Four years of Vegetation Monitoring on the Irvine Ranch Open Space Easements in Central Orange County, CA.

2010 Final Report
DATA FROM 2007-2010

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Executive Summary

Conservation easements are an important tool for protecting natural resources in the United States. Easements are designed to protect habitat for rare and threatened species, protect water resources, maintain ecosystem services, preserve biodiversity, and create buffers and/or corridors in fragmented landscapes. Easements comprise an increasing fraction of lands set aside for conservation, in part because the costs are lower than outright land purchases. Despite their popularity, serious concerns have been raised that conservation easements may not be delivering the promised conservation benefits. In order for easements to provide tangible and substantial conservation benefits, easements must be regularly monitored.

In 2004, The Nature Conservancy Conservation Easement Working Group called for improved monitoring of conservation easements. Natural systems are difficult to monitor because they result from complex relationships among species. Further, the natural and anthropogenic forces that drive them are inherently variable across space and through time. This project provides the statistical and ecological expertise necessary to establish a rigorous and cost-effective monitoring program of the Irvine Ranch conservation easements. The main objectives of this project are to describe status and trend of natural community conservation values, describe natural spatial and temporal variability, develop and test monitoring approaches that reliably separate natural variation from trend, and to provide an exemplar of how to develop a monitoring program from initial pilot studies, variance components analysis, and power analyses.

This report is based on field data collected in Orange County between 2007 and 2010. Our sample was stratified across coastal sage scrub (CSS), grassland and chaparral communities (oak woodlands were also surveyed but were analyzed separately). These three vegetation communities are described as conservation values in the easement agreement, and also support the largest percentage of species of conservation value compared to other vegetation types. The field protocols and data collection methods have been refined through four years of rigorous testing. In 2007, we used three assessment methods (point intercept, quadrat, and visual cover) on a 20m X 50m (0.1ha) plot. In subsequent years, we used a more efficient approach with point intercept and quadrats on 50m transects.

Over the four years, we made 109 plot visits to 58 different plots distributed across six sites in the two core regions of conserved lands in Orange County. We have recorded more than 200 taxa belonging to eight different functional groups of plants. In addition we collected information on eight different types of ground cover, estimated fuel loads, and measured shrub dieback.

Vegetation communities in central Orange County reflect the long history of invasion by non-native plants. The single most abundant species is the non-native grass *Bromus madritensis*. In fact, 11 of the 20 most abundant species are non-native grasses and forbs. The most common native species were the shrubs *Eriogonum fasciculatum*, *Salvia mellifera*, and *Artemisia californica*. These reflect the large number of plots that were placed in CSS vegetation. The most common chaparral shrub was *Adenostoma fasciculatum*.

Chamise chaparral is the dominant type of chaparral on the inland portion of the reserve. As expected, shrub cover is high and cover of non-native forbs and grasses is low. *Adenostoma fasciculatum* is the dominant shrub occurring in most plots. Non-native species play a larger role in CSS than they do in chaparral. CSS tends to be more invasible than chaparral because it has smaller shrubs and larger plant interspaces. As a result, the non-native grass *Bromus madritensis* has higher cover than any of the native shrub species. Orange county grasslands have much more non-native grass cover than any other
functional group including native grass cover. In fact, *Nassella pulchra* was the only native recorded in the top ten grassland species.

Trend analysis is crucial for detecting changes in conservation value, before changes are irreversible. Furthermore, separating trend from inter-annual variability is key to understanding the trajectory of the easements under current conditions. Species richness was uniformly low across all vegetation types in 2007. Richness was much higher in 2008 and remained high for 2009 and 2010. Native forb species were the largest component of species diversity. As expected, native shrub cover is high in both CSS and chaparral communities but largely absent from grasslands. Native shrub cover is relatively stable through time, but varies significantly at multiple spatial levels. Non-native grass cover appears to be increasing over time in all three vegetation communities. Despite a high degree of spatial variability, this increase is significant and substantial.

We quantified different sources of variability by estimating the components of variance. We present variance decomposition for seven functional groups or metrics including: shrub cover, native grass cover, non-native grass cover, total species richness, native-forb richness, bare ground, and standing dead material. The seven variables we selected for analysis all showed a large degree of spatial patchiness across all habitat types. In contrast, much of the variance in non-native grass cover and both measures of species richness is attributable to time. Across most metrics, variability due to teams and methods is fairly small.

We used the pilot data and variance components analysis to design a long-term monitoring program and to evaluate power. Power calculations are an excellent tool for developing and refining monitoring programs. However they are complex and require clear goals and objectives, strong data, and substantial statistical expertise. We recommend sampling eight plots in chaparral, 22 plots in CSS and 12 plots in grasslands. This will provide adequate power to detect meaningful changes in conservation values. Based on previous field work, this would take about 17 field days (4 weeks) for an experienced two-person team.

In order for conservation easements to provide tangible and substantial conservation benefits, TNC must rely on a flexible and self-correcting feedback loop of implementation, experimentation and adjustment which is fueled by a rigorous monitoring program. This project serves the immediate need to monitor natural communities and biodiversity on the Irvine Ranch Conservation Easements. This project focuses on natural community level conservation values as proposed by the TNC Conservation Easement Working Group. In addition this project is the first step in developing strategic guidelines about when and how to monitor and addresses the necessity for economy and scientific rigor.

A study of this size and scope has the potential to serve the broader scientific and conservation communities across California, the US, and other organizations actively involved in conservation and monitoring. Although our specific recommendations are restricted to the Irvine Ranch Conservation Easements, this project is a powerful case study that clearly demonstrates the process of using advanced statistical techniques to develop and refine cost effective monitoring programs.
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Introduction
Conservation easements are an important tool for protecting natural resources in the United States. Easements are designed to protect habitat for rare and threatened species, protect water resources, maintain ecosystem services, preserve biodiversity, and create buffers and/or corridors in fragmented landscapes (Pocewicz et al. 2011). The Nature Conservancy (TNC) alone has protected over 3 million acres using easements, an area that is comparable to that of the state of Connecticut (Kiesecker et al. 2007, Fishburn et al. 2009). They enable conservation efforts to move forward in areas where, for financial or socio-political reasons, land cannot be purchased outright for the purposes of conservation. Easements comprise an increasing fraction of lands set aside for conservation, in part because the costs per acre are approximately 2/3rds lower than outright land purchases (Fishburn et al. 2009).

Serious concerns have been raised that conservation easements may not be delivering the promised conservation benefits (Bendrick et al. 2004, Morris and Rissman 2009). Little systematic quantitative information is being collected to ensure that conservation values provided by easements are protected (Kiesecker et al. 2007, Morris and Rissman 2009, Pocewicz et al. 2011). In order for easements to provide tangible and substantial conservation benefits, easements must be regularly monitored (Bendrick et al. 2004, Kiesecker et al. 2007).

“Ecological monitoring can provide a feedback loop that documents threat abatement, is useful for modifying our approach to locating and drafting easements and supports our ability to assess the success of our conservation strategies in protecting the portfolio”  (Bendrick et al. 2004)

Monitoring for Conservation
The key impetus for conservation monitoring is often to meet legal requirements or organizational directives. As a result monitoring programs are often simplistic and focus solely on ensuring the owner has not violated the legal terms defined in the conservation easement. As monitoring of easements focuses on legal compliance, ecological monitoring occurs infrequently (Kiesecker et al. 2007, Morris and Rissman 2009) and generally faces the same shortcomings of monitoring programs designed for publically managed natural lands. Too often, data are collected using multiple protocols and effort fluctuates across space and over time. Under these conditions, the data are unreliable making it impossible to understand the status and trend of conservation values (Kull et al. 2008, Lengyel et al. 2008, Schmeller et al. 2008, Fancy et al. 2009). Many such monitoring programs have been criticized as naïve, inefficient, and even dangerous because they give the illusion that something useful has been done (Peterman 1990, NRC 1995, Legg and Nagy 2006, Rahn et al. 2006).

Common Weaknesses in Conservation Monitoring
Ecological systems are difficult to monitor because they result from processes that are inherently variable across space and through time. The complex relationships among species and the natural and anthropogenic forces that drive them makes prediction extremely difficult (Atkinson et al. 2004, Bormann et al. 2007, Lyons et al. 2008). Communicating this uncertainty in partnerships involving landowners and the public is particularly difficult (Kinzig et al. 2003, Haberl et al. 2006, Walters 2007).

Monitoring failures can arise when data do not speak directly to monitoring objectives (Ferretti 2009). For example a high level of “diversity” and “function” are often used to describe desired conservation outcomes. These terms are vague and thus cannot be measured without the ideas being refined. Monitoring personnel must then use their best judgment about what measurable aspects of the
ecosystem best describe “function.” In the absence of clearly defined objectives, it is not possible to evaluate the appropriateness of the statistical design because the population or phenomena being measured has not been adequately defined.

When a monitoring program does not have a sound design and analysis plan before it is implemented, the data are often inadequate to address the key objectives of the overall program (Schmeller et al. 2008, Ferretti 2009). Often statistical advice is sought after program implementation when the data yield unexpected or inconclusive results (Marsh and Trenham 2008). At that point even the most skillful analysis may not salvage useful information from the data. As a result, data analysis is often boiled down to basic statistical summaries, and yields an overly simplistic characterization of the process of interest (Kull et al. 2008).

**Best Practices in Conservation Monitoring**

There is growing consensus that the design and implementation of a conservation monitoring program is a multi-step process (Noss 1990, Atkinson et al. 2004, Franklin et al. 2011). Successful conservation monitoring requires clear goals and objectives, an understanding of the system being studied, and comprehensive baseline data. Collection and analysis of monitoring data allows for feedback so that each of the steps can be updated in the light of new information (Figure 1).

![Flowchart](image)

**Figure 1: Best Practices in Conservation Monitoring.** The flowchart is adapted from Franklin et al. 2011. The table contains some of the key papers.

Ideally, monitoring data should be both precise and accurate. Although these terms are synonymous in common usage, they are not identical (See Table 1). The accuracy of a sample is very difficult to assess since we rarely know the true value of the variable of interest. However precision is easy to quantify (Kercher et al. 2003, Milberg et al. 2008). Although accuracy is highly desirable, trend detection -- the key to understanding if an easement is working-- relies on precision which allows relative change to be measured (Milberg et al. 2008).
Table 1: Precision V. Accuracy.

<table>
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<th>Definition</th>
<th>Example</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Precision</strong></td>
<td>The repeatability of measurements under the same conditions.</td>
</tr>
<tr>
<td><strong>Accuracy</strong></td>
<td>How close a sample estimate is to the actual population value.</td>
</tr>
<tr>
<td></td>
<td>Species richness measurements would be precise if they yielded very similar values (e.g. 20, 19, 20, 21 species) on repeated estimates made at the same location.</td>
</tr>
<tr>
<td></td>
<td>In the example above, the species richness estimates could be precise and still be inaccurate if they were far from the true value (e.g. if there were really 30 species).</td>
</tr>
<tr>
<td></td>
<td>An estimate that is precise but not accurate is said to have bias (repeatedly under- or over-estimating the true value)</td>
</tr>
</tbody>
</table>

Maximizing precision requires minimizing sampling error introduced by how and who collected the data (Ferretti 2009). Inter-observer variability has been examined by numerous studies, and there is general agreement that it is an important source of error (Kercher et al. 2003, Vittoz and Guisan 2007, Whitacre et al. 2007). This effect can be compounded by methods that are subjective, such as visual cover estimation in vegetation sampling (Milberg et al. 2008). This is particularly true if the observers are different each year (as is common with many monitoring programs) and if qualified and experienced data collectors are a rarity (Kercher et al. 2003, Schmeller et al. 2008). Yet even the most skilled observer will fail to make some observations (Milberg et al. 2008). Although “Such errors cannot be eliminated... they can be controlled” (Ferretti 2009).

In order to control error a good deal of effort must be spent up front during the design phase (Ferretti 2009). Appropriate sampling and response designs must be selected to collect data with a defensible relationship to the conservation value of interest. Then the data analysis procedure must be linked back to desired outcomes, and should include advanced statistical techniques (Atkinson et al. 2004, Kull et al. 2008, Marsh and Trenham 2008, Schmeller et al. 2008, Fancy et al. 2009, Ferretti 2009). The monitoring design must be robust across numerous observers, yet efficient while collecting the necessary data with precision (Kercher et al. 2003, Fancy et al. 2009). Knowing the degree to which differences between plots and sites can be attributed to natural spatial and temporal variation rather than methodological error is also important (Kercher et al. 2003, Schmeller et al. 2008). Data quality objectives can then be defined based on thresholds for conservation values (e.g. trend analysis) and power analysis (Milberg et al. 2008, Ferretti 2009). With this information in-hand, a standard operating procedure detailing data collection, data processing and quality checking can be defined, further reducing error (Milberg et al. 2008, Ferretti 2009).

This planning phase can be a costly investment, however having a clear statistically sound monitoring design and analysis approach saves money over the long term (Kiesecker et al. 2007, Lengyel et al. 2008, Marsh and Trenham 2008, Fancy et al. 2009).

**Monitoring Conservation Easements**

Monitoring provides an important platform for evaluating the efficacy of easement restrictions in protecting conservation values, and compatibility of management decisions made by the landowner (Bendrick et al. 2004, Kiesecker et al. 2007, Kull et al. 2008, Ferretti 2009). In order to optimize easement conditions aimed at protecting conservation values, TNC must rely on a flexible and self-correcting feedback loop of implementation, experimentation and adjustment fueled by data collection...
and analysis (Pollak 2001, Olsson et al. 2004, Lyons et al. 2008, Carpenter et al. 2009, Morris and Rissman 2009). The need for such self-assessment is highlighted by anecdotal reports of easements not serving any clear conservation purpose (Kiesecker et al. 2007). Monitoring also provides a basis upon which to assess the benefits of an easement, and as a result can help rebuff criticism (Bendrick et al. 2004, Kiesecker et al. 2007). Optimizing monitoring results is key because time, funding and expertise often comes out of a small pool (Bendrick et al. 2004, Marsh and Trenham 2008, Fancy et al. 2009, Ferretti 2009). In addition, data analysis is the crucial link between monitoring and information useful for those enforcing easements, developing new easements and other end users (Lengyel et al. 2008).

Monitoring also serves as one method for maintaining a positive relationship with the landowner, identifying issues which could potentially conflict with the terms of the easement. When other courses fail, it serves as a readily defensible basis for enforcement actions (Bendrick et al. 2004). Rigorous monitoring is required to ensure that lapses of easement terms or degradation of conservation values are detected promptly and corrected prior to major disagreements or legal action (Bendrick et al. 2004).

“The effectiveness of even the toughest easement will be severely compromised by weak or non-existent monitoring” (Bendrick et al. 2004)

Monitoring the Irvine Ranch Easements in Orange County, CA
In 2004, the TNC Conservation Easement Working Group (hereafter CEWG) called for the “engagement of scientific staff applied in developing measurable ways to track the effectiveness of different easement restrictions at protecting different biodiversity targets” (Bendrick et al. 2004). Yet the working group also recognized that scientific expertise is itself a limited commodity. The role of this project is to provide the statistical and ecological expertise necessary to establish a scientifically defensible monitoring program of the Irvine Ranch conservation easements which minimizes cost, and maximizes information gained.

This project serves the immediate need to monitor natural communities and biodiversity on the Irvine Ranch conservation easements. This project operates on the natural community level conservation values as proposed by the CEWG focusing on scrub, chaparral, oak woodland and grassland communities. In addition this project is the first step in developing strategic guidelines about when and how to monitor as recommended by Kiesecker et al. (2007) and addresses the necessity for economy and scientific rigor.

This report satisfies the requirement for documentation of the monitoring process. The report uses rigorous field-tested methods, data quality assurance procedures, and high-level statistics to assess status and trend. Information from these analyses are linked back to target conservation values on the Irvine Ranch conservation easements in Orange County. By satisfying these requirements, this work serves as a model for best-monitoring practices as proposed in the CEWG report (Bendrick et al. 2004).
The objectives of this project are to:

1. Describe the status and trend of key natural community conservation values on the Irvine Ranch conservation easements in Orange County.

2. Describe the natural spatial and temporal variability of monitoring targets in order to establish a rigorous monitoring program covering the largest number of conservation values possible.

3. Clearly communicate the role uncertainty plays in the system.

4. Design the most cost effective, repeatable monitoring program that reliably separates trend from natural temporal and spatial variation.

5. Provide a case example for using pilot studies, variance decomposition and power analysis to guide development of a monitoring program.

**TNC Conservation Values**

Ecological monitoring as called for in the CEWG must focus on units of biological diversity — “typically at the natural community or population level...” (Bendrick et al. 2004). Conservation values found on the Irvine Ranch conservation easements include open space, habitat, habitat linkages, eight natural communities, and 20 sensitive, rare or endangered species dependent upon those communities and linkages (Table 2; County of Orange 2002). This project operates at the natural community level, and targets coastal sage scrub, chaparral, oak woodlands, and grasslands. In addition to being described as conservation values in themselves, these vegetation communities also provide habitat for the majority of individual species listed as conservation values (Table 3).

Table 2: Conservation values on the Irvine Ranch conservation easements, listed by type.

<table>
<thead>
<tr>
<th>Community Or Group</th>
<th>Specific Conservation Values</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vegetation Communities</td>
<td>Coastal sage scrub, chaparral, grasslands, oak woodlands, rock outcrops, Tecate cypress forest, riparian forest, aquatic communities</td>
</tr>
<tr>
<td>Plants</td>
<td>Chaparral beargrass, many-stemmed dudleya, Catalina mariposa lily, Humboldt lily, Tecate cypress, big-cone spruce, heart-leaved pitcher sage</td>
</tr>
<tr>
<td>Large Mammals</td>
<td>Mule deer, mountain lion, bobcat, coyote</td>
</tr>
<tr>
<td>Bats</td>
<td>Mexican free-tailed bat, California mastiff bat, pallid bat</td>
</tr>
<tr>
<td>Herptofauna</td>
<td>San Diego mountain king snake, speckled rattlesnake</td>
</tr>
<tr>
<td>Birds</td>
<td>Bell's sage sparrow, coastal California gnatcatcher, black-chinned sage sparrow</td>
</tr>
<tr>
<td>Invertebrates</td>
<td>San Diego fairy shrimp</td>
</tr>
</tbody>
</table>
Table 3: Single species conservation values listed by natural communities also identified as conservation values.

<table>
<thead>
<tr>
<th>Vegetation Community</th>
<th>Species</th>
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<tr>
<td>Coastal Sage Scrub</td>
<td>Many-stemmed dudleya, Catalina mariposa lily, Bell's sage sparrow, coastal California gnatcatcher, black-chinned sage sparrow, mule deer, mountain lion, bobcat, coyote</td>
</tr>
<tr>
<td>Chaparral</td>
<td>Chaparral beargrass, many-stemmed dudleya, Humboldt's lily, bigcone spruce, heart-leaved pitcher sage, speckled rattlesnake, Bell's sage sparrow, black-chinned sage sparrow, mule deer, mountain lion, bobcat, coyote</td>
</tr>
<tr>
<td>Native Grasslands</td>
<td>Many-stemmed dudleya, Catalina mariposa lily, mule deer, mountain lion, bobcat, coyote</td>
</tr>
<tr>
<td>Oak Woodlands</td>
<td>Mule deer, mountain lion, bobcat, coyote, San Diego Mountain king snake, and several species of bats.</td>
</tr>
<tr>
<td>Rock Outcrops</td>
<td>Mexican free-tailed bat, California mastiff bat, pallid bat, speckled rattlesnake</td>
</tr>
<tr>
<td>Tecate Cypress Forest</td>
<td>Tecate cypress, heart-leaved pitcher sage, mule deer, mountain lion, bobcat, coyote</td>
</tr>
<tr>
<td>Riparian Forest</td>
<td>Mule deer, mountain lion, bobcat, coyote, San Diego Mountain king snake</td>
</tr>
<tr>
<td>Aquatic Communities</td>
<td>San Diego fairy shrimp</td>
</tr>
</tbody>
</table>

Functional Indicators for TNC Conservation Values

In 2007, Diffendorfer et al. constructed an Index of Biological Integrity based on a large multi-taxon data set collected, in part, in Orange County’s coastal NCCP reserve. Although Diffendorfer et al., acknowledged that large, multi-taxon IBI’s are expensive to build and sustain, they achieved a reproducible index using a large number of plant and animal species. They showed that the response of individual species and taxa to disturbance generally did not correlate to the responses of other species and taxa across space and through time. This refutes the use of single-species to monitor the total system. Furthermore, they described the dynamic interplay of rainfall (year) and the multi-scaled nature of coastal sage scrub and grass invasion that also diminishes the utility of single umbrella species monitoring (Diffendorfer et al. 2007).

Although they argue there is no single species that can serve as an umbrella for conservation, Diffendorfer did show that the structure and composition of the vegetation was a reliable indicator of ecosystem function for multiple taxa. Their work showed that CSS-specific taxa decrease in richness and/or relative abundance as the level of exotic plant cover increases and as native shrub cover decreases (Diffendorfer et al. 2007). The central role that vegetation metrics play suggests that vegetation monitoring is an efficient way to capture information relevant to multiple conservation values. In addition, if the richness or abundance of other species and/or conservation values are
declining, habitat information (based on vegetation metrics) is one of the most rational places to start looking for explanatory co-variates in order to develop a response to the perturbation.

**Project Time Line**

We began development and evaluation of our monitoring program in 2007 (Table 4). In the first year, we began with a modest sample size of eight plots. Each of the eight plots was sampled multiple times using several different teams on TNC easements. Plots were large, 20m X 50m rectangular plots, which utilized three different data collection methods based on several key papers published in ecology (Stohlgren et al. 1995, Keeley and Fotheringham 2005, CNPS 2007, Sawyer et al. 2009). Sampling for this project was coordinated with work on NROC NCCP lands in Orange County, and throughout MSCP lands in San Diego County, using the same techniques and with the same goals. These data in combination were leveraged to yield a large sample, and improved statistical power, which enabled the initial stages of the project to advance rapidly.

Analysis of the 2007 field data provided insight into the important sources of variation due to methods and teams. As a result, we were able to eliminate one of the three field protocols as well as improve team training (see the 2008 Final Report for this project for more detail). Data collected in 2008 improved our understanding of the strengths and weaknesses of our two protocols. Rainfall levels in 2007 were extremely low and as a result herbaceous cover and diversity were very low. In 2008, rainfall levels were closer to average and the resulting increase in species richness and cover highlighted the differences between transect and quadrat methods. Despite the increased species richness, our data showed that we dramatically reduced team to team variability by developing an expanded training program.

In 2008 and 2009, we focused on quantifying the spatial variation inherent in the system, and as a result doubled the number of plots in 2008 and again in 2009. In addition, we added a similar number of plots from NROC’s NCCP project. By the end of 2009 we had a good understanding of the spatial variability of the target communities, and had amassed a strong baseline dataset on vegetation community composition and diversity. Despite our significant progress, we still do not have an adequate handle on temporal variability. Ecosystems in southern California show pronounced inter-annual variation driven by rainfall and large changes in response to fire. Inter-annual variation will take longer to describe the full impact of these external drivers.
Throughout the project we have been developing standard operating procedures for field protocols, staff training, data quality assurance, and database management (Table 4). As expected, this is an iterative process that requires careful documentation, detailed analyses, and improved training. We have arrived at a set of simple, easily replicated procedures for field work, although there is always room for improvement in any process. In 2010, we moved our data into an expanded and revised SQL database. In the process, we began to automate some of the quality assurance processes as well as improve our ability to search, retrieve, and structure data from the rapidly expanding database. While this approach is superior, it is a difficult and lengthy process that requires additional expertise and time for adjustment and refinement.

Likewise, because temporal variation plays a dramatic effect on the system, several of our project objectives can only be achieved gradually as our understanding of temporal variation increases. These objectives include: developing a power analysis to detect trend, establishing data quality objectives, developing strategic guidelines for monitoring vegetation in Orange County and creating a generalizable template for designing monitoring programs.

Table 4: Stepwise progress made by this project toward achieving an optimal monitoring design, and a template for best-monitoring-practices. Darker shading indicates the level of completeness. Black indicates that the step has been completed.

<table>
<thead>
<tr>
<th>Step</th>
<th>2007</th>
<th>2008</th>
<th>2009</th>
<th>2010</th>
<th>2011</th>
<th>Future</th>
</tr>
</thead>
<tbody>
<tr>
<td>Minimize Cost/Effort</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Quantify Methodological Variation</td>
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<td></td>
<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Assess status/ Collect Baseline Data</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Quantify spatial variation</td>
<td></td>
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</tr>
<tr>
<td>Establish Data Collection SOP</td>
<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Maximize Precision</td>
<td></td>
<td></td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Finalize Response Design</td>
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<td></td>
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<tr>
<td>Finalize Sampling Design</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Develop Strategic Guidelines for OC*</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Develop Template for Monitoring Program Design*</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Apply high-level statistics to assess trend</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Quantify temporal variation</td>
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<td></td>
</tr>
<tr>
<td>Power Analysis</td>
<td></td>
<td></td>
<td></td>
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<td></td>
<td></td>
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<tr>
<td>Establish Data Quality Objectives</td>
<td></td>
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<td></td>
</tr>
<tr>
<td>Establish Thresholds for Conservation Values*</td>
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</tr>
<tr>
<td>Establish Data Processing and Analysis SOP</td>
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</tr>
</tbody>
</table>

*In Consultation with TNC Staff
Methods
The current study was part of a larger effort including sampling on NROC NCCP lands in Orange County, and throughout MSCP lands in San Diego County. Understanding the composition, diversity, and ecosystem function of CSS, chaparral, and grassland communities is a shared goal of all three preserve systems. We used the same techniques and field staff on all three projects which allows us to use the much larger dataset to address the key methodological questions.

Sampling Design
This year (2010) we sampled 23 plots on Irvine Ranch conservation easements, and an additional 6 plots on NCCP lands (Figure 2; Table 5). 2009 represented our peak effort at 44 plots which were nearly evenly distributed between TNC and NROC interests, and a great deal of our understanding of spatial variability resulted from the combined effort (Figure 2). Our sample was stratified across CSS, grasslands and chaparral (oak woodlands were also surveyed but will be reported on separately). These three vegetation communities are described as conservation values in the easement agreement, and also support the largest percentage of species of conservation value compared to other vegetation types.

Figure 2: Sampling effort on Irvine Ranch conservation easements and NROC/NCCP lands.
Table 5: Number of plots and plot visits across inland and coastal areas of central Orange County.

<table>
<thead>
<tr>
<th>Inland</th>
<th>Total Plots</th>
<th>2007</th>
<th>2008</th>
<th>2009</th>
<th>2010</th>
<th>Total Visits</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chaparral</td>
<td>7</td>
<td>3</td>
<td>5</td>
<td>6</td>
<td>5</td>
<td>19</td>
</tr>
<tr>
<td>CSS</td>
<td>17</td>
<td>3</td>
<td>9</td>
<td>12</td>
<td>11</td>
<td>35</td>
</tr>
<tr>
<td>Grassland</td>
<td>14</td>
<td>2</td>
<td>6</td>
<td>9</td>
<td>6</td>
<td>23</td>
</tr>
<tr>
<td>Total</td>
<td>38</td>
<td>8</td>
<td>20</td>
<td>27</td>
<td>22</td>
<td>77</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Coastal</th>
<th>Total Plots</th>
<th>2007</th>
<th>2008</th>
<th>2009</th>
<th>2010</th>
<th>Total Visits</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chaparral</td>
<td>4</td>
<td>-</td>
<td>-</td>
<td>4</td>
<td>1</td>
<td>5</td>
</tr>
<tr>
<td>CSS</td>
<td>11</td>
<td>-</td>
<td>5</td>
<td>10</td>
<td>4</td>
<td>19</td>
</tr>
<tr>
<td>Grassland</td>
<td>5</td>
<td>-</td>
<td>2</td>
<td>4</td>
<td>2</td>
<td>8</td>
</tr>
<tr>
<td>Total</td>
<td>20</td>
<td>0</td>
<td>7</td>
<td>18</td>
<td>7</td>
<td>32</td>
</tr>
</tbody>
</table>

Response Design
The field protocols and data collection methods we use have evolved over several years of rigorous testing and refinement. In 2007, we began with a 20m X 50m (0.1ha) plot, based on a modified Whittaker design (Whittaker 1977, Stohlgren et al. 1995) evaluated by Keeley and Fotheringham (Keeley and Fotheringham 2005, Deutschman et al. 2008) and implemented as described in the report for the 2007 field season. We also used three common data collection techniques: visual cover (CNPS 2007, Sawyer et al. 2009), point intercept transects and 1m² quadrat methods. All three techniques are commonly used and have been evaluated in the literature (Heady et al. 1959, Jorgensen et al. 2000, Kercher et al. 2003, Godinez-Alvarez et al. 2009). Based on post-hoc analysis, we discovered that plot size could be reduced without substantial loss of precision. In addition, visual cover estimates did not contribute any additional information but had a higher degree of inter-observer variability than the other two methods (Deutschman et al. 2008).

In 2009 and 2010, we continued using the 50m linear design and improved training methods that were first tested in 2008. We continued to focus on spatial coverage outside of areas burned in 2007, and improved our representation of grasslands in our sample. In 2011, we plan on revisiting all plots which were burned in 2007.

Response Variables and Data Handling
We defined several key variables of interest based on previous work conducted for scrub communities in Orange and San Diego Counties (Franklin et al. 2006, Deutschman et al. 2007, Diffendorfer et al. 2007, Hierl et al. 2008). We analyzed species richness as a simple measure of diversity. We also analyzed cover based on categorization of species into several functional groups based on life form (shrub, forb, grass) and origin (native, non-native). Previous work suggests that healthy vegetation communities have high species richness and cover of native shrubs and/or native grasses. Degraded communities are dominated by non-native herbaceous cover, particularly non-native grasses (Diffendorfer et al. 2007).
In previous years we analyzed the vegetation data with an emphasis on designing a long-term monitoring program. As a result we focused on describing natural variability among plots (variance decomposition) and cost. Variance decomposition allows us to better understand vegetation community dynamics, and tailor data collection techniques to best reflect and capture those dynamics (Urquhart et al. 1998, Larsen et al. 2001, Sims et al. 2006, Deutschman and Strahm 2009). These considerations are still important but have been well described in previous reports (Deutschman et al. 2008, Deutschman and Strahm 2009).

This year we have expanded and revised our statistical analysis approach. We have separated the analysis of each vegetation community (chaparral, CSS, grassland). Although these communities are variable and sometimes grade into one another, the distinction is still valuable. Management efforts differ among the three communities and their separation improves the match between our analysis and the conservation values of the easements. We added a coarse-scale spatial distinction between coastal lands (W of I-5) and inland lands (E of I-5). These lands differ in micro-climate, grazing, fire history, and ownership. Most coastal lands are owned and managed by NROC while most inland lands are part of TNC easements.

**Results**

These data were collected on 109 plot visits to 58 different plots distributed across six sites in the two core regions of conserved lands in Orange County. Most of the coastal sites are located on land which NROC has an obligation to monitor as part of the NCCP implementation agreement. Most inland plots are located on the Irvine Ranch conservation easements which TNC must monitor to ensure the easement terms are being met. Since both entities share similar goals, and desire similar conservation outcomes, we have combined the data to improve the power of the analysis. Although San Diego data was used to make initial design choices, we have excluded it from the analysis presented throughout this section.

During the four years of this study, we have recorded the cover of more than 200 taxa belonging to eight different functional groups of plants. In addition we collected information on eight different types of ground cover, estimated fuel loads, and recorded standing dead material. An exhaustive analysis of all these data (250 variables at minimum) is not practical. The analysis must be guided by the over-arching goals of the conservation programs. We analyzed total species richness, native forb richness, native shrub cover, non-native grass cover, bare ground and standing dead material. We chose these variables based on TNC’s conservation values, the goals in NROC’s Umbrella Monitoring Plan, the best available science on what constitutes ecosystem integrity in Southern California, and the commonly expressed views of land managers. We chose to evaluate richness as part of TNC’s commitment to biodiversity. We also selected non-native grass cover and native shrub cover as two relevant measures which reflect the use of vegetation by multiple animal taxa, and as responders to perturbation such as fire and grazing (Keeley 2002, Diffendorfer et al. 2007).

**Vegetation Community Status**

We assessed the status of chaparral, CSS and grasslands, by looking at dominant species and species richness in each community. This assessment was also divided between inland and coastal plots because each region has its own distinct climate and history (for example, coastal plots get more moisture, more stable temperature, and grazing was stopped long ago). Furthermore, most of the coastal plots fall within NROC’s project area, while most inland plots are located on TNC easements, so this division also reflects jurisdictional boundaries.
Common Species in central Orange County
Vegetation communities in central Orange county reflect the long history of invasion by non-native plants (Figure 3). The single most abundant species is the non-native grass Bromus madritensis. In fact, six of the 20 most abundant species are non-native grasses. An additional five species are non-native forbs including Erodium botrys and E. circutarium as well as several others. The most common native species were the shrubs Eriogonum fasciculatum, Salvia mellifera, and Artemisia californica. These reflect the large number of plots that were placed in CSS vegetation. The most common chaparral shrub was Adenostoma fasciculatum.

![Figure 3: Absolute cover of the top 20 species across all habitats and plots. Dark green indicates shrubs, light green indicates native herbs, red indicates non-native herbs, and crosshatching distinguishes grasses from forbs.](image)

Richness and Species Accumulation Curves
Species accumulation curves are helpful for assessing the relationship between richness and cover. Across all three vegetation communities studied, the majority of cover can be accounted for by a handful of common and dominant species (Figure 4). Although we recorded more than 200 taxa, more than 50% of all vegetation cover can be attributed to the top eight species. The cover of the top 44 species accounts for 90% of vegetation cover.
Most of the species richness in these communities resides in the uncommon plants on the tail of the species richness distribution. This information has significant ramifications for project design, from methodology to what qualifications are needed by an observer, based on the main objectives of the project. For example, in most habitat types, field crews will need to know about the top 20-40 species in order to account for 90% of the cover (Table 6). Estimates of cover by functional group will be insensitive to the proper identification of the less common species. On the other hand, describing biodiversity relies heavily on the myriad of species that comprise the final 10% of cover. Ultimately, the goals and objectives of the project will determine the relative importance of cover and diversity.
Table 6: Species richness in chaparral, CSS, and grassland plots.

<table>
<thead>
<tr>
<th></th>
<th>Chaparral (52 plot visits)</th>
<th>CSS (109 plot visits)</th>
<th>Grassland (46 plot visits)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Number of Taxa</strong></td>
<td>119</td>
<td>170</td>
<td>101</td>
</tr>
<tr>
<td>(Species Richness)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong># Species to reach</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>50% Cover</td>
<td>4</td>
<td>5</td>
<td>4</td>
</tr>
<tr>
<td>75% Cover</td>
<td>10</td>
<td>16</td>
<td>11</td>
</tr>
<tr>
<td>90% Cover</td>
<td>25</td>
<td>38</td>
<td>21</td>
</tr>
<tr>
<td><strong>Dominant Species</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(in order)</td>
<td>Adenostoma fasciculatum</td>
<td>Bromus madritensis</td>
<td>Bromus diandrus</td>
</tr>
<tr>
<td></td>
<td>Bromus madritensis</td>
<td>Eriogonum fasciculatum</td>
<td>Bromus madritensis</td>
</tr>
<tr>
<td></td>
<td>Salvia mellifera</td>
<td>Salvia mellifera</td>
<td>Nassella pulchra</td>
</tr>
<tr>
<td></td>
<td>Rhus integrifolia</td>
<td>Artemisia californica</td>
<td>Erodium botrys</td>
</tr>
<tr>
<td></td>
<td>Malosma laurina</td>
<td>Malosma laurina</td>
<td>Lolium multiflorum</td>
</tr>
<tr>
<td></td>
<td>Hirschfeldia incana</td>
<td>Vulpia myuros</td>
<td>Avena barbata</td>
</tr>
<tr>
<td></td>
<td>Nassella lepida</td>
<td>Erodium botrys</td>
<td>Bromus hordeaceus</td>
</tr>
<tr>
<td></td>
<td>Ceanothus tomentosus</td>
<td>Lotus scoparius</td>
<td>Brachypodium distachyon</td>
</tr>
<tr>
<td></td>
<td>Eriogonum fasciculatum</td>
<td>Erodium cicutarium</td>
<td>Erodium cicutarium</td>
</tr>
<tr>
<td></td>
<td>Eriodictyon crassifolium</td>
<td>Centaurea melitensis</td>
<td>Brassica nigra</td>
</tr>
</tbody>
</table>

We encountered 51 species in coastal chaparral. The majority of the cover was accounted for by large shrubs, and a few non-native species. Inland chaparral was more rich (103 species total), and required many more species to reach 90% cover. This is because the major shrubs occur at far lower cover than they do on the coast, giving other species an opportunity to play substantial role in the vegetation type. This striking difference could be due to differences in climate, history and fire.

CSS is more species rich than chaparral in Orange County. We encountered about 100 species in coastal CSS, and 150 species inland. As observed in chaparral, we measured far higher native shrub cover in the coastal reserves, and lower species richness. It may be that the high cover in the coastal reserves is excluding some of the smaller native forbs that account for so much of the richness inland.

The accumulation curves for grasslands are comparable to those of other vegetation types (18 and 17 species to reach 90% coastal and inland), except the top species are non-native. Native forbs do exist, accounting for most of the richness, however they fail to account for very much cover. We encountered about 69 species in coastal grasslands, 26 (37%) of which accounted for the final 1% cover. Inland we encountered 89 species in grasslands, 40 (45%) of which accounted for the last 1% of cover.
Chaparral
Chamise chaparral is the dominant type in the inland portion of the reserve. As expected, *Adenostoma fasciculatum* is the dominant shrub occurring in most plots (Figure 5). Other shrub species occur either in small, but dense patches inside the matrix of chamise chaparral or as scattered individuals across the landscape. Although they occur at far lower cover, most of the other dominant species are large native shrubs, including *Malosma laurina*, *Eriodictyon crassifolium*, and species of *Ceanothus*. *Salvia mellifera* is also a major constituent but is of a lower stature than the other shrubs. *Bromus madritensis* is the second largest constituent of cover in the chaparral of the central reserve.

The vegetation community we described as chaparral on the coast consists of many tall-statured shrubs known to regrow from root crowns after fire, such as *Rhus integrifolia* and *Malosma laurina* (Figure 5). This community could also be characterized as a different phase of coastal sage scrub, however it provides the same structural role as chamise in inland chaparral. It also contains a number of typical CSS shrubs, such as *Artemisia californica*, *Eriogonum fasciculatum* and *Salvia mellifera*. In general, coastal shrubs are tall and dense. The understory often contains a large fraction of native and non-native grasses, leaving few bare interspaces.

![Inland Chaparral](image1)

**Inland Chaparral**
- 19 Visits to 7 Plots
- Avg Cover = 90.4%
- Avg Richness = 15.6

![Coastal Chaparral](image2)

**Coastal Chaparral**
- 5 Visits to 4 Plots
- Avg Cover = 157.3%
- Avg Richness = 17.9

**Figure 5:** Absolute cover of the top 10 species in chaparral. Dark green indicates shrubs, light green indicates native herbs, red indicates non-native herbs, and crosshatching distinguishes grasses from forbs.
Coastal Sage Scrub
Non-native species play a larger role in CSS than they do in chaparral. CSS tends to be more invasive than chaparral because it has characteristically large plant interspaces and the diagnostic species are smaller in stature, and do not create an interlocked canopy capable of excluding non-native forbs and grasses. These characteristics are true of both coastal and inland areas in Orange County (Figure 6). However, the coastal reserve has dramatically higher cover for most of its major shrub species. *Eriogonum fasciculatum* is clearly dominant on the coast, although other shrubs occur at high cover as well. Inland, the cover of common shrubs is more even. In addition, *Bromus madritensis* is the fourth species in terms of cover in the coastal reserve and the first in the inland reserve. These differences, including the size/cover of shrubs, the level of invasion, and different levels of richness likely arise from differences in climactic conditions, fire and grazing history. The inland reserve is not only drier, but has experienced fire more recently, and was grazed until recently. Although we cannot tease these factors apart, the clear differences in inland and coastal CSS demonstrate that vegetation monitoring can detect differences relevant to management.

![Figure 6: Absolute cover of the top 10 species in Coastal Sage Scrub.](image_url)
Grasslands
Orange county grasslands have much more non-native grass cover than any other functional group including native grass. In fact, the only native among the top ten species was *Nassella pulchra* (Figure 7). These conditions are fairly typical of most southern California grasslands which have been heavily grazed, burnt and otherwise impacted since Spanish Missionaries arrived hundreds of years ago. Although highly invaded grasslands tend to support lower biodiversity and present a number of management challenges, they are also not devoid of conservation value, since they provide important habitat and foraging areas for animals, particularly raptors and large mammals.

Coastal grasslands are dominated by *Bromus diandrus* which typically occurs in monoculture in many areas. *Bromus madritensis* is also important. On average, coastal grasslands have about 7% *Nassella pulchra* cover. There is actually more native grass in coastal chaparral than there is in grasslands, although the species in coastal chaparral are different. *Bromus madritensis* is the dominant grass inland, however *Bromus diandrus* is much less important. *Nassella pulchra* falls in second behind *Bromus madritensis* at around 18% cover. This is actually high native grass cover for southern California grasslands in general.

![Inland Grassland](image)

**Inland Grassland**
- *Bromus madritensis*
- *Nassella pulchra*
- *Erodium botrys*
- *Avena barbata*
- *Lolium multiflorum*
- *Bromus diandrus*
- *Avena fatua*
- *Bromus hordeaceus*
- *Brassica nigra*
- *Erodium moschatum*

![Coastal Grassland](image)

**Coastal Grassland**
- *Bromus diandrus*
- *Bromus madritensis*
- *Erodium botrys*
- *Avena barbata*
- *Lolium multiflorum*
- *Brassica nigra*
- *Bromus hordeaceus*
- *Avena fatua*
- *Erodium cicutarium*

Figure 7: Absolute cover of the top 10 species in grasslands.
**Spatio-Temporal Trends**

Trend analysis is the crucial step to detecting changes in habitat value, before negative changes are irreversible. Furthermore, separating trend from inter-annual variability is key to discerning significant change from natural fluctuations, and the trajectory of the reserve under current management regimes. With a good understanding of temporal and spatial variability, it should be possible to detect changes in conservation values and key habitat metrics. In the following sections, we describe inter-annual variation in several metrics including species richness, functional group cover, and community structure.

**Total Species Richness**

Richness was uniformly low across all vegetation types in 2007 (Figure 8). Richness was so low that we saw relatively little spatial variability that year. Richness was much higher in 2008 and remained high for 2009 and 2010. The relatively low species richness in 2007 was likely due to a dry growing season which did not support the native forb component. With increased sampling and higher average richness we were able to document significant differences among plots. In 2009 and 2010, observed richness varied two and three fold between species poor plots and species rich plots.

![Figure 8: Total richness across plots and through time. Plots were sampled in coastal (open triangles) and inland sites (gray circles). Plots resampled in consecutive years are connected by lines (dotted line for coastal sites, solid line for inland sites). The thick line traces the grand mean for each vegetation community.](image)

In chaparral, temporal fluctuations in species richness appear to be fairly coherent over time (parallel lines on left panel of Figure 8). When richness goes up at one plot, it goes up at most, although the actual number of species at each plot varies. This coherence suggests that some form of rotating panel design may provide precise estimates of change with lower costs.

CSS plots displayed pronounced spatial variability (middle panel of Figure 8). In contrast to the chaparral plots, temporal changes in richness were themselves highly variable. While some plots increased in species richness, others decreased at the same time. The only period for which all plots saw a uniform increase in richness was the change from the dry growing season in 2007 to a more productive one in 2008. In general, inland plots of CSS tended to be more rich than coastal plots, indicating that spatial variation occurs on both the broad scale (region) and fine scale (plot). Greater levels of variability in CSS plots across space and through time will likely require increased replication in order to estimate trend with adequate precision.
The spatial and temporal signals for richness in grasslands are more difficult to see given our limited sampling. In 2007, we sampled only two grassland plots, one of which burned in the Santiago fire. In 2008 and 2009 we expanded our grassland effort, however due to budget cuts (and in one case lost plot markers) we were unable to follow many of these plots into 2010. Based on the data we were able to collect, richness in grasslands appears to vary across plots a moderate amount, and the temporal signal does not seem to be coherent.

**Species Richness of Native Forbs**

Much of the total species richness observed in these plots was comprised of native forbs. More than 100 species of native forb were recorded, but they totaled around 10% of cover across all plots. In contrast, the 17 species of non-native grass recorded accounted for more than 31% of total cover.

Changes in the richness of native forbs mirrored the pattern observed in total richness (or perhaps total richness tracked changes in native forb richness). Richness was low in 2007 and increased in 2008 (Figure 9). Changes in chaparral plots were coherent, but less so in CSS and grassland plots.

![Figure 9: Native Forb Richness across plots and through time. Symbols and line styles are the same as in the previous figure.](image)

**Native Shrub Cover**

As expected, native shrub cover is high in both CSS and chaparral communities but largely absent from grasslands. In fact, this is guaranteed since our designation of plots reflects the composition and structure of the shrub communities. In the following paragraphs, we only present the results from CSS and chaparral communities.

Native shrub cover is relatively stable through time, but varies significantly at multiple spatial levels. On the regional scale, coastal plots, regardless of vegetation community, almost always have more shrub cover than inland plots (Figure 10). CSS has a wider range of shrub cover values, and is more spatially patchy at both large and fine spatial scales. Interestingly, CSS and chaparral have remarkably similar total shrub cover values, despite real structural differences between the two. This is because our methods measure the cover of single species as two-dimensional metrics, and, as a result higher richness plots often have higher cover when functional groups are taken as a composite value.
The relative absence of temporal variability in shrub cover is expected, since shrubs are slow-growing perennials. The only circumstance when we would expect to see rapid temporal change in native shrub cover would be following some major disturbance, and throughout the recovery process, in which shrub cover should trend upward slowly, and probably coherently across all plots.

If the sole objective of a monitoring program was to track changes in shrub cover, than a single, high intensity, widely dispersed effort could be used with a very long interval between surveys (possibly on a decadal scale), except in areas recovering from disturbance. Alternatively a rotating panel design could be used that sampled a small fraction of the selected plots each year. The rotation of sampling effort should be designed to visit all plots every 5 or 10 years. The rotating panel design spreads the effort and cost out over multiple years which may make it more practical.

Figure 10: Shrub cover across plots and through time.

**Non-native Grass Cover**
Non-native grass cover appears to be increasing over time in all the three vegetation communities. Despite a high degree of spatial variability, this increase is significant and substantial. Although the rate of increase is different in each community—low in chaparral, moderate in CSS and fairly high in grasslands, it is generally coherent across time (Figure 11).

Non-native grass cover was relatively low in 2007, probably due to the dry growing season. It increased by a small amount in chaparral and CSS in 2008, then increased again in 2009 and 2010. Non-native grass cover in grassland communities increased from only 35% to more than 100% in 2009 and 2010. Since our point-intercept method records all species that intersect with the dowel, values over 100% are common in diverse communities with high cover. It should be noted that these data do not include plots from areas burned during the Santiago fire, and cannot be attributed to a single catastrophic event.
Figure 11: Non-native grass cover across plots and through time.

Although this is not the type of trend we hope to see in conservation areas, it is the kind that we must be able to detect in order to decide if action is necessary, and to assess different options. This trend could still be nested inside the normal range of temporal variability, as our data only extends four years. Moreover, rainfall increased throughout the study period. As a result, we cannot rule out the possibility that non-native grass cover might reset to low levels during a very dry year. It seems more likely, however, that the dramatic increase is due to real and gradual change in the system. This could result from a growing non-native grass seed bank as each subsequent generation succeeds. It could also be the result of a deepening layer of grass thatch promoting grasses and excluding other species. The ground cover data collected for this project could help determine if thatch removal is a potential management option.

**Bare Ground**

We measured the major components of ground cover using our point intercept technique. The two dominant classes were litter and bare ground. Here we present the analysis for bare ground since it may indicate potential for recruitment of native shrubs and perennial herbs. Despite the large increases in non-native grass cover, the surface area of bare ground did not change greatly through time (Figure 12). Bare ground was higher in CSS and chaparral plots and very low in grasslands. In addition, bare ground was substantially higher in inland plots than coastal plots of the same community.

Figure 12: Area of bare ground across plots and through time.
Standing Dead Material
We also started measuring standing dead material in 2009 and 2010. Standing dead material is an important structural aspect of CSS and chaparral communities. It is also possible that tracking changes in standing dead material will help explain changes in woody shrub cover.

Standing dead material was generally low (around 5% cover in chaparral and CSS; Figure 13). Plots differed significantly and may have decreased slightly from 2009 to 2010. It will be several more years before we have a long enough record of standing dead material to understand temporal dynamics.

![Figure 13: Area of bare ground across plots and through time.](image)

Variance Components Analysis
We quantified different sources of variability by estimating the different components of variance (Urquhart et al. 1998, Larsen et al. 2001, Sims et al. 2006). This variance decomposition along with the effort analysis are necessary to develop an optimal (or at least near optimal) monitoring plan and to estimate statistical power.

We present variance decomposition for seven functional groups or metrics including: shrub cover, native grass cover, non-native grass cover, total species richness, native-forb richness, bare ground, and standing dead material. The seven variables we selected for analysis all showed a large degree of spatial patchiness across all habitat types (Figure 14). There were some differences in how much variance was introduced on each scale. Plot to plot variability (fine scale) was usually substantial; however region (larger scale) was an important source of variation in both shrub cover and bare ground (pink and red bars in Figure 14).

As discussed in previous sections, the variance attributable to region is likely due to a combination of climatic conditions, fire and grazing history, all of which account for inland plots having lower shrub cover and greater bare ground. The overall spatial variability of CSS may be attributable to this community’s high species diversity, and tendency to occur in numerous phases depending on local microclimate and topological conditions.

In contrast, much of the variance in non-native grass cover and both measures of species richness is attributable to time (black bars in Figure 14). This is true regardless of the vegetation type, because these metrics are driven by annual life forms (native forbs, in the case of richness). Although chaparral...
is characterized by high and stable shrub cover, it’s herbaceous component fluctuates dramatically over time. This may be because the cover of native and non-native annuals in chaparral is low. As a result even a small change (for example going from 1% to 3%) can represent many times more cover (for example a tripling) than what was found initially. A large absolute change in a community with high richness (CSS) or high native grass cover (grasslands) may still represent a much smaller relative change from one year to the next.

Across most metrics, variability due to teams and methods is fairly small (light and dark blue bars in Figure 14). Method (point intercept versus quadrat) is many times more important to estimating species richness than it is to estimating cover, although it plays a role in both. For richness, point intercept transects pick up many fewer species, because they are sampling such a small area. Fifty ½” diameter point intersections sample a two dimensional area equivalent to 0.006m². Ten 1m² quadrats sample an area over 1500 times that. Not surprisingly point intercept transects pick up fewer small species, but return unbiased estimates for shrub cover. In grasslands, method was an important source of variance in non-native grass cover. This is likely due to layering of multiple species of non-native grasses, which tends to depress qualitative estimates as teams struggle to visualize and assess each layer in a single quad rat. The same is true for estimating shrub cover, except in addition to layering, teams must also struggle with identifying multiple grass species, many of which look similar.

Team generally accounts for relatively little variance in our data. This source of variability is not absent, but has been minimized by using experienced field crews and improving our training in each successive field season.
Figure 14: Variance decomposition for native shrub cover, non-native grass cover, and total richness. Reading across panes compares variables across a single vegetation community. Reading vertically allows for the comparison of the same variable across different vegetation communities.
**Example Power Calculations**

Power calculations are an excellent tool for developing and refining monitoring programs (Legg and Nagy 2006, Nielsen et al. 2009). Power must be high enough that it is very likely that the monitoring program can produce timely and accurate information about conservation values. Conversely, monitoring programs should be designed to avoid excessively high power, since this results from wasteful and costly over sampling (Archaux and Berges 2008). Power calculations are under-utilized in many conservation programs because they rely on accurate baseline data and the calculations are mathematically complex.

Power calculations involve a number of professional judgments and are best informed by clearly articulated goals and objectives, and conducted by an expert data analyst. First a quantifiable variable linked to the goals and objectives of the project must be selected and baseline data collected. Then the analyst must consider 7 different factors (see Table 7), using the project goals and objectives as a guide. This includes choosing an effect size based on critical thresholds for conservation values. For our purposes, effect sizes were chosen in consultation with TNC staff.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Decision Making Factors</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\alpha$ (Type I error)</td>
<td>Type I error is defined as the probability of rejecting a null hypothesis <em>(a decision based on data)</em> when the hypothesis is actually true <em>(the true state of the system)</em>. In conservation, this is (1) the risk of concluding that a change has occurred when it has not or (2) the risk of taking management action when it was not really needed. Type I error is generally set at 0.05 (=5%) in traditional statistics. However, it is often set at 0.10 or even higher in conservation applications. These larger values are warranted because the cost of inaction is extremely high. Inaction could lead to the loss of a rare species or the permanent reduction in conservation value. This is call the “cautionary principle”</td>
</tr>
<tr>
<td>$\beta$ (Type II error)</td>
<td><strong>Type II error</strong> is defined as the probability of accepting a null hypothesis <em>(a decision based on data)</em> when the hypothesis is actually false <em>(the true state of the system)</em>. In conservation, this is (1) the risk of concluding that no change has occurred when it has or (2) the risk of not taking management action when it was needed. <strong>Power</strong> is the opposite of Type II error. Power is the probability of rejecting a hypothesis when it is false. Ideally, we would like this to be very close to 100%. Type II error is generally set at 0.20 (=20%; Power = 80%) in traditional statistics. However, it is often set at 0.10 (power = 90%) in conservation applications. Low values of Type II error (and thus high power) are needed because the lack of a timely response could lead to permanent degradation of the system.</td>
</tr>
<tr>
<td>The opposite of power.</td>
<td></td>
</tr>
</tbody>
</table>

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**Table 7: Factors required to conduct a power analysis.**

- **$\alpha$ (Type I error)** is defined as the probability of rejecting a null hypothesis *(a decision based on data)* when the hypothesis is actually true *(the true state of the system)*. In conservation, this is (1) the risk of concluding that a change has occurred when it has not or (2) the risk of taking management action when it was not really needed. Type I error is generally set at 0.05 (=5%) in traditional statistics. However, it is often set at 0.10 or even higher in conservation applications. These larger values are warranted because the cost of inaction is extremely high. Inaction could lead to the loss of a rare species or the permanent reduction in conservation value. This is call the “cautionary principle”.

- **$\beta$ (Type II error)** is defined as the probability of accepting a null hypothesis *(a decision based on data)* when the hypothesis is actually false *(the true state of the system)*. In conservation, this is (1) the risk of concluding that no change has occurred when it has or (2) the risk of not taking management action when it was needed. **Power** is the opposite of Type II error. Power is the probability of rejecting a hypothesis when it is false. Ideally, we would like this to be very close to 100%.

- **The opposite of power.**
| Parameter Estimate and Variability (Mean and SD) | Power analyses are linked to the goals and objectives of a monitoring program through the choice of parameters to monitor. Often, the parameter of interest is an average of population size or cover. It can also be the rate of change over time (e.g. annual change in population size).

Calculating power requires a credible estimate of the parameter of interest as well as a measure of its natural variation. Preferably the values are based on pilot data, when available, but can also be based on best available science or expert opinion. |
|---|---|
| ES (Effect size) | The effect size is the magnitude of the change that you want to be able to detect in the parameter specified above. This can be set based on enforcement or management triggers where available. It may be based on easement terms when available, best available science, or when other information is not available, expert opinion.

Effect size is very important to the power calculation. Generally large effects are easy to detect and often result in high power. On the other hand, very small effects are extremely difficult to detect and often result in low power. |
| Type of Statistical Analysis | The power to detect change depends on the statistical analysis used. This, in turn, is determined by the nature of the monitoring program and the question being asked.

For example, a two-sample t-test could be used to test for a difference in the mean of a variable in two successive years of sampling. This would be appropriate if the two samples (year 1, year 2) were independent random samples. If the plots were chosen randomly in year 1 and then the exact same plots were revisited in year 2, a paired t-test would be appropriate. Usually, the paired t-test is far more powerful than the two-sample t-test because the statistic is based on the change observed at each plot (formally a difference). As a result, each plot serves as its own control or reference site and there is much less variability in the parameter of interest. |
| Maximum Effort Possible | It is important to compare the projected sample sizes from the power calculations to the scope of the monitoring program. Monitoring programs are often limited in total budget, field personnel, or access to remote sites. These constraints impose an upper limit on the sampling program.

Ideally, it will be possible to reach the desired power or effect size within the limits of the project. However, if the effect size or type 1 error must be increased dramatically, or power is unacceptably low, it might be preferable to select a different design or to refocus those resources elsewhere. |
In the following table (Table 8) we summarize the parameters used in the following power analyses. We made some simplifying assumptions since it was not feasible to conduct all possible power analyses. We discussed these assumptions as well as our parameter choices with TNC staff.

**Table 8: Parameters chosen for the power analyses.**

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Decision Making Factors</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\alpha$ (Type I error)</td>
<td>Type I error rate was set at 0.10 (10%). Thus, our calculations are based on having a 10% chance of concluding a change has taken place, when in fact it has not. This value is consistent with the application of power calculations in conservation.</td>
</tr>
<tr>
<td>Power (1−$\beta$) (1 - Type II error)</td>
<td>We calculated expected power (1-$\beta$) for a range of sample sizes. We generally evaluated models based on achieving 80% power (that is, type II error set at 0.20 (=20%). This value is commonly used in power calculations. In many conservation applications, it may be important to evaluate models based on achieving 90% power. We present power curves so that the reader can evaluate the model at any desired level of power.</td>
</tr>
<tr>
<td>Parameter Estimate and Variability (Mean and SD)</td>
<td>We based our calculations on mean (average) values for several parameters including measures of diversity (total species richness, native forb richness) measures of cover and composition (native shrub cover, non-native grass cover) and measures of structure (bare ground and standing dead shrub cover). Estimates of natural variability were derived from the Mean Squared Error term from the variance decomposition.</td>
</tr>
<tr>
<td>ES (Effect size)</td>
<td>Our default effect sizes was set at 20% of the observed mean. For example, if mean species richness was 25 species, then a 20% change would be the addition or loss of 5 (=0.20 * 25) species. For some parameters, this resulted in unreasonable effect sizes. Effect sizes for these parameters were adjusted after discussions with several TNC ecologists. For example, we set the effect size for bare ground at 10% absolute cover. Thus we would be able to detect a change from 15% to 25% (or 15% to 5%) absolute cover.</td>
</tr>
<tr>
<td>Type of Statistical Analysis</td>
<td>We based our power calculations on 1-sample t-tests. This makes several simplifying assumptions. First, it assumes that we are testing for a departure in next year’s data from our long-term average (taken as known without error). This assumption is optimistic and could lead to overestimating power. On the other hand, we are calculating power based on detecting a significant change in a single year. This assumption is very restrictive (conservative) and could lead to underestimating power. We feel these sources of error are largely balanced and that the resulting power curves are appropriate.</td>
</tr>
</tbody>
</table>
Power Calculations: Results
We present power calculations from six response variables across three vegetation communities. In the first section, we present the results for each response variable and compare power across the three vegetation communities. In the second section, we present the results for each vegetation community and compare power across the response variables. This facilitates comparisons among variables and vegetation communities and allows us to make decisions about minimum acceptable sample sizes for each community.

Species Richness
Species richness is a measure of biodiversity and is often thought of as an indicator of vegetation community health. In southern California shrub systems, richness is generally driven by specialist species which require high-quality habitat, but do not preclude the common species responsible for the overall vegetation structure.

We based the power analysis for total species richness on an effect size of 20% of the mean. This translated to a gain or loss of four species in chaparral, five species in CSS and four species in grasslands, which is a small but meaningful change. In order to reach 80% power, a minimum of six plots are necessary in CSS, seven in chaparral and eight in grasslands (See Figure 15).

The native forb community is highly diverse and can fluctuate dramatically over time and space. In addition, the native forb community is strongly depressed in grasslands, where non-native grasses outcompete most other species for space. As a result native forb richness provides more specialized insight by using a smaller group of more sensitive species. We chose a slightly larger effect size (25% of the mean) in order to provide guidance about a biologically meaningful change in the number of species. This is approximately two native forb species in each vegetation community (when rounded). In order to detect this change with 80% power in chaparral seven plots are needed. Eight plots are required for CSS and 12 for grasslands (See Figure 16).

Native Shrub Cover
Native shrub cover provides necessary habitat for numerous animal species, and is one of the key indicators of biological integrity described by Diffendorfer (Diffendorfer et al. 2007). Our power analysis was based on a change of 20% of the mean in chaparral and CSS, equating to 12% absolute cover change in chaparral and 10% absolute cover change in CSS. At 80% power it takes about five plots in chaparral and six in CSS to detect such a change (See Figure 17). Grasslands were omitted because they, by definition, do not have a significant number of shrubs, and shrub cover is not considered biologically important.
Figure 15: Power analysis of total richness for chaparral, CSS and grasslands. Results indicate that seven, six and eight plots respectively are needed to achieve 80% power with a type-I error rate of 10%.

Figure 16: Power analysis of native forb richness for chaparral, CSS and grassland. Results indicate that seven, eight and 12 plots respectively are needed to achieve 80% power with a type-I error rate of 10%.
Did you mean: "Non-native Grass Cover"

Non-native grass cover is another effective indicator of biological integrity (Diffendorfer et al. 2007). Non-native grass competes with native shrub recruits and forb species, and can be a significant obstacle for many native animal species. Invasion by non-native grasses can lead to wholesale degradation of habitat value and vegetation community quality. In addition, non-native grass invasion shortens the fire return interval and repeated burns contribute to a positive feedback loop ultimately leading to type conversion (D’Antonio and Vitousek 1992).

We based our power analysis on a change of 20% of the mean in all three vegetation communities. For grasslands this change is about 16% absolute cover, and is relatively easy to detect, requiring about nine plots (Figure 18). For CSS the change would be about 6% absolute cover requiring 23 plots. This discrepancy is due to the fact that CSS has far less non-native grass cover than grasslands, and is more variable in general. The smaller effect size coupled with higher natural variability requires significantly more effort. In chaparral the effect size equates to about 4% absolute cover change. This is so small it would take far too many plots than is feasible (71I). As a result, another power analysis could be conducted based on a fixed absolute change that was considered biologically meaningful (for example 10% absolute cover) or a larger percentage of the mean could be selected (for example 50 or 100% relative to the mean).

Figure 17: Power analysis of native shrub cover for chaparral and CSS. Results indicate that five and six plots respectively are needed to achieve 80% power with a type-I error rate of 10%.

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Figure 18: Power analysis of non-native grass cover for chaparral, CSS and grassland. Results indicate that grasslands can be monitored to 80% power with a type-I error rate of 10% with nine plots, and 23 plots in CSS. It is impractical to base a monitoring program on non-native grass in chaparral because the average cover of this functional group is typically low.

Bare Ground
Bare ground (here the combined cover of bare, cryptogrammic crust, and moss) is an important factor affecting vegetation communities. Bare ground represents open space available to plant species for colonization. While such spaces can encourage invasion, they also provide spaces for small native plant species to germinate and represent the large plant interspaces characteristic of undisturbed CSS and native forb-lands. Anecdotally we observe a great deal of native forb richness in areas with bare ground, even though the size of such spots may be very small. Areas with bare ground and crusts are necessary for herbaceous host plants for animal species (e.g. Plantago erecta and Quino checkerspot butterfly (Euphydryas editha quino)), and are also critical parts of the habitat structure for numerous animal taxa.

We based our power analysis on a change of 10% absolute cover. This benchmark is an example of choosing biologically meaningful effect size directly, instead of defining the effect size relative to the mean. This approach can be helpful when a resource is very rare. For example, bare ground averages about 5% in grasslands a change of 20% relative to the mean would equate to 1% absolute cover. This is an extremely small effect and likely within the background variance of the data collection process. Even if a small change was detected with adequate power, it would probably not be important enough to trigger some sort of practical corrective action. Instead, if we base our power analysis on 10% absolute change, we suspect that something truly significant has changed in the system.
Using this effect size, only three plots are needed to detect a change in grasslands because 10% absolute cover represents a 150% relative change in bare ground (Figure 19). In CSS bare ground averages about 21% cover, so a 10% change in absolute cover is a 50% relative change. As result it will only take about four plots to detect. In chaparral bare ground averages 33%, so a 10% absolute change is close to a 30% relative change in bare ground. In chaparral detecting such a change will take about eight plots.

![Bare Ground (Effect Size 10% Absolute)](image)

*Figure 19: Power analysis of bare ground cover for chaparral, CSS and grassland. Results indicate that 8, 4 and 3 plots respectively are needed to achieve 80% power with a type-I error rate of 10%.

**Standing Dead Material**
In 2009, we began recording the cover of standing dead material. Our hope was to measure the cover of shrubs succumbing to disease or some other physiological stress above and beyond the rain-drought cycle for which they are adapted. As a result, we only record dead cover if the plant has a dead primary or secondary stem, and assume that small and feathery tertiary branches are generally shed as part of a drought-deciduous adaptation.

As a result, our averages of standing dead material are relatively small (4% in CSS and 8% in chaparral), so again we are confronted with a need to arrive at a threshold for the power analysis that is both biologically meaningful and logistically feasible. We chose a 50% relative increase, which would mean detecting an additional 2% relative cover in CSS and 4% in chaparral. Ten plots are necessary in CSS, and five in chaparral (Figure 20).
An alternative way to present the results from these power analyses is to compare the sampling effort needed across all the key response variables at once. Since these variables represent different aspects of the conservation value of each community, this approach lends itself to a more holistic interpretation. In the following pages, we present the power curves for the three main vegetation communities.

**Chaparral**

In chaparral, we will have adequate power to detect changes with eight plots (Figure 21). This level of effort will allow us to detect important changes in all variables except non-native grass. As mentioned earlier, non-native grasses exist at low cover values thus the effect size used was very small. For this variable, we would need 71 plots in chaparral alone, an effort which is not logistically feasible nor biologically justifiable. Eight plots is a reasonable investment and would require about one week of effort for a two-person team with at least one experienced biologist.
Figure 21: Power analysis of multiple variables in chaparral indicate that eight plots will allow the largest numbers of variables to be collected with 80% power and a 10% type-I error rate. The vertical dashed line represents the maximum number of plots a two-person team can complete in a week.

CSS
In coastal sage scrub we can detect important changes in each variable addressed in the last section with 22 plots (Figure 22). This represents about two weeks of effort by a single team of two people. This is a significant investment, but is logistically feasible. Non-native grass cover continues to be the most difficult variable. Only ten plots would be needed to achieve 80% power in the other variables. In CSS, non-native grasses are very important since they tend to fill the interspaces between the shrubs. The density of these grasses can significantly alter the structure and function of the system. As a result, it is important that a monitoring program be able to detect changes in non-native grass cover in CSS.

Grassland
Twelve grassland plots will adequately capture the four variables relevant to this vegetation type (Figure 23). One two-person team should be able to finish this effort in about one week. Monitoring in grasslands is faster than in either CSS or chaparral since moving through a landscape without dense shrub cover is much easier.

It is important to realize that the minimum sample size would drop to eight plots if monitoring richness of native forbs is less important. The three remaining variables, total richness, bare ground and non-native grass cover capture important information about the structure of the grassland. In addition, it is probably advisable to conduct an additional power analysis for *Nasella spp* cover, as this is the dominant native species in the system.
Figure 22: Power analysis of multiple variables in CSS indicate that 22 plots will allow the largest number of variables to be collected with 80% power and a 10% type-I error rate. The vertical dashed line represents the maximum number of plots a two-person team can complete in a week.

Figure 23: Power analysis of multiple variables in grasslands indicate that 12 plots will allow all of the variables to be collected with 80% power and a 10% type-I error rate. The vertical dashed line represents the maximum number of plots a two-person team can complete in a week.
Recommended Sample Sizes

Recommended sample sizes reflect the goals and objectives of the monitoring program and the power calculations presented above. We recommend eight plots in chaparral, 22 plots in CSS and 12 plots in grasslands (See Table 9). Based on previous field work, this would take about 17 field days (4 weeks) for an experienced two-person team.

Table 9: Recommended sampling effort and power.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Chaparral</th>
<th>CSS</th>
<th>Grassland</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Recommended Sample Size</strong></td>
<td>8 plots</td>
<td>22 plots</td>
<td>12 plots</td>
</tr>
<tr>
<td>[min, max]</td>
<td>[7 – 10]</td>
<td>[8 – 24]</td>
<td>[8 – 15]</td>
</tr>
<tr>
<td><strong>Effort (2-person team)</strong></td>
<td>4 days</td>
<td>9 days</td>
<td>4 days</td>
</tr>
<tr>
<td>[min, max]</td>
<td>[4 - 5]</td>
<td>[4 - 10]</td>
<td>[3 – 5]</td>
</tr>
</tbody>
</table>

**Power at Recommended Sample Size**

- Native Shrub Cover: 98% (Chaparral), 99% (CSS), - (Grassland)
- Non-Native Grass Cover: 20% (Chaparral), 80% (CSS), 93% (Grassland)
- Total Species Richness: 89% (Chaparral), 99% (CSS), 98% (Grassland)
- Native-Forb Richness: 87% (Chaparral), 99% (CSS), 81% (Grassland)
- Standing Dead Material: 99% (Chaparral), 99% (CSS), - (Grassland)
- Bare Ground: 85% (Chaparral), 99% (CSS), 99% (Grassland)

Discussion and Recommendations

In order for conservation easements to provide tangible and substantial conservation benefits, TNC must rely on a flexible and self-correcting feedback loop of implementation, experimentation and adjustment which is fueled by a rigorous monitoring program. Internal documents and peer-reviewed papers published by TNC staff call for the “engagement of scientific staff applied in developing measurable ways to track the effectiveness of different easement restrictions at protecting different biodiversity targets” (Bendrick et al. 2004). In working conditions where qualified personnel and funding are limited resources, this project provides the statistical and ecological expertise necessary to establish a scientifically defensible monitoring program which minimizes cost and maximizes information gained.

This project serves the immediate need to monitor natural communities and biodiversity on the Irvine Ranch conservation easements. This project focuses on natural community level conservation values as proposed by the CEWG focusing on scrub, chaparral, and grassland communities. In addition this project is the first step in developing strategic guidelines about when and how to monitor (as recommended by Kiesecker et al. (2007) ) and addresses the necessity for economy and scientific rigor.
The overarching objectives of this project are to:

1. Describe the status and trend of key natural community conservation values on the Irvine Ranch conservation easements in Orange County.  
   See Vegetation Community Status, page 19

2. Describe the natural spatial and temporal variability of monitoring targets in order to establish a rigorous monitoring program covering the largest number of conservation values possible.  
   See Spatio-Temporal Trends, page 26

3. Clearly communicate role uncertainty plays in the system.  
   See Variance Components Analysis, page 30

4. Design the most cost effective, repeatable monitoring program, that reliably separates trend from natural temporal and spatial variation.  
   See Example Power Calculations, page 36

5. Provide an example of the role of pilot studies, variance decomposition and power analysis to guide the development of a rigorous and cost-effective monitoring program.  
   This Report

Using an adaptive process we evaluated potential monitoring strategies for precision and cost effectiveness, developed a training program for field staff, quantified variability in the system and fine-tuned the sampling strategy based on our results.

This study has been conducted over a large area and across multiple field seasons. It has yielded hundreds of thousands of rows of high-quality data, all of which has been used to develop a scientifically credible monitoring program for the Irvine Ranch conservation easements. This project represents the concrete and successful implementation of the CEWG’s recommendations made in 2004.

A revisit design is necessary in order to detect trends over time. Because the system is so spatially variable a new random sample each year would confound the temporal and spatial signals. Until monitoring goals and critical thresholds for conservation values are updated, we suggest an annual monitoring program of at least eight chaparral plots, 22 CSS plots and 12 grassland plots every spring. The proposed sampling effort of eight, 22, and 12 plots is somewhat larger than the samples we have been able to collect on easement lands. Between 2008 and 2010, we averaged six, ten, and seven plots respectively.

At this time we have enough information to characterize an optimal monitoring program for the Irvine Ranch conservation easements, based on variance decomposition and power calculations. In future years we will no longer be testing a monitoring program. Instead we will be implementing a monitoring program with high power to detect relevant change.

Despite the fact we are able to make recommendations at this time, it is critical to understand that even a project of this size may not capture the full range of environmental variability possible in this system. Southern California ecosystems are buffeted by extreme events like fire, drought, heavy rain, and human disturbances from nutrient pollution to introduction of exotic species. As a result, the system is still capable of surprising behavior. The recommended monitoring program will allow us to detect and measure such perturbations and make adjustments proactively.
Conclusions
A study of this size and scope has the potential to serve the broader scientific and conservation communities across California, the US, the European Union and numerous other organizations actively involved in conservation and monitoring. Although our specific recommendations are restricted to the Irvine Ranch conservation easements, this project is a powerful case study that clearly demonstrates the process of using advanced statistical techniques to develop and refine cost effective monitoring programs.

In consultation with TNC staff, we propose co-authoring a paper for submission to a peer reviewed conservation journal. This would benefit both SDSU and TNC by demonstrating that the monitoring program is capable of passing independent and informed peer-review. This will provide the ultimate quality check by scientists working in this field.
Literature Cited


