.9.1. Number of each type of record for each species covered by the plan. "Total" = total number of records for each species. "Private" = number of records on private property. "Public" = number of records on public lands. "Duplicate" = number of records on public land that

describe the same area. "Unique Public" = number of records on public land minus any duplicate records.

	Grand Total	Private	Public	Duplicate	Unique Public
Coachella Valley milkvetch	61	51	10	5	5
Little San Bernardino Mtns linanthus	10	8	2	0	2
Mecca aster	58	15	43	30	13
Orocopia sage	51	24	27	11	16
Triple-ribbed milkvetch	26	13	13	6	7
Total	206	111	95	52	43

Site Visits

Utilizing the geo-referenced coordinates and personal GPS units, an attempt was made to visit each record located on public land. Each location was searched for the presence of covered species; if the species was located, a $500m^2$ plot was established around the center of the population. If the species was not located, the plot was established around the original coordinates, or around the nearest suitable habitat. Within this $500m^2$ plot, the total number of individuals of the target species present was counted, or estimated if the species was too numerous. Finally, the number of individuals of the target species in the overall population was recorded.

Results

Coachella Valley Milkvetch, *Astragalus lentiginosus* var *coachellae*

A total of 61 records were identified for Coachella valley milkvetch (CMVM), clustered mainly in a large strip in the center of the plan area from approximately Desert Hot Springs SW to Coachella (Figure 4.9.1). Unfortunately, much of this area is privately held, and therefore inaccessible for purposes of this study. Of the 61 historic records for CVMV, 51 were determined to be on private property, eliminating the possibility of surveys on these sites. UCR researchers were successful at identifying only five unique records located on public land for CVMV. Over the three years of the study, researchers were able to visit all five locations. Of the five locations, three were found to be occupied by CVMV, corresponding to a 60% occupation rate. At two locations, surveyors were unable to locate any CVMV individuals. Of the three occupied locations, however, only one location was occupied by significant numbers of CVMV. This location occurs near Windy Point, and appears to be ideal habitat for the species. Table 4.9.2 displays population data for the three locations at which CVMV was present during at least one year. One location (# 2) experienced an increase from two individuals in 2004 to six individuals in 2005. This location was not visited in 2003. Record 1270, on the other hand, experienced an extirpation of the species; although five individuals had been located in 2003, none were located in 2004 or 2005. Finally, record 3672, located near Windy Point, also experienced a decrease in population from an estimate of 500 individuals in both 2003 and 2004, to only 200 individuals in 2005.

Figure 4.9.1. Historic distribution and survey sites for Coachella Valley milkvetch. Rose polygon is generalized distribution based on historic records. Blue points are historic locations where CVMV was present during the study; yellow points are locations where CVMV was not observed

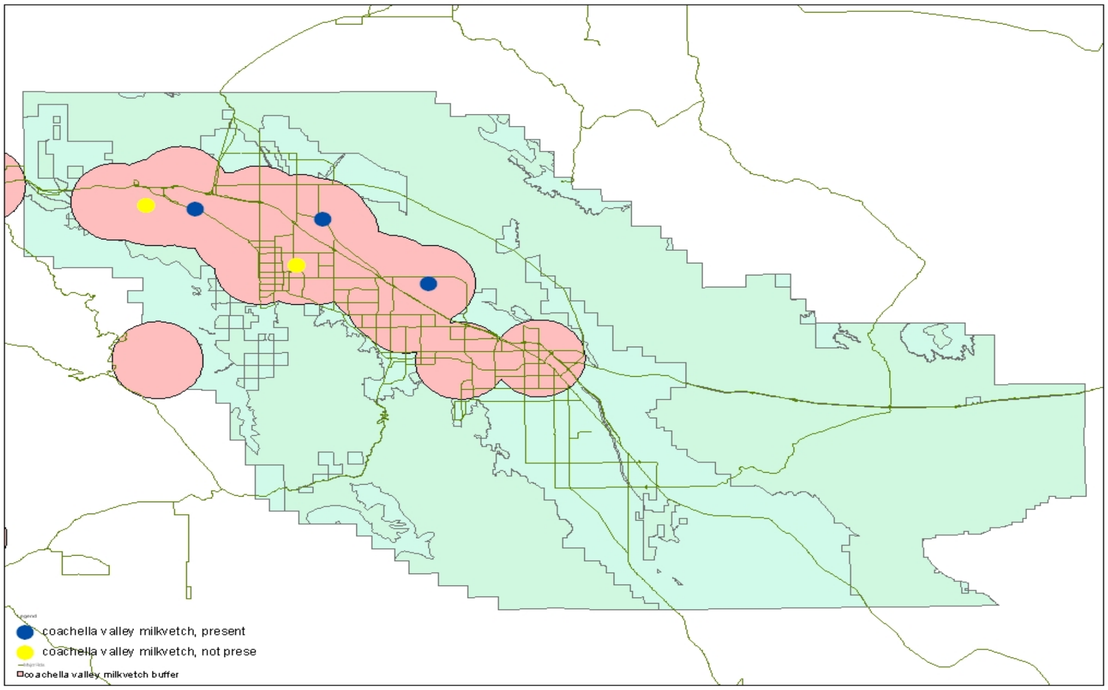


Table 4.9.2. Population data for three locations at which CVMV was present. Numbers for each year represent the number of individuals estimated at each site.

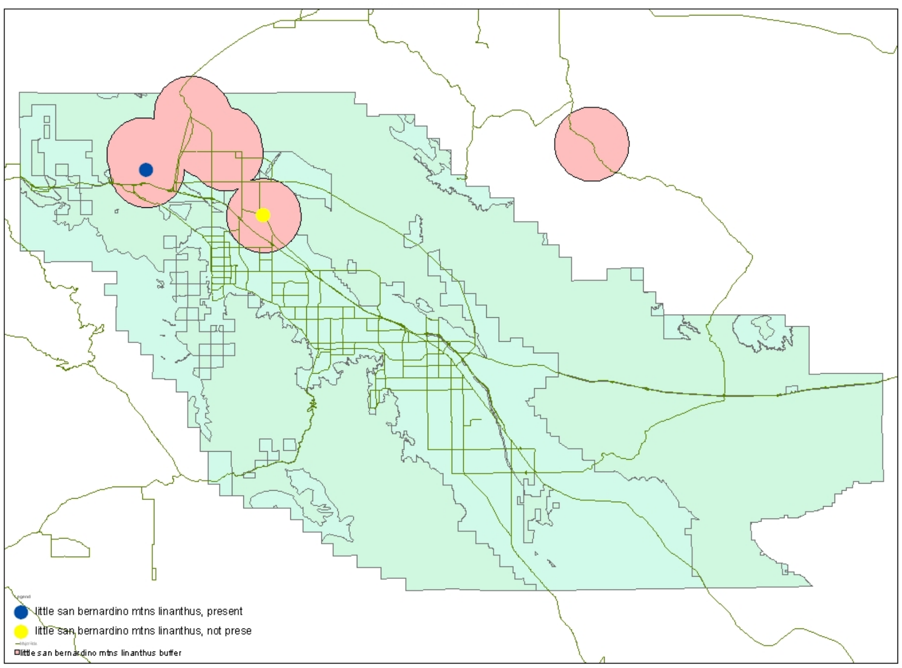
Record #	2003	2004	2005
2	--	2	6
1270	5	0	0
3672	500	500	200

Due to the small number of public-access locations, it is difficult to draw solid conclusions about this species. However, it appears that CVMV may be present in large numbers only in a much smaller sub-area of its overall range, and that over the last three years, some populations have seriously declined.

<u>Little San Bernardino Mountains Linanthus, *Linanthus (Gilia) maculata*</u>

Nearly every known location of this plant within the plan area occurs in a restricted area along the Dry Morongo and Mission Creek drainages (Figure 4.9.2). Unfortunately, this area is predominately privately held, and only two unique public locations were identified. Individuals of the Little San Bernardino Mountains Linanthus (LSBML) were observed at only one location, leaving a 50% occupation rate for this species. Fortunately, at the location where LSBML was observed, a large increase in estimated population size over the three years of the study was recorded: although no individuals of LSBML were observed at this site in

Figure 4.9.2. Historic distribution and survey sites for Little San Bernardino Mountains linanthus (LSBML). Rose polygon is generalized distribution based on historic records. Blue points are historic locations where LSBML was present during the study; yellow points are locations where LSBML was not observed.



2003, 1781 individuals were observed in 2004 and 2800 individuals were observed in 2005.

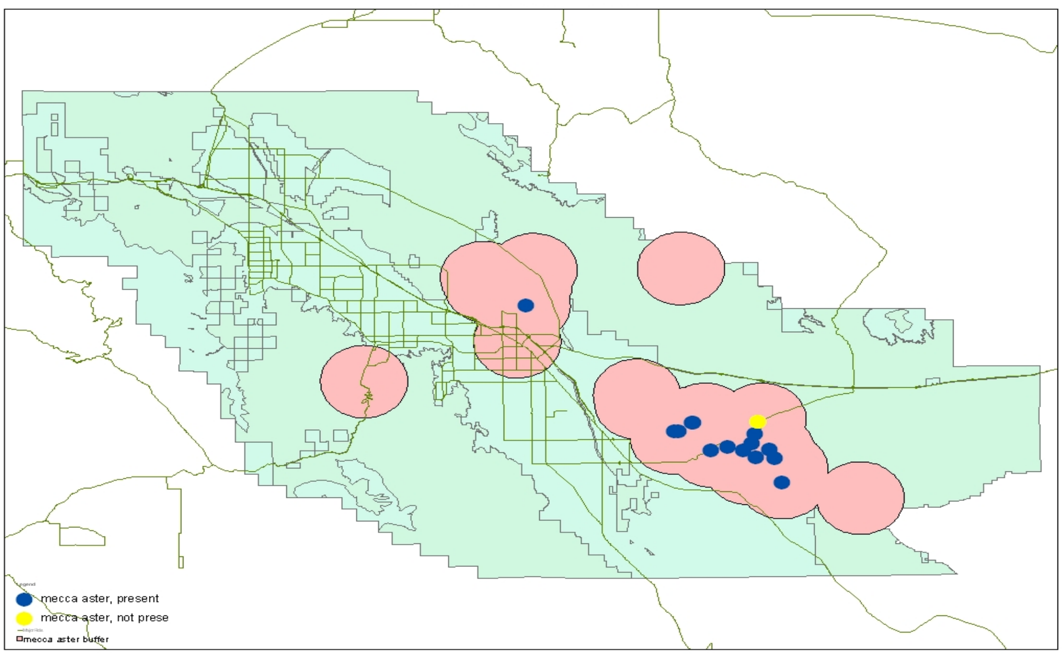
Again, it is problematic to draw conclusions about this species from two records. It is clear, however, that the plant is very restricted in distribution. UCR researchers were successful at locating another population of LSBML outside the plan area, within Joshua Tree National Park. The existence of this population indicates the likelihood that other populations may exist elsewhere in both the national park and in the plan area. There are almost certainly other populations located on private property within the plan area.

Mecca Aster, *Xylorhiza cognata*

Based on historic records, distribution of Mecca aster (MA) involves two main clusters around the Indio Hills and Mecca Hills regions (Figure 4.9.3). The majority of records are from the Mecca Hills area; the Indio Hills contained only one location on public land.

Over the course of the three years, researchers were able to visit all 13 unique public locations indicated by the historic record database. Of these 13 records, individuals of MA were observed at 12 locations, giving a 92% occupation rate. In addition, researchers located two new unique locations for this plant. Population sizes at all occupied locations varied from six individuals to 377, and averaged about 80 plants per location.

Figure 4.9.3. Historic distribution and survey sites for mecca aster (MA). Rose polygon is generalized distribution based on historic records. Blue points are historic locations where MA was present during the study; yellow points are locations where MA was not observed.



MA ranges from sparse to plentiful over its distribution area. It is present at the majority of historic locations visited, and the majority of populations are located within the Mecca Hills area. The Indio Hills cluster appears to be very restricted, with few individuals present.

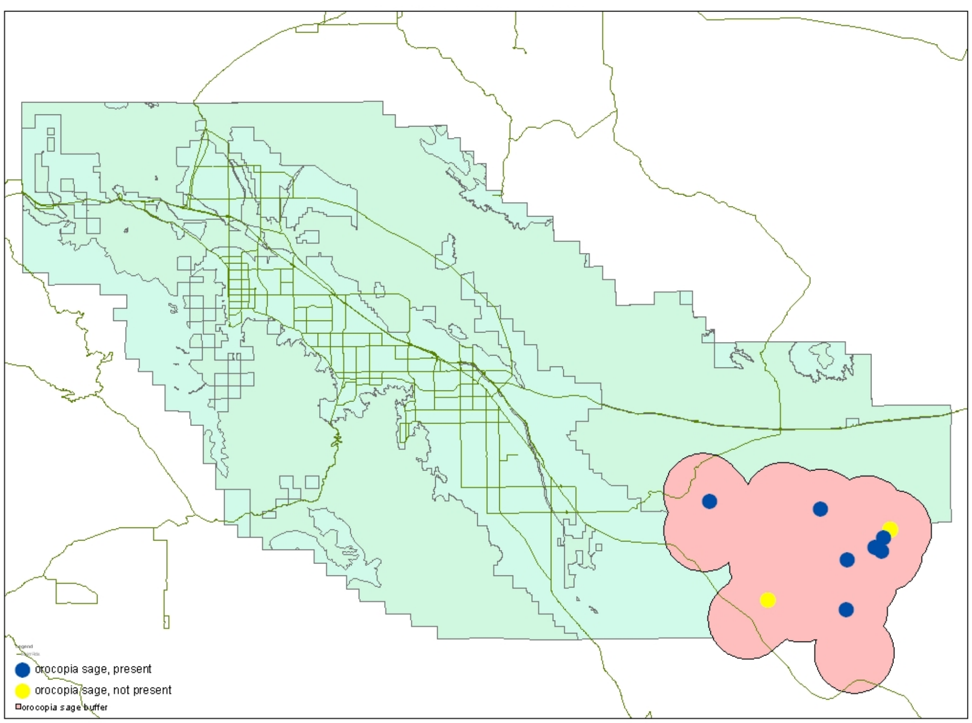
Orocopia Sage, *Salvia greatae*

Historically, Orocopia sage (OS) is known from only the Orocopia and Chocolate mountains, with a few records from the Mecca Hills area (Figure 4.9.4). Both these areas are currently protected by the BLM as Wilderness Areas.

OS was found to have 16 unique historic records on public land. Of the 16 locations, ten locations were visited to survey for OS presence, and locations described by six records were considered to be either too vague or too remote for further investigation. Of the ten locations visited, OS individuals were observed at seven locations, leading to a 70% occupation rate. Populations varied, but averaged over 200 individuals per hectare at each location. Indeed, several locations appeared to actually be sampling continuous population patches, covering large areas.

OS appears to be present and fairly numerous over large areas of its range. There has been some discussion, supported by observations at one site, that additional populations of this plant may exist at upper elevation sites in the Orocopia Mountains, but due to the remoteness and ruggedness of the mountains, such sites have not been adequately surveyed for the species.

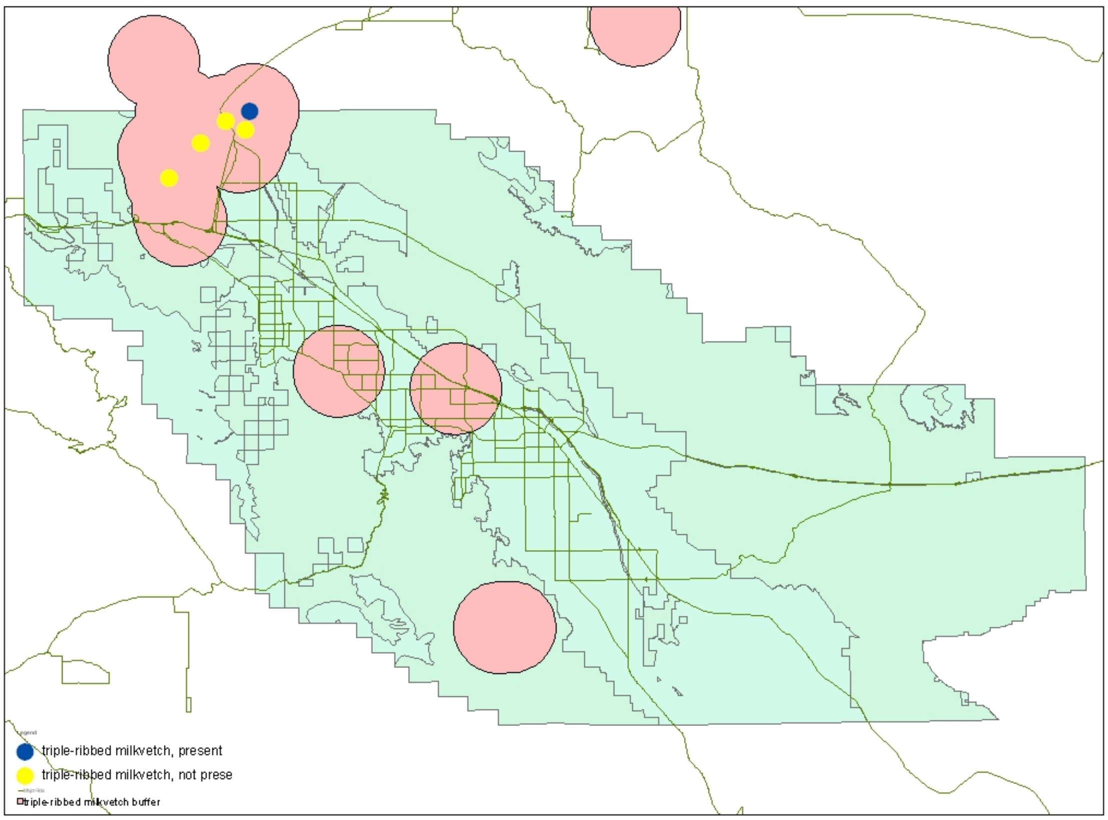
Figure 4.9.4. Historic distribution and survey sites for orocopia sage (OS). Rose polygon is generalized distribution based on historic records. Blue points are historic locations where OS was present during the study; yellow points are locations where OS was not observed.



<u>Triple-ribbed Milkvetch, *Astragalus tricharinatus*</u>

Historic distribution for triple-ribbed milkvetch (TRMV) is clustered in a small area at the northern portion of the plan area, with few records in other areas of the plan (Figure 4.9.5).

Figure 4.9.5. Historic distribution and survey sites for triple-ribbed milkvetch (TRMV). Rose polygon is generalized distribution based on historic records. Blue points are historic locations where TRMV was present during the study; yellow points are locations where TRMV was not observed



Locations corresponding to five of the seven unique public locations identified for triple-ribbed milkvetch were visited. The two records not visited were too vague for further investigation. TRMV was observed at only one location, resulting in a 20% occupation rate. This location, in the Morongo Canyon reserve, had only one individual in 2004 and two TRMV seedlings in 2005. No other historic location was found to be occupied by TRMV during the period of this study.

In 2003, CCB researchers accompanied a group of botanists to document a large, newly-discovered population of TRMV. This population, while neither in the plan area nor in Riverside County, contained several hundred individuals. This discovery indicates that there may be other as of yet unknown large populations of this species at locations within the plan area. Perhaps more importantly, the newly located population occurs outside of the habitat gestalt previously described for TRMV. Previous populations have been in areas of high disturbance, primarily the bottoms or edges of desert washes. Another small population had been found in an area that had recently burned in a wildfire. The population discovered in 2003 occurred in what appeared to be a distinct soil type, high up on the slope, well away from a wash area. The new location indicates earlier views of this species' habitat were too limiting, and may have focused on population "sink" habitats rather on population source habitats. Future surveys will need to take a much broader view of potential habitat for this species.

Conclusions

The majority of records received during the database creation phase of this project described locations on private property. Because these locations were not included in the survey effort, there is little information about whether plant species covered by the plan still exist on those sites. However, it is likely that many populations of these plants persist within their historic distribution area on private property.

During the study, CCB researchers were successful at identifying 45 unique, public locations from historic records for the five plants covered by the Plan, and were able to visit 37 of them. Of these 37 locations, target species were observed at 26 locations, resulting in an overall occupation rate of 70%. Actual species-specific occupation, however, varied greatly, with some species (MA, OS) apparently present at the majority of historic locations, while TRMV was absent from most historic locations visited. In addition, for two species (LSBML & TRMV) there is reason to believe that there may be significant populations of these plants within the plan area that are not accounted for in our understanding of either the historic or current distribution.

CHAPTER 5

BOUNDARY PROCESSES BETWEEN A DESERT SAND DUNE COMMUNITY AND AN ENCROACHING SUBURBAN LANDSCAPE

ABSTRACT - In contrast to the body of work in more mesic habitats, few studies have examined boundary processes between natural and anthropogenic desert landscapes. Our research examined processes occurring at boundaries between a desert sand dune community and an encroaching suburban habitat. We measured responses to an anthropogenic boundary by species from multiple trophic levels, and incorporated measures of habitat suitability, temporal variation, and spatial scales. At an edge versus core habitat scale the only aeolian sand species that demonstrated an unambiguous negative response to the anthropogenic habitat edges was the flat-tailed horned lizard (*Phrynosoma mcallii*). Conversely loggerhead shrikes (*Lanius ludovicianus*) demonstrated a positive response to that edge. At a finer scale, species that exhibited a response to a habitat edge within the first 250 m included the horned lizards along with Coachella Valley fringe-toed lizards (*Uma inornata*) and desert kangaroo rats (*Dipodomys deserti*). The latter two species' response was similar and confined to 25 m from the edge. For the flat-tailed horned lizard, edge effects were measured up to 150 m from the habitat boundary. Three potential causal hypotheses were explored for explaining the edge effect on horned lizards: 1) potential invasions of exotic ant species reducing potential prey for the lizards; 2) road avoidance and road associated mortalities; and, 3) predation from a suite of avian predators whose occurrence and abundance may be augmented by resources available in the suburban habitat. We rejected exotic ant hypothesis due to the absence of exotic ants within the boundary region, and because native ant species (prey for horned lizards) did not show an edge effect. Our data supported the predation hypothesis and may support a road mortality hypothesis as well. Mechanisms for regulating population dynamics of desert species are often "bottom-up," stochastic processes driven by precipitation. The juxtaposition of an anthropogenic edge appears to have created a shift to a "top-down," predator mediated dynamic for these lizards.

Primary mechanisms that distinguish processes at habitat boundaries include: 1) abiotic gradients unique to those boundaries, 2) access to spatially separated resources, and 3) species interactions (Wiens et al. 1985, Murcia 1995, Laurence et al. 2002, Ries et al. 2004). Collectively these mechanisms create a conceptual framework for understanding ecological boundary responses. Additionally, understanding factors that control the occurrence and fluctuations of populations in relatively unfragmented habitat patches provide a context from which to evaluate how those drivers are impacted at boundaries.

In arid ecosystems highly variable and unpredictable precipitation often regulates biological processes (Noy-Meir, 1973). Support for this axiom can be found across a broad range of taxa and regions (Mayhew 1965, 1966; Pianka 1970; Ballinger 1977; Whitford and Creusere 1977; Seely and Louw 1980; Dunham 1981; Abts 1987; Robinson 1990; Brown and Ernest 2002; Germano and Williams 2005). Population dynamics of desert species are thus often characterized as being regulated from the bottom-up, by resource availability mediated by annual rainfall (Brown and Ernest 2002). In contrast, Faeth et al. (2005) described a shift in the processes controlling population dynamics in a suburban desert environment. There irrigated landscapes fostered the establishment of exotic species, regulated productivity, dampened seasonal and annual population fluctuations, and resulted in a predation controlled, top-down community. These different population regulating processes meet at the boundary between natural desert and suburban desert habitats. The extent to which processes generated by suburban habitats encroach on the natural desert and impact components of that community is the subject of this paper.

In contrast to the body of work in more mesic habitats, few studies have examined boundary processes between natural and anthropogenic desert landscapes (e.g., Germaine et al. 1998, Germaine and Wakeling 2001, Boal et al. 2003). Here we examined processes and species occurring at boundaries between an aeolian sand landscape and an encroaching suburban habitat. Species included taxa from multiple trophic levels. Distinguishing between variance imposed by the heterogeneity of the available habitats and what if any effects the proximity of an edge has on the distribution of native species is critical in determining the ecological importance of those edges (Bolger et al. 1997, Fagan et al. 2003). We incorporated measures of habitat suitability, temporal variation, and spatial scales to identify whether components of an aeolian sand community have altered their distributions in response to the presence of anthropogenic habitat edges.

Much of the previous research on edges has focused on temperate and tropical habitats (Janzen 1983, Wilcove 1985, Laurence 1991, Murcia 1995, Laurence et al. 2002) where boundary-mediated ecological flow processes extend from 10-400 m into interior habitats (i.e., Kapos 1989, Camargo and Kapos 1995, Laurence et al. 2001). Fewer studies have investigated edge effects in more xeric environments, with much of that work focusing on semi-arid coastal sage scrub in southern California (Bolger et al. 1991, Bolger et al. 1997, Kristan et al. 2003). In this habitat, moisture gradients at suburban-natural community boundaries have limited the invasion of non-native ants to 100 m or more into the natural communities from mesic refuges in the suburban landscape, with a corresponding negative cascade affecting overall native species richness (Suarez et al. 1998). Increased predation is another factor identified at sage scrub boundaries (Bolger et al. 1991, Bolger et al. 1997, Crooks and Soulé 1999, Suarez et al. 2000, Suarez and Case 2002, Unfried 2003). Collectively these findings define the range of anthropogenic boundary impacts described to date. Our objective was to determine whether any of these impacts also influence the distribution and abundance of species in desert habitats.

Methods

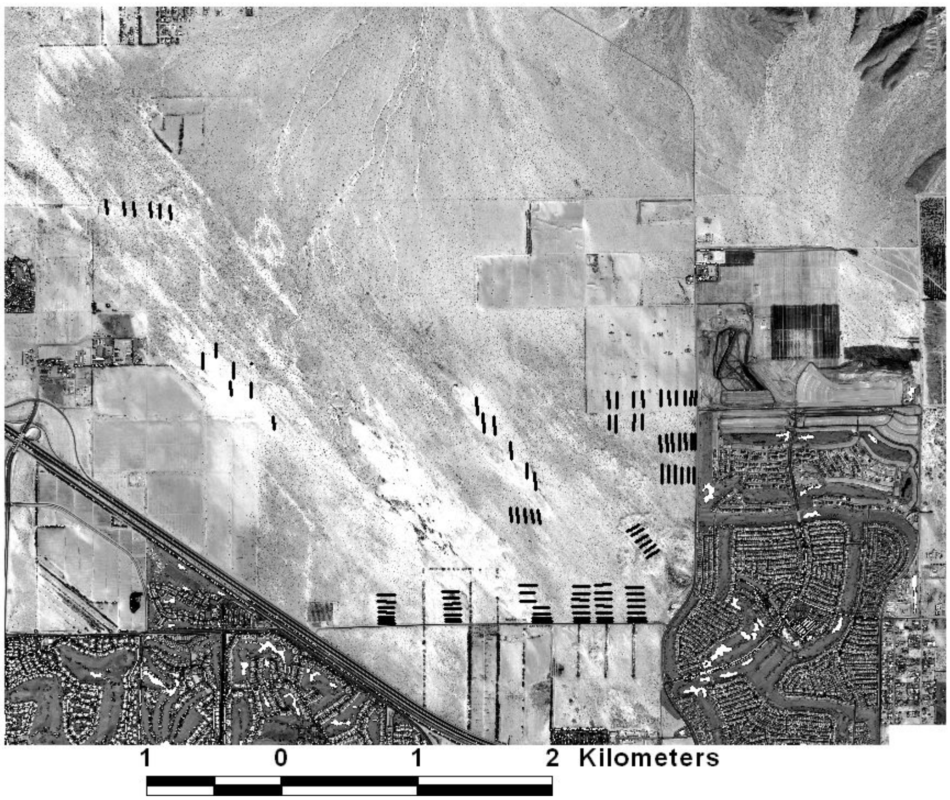
Study Sites

Sand dunes and sand hummocks were studied within the Thousand Palms Preserve (33º 47' N, 116º 20' W*)* in the Coachella Valley near Palm Desert, Riverside County, California. The Preserve is roughly 10,400 ha in size, including approximately 1,300 ha of contiguous aeolian sand habitats. The Coachella Valley is an extremely arid shrub desert with a mean annual rainfall of 79 to 125 mm (most recent 60 year means, Western Regional Climate Center, Palm Springs and Indio reporting stations). The lowest rainfall year occurred in 2002, with just 4 to 7 mm recorded across the valley floor. Temperatures show similar extremes ranging from a low approaching 0o C in the winter to highs exceeding 45o C commonly recorded during July and August.

Study plots were designed to enable analyses at both a coarse scale (edge versus interior plots) and at a finer scale along the habitat edges (within plot distance from the habitat edge). Additionally, study plots were also established to identify edge effects from two separate edge types. Fourteen study plots were established within the Preserve: three were located along a 2.4 km boundary with a suburban golf course community, six were located along a 3.2 km boundary with an abandoned agricultural area and sparse rural housing (Fig. 5.1), and five control plots were centrally located in “core” habitat, greater than 500 m from roads. There was a four-lane paved road separating the Preserve from the suburban habitat and a two-lane paved road separating the Preserve from an area of abandoned agriculture. All study plots were located in a stratified random manner. Plots were stratified so as to include both active sand dune and sand hummock habitat in a proportion corresponding to the aerial extent of those different habitat types. Edge plots were established adjacent to paved roads, but randomly located along the roadway.

Each of the 14 study plots consisted a cluster of 5-8, 10 m x 100 m belt transects. Edge plot included seven transects, with the first centered on a barbed wire boundary fence and running parallel to the fence and adjacent paved road. A second transect was established parallel to the first, but was 25 m interior from the edge. Additional parallel transects were placed at 50, 100, 150, 200, and 250 m from the edge. Interior or core study plots consisted of similar clusters of belt transects with the same dimensions as the edge sites. Core plots were >500 m from any roadway, residence, or habitat discontinuity and included five to eight parallel belt transects separated by 50-150 m. Each transect was marked with a short wooden stake at the beginning, middle, and end so that their position with respect to the boundaries of the belt transect could be readily determined. The stakes were shorter than the surrounding vegetation so that they would not become attractive perches for predatory birds. Each study plot covered approximately 2.5 ha.

Figure 5.1 Satellite image depicting distribution of plots, extent of aeolian sand habitat, juxtaposition of suburban golf course development and abandoned agricultural fields, and roads.



Surveys were repeated six times at each plot between June and July each year from 2002 through 2004. Data collected in 2002 focused on flat-tailed horned lizards, *Phrynosoma mcallii*. Data collected in 2003 and 2004 included all species encountered.

<u>Survey Protocol</u>

The fine aeolian sand of the Thousand Palms Preserve presented an opportunity unique to sand dunes to quantify the occurrence and abundance of all terrestrial species occurring along transects with more or less equal detectability. Each vertebrate species, and many arthropods, could be identified to species and age class by their diagnostic tracks left in the sand. Ground-based species left easily identifiable tracks, and so their ability to avoid detection by differences in activity times, cryptic coloration, or stealthy behavior was nullified. Because late afternoon and evening breezes would wipe the sand clean the next day's accumulation of tracks could not be confused with those from the previous day.

Surveys would begin after the sand surface temperature had risen sufficiently so that diurnal reptiles were observed to be active. Surveys continued until late morning when the high angle of the sun reduced the observer's ability to distinguish and identify tracks. One or two observers working in tandem completed a survey on a given study plot in 30-45 minutes, recording all fresh tracks observed within the 10 m wide belt of each 100 m transect. Tracks were followed off transect if it was necessary to confirm a species' identification and to insure that the same individual was not crossing the same transect repeatedly, thus avoiding an inflated count of the individuals active on that transect. Data for each transect were considered independently, although in rare instances an individual species could move from one transect to another and be recorded as occurring on both transects. In addition to tracks, we recorded any sightings of animals along transects and recorded any bird vocalizations heard during a survey. For the purpose of this analysis, wide ranging predators such as coyotes (*Canis latrans*), greater roadrunners (*Geococcyx californianus*), American kestrels (*Falco sparverius*), and loggerhead shrikes (*Lanius ludovicianus*), were recorded as present on a study plot, rather than on individual transects.

Arthropods were sampled using dry pitfall traps in April of each year. Three pitfall traps was placed on each transect; one at both ends and another at the transect middle. The traps were collected within 24 hrs of being set out to avoid any mortality of vertebrates that happened into the traps. All arthropods were identified to the species level.

Habitat Measures

Vegetation density and plant species composition were measured on each transect each year. All perennial shrubs were counted within the 10 m x 100 m belt transects. Annual plants were counted and cover estimated in a 1 m^2 sampling frame placed at 12 locations along the midline of the belt transect. Four samples were taken on alternating sides of the center line at each end point, and two samples were taken on each side of the center point.

Sand compaction has been described as a key habitat variable for Coachella Valley fringe-toed lizards, *Uma inornata*, (Barrows 1997), and is likely important for other psamnophilic species as well. Sand compaction was measured at 25 points, approximately four m apart, along the midline of each belt transect, each year, using a hand-held pocket penetrometer with an adapter foot for loose soils (Ben Meadows Company, Janesville, WI, USA).

Data Analysis

Data for each year were combined for each species to conduct statistical analyses. A one-way analysis of variance (ANOVA) was employed to conduct coarser scale analyses, examining edge versus core differences, and to include wider ranging bird species. Here edge plots adjacent to the preserve edge were compared with core clusters (plots > 500 m from the preserve edge). A two-way ANOVA was conducted to partition finer scale variance in species abundance between the treatment (distance from the preserve edge, 0 – 250 m) and variance associated with habitat heterogeneity between each of the edge plots.

For the nine edge plots, those species that showed statistically significant variation with respect to treatment were then subjected to a linear regression to determine whether environmental variation coincident with the edge distance could explain that observed variance. All variables were tested for normality and transformed with natural logs when necessary. Dependent variables were the mean of the six surveys on each transect per year for the each species Independent variables included measures of sand compaction (kg / cm^2) for each year, shrub density (shrubs / m^2), and linear distance from the Preserve edge Linear regression analyses were performed using SYSTAT 10.0 (SYSTAT, Wilkinson, 1990). A threshold of $\alpha = 0.05$ for statistical significance was used throughout this paper.

Results

Of the nine species tested with ANOVAs at the edge versus core scale, only the flat-tailed horned lizard and the loggerhead shrike showed a statistically significant effect, although their responses were opposite (Table 5.1). Shrikes were more common along the edge whereas the horned lizards were more abundant in the core. At the finer scale, for those nine plots situated along the Preserve boundary, distance from the Preserve edge was found to be a significant source of variance for the flat-tailed horned lizard, and the Desert kangaroo rat, *Dipodomys deserti* (Table 5.2).

Table 5.1 Analysis of variance (ANOVA) of the abundance of nine species at the larger, edge versus core, scale. The error term represents variation among plots. P-values ≤ 0.05 indicate a statistically significant amount of the variance in the distribution of that species is explained by that treatment (edge effect).

SPECIES	*SOURCE OF VARIATION*	*SS*	*df*	*MS*	*F*	*P-value*
Coachella Valley fringe-toed lizard	Edge effect	1.404	1	1.404	0.871	0.361
	Error	33.850	21	1.612		
Flat-tailed horned lizard	Edge effect	1.294	1	1.294	8.464	0.007
	Error	3.975	26	0.153		
Sidewinder	Edge effect	0.008	1	0.008	0.564	0.465
	Error	0.208	14	0.015		
Shovel-nosed snake	Edge effect	0.032	1	0.032	0.211	0.650
	Error	3.344	22	0.152		
Round-tailed ground squirrel	Edge effect	0.302	1	0.302	3.941	0.063
	Error	1.379	18	0.077		
Desert kangaroo rat	Edge effect	0.078	1	0.078	0.125	0.727
	Error	11.781	19	0.620		
Harvester ants	Edge effect	13.209	1	13.209	0.551	0.467
	Error	455.486	19	23.973		
Greater roadrunner	Edge effect	0.009	1	0.009	0.096	0.760
	Error	2.169	22	0.099		
Loggerhead shrike	Edge effect	1.131	1	1.131	18.871	0.0002
	Error	1.558	26	0.060		

Table 5.2 Two-way ANOVA were employed to determine sources of variance at a smaller, within edge plot, scale. Here variance is partitioned between edge effects and between plots occurring along two boundary types. Coachella Valley fringe-toed lizards did not occur along the boundary that included the four-lane road, so only a one-way ANOVA was calculated for edge effect. P-values ≤ 0.05 indicate a statistically significant amount of the variance in the distribution of that species is explained by that treatment (edge effect or boundary type).

SPECIES	*SOURCE OF VARIATION*	*SS*	*df*	*MS*	*F*	*P-value*
Coachella Valley fringe-toed lizard	Edge Effect	11.569	6	1.928	1.629	0.150
	Within Group (Error)	91.107	77	1.183		
Flat-tailed horned lizard	Edge Effect	1.549	6	0.258	9.545	0.007
	Boundary Type	0.319	1	0.319	11.810	0.014
	Error	0.162	6	0.027		
Sidewinder	Edge Effect	0.008	6	0.001	0.585	0.735
	Boundary Type	< 0.0001	1	< 0.0001	0.010	0.923
	Error	0.014	6	0.002		
Shovel-nosed snake	Edge Effect	0.109	6	0.018	2.073	0.198
	Boundary Type	0.005	1	0.004	0.550	0.486
	Error	0.053	6	0.009		
Round-tailed ground squirrel	Edge Effect	0.075	6	0.013	1.345	0.364
	Boundary Type	0.197	1	0.197	21.085	0.004
	Error	0.056	6	0.009		
Desert kangaroo rat	Edge Effect	2.683	6	0.447	15.529	0.002
	Boundary Type	3.323	1	3.323	115.400	< 0.0001
	Error	0.173	6	0.029		
Harvester ants	Edge Effect	8.789	6	1.465	1.890	0.229
	Boundary Type	13.114	1	13.114	16.921	0.006
	Error	4.650	6	0.775		

These statistical results are corroborated by the patterns of temporal and spatial species' abundance for the seven sand dune occurring species included in our analysis (Figs. 5.2a-2g). There were no consistent responses to proximity of the habitat boundary for Coachella Valley round-tailed ground squirrels (*Spermophilus tereticaudus chlorus*), sidewinders (*Crotalus cerastes*), western shovel-nosed snakes (*Chionactis occipitalis*), and harvester ants (*Pogonomyrmex* spp., including *P. californicus* and *P. magnacanthus*). The abundance of both fringe-toed lizards and desert kangaroo rats was reduced along the immediate habitat edge in both 2003 and 2004, but not at distances ≥ 25 m from that boundary in either year. In contrast, flat-tailed horned lizards' abundance was reduced at distances from the habitat edge of 150 m in 2002, and 100 m in 2003 and 2004.

Figure 5.2A-5.2G Mean counts and one standard error (indicated by the error bar) of species occurring on sand dunes and sand hummocks in the Coachella Valley at various distances from an anthropogenic habitat edge. Data for each year are the combined means for the plots on which the species occurred, with six repetitions per transect per plot. Data collected at >500 m represent the combined core plots.

Fig. 5.2A

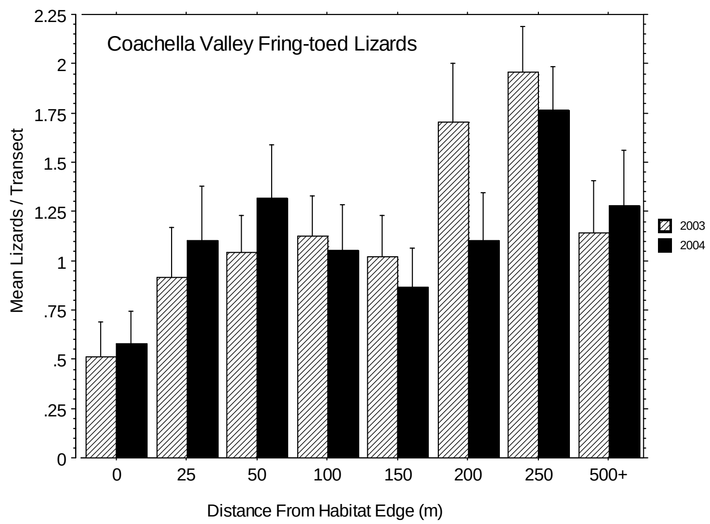


Fig. 5.2B

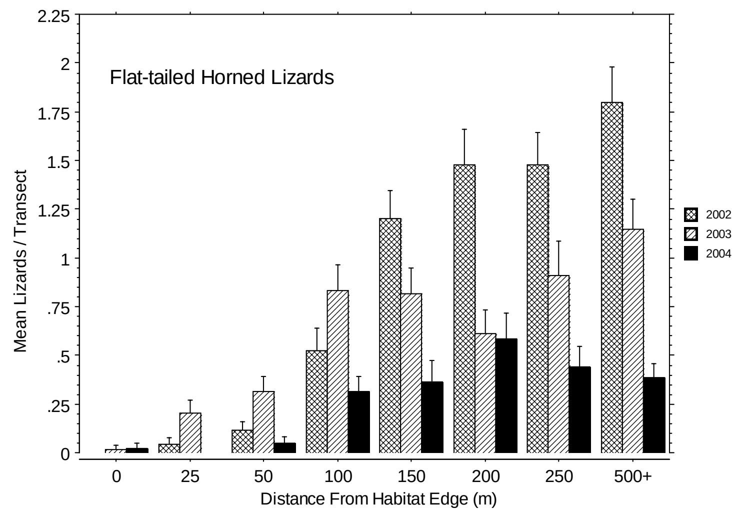


Fig. 5.2C

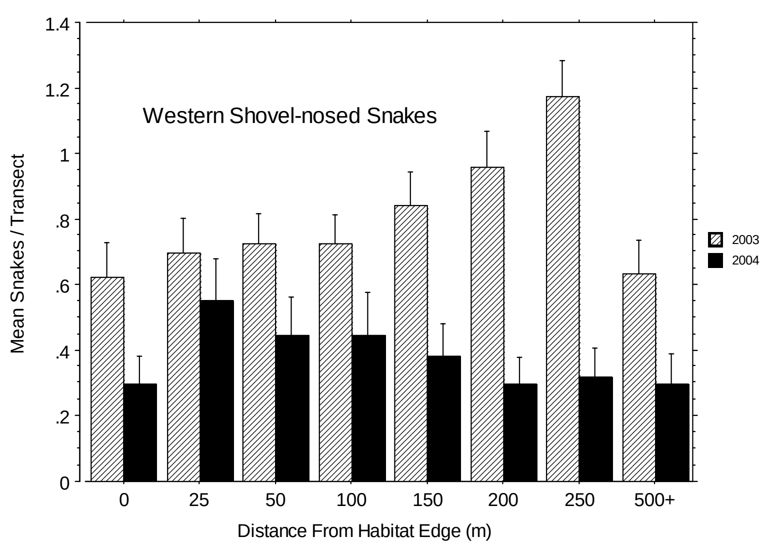


Fig. 5.2D

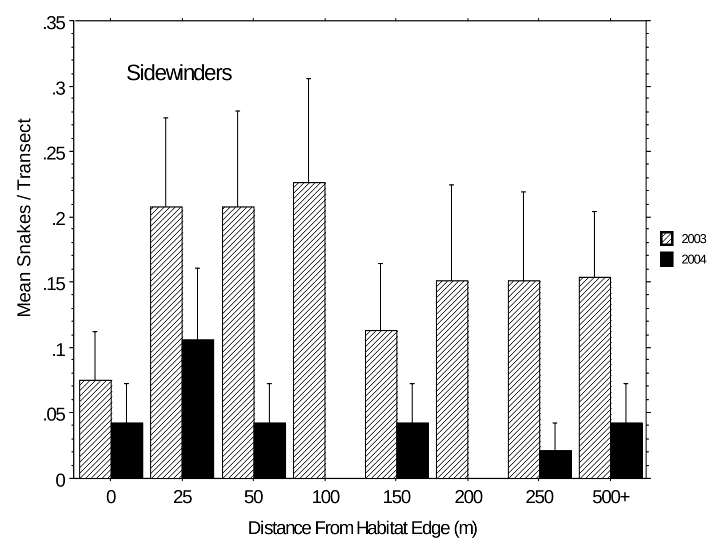


Fig. 5.2E

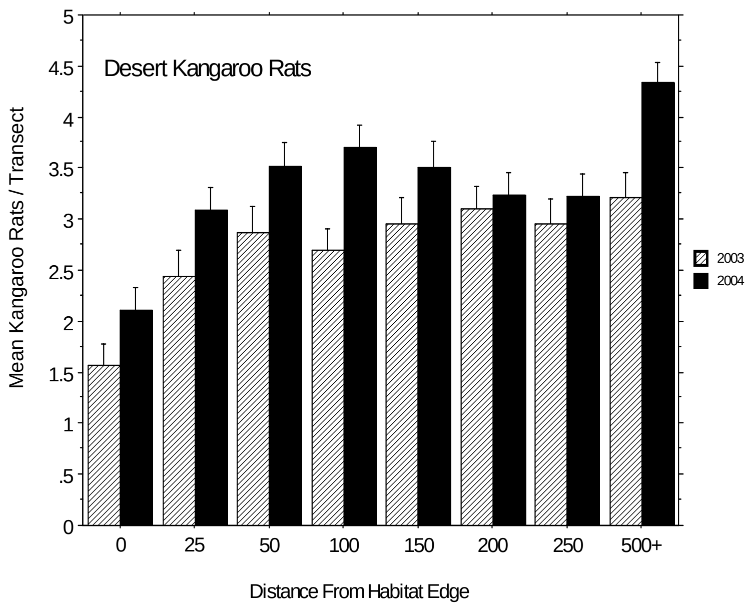


Fig. 5.2F

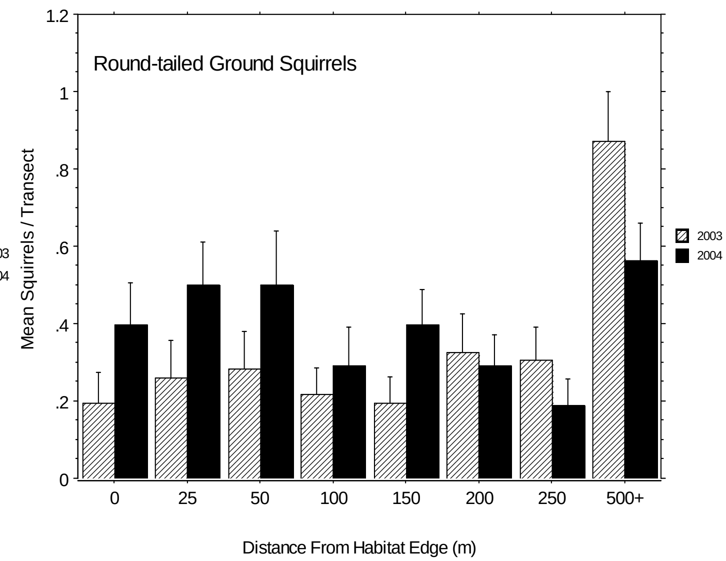
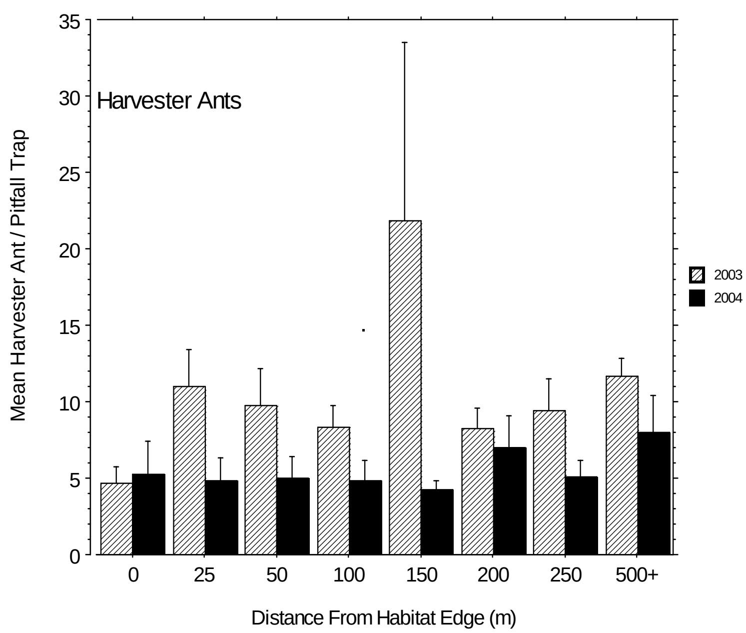
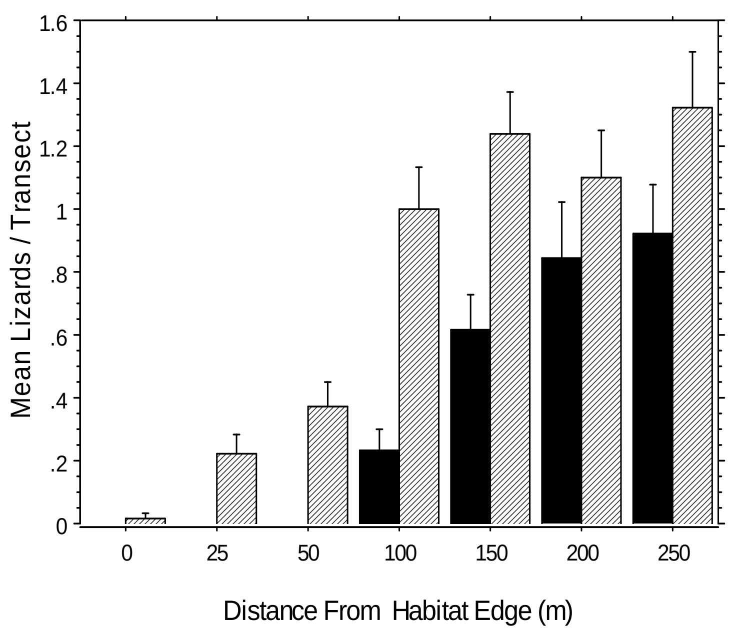


Fig. 5.2G



For the nine edge plots, Pearson's correlations were calculated for distance from the habitat edge and sand compaction and shrub density. Distance was not correlated with sand compaction (r = -0.001 to -0.135, all P = .0335 to 0.995), and was only moderately negatively correlated with shrub density (r = - 0.235, P = 0.043). Despite the lack of an overall correlation, sand was consistently more compacted along the immediate Preserve boundary than it was 25 m interior of that boundary (paired t-test, $p = 0.048$).

Figure 5.3 Mean counts and one standard error (indicated by the error bar) of flat-tailed horned lizards at distances from two boundary types. Solid black bars represent data summarized from three plots adjacent to a four-lane road, with curbs, bounded by a suburban golf course community. Diagonally lined bars represent data summarized from five plots adjacent to a two-lane curbless road, bounded by abandoned agricultural fields and tree-row windbreaks. Both summaries include data combined from 2002 and 2003. Data for each year are the combined means for the plots on which the species occurred, with six repetitions per transect per plot.



Regression models were run for the two species for which the within-plot ANOVAs indicated significant edge interactions (Table 5.3). Shrub density did not explain a significant amount of the variance in abundance for either species, and so was not included in the models. For each species, a single variate model using distance as the independent variable yielded statistically significant linear relationships. However, only the horned lizard's edge distance model yielded a R^2 above 0.100. A single variate model using sand compaction as the independent variable also yielded a significant relationship for the horned lizard.

Boundaries between the natural desert and anthropogenic landscapes evaluated here were of two types. One was adjacent to a suburban golf course community, but separated by a well used four-lane road with curbs. The other boundary was adjacent to abandoned agricultural fields with tree rows surrounding each parcel, and was separated by a low use, two-lane road without curbs. The abundance of flat-tailed horned lizards, round-tailed ground squirrels, desert kangaroo rats and harvester ants differed between habitats adjacent to the two boundary types (Table 5.2). There was insufficient loosely compacted sand habitat along the

Table 5.3 Results of linear regressions, with species as the dependent variable and two habitat metrics as the independent variables, included here as two separate one-variable models and together as a two-variable multiple regression model. Regression coefficients, R^2, and p-values are included.

Species		*Edge distance*	*Sand compaction*	*Edge distance and sand compaction*
Flat-tailed horned lizard	p	< 0.0001	< 0.0001	< 0.0001
	R^2	0.345	0.127	0.406
	Regression Coefficient	0.003	-0.241	.003/-0.169
Desert kangaroo rat	p	0.04	0.952	0.108
	R^2	0.038	< 0.0001	0.04
	Regression Coefficient	0.003	-0.669	0.001/-0.643

suburban golf course boundary to support fringe-toed lizards, so they were not included in this analysis. For those species other than the horned lizards, the differences in abundance between the boundary types were reflected in a generalized response across the plots. For the horned lizards there were also differences in abundance with respect to the Preserve edge. No horned lizards were located closer than 100 m from the boundary adjacent to the suburban landscape; here lizard abundance didn't reach an apparent asymptote until 200 m from the preserve edge (Fig. 5.3). Some horned lizards were located right to the edge of the boundary along the abandoned agricultural fields. Abundance appeared asymptotic 100 m from the preserve edge.

Discussion

The aeolian sand species that demonstrated negative responses to anthropogenic boundaries included Coachella Valley fringe-toed lizards, flat-tailed horned lizards, and desert kangaroo rats. Of those species, data for the horned lizards were the most consistent from the standpoint of different scales (edge versus core plots and within-plot edge distances) and linear regression results. For the kangaroo rat and fringe-toed lizard edge effects were disclosed only when using the finer scale, within-plot analyses and relatively weak regression results. This pattern may be explained by environmental variation associated with Preserve habitat boundary. Historic road grading created low berms along the road-Preserve boundaries. Rare flood events create pooled standing water and silt deposition along those berms, resulting in significantly more compacted sediments within 10-20 m of that boundary. Fringe-toed lizards avoid increased sand compaction (Barrows 1997), and their patterns of abundance showed a strong association with sand compaction in our analyses. The edge effect for both the fringe-toed lizard and desert kangaroo rat appeared to be confined to < 25 m from the Preserve boundary, coincident with the effects of roadside berms.

Flat-tailed horned lizards are found well within the sand compaction conditions found throughout the nine edge plots., Edge effects for this species were measured up to 150 m from the habitat boundary, well beyond the impact of the roadside berms. The 150-m edge effect effectively eliminates over 80 ha (150 m x 5500 m along the current boundary of the Preserve adjacent to roads) of habitat for this species This lizard's range has been reduced and fragmented in recent years (Turner and Medica 1982) and this preserve may represent the only remaining habitat for flat-tails in the northern one-third of their original distribution. The loss of an additional 80 ha, which is roughly 10% of its remaining occupied habitat in the Coachella Valley is not trivial. Deciphering causal factors for the flat-tail's absence along the preserve boundary may provide important directions for future management and preserve design strategies. Three non-exclusive hypotheses were evaluated to explain this edge effect.

1) <u>Road Mortality – Road Avoidance Hypothesis</u> - Like many reptiles, flat-tailed horned lizards will use the margins of paved roads, most likely for thermoregulation (Norris 1949, Turner and Medica 1982). Impacts of roads on wildlife populations include direct mortality and road avoidance (Forman and Alexander 1998). If there is a road impact here we would expect the response from the lizards to be stronger adjacent to larger, busier roadways. In fact, we found consistent differences in lizard-edge relationships between edges adjacent to a busy four-lane road and a less used two-lane road. While edge effects were apparent along each road type, lizards adjacent to the four-lane road demonstrated a more pronounced and abrupt edge effect than those along the two-lane road, and so the data are consistent with a road effect hypothesis. However, these results are confounded by differences in the type of anthropogenic landscape beyond each roadway. The four-lane road was bounded by a suburban golf course community whereas the two-land road was bounded by abandoned agricultural fields. The former represented a "hard" boundary with no usable habitat for the lizards across the road, whereas the latter was a "softer" boundary with potential lizard habitat on the other side. Flat-tailed horned lizards were observed to occasionally cross the two-lane road to gain access to habitats off of the preserve, indicating the road was not a habitat feature avoided by the lizards. Analyzing variance in the distribution of the lizards associated with a road effect versus an adjacent habitat effect was not possible with the existing road-habitat configuration.

If there is a road effect, we would also expect that effect to be observed in other sympatric reptiles. Rosen and Lowe (1994) reported a strong road effect in a congeneric species (*C. palarostris*) of the shovel-nosed snake included in our analyses. The lack of a clear edge effect for this, or any of the other reptiles studied here, failed to support a road mortality hypothesis.

2) <u>Invasive Alien Ant Hypothesis</u> - Flat-tailed horned lizards' prey is almost exclusively harvester ants (Pianka and Parker 1975, Turner and Medica 1982). The reduction in

harvester ants from 2003 to 2004 in the aeolian sand habitat, which coincided with a similar reduction in flat-tails, supports a hypothesis that the population dynamics of these two taxa are linked. A potential reduction in the abundance of this prey due to a potential invasion of exotic ants or other habitat changes along the habitat edge would have a direct impact on horned lizard populations.

Suarez and Case (2002) and Fisher et al. (2002) have identified the invasion of non-native Argentine ants (*Linepithema humile*) as a leading factor in the disappearance of coast horned lizards (*P. coronatum*) from fragmented habitats in coastal southern California. Suarez et al. (1998) described Argentine ants being able to invade up to 100 m into semi-arid natural habitats, greatly reducing native ant populations within that same 100 m belt. Coast horned lizards that were limited to Argentine ants for prey had negative or zero growth rates, and so could not maintain populations unless native ant populations were present (Suarez et al. 2000, Suarez and Case 2002).

Although Argentine ants were known to occur in adjacent suburban golf course communities, none (nor any other non-native ant species) were collected within any plots on the Thousand Palms Preserve. The extreme aridity of this habitat may be a barrier to invasion of ant species otherwise problematic to more mesic habitats. These data, and the lack of any edge effect apparent in the native harvester ants, indicate that alien ant invasions are not a cause for the observed edge effect in the horned lizard population.

3) Enhanced Predation Hypothesis - Increased predation along habitat edges is often identified as a causal factor for reducing nesting success for birds along forest edges (Andrén et al. 1985, Wilcove 1985, Angelstam 1986, Andrén and Angelstam 1988, Burkey 1993, Aquilani and Brewer 2004). If increased levels of predation along the habitat margins are responsible for the reduced flat-tail numbers there, then increased numbers of predators should be evident.

Comparing edge versus core plots, counts of loggerhead shrikes were consistently higher on edge of the aeolian sand habitat. The higher numbers of shrikes at edge plots versus core locations in our study area was consistent with an enhanced predator hypothesis. However, if predation rates are an important causal factor, then why were other species not similarly impacted? Of the six vertebrate species measured, three are nocturnal and so would not be subjected to predation pressure from the diurnal shrikes. Of the diurnal potential prey species, the ground squirrel's large size puts them well outside of the prey range of shrikes. The two lizards are within the shrikes' prey size, and flat-tailed horned lizards are regularly preyed on by shrikes (Young et al. 2004). Whereas both lizards are cryptically colored, flat-tailed horned lizards are slower moving and often respond to threats by remaining motionless (Norris 1949). Fringe-toed lizards respond to threats by running extremely fast or diving into the loose sand (Stebbins 1944), thus often becoming unavailable to a pursuing predator.

Although predators were not quantified in 2002, flat-tailed horned lizards were commonly observed being preyed upon by American kestrels during site visits that year. Carcasses of marked horned lizards that had disappeared from study plots were located 0.7 km away in a palm tree planted on the edge of a golf course and frequented by kestrels as a roost site. In 2003 and 2004 when predator occurrence was quantified, there were few observations of kestrels, but shrike observations were common. While kestrels and shrikes are native to the deserts of southern California, their abundance in the sand dune habitats of the Coachella Valley is enhanced by suburban development. In a pre-development landscape there were no trees growing in or around the Coachella Valley sand dunes. American kestrels are obligate hole or ledge nesters. Whereas there were once no nest sites for kestrels within 10 km of the dunes, today palm trees and other exotic vegetation planted in the neighboring suburban developments provide abundant nest sites on ledges formed by the large leaf petioles and in the thick "skirts" of dead palm leaves that remain attached to the bole. While shrikes will nest in native shrubs, trees in suburban areas as well as tree windbreaks planted at the margins of now abandoned agricultural fields provide more sheltered nest sites. Power poles bordering the preserve provide elevated perch sites for both the kestrels and shrikes to see prey and then launch their hunting sorties. Flat-tailed horned lizards are being subjected to levels of predation along edges they would not likely have experienced in a pre-development landscape.

By collecting data on multiple species from multiple trophic levels we have rejected the alien ant hypothesis and found support for the predation hypothesis; however the strength of the road affect hypothesis was equivocal. Edge effects are often idiosyncratic, being dependent on the matrix within which the habitat patch occurs (Donovan et al. 1997, Haila 2002, Kristan et al. 2003, Unfried 2003). The specific land uses within the matrix surrounding the Thousand Palms Preserve facilitated the occurrence of small, diurnal, avian predators.

Dynamics of the flat-tailed horned lizard population occupying a 100-200 m boundary region of the available habitat appears to have shifted from a bottom-up process where the lizard numbers are regulated by native ant abundance, to a top-down process where the lizards are limited by predation, and possibly road mortality. This shift in regulatory processes may contribute to a habitat "sink" (Pulliam 1988) along the preserve boundary. For 2003 and 2004 combined, the horned lizards' mean reproductive success ranged from 0 – 0.2 hatchlings/adult at distances from 0 to 150 m from the habitat edge; at 200 m from the edge and in core plots, mean reproductive success averaged 0.8 hatchlings/adult (Barrows, unpubl. data). Without immigration from the preserve core, flat-tailed horned lizards may not be able to sustain populations in the boundary region.

These results demonstrate the utility of community based research designed to evaluate hypotheses regarding processes that regulate the abundance of species (Barrows et al. 2005). Rather than having broad, multispecies impacts from indeterminate causes, boundary effects here were found to have a narrow scope and likely causes were identified. These findings can allow managers to focus adaptive management strategies aimed at reducing the boundary effect for flat-tailed horned lizards and so improve the viability of this remnant population.

In the face of increasing suburban expansion into natural desert communities in the southwestern U. S. and elsewhere in arid regions of the world, managers otherwise face decisions with little or no baseline from which to predict species responses.

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CHAPTER 6

CONCLUSIONS AND RECOMMENDATIONS

The primary goal of the Center Conservation Biology's (CCB) participation in the Coachella Valley Multiple Species Habitat Conservation Plan (CV MSHCP) has been to develop and test a framework for evaluating the biological efficacy of multiple species conservation programs. This is a deceptively complex endeavor, as it requires the ability to distinguish the symptoms of threats to the viability of a species from the otherwise natural, non-equilibrium dynamics that species' populations undergo on a regular basis. It also requires that the ability to extract species specific population data in order to provide compliance to the Endangered Species Act, while at the same time focusing at larger scale potential stressors to ecosystem function.

The multiple species monitoring framework drafted by the CCB has created a means of applying the scientific method to monitoring, changing it from enumeration of individuals within a population, to a hypothesis driven, and scientifically sound process. Rather than reporting how many of species X or Y there were in year Y or Z, our framework creates predictive models as how those species should react to environmental change. The "Rosetta Stone" of biological monitoring is the identification of meaningful thresholds as to if and when adaptive management should be employed. A simple annual count will not alone provide insights as to what constitutes a meaningful threshold. Predictive models can. When the models incorporate known threats then population metrics can be compared to modeled population behavior in response to those threats. If responses are inconsistent with modeled cause and effect predictions, then focused research can be initiated to improve the models or identify previously unrecognized threats. In any case there will be an unambiguous management response dictated by the data. We are still in the formative stages of creating a multiples species monitoring framework. Our models a largely conceptual, but through time the will become increasingly quantitative. Importantly, the models are heuristic, promoting further learning as to the nature of the drivers of the dynamics of these natural systems.

Over the past four years we have developed protocols for surveying 14 of the focal species covered under the CV MSHCP. These protocols are community based, collecting data on tens if not hundreds of additional species so that changes in the abundance of those focal species can be put in the context of a larger species assemblage. At the same time, the protocols have demonstrated sufficient precision to be able to statistically tease out the effects of even minor stressors and therefore begin the process of developing those quantitative models. The community based nature of our sampling regimes easily allows us to scale our data from landscape to community to population levels. Stressors can have differential impacts at each of these levels, and so having appropriately scaled data is essential to identify when those stressors have effects that warrant management responses.

These data also allow us to evaluate the California Department of Fish and Game's Natural Community Conservation Plan (NCCP) for this region as it is, by definition, community based.

An important strength of our monitoring framework is its hypothesis driven approach. Not only does this give context and meaning to data, but it promotes the application of increasingly sophisticated science. In the near future there are opportunities to incorporate a wide array of remote sensing tools to strengthen our understanding of these natural systems. With future involvement in the development of our monitoring framework we hope to:

- Use archived multi-spectral imagery to evaluate the trajectory of weed invasions in the Coachella Valley. Some of these weeds have been present here since 1927 or before. These weeds reach "plague" levels only in wetter rainfall years. Are these weeds expanding their distribution with each wet year, or have they reach limits imposed by their own adaptive capabilities? What influence does air pollution, primarily nitrogen deposition, have in providing a competitive advantage to weed invasions here? What are the scope of impacts the weeds have on biophysical processes that maintain the levels of biodiversity in this region? How effective are weed control efforts?

- Many, though not all, of the honey mesquite thickets in the Coachella Valley appear to have reduced their extent in recent decades. Is this decline associated with prolonged drought or is it due to reduced water table levels? Can water isotope analysis discern between the origins of the water utilized by the mesquite roots? We need to install soil moisture data loggers within the root levels of healthy and less healthy mesquite to more fully explore these questions.

- Of all species monitored over the past four years, flat-tailed horned lizards are the only focal species to have shown a precipitous decline. This decline has been coupled with a decline in harvester ants, their preferred food. What is driving the ant decline? Is there a competitive relationship here between harvester ants and rodents, as has been shown elsewhere? What are primary factors influencing harvester ant dynamics?

- Brown-headed cowbirds occur at all riparian bird habitats in the Coachella Valley. Are they having a measurable impact on community structure and recruitment of focal species? How effective is trapping versus nest monitoring and egg addling for controlling cowbird impacts?

This is not an exhaustive list of next steps, but it illustrates the scope and complexity of the questions that need to be addressed in order to provide an appropriately precise barometer of the condition of Coachella Valley's biodiversity. In each case experiments will need to be designed and in some instance new technologies may need to be developed. In moving biological monitoring from a mundane collection of data with no context to a scientific process we are also increasing the expectations for an outcome that truly informs and directs adaptive management activities. We are also increasing the levels of expertise needed to implement this sort of framework.

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APPENDIX 1
COVERED SPECIES UNDER THE CV MSHCP

Source: Coachella Valley Association of Governments. 2004. *Draft Coachella Valley MSHCP*. Retrieved from: http://www.cvag.org/.

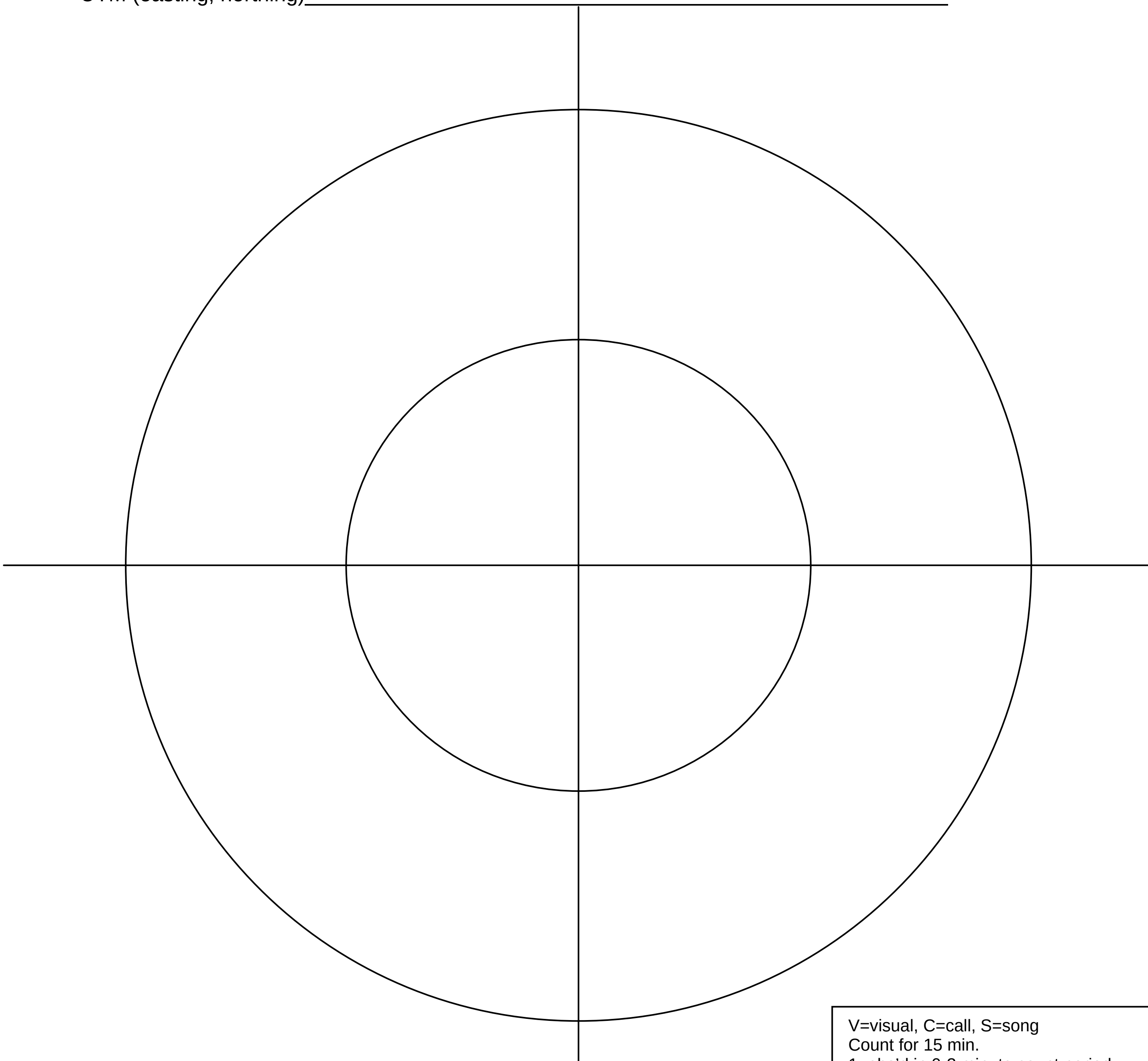
Plants	
Mecca aster	*Xylorhiza cognata*[1]
Coachella Valley milkvetch	*Astragalus lentiginosus* var. *coachellae* [FE,SE]
Triple-ribbed milkvetch	*Astragalus tricarinatus* [FE]
Orocopia sage	*Salvia greatae*[1]
Little San Bernardino Mountains linanthus	*Linanthus maculates* (or *Gilia maculate*)[1]
Invertebrates – Insects	
Coachella Valley giant sand-treader cricket	*Macrobaenetes valgum*
Coachella Valley Jerusalem cricket	*Stenopelmatus cahuilaensis*
Fish	
Desert pupfish	*Cyprinodon macularius* [FE,SE]
Amphibians	
Arroyo toad	*Bufo californicus*[FE, SSC]
Reptiles	
Desert tortoise	*Gopherus agassizii*[FT, ST]
Flat-tailed horned lizard	*Phrynosoma mcallii* [SSC]
Coachella Valley fringe-toed lizard	*Uman inornata*[FT, SE]
Birds	
Yuma Clapper Rail	*Rallus longirostris yumanensis*[FE, ST, SFP]
California Black Rail	*Laterallus jamaicensis*[ST, SFP]
Burrowing Owl	*Athene cunicularia*[SSC]
Southwestern Willow Flycatcher	*Empidonax traillii extimus*[FE, SE]
Crissal Thrasher	*Toxostoma crissale*[SSC]
LeConte's Thrasher	*Toxostoma lecontei*[SSC]
Least Bell's Vireo	*Vireo bellii pusillus*[FE, SE]
Gray Vireo	*Vireo vicinior*[SSC]
Yellow Warbler	*Dendroica petechia brewsteri*[SSC]
Yellow-Breasted Chat	*Icteria virens*[SSC]
Summer Tanager	*Piranga rubra*

Mammals	
Southern yellow bat	*Lasiurus ega* or *xanthinus*[1]
Coachella Valley round-tailed ground squirrel	*Spermophilus tereticaudus chlorus*[SSC]
Palm Springs pocket mouse	*Perognathus longimembris bangsi*[SSC]
Peninsular bighorn sheep	*Ovis candadensis nelsoni*[FE, ST, SFP]

[FE] Federal Endangered
[FT] Federal Threatened
[SE] State Endangered
[SFP] State Fully Protected
[SSC] Species of Special Concern
[ST] State Threatened
[1] No current federal or state listing; however, they are included because of their rarity and decline and potential elevation to listing status in forthcoming years.

Appendix 2.1. Point Count Data Form: Riparian Birds

Location ________________________ Date _____________Observer ______________________
Time Start_________Time End________ Point ____ of ______ Visit # ________
Temp (C) _______Wind _________________ Sky_________________________
UTM (easting, northing)____



Fly-overs: ____
____

V=visual, C=call, S=song
Count for 15 min.
1=obs'd in 0-3 minute count period
2=" 3-5 min.
3=" 5-10 min.
4="10-15 min.
Ring 1=25m, Ring 2=50m
Outside Ring 2 is any detection >50m
Magnesia Spring Canyon

Appendix 2.2. Point Count Vegetation Form: Riparian Habitat

Project: *CV-RIPARIAN* Site: Point#: Date: Obs:

UTM

HABITAT1 *RIPARIAN* HAB1% HAB2 HAB2%

LAYER	TOTAL COV%	SPECIES	REL SP COV%	HIGH AVG	LOW AVG	DBH MAX	DBH MIN	Condition/ Notes
TREE								
(>5m)								
		misc.						
PALM		*WAFI*	*100*					
SHRUB								
(0.5-5m)								
		misc.						

HERB			
(<0.5m)			
		misc.	

WATER

standing? running?

water condition or moisture content:

		Ground Cover	RelCov%	Ground Cover	RelCov%
GROUND	*100*				
		water		*leaf litter*	
		vegetation		*coarse wood*	
		sand		*trail/road (compacted)*	
		rock			
		soil			

#Snag>10 #Snag<10 #Logs: Slope: Aspect:

Invasive species:

Surrounding landscape:

Human activity:

Other:

Appendix 2.3 Riparian Habitat Assessment Form.

RIPARIAN HABITAT ASSESSMENT FORM

Point location:___________________________ Date:_____________ Observer: __________

	grade (0-3)	Description (distance from point; amount; quality)
SURFACE WATER		

	Apx. total width	Apx. total length	% of 50m circle	% cov by veg
RIPARIAN HABITAT				
Description of riparian habitat (community type; quality)				

DOMINANT VEG SPECIES	% rel cov	Avg. height	Notes/other species:
1)			
2)			
3)			
4)			

	N	S	E	W
DENSIOMETER/PHOTOS				

HUMAN ACTIVITY	grade (0-3)	Description (recent/old activity; extent; in/near habitat)
trash/litter		
damaged/removed vegetation		
vehicle tracks/presence		
paved roads/structures		
human footprints/presence		
other (describe)		

OTHER DISTURBANCES	grade (0-3)	Description (recent/old damage; extent; in/near habitat)
cattle tracks/presence		
flood damage		
fire damage		
other (describe)		

INVASIVE NON-NATIVES	grade (0-3)	Description (species; extent; in/near habitat)
tamarisk		
arundo		
fountain grass		
other shrubs and trees		
other grasses and herbs		

LANDSCAPE (other habitat types <50m; known habitat types and disturbances 50m-1km; description):

OTHER NOTES:

(grades: 0=absent, 1=small amount, 2=moderate amount, 3=large amount or substantial)

Appendix 2.4. Locations of Bird Surveys, site ownership, dates visited, field work performed, and Target Species Detected, 2002-2004

Site	Land ownership/ management	Date visited	Field work performed[1]	Target riparian bird species observed
Andreas Canyon	Agua Caliente Tribe	5/17/02	Reconnaissance survey	Bell's Vireo
		5/18/02	Point counts (5)	Bell's Vireo Summer Tanager
		6/26/02	Follow-up visit	Bell's Vireo Summer Tanager
		6/9/03	Reconnaissance survey	No target species detected
		6/10/03	Point counts (5); first round	Bell's Vireo Summer Tanager **Brown-headed Cowbird**
		7/8/03	Point counts (5); second round	Bell's Vireo Summer Tanager **Brown-headed Cowbird**
		8/11/03	Vegetation/Arthropod sampling	Summer Tanager
		8/14/03	Vegetation/Arthropod sampling	Bell's Vireo Summer Tanager
		5/27/04	Point counts (5); first round	Bell's Vireo Yellow Warbler Summer Tanager **Brown-headed Cowbird**
		7/12/04	Point counts (5); second round; arthropod sampling	Bell's Vireo Summer Tanager
		7/15/04	Arthropod sampling	Summer Tanager
Chino Canyon	Private (Steve Nicols, Palm Springs Tramway)	4/19/02	Reconnaissance survey	Bell's Vireo **Brown-headed Cowbird**
		6/9/02	Point counts (5)	Willow Flycatcher Bell's Vireo Yellow Warbler **Brown-headed Cowbird**
		7/5/02	Follow-up survey	Bell's Vireo Summer Tanager **Brown-headed Cowbird**

Site	Land ownership/ management	Date visited	Field work performed[1]	Target riparian bird species observed
		6/5/03	Point counts (6); first round	Bell's Vireo Yellow Warbler Summer Tanager Brown-headed Cowbird
		7/3/03	Point counts; aborted due to strong winds	Bell's Vireo Yellow Warbler Summer Tanager
		7/15/03	Point counts (6); second round	Bell's Vireo Yellow Warbler Summer Tanager **Brown-headed Cowbird**
		8/25/03	Vegetation/Arthropod sampling	Willow Flycatcher Bell's Vireo Summer Tanager
		8/28/03	Vegetation/Arthropod sampling	No target species detected
		5/13/04	Point counts (6); early round	Bell's Vireo Yellow Warbler Yellow-breasted Chat Summer Tanager Brown-headed Cowbird
		6/8/04	Point counts (6); first round	Bell's Vireo Yellow Warbler Yellow-breasted Chat Summer Tanager Brown-headed Cowbird
		6/22/04	Point counts (4); first round	Summer Tanager **Brown-headed Cowbird**
		7/8/04	Point counts (4); second round	Yellow-breasted Chat Summer Tanager **Brown-headed Cowbird**
		7/21/04	Point counts (6); second round	Bell's Vireo Yellow-breasted Chat Summer Tanager
		7/30/04		No target species detected
		8/2/04	Vegetation/Arthropod sampling Arthropod sampling	Yellow-breasted Chat

Site	Land ownership/ management	Date visited	Field work performed[1]	Target riparian bird species observed
Cottonwood Springs (Joshua Tree National Park)	United States National Park Service	n/a	Surveys were not conducted in 2002	n/a
		6/15/03	Reconnaissance	No target species detected
		6/19/03	Point counts (Pts 1-2)	Willow Flycatcher **Brown-headed Cowbird**
		8/26/03	Vegetation/Arthropod sampling	Willow Flycatcher
		8/29/03	Vegetation/Arthropod sampling	No target species detected
		6/3/04	Point counts (3); first round	Willow Flycatcher **Brown-headed Cowbird**
		6/25/04	Point counts (3); second round	**Brown-headed Cowbird**
		7/26/04	Arthropod sampling	No target species detected
		7/29/04	Arthropod sampling	No target species detected
Dead Indian Canyon	California Dept. of Fish and Game	n/a	Surveys were not conducted in 2002	n/a
		7/21/03	Reconnaissance survey	No target species detected
		7/22/03	Point counts (4)	No target species detected
		9/30/03	Vegetation/Arthropod sampling	No target species detected
		10/3/03	Vegetation/Arthropod sampling	No target species detected
		n/a	Surveys were not conducted in 2004	n/a
Dos Palmas Preserve (Dos Palmas Oasis and Andres Oasis)	Bureau of Land Management	5/10/02	Reconnaissance survey	**Brown-headed Cowbird**
		6/16/02	Point counts: Dos Palmas (6), Andres (2)	**Brown-headed Cowbird**
		5/28/03	Reconnaissance	Yellow Warbler Yellow-breasted Chat **Brown-headed Cowbird**
		5/29/03	Point counts: Dos Palmas (4), Andres (2); first round	Willow Flycatcher Yellow-breasted Chat **Brown-headed Cowbird**
		6/1/03	Reconnaissance survey	Yellow-breasted Chat
		6/3/03	Point counts: Dos Palmas (6); first round	**Brown-headed Cowbird**

Site	Land ownership/ management	Date visited	Field work performed[1]	Target riparian bird species observed
				Yellow-breasted Chat **Brown-headed Cowbird**
		7/1/03	Point counts: Dos Palmas (10), Andres (2); second round	Yellow-breasted Chat **Brown-headed Cowbird**
		8/18/03	Vegetation/Arthropod sampling	Willow Flycatcher Yellow Warbler Yellow-breasted Chat **Brown-headed Cowbird**
		8/21/03	Vegetation/Arthropod sampling	No target species detected
		5/14/04	Point counts (6); early round	Willow Flycatcher Yellow Warbler Yellow-breasted Chat **Brown-headed Cowbird**
		6/1/04	Point counts (6); first round	Willow Flycatcher Yellow-breasted Chat **Brown-headed Cowbird**
		6/2/04	Point counts (6); first round	Willow Flycatcher Yellow-breasted Chat **Brown-headed Cowbird**
		6/28/04	Point counts (6); second round; arthropod sampling	Yellow-breasted Chat **Brown-headed Cowbird**
		6/30/04	Point counts (6): second round; arthropod sampling	Yellow-breasted Chat **Brown-headed Cowbird**
Magnesia Spring Canyon	California Dept. of Fish and Game	n/a	Surveys were not conducted in 2002	n/a
		7/21/03	Reconnaissance survey	**No target species detected**
		7/22/03	Point counts (6); first round	No target species detected
		9/30/03	Vegetation/Arthropod sampling	No target species detected
		10/1/03	Vegetation/Arthropod sampling	No target species detected
		n/a	Surveys were not conducted in 2004	n/a

Site	Land ownership/ management	Date visited	Field work performed[1]	Target riparian bird species observed
Mission Creek	Wildlands Conservancy	4/14/02	Reconnaissance survey	**Brown-headed Cowbird**
		4/28/02	Point counts (3) single-observer	Bell's Vireo Yellow Warbler **Brown-headed Cowbird**
		6/1/02	Point counts (3)	Willow Flycatcher Bell's Vireo Yellow Warbler Yellow-breasted Chat Summer Tanager **Brown-headed Cowbird**
		7/6/02	Follow-up survey	No target species detected
		6/6/03	Point counts (3); first round	Willow Flycatcher Bell's Vireo Yellow-breasted Chat **Brown-headed Cowbird**
		7/7/03	Point counts (3); second round	Bell's Vireo **Brown-headed Cowbird**
		8/25/03	Vegetation/Arthropod sampling	Willow Flycatcher Bell's Vireo
		8/28/03	Vegetation/Arthropod sampling	Willow Flycatcher Bell's Vireo Summer Tanager
		5/19/04	Point counts (3); early round	Willow Flycatcher Bell's Vireo Yellow Warbler **Brown-headed Cowbird**
		6/10/04	Point counts (3); first round	Willow Flycatcher Bell's Vireo
		7/7/04	Point counts (3); second round	No target species detected
		7/27/04	Arthropod sampling	No target species detected
		7/30/04	Arthropod sampling	No target species detected
Murray Canyon	Agua Caliente Tribe	6/27/02	Point counts (7)	**Brown-headed Cowbird**
		6/11/03	Point counts (7); first round	**Willow Flycatcher** Brown-headed Cowbird
		7/10/03	Point counts (7); second round	No target species detected
		8/11/03		No target species detected

Site	Land ownership/ management	Date visited	Field work performed[1]	Target riparian bird species observed
		8/14/03	Vegetation/Arthropod Sampling Vegetation/Arthropod Sampling	No target species detected
		5/28/04 7/12/04 7/16/04	Point counts (7); first round Arthropod sampling Point counts (7); second round; arthropod sampling	**No target species detected** No target species detected No target species detected
Oasis de los Osos	University of California	7/12/02	Point counts (2)	No target species detected
		5/27/03 6/26/03 8/22/03 8/25/03	Point counts (2); first round Point counts (2); second round Vegetation/Arthropod sampling Vegetation/Arthropod sampling	No target species detected No target species detected No target species detected No target species detected
		6/21/04 7/19/04 7/22/04	Point counts (2); first round Arthropod sampling Point counts (2); second round; arthropod sampling	No target species detected No target species detected **Brown-headed Cowbird**
Palm Canyon	Agua Caliente Tribe	5/17/02 6/28/02	Reconnaissance survey Point counts (7)	Summer Tanager Summer Tanager **Brown-headed Cowbird**
		6/9/03 6/13/03 6/16/03 7/9/03 8/12/03 8/15/03	Reconnaissance survey Point counts (5); first round Point counts (5); first round Point counts (10); second round Vegetation/Arthropod sampling Vegetation/Arthropod sampling	**No target species detected** Summer Tanager **Brown-headed Cowbird** Summer Tanager **Brown-headed Cowbird** **Brown-headed Cowbird** No target species detected No target species detected
		6/15/04 6/16/04	Point counts (5); first round Point counts (5); first round	Brown-headed Cowbird **Brown-headed Cowbird**

Site	Land ownership/ management	Date visited	Field work performed[1]	Target riparian bird species observed
		7/13/04 7/15/04 7/16/04	Point counts (5); second round; arthropod sampling Point counts (5); second round Arthropod sampling	Summer Tanager **Brown-headed Cowbird** **Brown-headed Cowbird** No target species detected
Pushwalla Canyon (Coachella Valley Preserve)	Bureau of Land Management and California Dept. of Parks and Recreation	n/a	Surveys were not conducted in 2002	n/a
		6/18/03 6/27/03 7/17/03 8/5/03 8/8/03	Reconnaissance survey Points counts (5); first round Point counts (5); second round Vegetation/Arthropod sampling Vegetation/Arthropod sampling	No target species detected No target species detected No target species detected No target species detected No target species detected
		n/a	Surveys were not conducted in 2004	n/a
Stubbe Canyon	Private (Dr. Jane Smith)	n/a	Surveys were not conducted in 2002	n/a
		n/a	Surveys were not conducted in 2003	n/a
		5/17/04 5/20/04 6/17/04 6/18/04 7/9/04 7/20/04 7/27/04 7/30/04	Reconnaissance survey Point counts; aborted due to strong winds Point counts (6); first round Point counts (6); first round Point counts (6); second round Point counts (6); second round Vegetation/Arthropod sampling Vegetation/Arthropod sampling	Summer Tanager Summer Tanager Summer Tanager **Brown-headed Cowbird** Summer Tanager **Brown-headed Cowbird** Summer Tanager Summer Tanager **Brown-headed Cowbird** Summer Tanager Summer Tanager

Site	Land ownership/ management	Date visited	Field work performed[1]	Target riparian bird species observed
Tahquitz Canyon	Agua Caliente Tribe	n/a	Surveys were not conducted in 2002	n/a
		6/9/03	Reconnaissance survey	No target species detected
		6/12/03	Point counts (4); first round	Summer Tanager
		7/11/03	Point counts (4); second round	No target species detected
		8/12/03	Vegetation/Arthropod sampling	No target species detected
		8/15/03	Vegetation/Arthropod sampling	No target species detected
		6/14/04	Point counts (4); first round	**Brown-headed Cowbird**
		7/16/04	Arthropod sampling	No target species detected
		7/19/04	Point counts (4); second round; arthropod sampling	No target species detected
Thousand Palms Oasis (Coachella Valley Preserve)	Center for Natural Lands Management and The Nature Conservancy	4/14/02	Reconnaissance	No target species detected
		5/5/02	Reconnaissance	**Brown-headed Cowbird**
		6/15/02	Point counts (5)	Yellow-breasted Chat **Brown-headed Cowbird**
		5/23/03	Practice point counts (5)	Willow Flycatcher Yellow Warbler **Brown-headed Cowbird**
		6/17/03	Point counts (5); first round	Willow Flycatcher Bell's Vireo Brown-headed Cowbird
		7/16/03	Point counts (5); second round	No target species detected
		8/5/03	Vegetation/Arthropod sampling	No target species detected
		8/8/03	Vegetation/Arthropod sampling	No target species detected
		5/12/04	Point counts (5); early round	Willow Flycatcher Yellow Warbler Yellow-breasted Chat **Brown-headed Cowbird**
		5/24/04	Point counts; aborted due to strong winds	Willow Flycatcher Brown-headed Cowbird
		6/11/04	Point counts (5); first round	Willow Flycatcher

Site	Land ownership/ management	Date visited	Field work performed[1]	Target riparian bird species observed
				Brown-headed Cowbird
		7/14/04	Point counts (5); second round	No target species detected
		7/29/04	Arthropod sampling	No target species detected
		8/2/04	Arthropod sampling	No target species detected
Whitewater Canyon	Private and Bureau of Land Management	4/17/02	Reconnaissance	Yellow-breasted Chat
		4/26/02	Point counts (5) single observer	**Brown-headed Cowbird**
		6/20/02	Point counts (5)	Yellow-breasted Chat **Brown-headed Cowbird**
		6/21/02	Point counts (5)	**Brown-headed Cowbird**
		5/30/03	Point counts (5); first round and reconnaissance	Willow Flycatcher Summer Tanager **Brown-headed Cowbird**
		6/2/03	Point counts (7); first round	Willow Flycatcher Yellow Warbler Summer Tanager Brown-headed Cowbird
		7/2/03	Point counts (6); second round	No target species detected
		7/14/03	Point counts (6); second round	Summer Tanager **Brown-headed Cowbird**
		8/1/03	Vegetation/Arthropod sampling	Summer Tanager
		8/4/03	Vegetation/Arthropod sampling	No target species detected
		8/8/03	Vegetation sampling	No target species detected
		9/10/03	Vegetation sampling	No target species detected
		5/17/04	Point counts (6); early round	Willow Flycatcher Yellow Warbler Summer Tanager **Brown-headed Cowbird**
		6/4/04	Point counts (6); first round	Willow Flycatcher Yellow Warbler Yellow-breasted Chat Summer Tanager

Site	Land ownership/ management	Date visited	Field work performed[1]	Target riparian bird species observed
				Brown-headed Cowbird
		6/7/04	Point counts (6); first round	Willow Flycatcher Summer Tanager **Brown-headed Cowbird**
		6/29/04	Point counts (6); second round; arthropod sampling	Summer Tanager **Brown-headed Cowbird**
		7/2/04	Point counts (6); second round; arthropod sampling	No target species detected
Whitewater Delta	Coachella Valley Water District and Torres-Martinez Tribe	5/11/02	Reconnaissance	Bell's Vireo Yellow Warbler **Brown-headed Cowbird**
		6/22/02	Point counts (6)	Bell's Vireo **Brown-headed Cowbird**
		7/30/03	Point counts (6) and reconnaissance	**Brown-headed Cowbird**
		7/31/03	Point counts (7)	Bell's Vireo **Brown-headed Cowbird**
		8/19/03	Vegetation/Arthropod sampling	No target species detected
		8/22/03	Vegetation/Arthropod sampling	No target species detected
		7/6/04	Point counts (7); first round	**Brown-headed Cowbird**
		7/23/04	Point counts (6); first round; vegetation/arthropod sampling	**Brown-headed Cowbird**
		7/26/04	Vegetation/Arthropod sampling	**Brown-headed Cowbird**
Willis Palms (Coachella Valley Preserve)	Bureau of Land Management	n/a	Surveys were not conducted in 2002	n/a
		6/17/03	Reconnaissance	No target species detected
		6/18/03	Point counts (2); first round	No target species detected
		7/16/03	Point counts (2); second round	No target species detected
		8/5/03	Vegetation/Arthropod sampling	No target species detected
		8/8/03	Vegetation/Arthropod sampling	No target species detected
		n/a	Surveys were not conducted	n/a

Site	Land ownership/ management	Date visited	Field work performed[1]	Target riparian bird species observed
			in 2004	
Willow Hole	Coachella Valley Mountains Conservancy, Center for Natural Lands Management, and Bureau of Land Management	7/12/02	Point counts (1)	No target species detected
		5/27/03 6/26/03 8/26/03 8/29/03	Point counts (1); first round Point counts (1); second round Vegetation/Arthropod sampling Vegetation/Arthropod sampling	No target species detected No target species detected No target species detected No target species detected
		6/21/04 7/19/04 7/22/04	Point counts (1); first round Arthropod sampling Point counts (1); second round; arthropod sampling	No target species detected No target species detected **Brown-headed Cowbird**

[1]Point count surveys are all double-observer, except where noted

Appendix 2.5. List of All Bird Species Detected During Riparian Surveys, Coachella Valley 2002-2004

Common name	Scientific name
Abert's Towhee	*Pipilo aberti*
Accipiter sp.	*Accipiter sp.*
Acorn Woodpecker	*Melanerpes formicivorus*
American Avocet	*Recurvirostra americana*
American Bittern	*Botaurus lentiginosus*
American Coot	*Fulica americana*
American Crow	*Corvus brachyrhynchos*
American Goldfinch	*Carduelis tristis*
American Kestrel	*Falco sparverius*
American Robin	*Turdus migratorius*
American White Pelican	*Pelecanus erythrorhynchos*
American Wigeon	*Anas americana*
Anna's Hummingbird	*Calypte anna*
Ash-throated Flycatcher	*Myiarchus cinerascens*
Barn Swallow	*Hirundo rustica*
Bell’s Vireo	*Vireo bellii*
Belted Kingfisher	*Ceryle alcyon*
Bewick's Wren	*Thryomanes bewickii*
Black Phoebe	*Sayornis nigricans*
Black Skimmer	*Rynchops niger*
Black Tern	*Chlidonias niger*
Black-chinned hummingbird	*Archilochus alexandri*
Black-chinned Sparrow	*Spizella atrogularis*
Black-crowned Night-Heron	*Nycticorax nycticorax*
Black-headed Grosbeak	*Pheucticus melanocephalus*
Black-necked Stilt	*Himantopus mexicanus*
Black-tailed Gnatcatcher	*Polioptila melanura*
Black-throated Gray Warbler	*Dendroica nigrescens*
Black-throated Sparrow	*Amphispiza bilineata*
Blue Grosbeak	*Passerina caerulea*
Blue-winged Teal	*Anas discors*
Bonaparte's Gull	*Larus philadelphia*
Brewer's Blackbird	*Euphagus cyanocephalus*
Brown Pelican	*Pelecanus occidentalis*
Brown-crested Flycatcher	*Myiarchus tyrannulus*

Common name	**Scientific name**
Brown-headed Cowbird	*Molothrus ater*
Bullock's Oriole	*Icterus bullockii*
Bushtit	*Psaltriparus minimus*
Cactus Wren	*Campylorhynchus brunneicapillus*
California Gull	*Larus californicus*
California Quail	*Callipepla californica*
California/Gambel's Quail	*Callipepla sp.*
California Thrasher	*Toxostoma redivivum*
California Towhee	*Pipilo crissalis*
Canyon Wren	*Catherpes mexicanus*
Caspian Tern	*Sterna caspia*
Cassin's Vireo	*Vireo cassinii*
Cattle Egret	*Bubulcus ibis*
Cedar Waxwing	*Bombycilla cedrorum*
Chipping Sparrow	*Spizella passserina*
Cinnamon Teal	*Anas cyanoptera*
Clapper Rail	*Rallus longirostris*
Cliff Swallow	*Petrochelidon pyrrhonota*
Common Barn Owl	*Tyto alba*
Common Ground-Dove	*Columbina passerina*
Common Moorhen	*Gallinula chloropus*
Common Poorwill	*Phalaenoptilus nuttallii*
Common Raven	*Corvus corax*
Common Yellowthroat	*Geothlypis trichas*
Cooper's Hawk	*Accipiter cooperi*
Costa's Hummingbird	*Calypte costae*
Double-crested Cormorant	*Phalacrocorax auritus*
Empidonax sp.	*Empidonax sp.*
European Starling	*Sturnus vulgaris*
Feral Pigeon	*Columba livia*
Forster's Tern	*Sterna forsteri*
Gambel's Quail	*Callipepla gambelii*
Gnatcatcher sp.	*Polioptila sp.*
Golden Eagle	*Aquila chrysaetos*
Gray Flycatcher	*Empidonax wrightii*
Great Blue Heron	*Ardea herodias*
Great Egret	*Casmerodius albus*

Common name	**Scientific name**
Great Horned Owl	*Bubo virginianus*
Greater Roadrunner	*Geococcyx californianus*
Great-tailed Grackle	*Quiscalus mexicanus*
Green Heron	*Butorides virescens*
Hammond's Flycatcher	*Empidonax hammondii*
Hermit Warbler	*Dendroica occidentalis*
Hooded Oriole	*Icterus cucullatus*
House Finch	*Carpodacus mexicanus*
House Wren	*Troglodytes aedon*
Hummingbird sp.	*Trochilidae sp.*
Hutton's Vireo	*Vireo huttoni*
Killdeer	*Charadrius vociferus*
Ladder-backed Woodpecker	*Picoides scalaris*
Ladder-backed/Nuttall's Woodpecker	*Picoides sp.*
Lark Sparrow	*Chondestes grammacus*
Laughing Gull	*Larus atricilla*
Lawrence's Goldfinch	*Carduelis lawrencei*
Lazuli Bunting	*Passerina amoena*
Least Bittern	*Ixobrychus exilis*
Lesser Goldfinch	*Carduelis psaltria*
Lesser Nighthawk	*Chordeiles acutipennis*
Loggerhead Shrike	*Lanius ludovicianus*
Long-eared Owl	*Asio otus*
MacGillivray's Warbler	*Oporornis tolmiei*
Mallard	*Anas platyrhynchus*
Marbled Godwit	*Limosa fedoa*
Marsh Wren	*Cistothorus palustris*
Mountain Chickadee	*Poecile gambeli*
Mourning Dove	*Zenaida macroura*
Nashville Warbler	*Vermivora ruficapilla*
Northern Flicker	*Colaptes auratus*
Northern Harrier	*Circus cyaneus*
Northern Mockingbird	*Mimus polyglottos*
Northern Rough-winged Swallow	*Stelgidopterix serripennis*
Nuttall's Woodpecker	*Picoides nuttallii*
Oak Titmouse	*Baeolophus inornatus*
Orange-crowned Warbler	*Vermivora celata*

Common name	**Scientific name**
Osprey	*Pandion haliaetus*
Pacific Slope Flycatcher	*Empidonax difficilis*
Phainopepla	*Phainopepla nitens*
Pied-billed Grebe	*Podilymbus podiceps*
Prairie Falcon	*Falco mexicanus*
Redhead	*Aythya americana*
Red-tailed Hawk	*Buteo jamaicensis*
Red-winged Blackbird	*Agelaius phoeniceus*
Ring-billed Gull	*Larus delawarensis*
Rock Wren	*Salpinctes obsoletus*
Ruby-crowned Kinglet	*Regulus calendula*
Ruddy Duck	*Oxyura jamaicensis*
Rufous-crowned Sparrow	*Aimophila ruficeps*
Sage Sparrow	*Amphispiza belli*
Say's Phoebe	*Sayornis saya*
Scott's Oriole	*Icterus parisorum*
Snowy Egret	*Egretta thula*
Snowy Plover	*Charadrius alexandrinus*
Song Sparrow	*Melospiza melodia*
Spotted Sandpiper	*Actitis macularia*
Spotted Towhee	*Pipilo maculatus*
Summer Tanager	*Piranga rubra*
Swainson's Thrush	*Catharus ustulatus*
Swallow sp.	*Hirundinidae sp.*
Thrasher sp.	*Toxostoma sp.*
Thrush sp.	*Catharus sp.*
Townsend's Warbler	*Dendroica townsendi*
Turkey Vulture	*Cathartes aura*
Verdin	*Auriparus flaviceps*
Warbling Vireo	*Vireo gilvus*
Western Grebe	*Aechmophorus occidentalis*
Western Kingbird	*Tyrannus verticalis*
Western Meadowlark	*Sturnella neglecta*
Western Sandpiper	*Calidris mauri*
Western Scrub Jay	*Aphelocomo californica*
Western Tanager	*Piranga ludoviciana*
Western Wood-Pewee	*Contopus sordidulus*

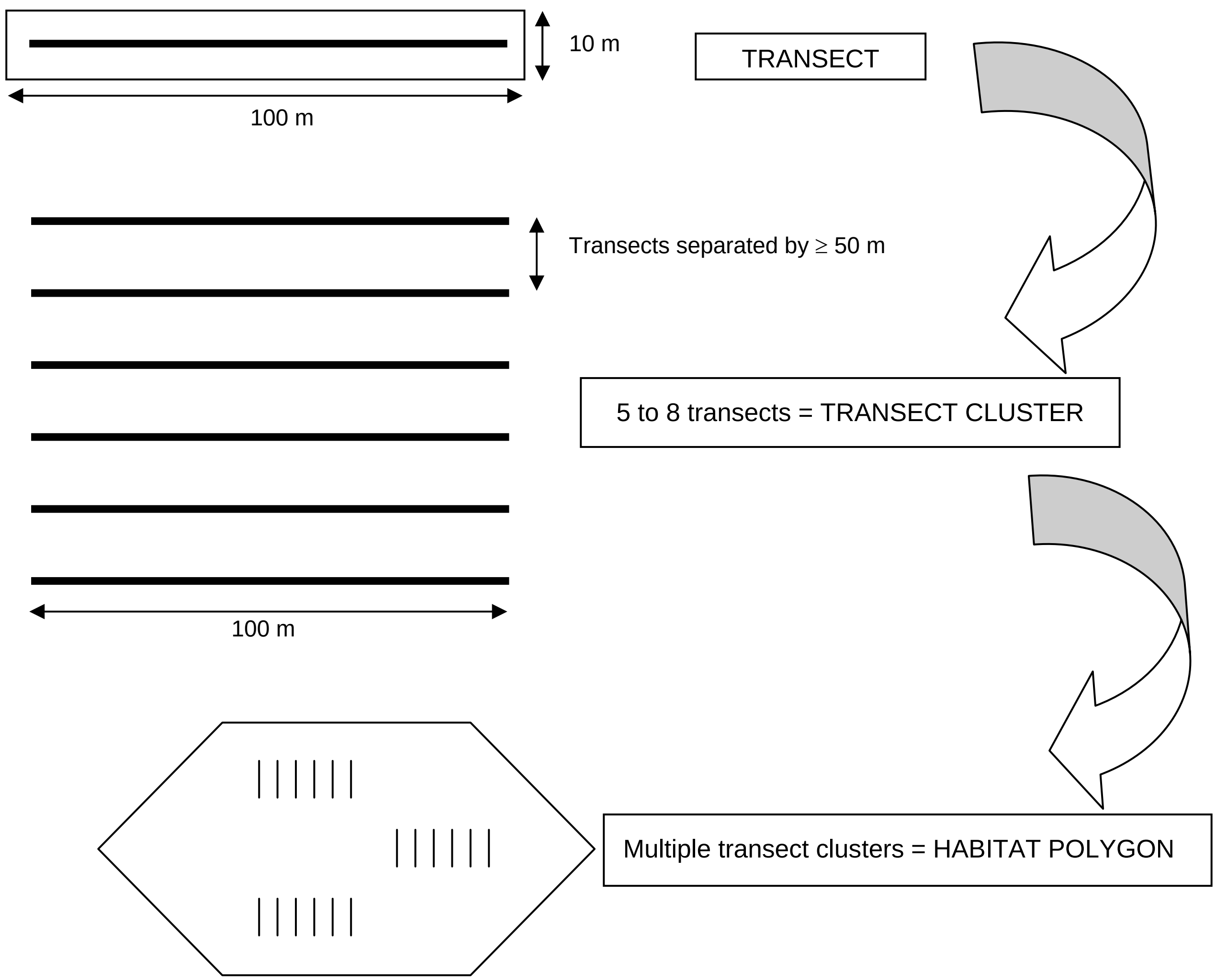
Common name	Scientific name
White-crowned Sparrow	*Zonotrichia leucophrys*
White-throated Swift	*Aeronautes saxatalis*
White-winged Dove	*Zenaida asiatica*
Willet	*Catoptrophorus semipalmatus*
Willow Flycatcher	*Empidonax traillii*
Wilson's Warbler	*Wilsonia pusilla*
Wood Stork	*Mycteria americana*
Wrentit	*Chamaea fasciata*
Yellow-breasted Chat	*Icteria virens*
Yellow-footed Gull	*Larus livens*
Yellow-headed Blackbird	*Xanthocephalus xanthocephalus*
Yellow-rumped Warbler	*Dendroica coronata*
Yellow Warbler	*Dendroica petechia*

APPENDIX 3

SAND HABITAT SURVEY PROTOCOL FOR VERTEBRATES

Sampling Scale

The objective is to sample simultaneously at multiple scales in an effort to determine which scale best characterizes the "sand communities", and which scales sensitive species are defining appropriate habitat. The scales at which we will sample includes the "transect" (0.1 ha belt transect), the "transect cluster" (2.0 - 5.0 ha), the habitat polygon (2.0 - 1000 ha), and the Coachella Valley landscape (ca. 26,000 ha).



Transect - Transect Cluster Locations

Transect clusters will be located in a random-stratified configuration within habitat polygons. Habitat polygons are presently defined by natural community maps developed through the Coachella Valley Multiple Species Habitat Conservation Planning (CVMSHCP) process. In the future those polygons may be redefined through a more refined habitat classification procedure that incorporates the data generated through these surveys.

Stratification of the cluster locations will depend in large part on specific questions being asked. If edge effects are being measured, then clusters will be located at varying distances from habitat polygon edges, or edges with respect to a hypothesized stress source (roads, urbanization). Adequate edge versus non-edge, or core, sampling sites will need to be established, that number determined by the between-cluster variances in community composition and/or abundance of target species. If a characterization of species abundance within a habitat polygon is a primary objective, then transect clusters need to be situated to capture gradients in temperature, precipitation, sand compaction, elevation, or any metric deemed important to the distribution of target species.

Initially 19 transect clusters have been established which include all the defined habitat types by the CVMSHCP, as well as the prominent east-west gradient in habitat features found within the Coachella Valley. These sites also include a series of edge clusters within the Thousand Palms Preserve unit, where edge effects are suspected to be reducing habitat availability for certain reptile species. As additional lands are acquired, additional clusters will be added; similarly, existing clusters deemed to be redundant might be eliminated.

Transects should be situated in a north-south direction. This allows optimal lighting conditions for sighting and identifying animal tracks in both directions along the transects. Otherwise, walking west to east, a low morning sun would also provide optimal lighting. Walking the opposite direction, from east to west, places the observer's shadow in front of them, obscuring track definition. East west transect placement may be appropriate when testing for edge effects, when the edge is situated in an east-west configuration. Depending on lighting conditions and the skill of the observers, those transects may be optimally surveyed walking from the west to the east.

Survey Conditions

Transect clusters should be surveyed between late May and mid August to ensure a consistent representation of all species. An additional round of surveys should also be conducted in September and October to assess reproductive success.

Transects should be surveyed during morning hours while the sun is low, creating shadows that help define animal tracks. In June, July and August, lighting conditions begin to deteriorate after 10:00 - 10:30 AM. In September and October, when surveys are aimed at quantifying reproductive success, the lower sun extends the surveys another hour. Usually two transect clusters can be completed by a survey team within optimal light conditions.

The time surveys can be initiated in the morning depends on specific conditions. The objective is to begin as early as possible after adult lizard species are active. The beginning time can be established by sighting active adult lizards in similar habitat off the transect and/or when air temperatures one centimeter above un-shaded sand exceeds 32-35^{o} C.

Surveys should not be conducted during windy conditions, when blowing sand erases fresh animal tracks, or excessively cloudy conditions when a lack of shadows make tracks difficult to locate and identify. Additionally, if there has been no wind over a several day period, tracks from multiple days can give a false impression of higher animal abundance. Unless the biologists are particularly adept at determining track ages under such conditions, surveys should be postponed until winds return. Fortunately, evening winds are common in the Coachella Valley.

Personnel

Transects are optimally surveyed with two to three biologists working as a team. An increasing number of surveyors allows for increasing task specialization. A single biologist will need to be versatile enough to identify all tracks, sight any active individuals, and record all observations. With three biologists, one can focus on tracks, one on sightings, and the other on data collection. Using three or more observers allows use of, and provides training for individuals lacking sufficient identification skills.

At the beginning of the survey period all transect surveyors will need training in animal track identification. Training should continue until surveyors are adept at identifying all vertebrate animal tacks to species, determine track ages, and identifying live animals by sight. Everyone learns at different rates, and as biologists demonstrate increased identification skills they may be given survey tasks appropriate to those skill levels.

Data Collection

Tracks

Data sheets (Appendix A) will be provided to each survey team. The survey team will attempt to identify all animal tracks within each 10 m x 100 m belt transect. We want to only count "fresh tracks". Defining what constitutes a fresh track will vary between species. Most diurnal lizards will be active as soon as light and temperature conditions are appropriate, although desert iguanas have higher thermal optima, so usually don't become active until mid morning and the onset of very warm conditions. This asynchrony in activity between the iguanas, and most other diurnal lizards means that early morning surveys will usually under-sample the iguanas. To compensate for this, iguana tracks from the previous day should be tallied -- as long as the tracks appear to have been made within the past 24 hrs.

Similarly, round-tailed ground squirrels tend to be late risers, so any track within the past 24 hours should be counted. Obviously, all nocturnal animal tracks will have been created the previous night and should be counted as long as they are not more than 24 hrs old.

Otherwise, for most diurnal lizards such as Coachella Valley fringe-toed lizards, flat-tailed horned lizards, western whiptails, long-tailed brush lizards, zebra-tailed lizards, leopard lizards and side-blotched lizards, their tracks should only be tallied if made the morning of the survey. Tracks of these lizards overprinted by nocturnal animals are one obvious indication that tracks are too old to be tallied.

Care must be taken to identify the number of individuals of each species that have occurred within the belt transect. Many of the lizards, such as the flat-tailed horned lizard, can travel hundreds of meters in a single morning, potentially crossing a transect several times. Care needs to be taken to be sure that individual is counted just once per transect (if it crosses a second transect it can then be counted for that transect as well). Surveyors should follow tracks to be sure they are distinct from other tracks before counting additional individuals of a species on a transect.

Due to the activity of the lizards trough the morning, there is a greater likelihood that a lizard will cross a transect the later in the morning that transect is surveyed. Therefore the order in which transects within a cluster are surveyed should be changed each time the cluster is sampled.

Sightings and Vocalizations

While the vast majority of data gathered will be from tracks, we want data gathered on sightings and vocalizations as well. This will aid in comparisons of sampling methods. When survey teams have three or more members, at least one person should be designated to focus on searching for animals by sight and sound. When there are less than three people, each team member needs to spend some time searching and listening for animals while otherwise searching for tracks.

A surveyor who is responsible for sightings and vocalizations can identify animals outside the 100 m x 10 m belt transect, as long as the surveyor remains within that belt while doing the searches. With binoculars the surveyor should scan all sand hummocks for basking lizards, and the tops of trees and bushes for bird silhouettes. They should also listen for singing birds.

All animal sightings and vocalizations should include the perpendicular distance that animal is from the centerline of the transect. Sightings that extend beyond the end of the transect should include the distance from the transect end point.

Transect Repetitions

Each transect cluster should be repeated, on separate days, at least six times. At the end of six repetitions the data should be checked for consistency. If there is a lack of consistency or any question of the data quality (i.e. poor sampling conditions) then additional repetitions should be conducted.

Arthropod Sampling Methodology for Sand Transects

Each transect has three markers, two endpoints and one midpoint. A pitfall is placed near each marker so all transects have three pitfalls. The traps are located with consideration for sampling microhabitats and minimizing sand fill. Since most nights produce some wind, blowing sand can fill a pitfall trap. Most traps are set on the lee side of a shrub either at its base or atop the little sand dune. All pitfalls in a transect cluster are sampled on the same morning.

The traps are dry pitfalls that collect live specimens. Each trap is constructed of two 24oz. plastic cups, the top of a 2-liter plastic soda bottle cut along the widest part of the curved edge to form a funnel, three 1" cubic wooden blocks, and a 6" square wooden board. At each pitfall location, the two cups are set one inside the other and buried so that the upper lip is flush with the ground. The funnel fits snugly inside the inner cup to create a sliding surface for arthropods to fall in and prevent their escape. The three cubic blocks are placed on the sand along the outside edge of the trap and the board is set on top weighted down with rocks or sand. The board provides cover to attract arthropods and helps prevent larger organisms from falling into the trap. A flag is placed near the trap to mark its position.

Ideally, traps are set in the afternoon and sampled the next morning in order to minimize the chance of capturing lizards. Traps are always sampled within 24 hours. The inner cup, containing the arthropods, is lifted from the outer cup and its contents inspected. The percent of sand filling the cup is recorded since a large amount of sand can negatively affect sampling. The arthropods are sampled by first looking into the cup, then slowly pouring the specimens into a white pillowcase. Large beetles are relatively easy to separate and identify, while ants and other small arthropods must be identified and tallied quickly before they scurry away. Clusters of dead warring ants must be teased apart to determine number and identity. Unknown specimens are collected in vials of alcohol for later identification in the lab. All individuals of all species are counted. Two or three technicians are required to dispense, identify, collect and record the sample accurately. Some experience and familiarity with the arthropod community is essential.

The inner cup can be set back inside the outer cup, filled with sand to prevent animals from entering, and marked by the flag. Alternatively, only one cup may be used and all materials collected after sampling. The double cup method marks the exact location for replication purposes although degradation or loss of the cups may occur. The single cup method requires resetting all the traps.

Vegetation Surveys

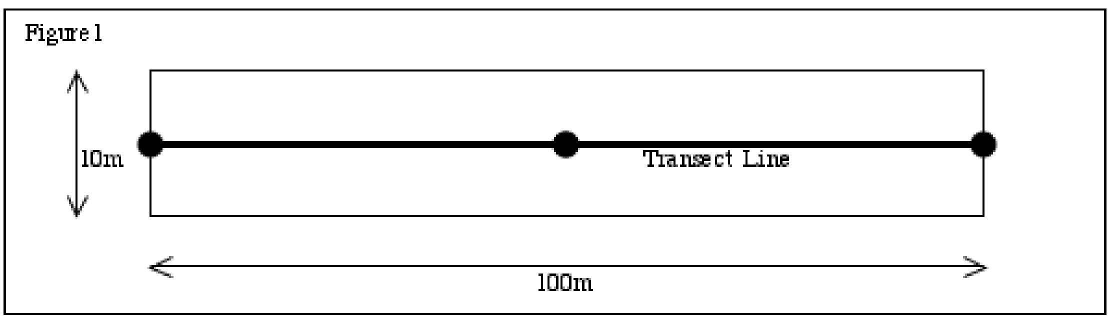
Plant Surveys

Plant surveys of the Coachella Valley Preserve consist of two main components, an annual plant survey and a perennial plant survey. The annual plant survey should be conducted before the perennial plant survey to minimize damage from foot traffic to the annual plant community. Soil compaction, and all plant surveys are to be conducted simultaneously for any cluster of transects. All surveys will be repeated once each year, as close as possible to the peak of annual plant growth.

Perennial Plant Community Survey Protocols

Sampling Area

The sampling area for each transect line is a 10m x 100m rectangle resulting in a 0. 1-hectare sample area for each transect. The transect line will bisect the rectangle lengthwise resulting in a 5m wide area on each side of the line (Figure 1). The transects are marked at end and midpoints with stakes mapped as the A, B, and C points for that transect.



All transect lines in any particular area are named and grouped in a set, from five to eight lines each, and are referred to as a cluster.

Definition of Perennial

The definition of a perennial plant, as defined by "The Jepson Desert Manual" is "A plant that is living for more than two years or growing seasons" (Baldwin et al. YEAR). The term "perennials" is defined similarly in this survey. Some perennial plants will be excluded from the perennial survey because they demonstrate habits of an annual plant in this environment and therefore will be included in the annual survey. Young perennial plants (demonstrating perennial habits) that are one year or younger will not be included in either survey. A species may only be included once, either in the perennial or the annual survey, never both.

Perennial Survey

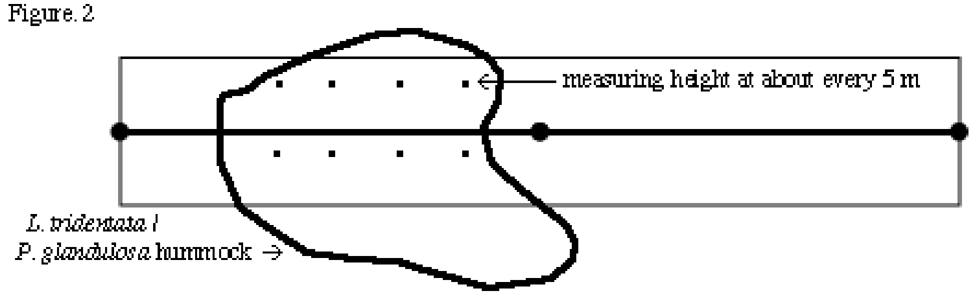
The heights of all live perennials within the sampling area are measured. A perennial is considered live if it has any living leaves. Only non-invasive methods are used to determine if a plant is dead or alive. The main trunk of a perennial plant must fall within the sampling area for it to be included in the survey. Dormant or dead plants are not included in the survey.

The survey can be conducted most effectively with two people, one to measure height and the other to record. The recorder should walk along the transect line in order to guide the person measuring. The person measuring height should proceed in a zigzag pattern along the length of the sampling area while using the recorder as a guide to keep within the sampling area. This is repeated for each transect in each cluster.

The height measurement is taken with a meter stick and recorded in centimeters. All measurements are rounded up to the nearest whole centimeter. The meter stick will be placed next to the main trunk of the plant and the tallest point of the plant will be recorded as the height. If the tallest point of the plant is dead, it will still be recorded as the plant's height. The meter stick is placed perpendicular to level ground, not parallel to the trunk of the plant. A plastic or wooden meter stick is recommended as metal heats up very quickly and causes unnecessary glare.

Special procedures are employed for *Larrea tridentata* and *Prosopis glandulosa* var. *torreyana* hummocks due the lack of distinction between individual plants. For any one hummock, a height measurement will be taken at about every five meters within the area of the hummock. Only the area of the hummock that is within the sampling area is to be included (Figure 2).

Figure. 2



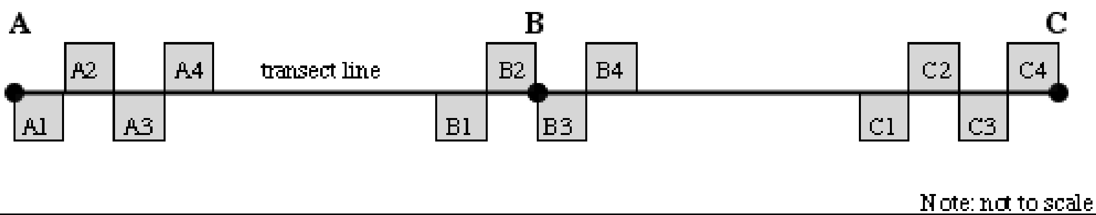
<u>Annual Plant Community Survey Protocols</u>

Sampling Area

The sampling area for each transect will consist of a series of 12, 1m^2 square samples. These samples will run along the length of the 100m transect line in a diagonally alternating pattern.

Each transect point (A-C) will contain four $1m^2$ square samples (numbered 1-4). The label of each square designates the transect point to which it belongs to and what number in the series it is. For example, "B3" is the third square of transect point B (Figure 3).

Figure. 3



Definition of Annual

The technical definition of an annual plant, as defined by "The Jepson Desert Manual" (Baldwin et al.) is "A plant that completes its life cycle (germination through death) in one year or growing season and is essential non-woody." The term "annuals" is defined similarly in this survey. All annual plants will be included in this survey. Rarely, an annual plant may live for more than one year and develop a woody growth form (i.e. Coachella Valley milkvetch), but will still be considered an annual plant. In this study, the annual definition applies to a species, not to individuals. In addition, all perennial plants that demonstrate habits of annual plants will also be included in the annual plants survey. This primarily includes perennials that survive for one year or less in this environment. Young perennial plants (demonstrating perennial habits) that are one year or younger will not be included in either survey. A species may only be included once, either in the perennial or the annual survey, never both.

Annual Survey

The square samples are delimited with a $1m^2$ quadrat. The quadrat is constructed out of 3/4" ID Polyvinylchloride piping, four 3/4" 90° elbow SXS connectors, and PVC cement. When constructing the quadrat, take note that the <u>interior</u> perimeter of the quadrat should be exactly $1m^2$.

All annual plants of the current year or growing season, whether dead or alive, are to be included in the survey. Current year/growing season is defined by the year/growing season in which the survey is taking place.

All annuals whose main shoot falls within the interior perimeter of the quadrat are to be counted. The number of plants for each species is counted and recorded separately for each $1m^2$ sample. The percentage of bare ground and the percentage of vegetation cover is approximated and recorded for each $1m^2$ sample.

The percentage of bare ground (PBG) is the percentage in the 1m^2 area consisting of bare ground. Bare ground is defined as any area in which there is a potential for plant growth to occur. This excludes, but is not limited to, ground that is covered with: a) dead or alive vegetation, up to 1 m above level ground, b) rocks greater than 3cm in length, and c) solid trash items. The PGB is determined by viewing the quadrat area from a point directly above the central point of the 1m^2 square. Estimating the PBG should take no longer than 30 seconds for each sampling area.

The percentage of vegetation cover (PVC) is the percentage in the 1m^2 area covered by vegetation. Vegetation cover is defined as any area in which there cover by vegetation of the current year/growing season. The PVC includes all above ground vegetation of any height. The PVC is determined by viewing the quadrat area from a point directly above the central point of the 1m^2 square. If this process is impeded by tall vegetation, then approximate the PVG as accurately as possible. Some techniques that may be helpful in determining the PVC of tall vegetation: 1) view tall vegetation from the ground up, 2) use the shadow of tall vegetation to assess total PVC for the sampling area. Estimating the PVC should take no longer than 30 seconds for each sampling area.

The survey can be conducted most effectively with two people. The first surveyor will be responsible for: a) identification of plant species, b) tally of plant species, and c) positioning the quadrat. The second surveyor will be responsible for: a) recording plant species tally, b) determining the PBG, c) determining the PVC and d) observing and assuring the accuracy of the first surveyor.

The first surveyor will position the quadrat according to fig. 3, starting at Point A of the transect line. The quadrat can be laid on either side of the line when sampling is initiated on each point (A-C) on the transect line. The subsequent sampling squares on that point will be determined by the position of the first sampling area. All subsequent sampling area must be laid diagonally from its neighboring squares, similar to that of a chessboard. In order to avoid bias, do not determine which side of the transect to start on based on vegetation diversity, density, species, etc. The survey can also be conducted in reverse order (point C-A) in a similar manner.

Identification and tally of species within the sampling square is performed simultaneously. The first surveyor will identify and tally plants, one species at a time, and relay this to the second surveyor, who will record it. An effective approach to this is to circle, in the sand, the plants that are being counted. This will allow the second surveyor to visually confirm the accuracy of the identification and the count. As the first surveyor is counting, the second surveyor should take time to determine the PBG and the PVC of the sampling area. At all times, the second surveyor should confirm with the first surveyor that the quadrat is laid in the correct position. This is repeated for the rest of transect until 14 samples are obtained, at which point the surveyors will switch roles and proceed on to the next transect line. This is repeated for all transects in all clusters.

Soil Compaction Survey

Sampling Area

The soil compaction survey will sample on or near (within two meters of) the transect line. Sampling will occur in a linear manner, along the length of the transect, with approximately 25 to 30 measurements taken per line.

Soil Survey

The soil compaction is measured with a penetrometer. To begin taking samples, reset the white plastic ring to zero and take note of where the zero falls on the ring. Most penetrometers will use the top edge of the ring as the zero mark. Take a sample at an undisturbed location in the sand surface by pressing lightly down on the sand with the penetrometer, until it gives way. Retract the penetrometer and record the measurement, to two decimal places, and reset the ring. This is repeated for the rest of the line.

Sand measures are most effectively conducted by one surveyor. The surveyor will start at Point A on the line and take the first measurement by the stake. This is repeated 25 to 30 times along the length of the transect. A helpful approach to obtaining between 25 to 30 measurements is for the surveyor to practice and recognize his/her own stride length. This allows for the surveyor to appropriately adjust his/her technique during sampling to obtain the correct amount of measurements. This is repeated for all transects in all clusters.