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Planning, implementing, and monitoring multiple-species habitat conservation plans[1]

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- *Premise of the study*: Despite numerous recommendations for various aspects of the design and monitoring of habitat conservation plans, there remains a need to synthesize existing guidelines into a comprehensive scheme and apply it to real-world conservation programs.
- *Methods*: We review tools for systematic conservation planning and elements for designing and implementing ecological monitoring in an adaptive management context. We apply principles of monitoring design to the San Diego Multiple Species Conservation Program (MSCP) in California, USA—one of the first multispecies habitat conservation plans, located in a landscape where high biodiversity and urban development converge.
- *Key results*: Tools for spatial conservation planning are aimed to conserve biodiversity, often in the context of a limited budget. In practice, these methods may not accommodate legislative mandates, budgetary uncertainties, and the range of implementation mechanisms available across consortia of stakeholders. Once a reserve is implemented, the question becomes whether it is effective at conserving biodiversity, and if not, what actions are required to make it effective. In monitoring plan development, status and threats should be used to prioritize species and communities that require management action to ensure their persistence. Conceptual models documenting the state of knowledge of the system should highlight the main drivers affecting status and trends of species or communities. Monitoring strategies require scientifically justified decisions based on sampling, response, and data design.
- *Conclusions*: Because the framework illustrated here tackles multiple species, communities, and threats at the urban–wildland interface, it will have utility for ecosystem managers struggling to design monitoring programs.

Key words: biodiversity; conservation reserves; ecological monitoring; habitat conservation plan; landscape; multiple species; prioritization.

Systematic conservation planning is at the forefront of conservation biology both in academic circles and on-the-ground conservation efforts. More than 30 years of development (Wilson and Willis, 1975) has yielded a comprehensive approach to the design of nature reserves founded on scientific principles and employing decision support tools (e.g., Pressey et al., 1999; Margules and Pressey, 2000; Carroll et al., 2003; Wilson et al., 2006; Moilanen et al., 2009). Nature reserves comprise geographical areas of the land or sea, managed for conservation, where protection of designated species or other ecosystem elements or processes is mandated, and therefore, certain human activities are limited or prohibited. Reserve design usually involves prioritizing areas for designation as a nature reserve, or part of a reserve network, based on explicit criteria, one of those criteria being a measure of the conservation value of that location (Margules and Usher, 1981).

Significant advances have been made in developing rigorous and spatially explicit methods for prioritization, viability analysis, species distribution modeling, and uncertainty/risk analysis as decision support tools for conservation planning (summarized recently in Moilanen et al., 2009). Although a great deal of emphasis in the literature and in practice has been placed on systematic spatial prioritization for the design of conservation reserves (the planning stage), these methods of assessment may also be used to inform decision-making about management actions "affecting conservation outcomes" (Ferrier and Wintle, 2009, p. 1) for any area managed at least in part for biodiversity conservation.

Once a nature reserve or conservation plan has been designed and established, the work is just beginning (Cowling et al., 1999; Barrows et al., 2005; Knight et al., 2006a; Barrows and Allen, 2007; Field et al., 2007). Ongoing monitoring is required of institutional entities responsible for managing reserves, because monitoring is critical for determining whether conservation plans are meeting their goals. Yet comprehensive monitoring programs to assess the efficacy of conservation plans to meet their stated objectives are often lacking (as noted by Kiesecker et al., 2007), despite the fact that the past decade

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has seen considerable advances in methods available for monitoring (discussed later). Oftentimes this lack of effective monitoring occurs because the goals of a conservation plan have not been specified in a way that makes management targets and monitoring objectives obvious, leaving managers without a clear and well-defined starting point.

As a result, monitoring activities often occur in an ad hoc and piecemeal fashion that lacks cohesion as well as the ability to assess the effectiveness of the conservation plan as a whole. And while many monitoring recommendations seem blatantly obvious at face value, their synthesis into a general framework that allows flexibility across regions, taxonomic groups, ecological communities and threats is far from obvious. The conservation biology literature has recently lamented the implementation crisis associated with conservation plans, in large part due to the failure to implement pragmatic and effective monitoring plans to assess and inform management (Knight et al., 2006a, b). For example, a recent comprehensive review of conservation prioritization methods only has one chapter on implementation (Moilanen et al., 2009), although there is a growing literature on monitoring conservation plans, which we will summarize.

This paper reviews major themes in biodiversity conservation planning and then focuses on a critical aspect of implementation—monitoring. We synthesize key recent developments in, and recommendations for, monitoring design for conservation plans, with a specific focus on regional multiple-species habitat conservation plans. We use San Diego's Multiple Species Conservation Program (MSCP) in southern California, USA, as a representative case study to demonstrate the pragmatism of our recommendations because it captures many of the issues and features of multispecies conservation plans—it was designed to conserve high biodiversity that is threatened by a multitude of human activities and altered natural processes. However, the monitoring recommendations outlined in this paper are flexible for application to general species-, community-, or ecosystem-based conservation plans.

KEY DEVELOPMENTS IN BIODIVERSITY CONSERVATION

Defining the concept of biodiversity so that it can be expressed in a form suitable for systematic conservation planning can lead to a multitude of features to be included in a conservation plan (Regan et al., 2007). Hence, by necessity, biodiversity value tends to be measured in terms of surrogates for biodiversity or focal species (Landres et al., 1988; Lambeck, 1997). Conservation plans, in turn, may be based on additional factors such as cost, connectivity, sociopolitical criteria, persistence, threats, representativeness, irreplaceability, resilience, ecosystem services, and/or likelihood of acquisition (Groves, 2003; Margules and Sarkar, 2007). A variety of approaches to conservation planning exist, ranging from sophisticated algorithms to less systematic approaches such as biodiversity hotspots, Key Biodiversity Areas and Important Bird Areas (Myers et al., 2000; Eken et al., 2004). The major formal conservation planning software tools currently in use are Marxan (Ball and Possingham, 2009), Zonation (Moilanen et al., 2005; Moilanen and Kujala, 2006), C-Plan (NSWNPWS, 2001; Pressey et al., 2005), ConsNet (Ciarleglio et al., 2009) and WorldMap (Williams and Gaston, 1994; Williams, 2000). Effective monitoring plans should support the objectives of the conservation plan, and so these objectives need to be explicit at the outset. Next, we briefly summarize these formal conservation planning tools and their underlying objectives; refer to Moilanen et al. (2009) for a more thorough treatment of quantitative conservation prioritization tools and their applications.

Marxan finds solutions to the problem of selecting a system of spatially aggregated sites that meet a suite of user-defined biodiversity targets while minimizing costs (defined as a weighted sum of management and acquisition costs and boundary length or connectivity costs). It addresses most objectives typically considered in conservation planning and can thus be applied to genes, species, vegetation classes, communities, or their surrogates. It can also include socially important features such as archaeological sites. Ecological processes, site condition, or sociopolitical influences may be included. The program delivers either several very good solutions to a mathematically defined problem, or information about the frequency with which planning units are selected from these very good solutions.

Zonation identifies areas important for retaining habitat quality and connectivity for multiple species, indirectly aiming for species' long-term persistence. Zonation prioritizes sites according to representativeness and persistence of species within cost and area constraints. Like Marxan, in Zonation, the species, genes, habitat, and community types can all serve as biodiversity targets in the conservation plan. Species can be weighted according to their importance in meeting conservation goals, and costs can be included, although Zonation does not directly incorporate sociopolitical considerations (except for costs and species weights). Zonation produces a hierarchical prioritization of the landscape based on the conservation value of sites, iteratively removing least valuable cells (accounting for complementarity) from the landscape while minimizing loss of both biodiversity and connectivity.

C-Plan maps the options for achieving explicit conservation goals based on the concept of irreplaceability. Three methods for estimating irreplaceability are implemented in C-Plan—site, summed and weighted average irreplaceability. C-Plan also uses a heuristic to maximize highly irreplaceable sites while minimizing impact on other land uses. C-Plan interfaces with Marxan to include measures of patch size, connectivity, and spread to refine reserve selection. Like Marxan and Zonation, C-Plan can be applied to genes, species, vegetation classes, communities, or their surrogates. Features can include geology and terrain, species assemblages, vegetation types, species, populations, cultural sites, or scenic sites, among others. C-Plan requires targets for all the included biodiversity features. Sociopolitical factors and costs can also be included.

ConsNet uses distribution maps of conservation features or surrogates for these features to prioritize places, based on rarity, complementarity, representation, and richness, in efforts to identify reserve systems that meet a conservation target. The conservation target can be specified in terms of adequate representation of all conservation features (such as species, vegetation classes, communities, or surrogates for any of these) or a desired maximum area or cost. Additional ecological and sociopolitical criteria can be invoked to refine the set of feasible alternatives identified by ConsNet. It computes the subset of nondominated alternatives in the set of feasible alternatives with respect to a set of specified criteria. Further refinement of alternatives can be achieved through dropping criteria or ranking alternatives using the standard or modified analytic hierarchy process. Unlike the three software packages already mentioned, ConsNet does not explicitly include connectivity as an objective for site selection.

WorldMap is the simplest conservation planning method of all those considered here, but it is included because it emphasizes spatial distributions of species in the context of their phylogenetic relationships. WorldMap is primarily designed as an interactive system for exploring aspects of spatial pattern, not for designing reserves. Worldmap does not explicitly address many objectives specified in conservation planning. The only constraints addressed are number of sites selected. Via maps, WorldMap displays the reserves that have the highest species richness or rarity. WorldMap then selects the sites with the highest species richness (hotspots or near-maximum coverage) or rarity (near-minimum coverage) iteratively, taking into account complementarity, until a user-defined number of sites is chosen. Only the presence of species, habitats, or surrogates and their relationships (e.g., phylogenetic, ecological similarity, habitat association) can be entered. Costs of planning units or site aggregation preferences cannot be accommodated. Threats or sociopolitical factors are difficult to incorporate.

In practice, reserves need to be designed around existing preserved land and other land use and political constraints; they need to obey legislative mandates where necessary, and should add value to or strengthen leverage with existing conservation initiatives in the region, many of which cannot be quantified (e.g., Greer, 2004). Once a reserve is implemented, the important question becomes whether it is effective at conserving biodiversity. In the following sections, multispecies habitat conservation plans (HCPs) are introduced, and then a case study is presented to illustrate the idiosyncratic nature of the HCPs, and a systematic approach to designing a monitoring and management program for such conservation areas is discussed.

MULTIPLE-SPECIES HABITAT CONSERVATION PLANS

Recent regulatory changes in the United States emphasize preservation of endangered species through the development of multiple-species habitat conservation plans (HCPs); they are the legislated mechanism for implementing reserve systems in the United States (see U. S. Fish and Wildlife Service, 1996; Smallwood, 2000). HCPs can potentially protect multiple species before they decline to the point of requiring a listing under the U. S. federal Endangered Species Act (ESA). They may also be preferred by land owners and managers because they provide some security by covering (i.e., authorizing lawful incidental "take" of) species that are currently or may become listed as threatened or endangered in the future (U. S. Fish and Wildlife Service, 1996). The number of multispecies HCPs has increased over time, and the number of species they cover has also steadily increased from a single species to over 200 species in plans currently under consideration (U. S. Fish and Wildlife Service, 2007). The multispecies HCPs are often designed using approaches borrowed from many types of conservation planning methods, including both formal and informal spatial prioritization tools, but it can be difficult to determine how a reserve was designed because this information is either buried in the gray literature and technical reports or not clearly articulated—such plans arise through a complex process of negotiation, politics, pragmatism, science, and expert and popular opinion, and they are often highly constrained by historic and existing priorities and plans. They are often implemented across vastly different jurisdictions, each with their own budgets and priorities (Greer, 2004). Nonetheless, these reserves are implemented, and they require transparent and systematic approaches for monitoring and management. Because they are governed by specific legislative requirements, they have idiosyncratic needs when it comes to implementation, monitoring, and management that transgress current recommendations expounded in the academic literature for reserves designed using the principles of scientifically based conservation planning.

An important distinction between the way many multispecies HCPs are designed and implemented and the reserves resulting from the systematic conservation planning tools summarized in the previous section is the way costs are represented. In the systematic approaches summarized in the previous section, costs are usually represented via a static constraint (although sometimes this can be dynamic), and they are usually assumed to be known at the outset (although sometimes this can be uncertain). However, this is idealistic for multispecies HCPs. Often, a tentative conservation plan needs to be on the table to negotiate an available budget, and this incremental development can entail many ad hoc modifications to the plan that do not necessarily obey the original objectives or constraints imposed on the plan at the outset. Furthermore, easements and partnerships across government jurisdictions, nongovernmental organizations, and private land-holders do not easily translate to a fixed budget, and they depend on ad hoc negotiation that is extraordinarily difficult to capture in the statement of the conservation planning problem or models of cost.

When monitoring and management of HCPs are considered, the challenges increase in complexity. Ideally, monitoring objectives should be derived from, and support, the objectives of the conservation plan. However, if the conservation plan objectives are specified very broadly, or if the plan has deviated from original goals, monitoring goals may need to be set post hoc. Furthermore, when regional monitoring and management are administered piecemeal across a consortium of government agencies, nongovernment organizations and private land owners, each with local monitoring objectives and mandates, systematic algorithms for broad-scale optimal monitoring design fail. Finally, monitoring and management often require more resources and commitment than are usually acknowledged at the planning stage. The total budget for monitoring activities is often unknown at the outset, changes over time, and depends on coordination and negotiation across the consortium. Citizen science groups may also play a role in monitoring and managing local biodiversity and can contribute to ongoing monitoring with little expense (Hoyer et al., 2001).

In the following sections, we analyze a multispecies habitat conservation plan in this context and provide a general strategy for developing monitoring plans when (1) the stated objectives of the conservation plan are insufficient, in and of themselves, to guide monitoring and management, (2) the available budget is unknown, temporally uncertain and subject to negotiation, and (3) multiple monitoring and management objectives must be met across a consortium of jurisdictions and institutions. While this is not an ideal basis for developing a general monitoring strategy, it is the realistic state of affairs for many multispecies HCPs.

CASE STUDY: SAN DIEGO MULTIPLE SPECIES CONSERVATION PROGRAM (MSCP)

Approved in 1997, the San Diego MSCP was the second multispecies HCP implemented in California, under the state's Natural Community Conservation Planning (NCCP) Act

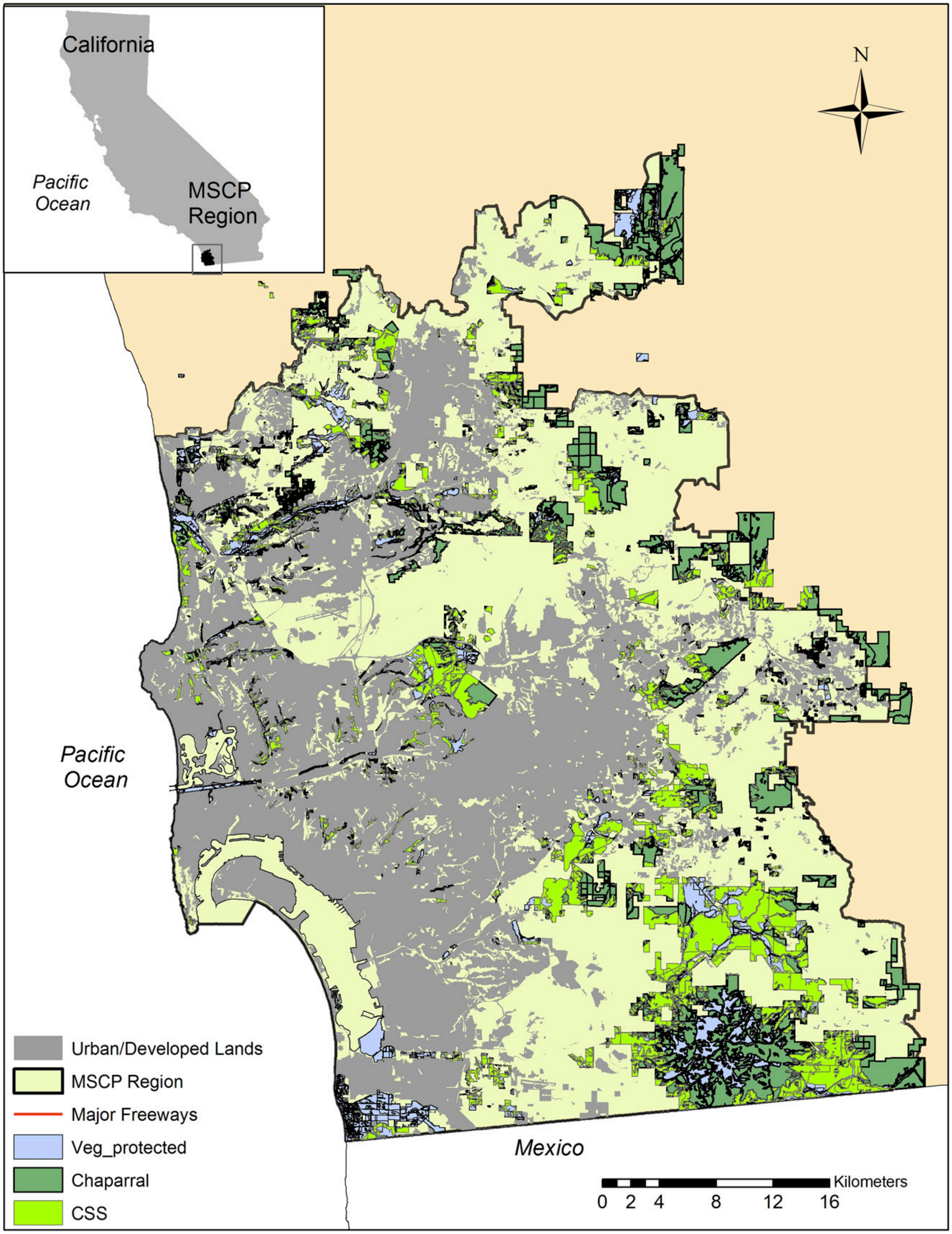

Fig. 1. Map of the San Diego Multiple Species Conservation Program (MSCP) Region in southwestern San Diego County, California, USA, showing the extent of urban development. The major plant communities in the reserve, chaparral, and coastal sage scrub (CSS), and all other natural communities (Veg_protected) are shown; together, they comprise the current configuration of lands acquired for the reserve.

(California Department of Fish and Game Code, 1991), and as then-Secretary of the Interior Bruce Babbitt stated, it was viewed as "a major milestone in America's conservation history and a model plan for communities nationwide" (Hanna, 1997). Located in southwestern San Diego County, the MSCP currently comprises about 500 km^2 within a 2300 km^2 conservation planning region (Fig. 1). The region has high biological diversity and varied terrain, spanning from mountains to coastal strand. It has been designated a biodiversity hotspot (Myers et al., 2000; Brooks et al., 2002; Underwood et al., 2009) because of the large number of endemic and rare species and the high levels of human population growth and urban development pressure in the region (Fig. 2). In particular, located within the California Floristic Province, a Mediterranean-type ecosystem, the region has especially high plant species richness and endemism (Cowling et al., 1996; Mittermeier et al., 1998). The MSCP covers 85 endangered, threatened, and rare plant and animal species and over 60 natural community types (Ogden Environmental and Energy Services, 1998). As noted by Greer (2004), land acquisition for the MSCP outpaced everyone's expectations, and he attributed that success to cooperation and cost-sharing among local, state, and federal governments.

The overarching emphasis of the San Diego MSCP was on the conservation of the coastal sage scrub ecological community. This flagship conservation plan is ideal for applying recently developed principles, providing an example for other conservation plans at the critical stage of monitoring plan design. The dual purpose of the MSCP—to protect both endangered plant and animal species and communities—contributes to its generality as a case study for the application of monitoring recommendations. Furthermore, the MSCP is well-established, and so potential monitoring entities and locations are known and unlikely to change, making it a fairly stable plan that is conducive to applying recent advances.

STEPS FOR MONITORING PROGRAM DEVELOPMENT

The general steps involved in designing a conservation plan monitoring program are outlined in Fig. 3. This framework, following Atkinson et al. (2004) with some key modifications, includes the following interconnected steps: (1) prioritization of species and communities (biodiversity elements) for monitoring, based on risk and representation; (2) development of conceptual models that identify specific conservation objectives, critical monitoring variables, important threats, and management responses; and (3) use of existing data to begin assessing key components of spatial and temporal variability in some of the monitoring variables. The advantage of these recommendations is that their common sense approach to monitoring is likely to gain traction among practitioners. Indeed, they are currently being adopted and implemented by a broad group of managers for the San Diego MSCP, with the view to expand application to multiple-species habitat conservation plans across California and beyond. The following sections review and summarize recent advances in monitoring design and are organized according to the steps shown in Fig. 3.

Identify habitat conservation plan goals—Clear and concise conservation goals leading to specific objectives are essential precursors to any successful ecological monitoring program (Gibbs et al., 1999; Mulder et al., 1999; Noon, 2003; Field et al., 2007). Bisbal (2001) emphasized the importance of simple and unambiguous goals, reflecting appropriate spatial and temporal scales that are feasible to assess through monitoring. Objectives should describe the desired impacts of management. Such objectives help determine what should be measured, where, and how often (Gibbs et al., 1999) and the statistical methods that should be used to analyze the data (Olsen et al., 1999).

Ambiguous objectives can lead to "the wrong variables being measured in the wrong place at the wrong time with poor precision or reliability" (Noss and Cooperrider, 1994, p. 304). Additionally, ambiguous conservation objectives can lead to differences in interpretation, resulting in loss of resources (time and money) while various groups debate about the intent of the original plan. The goals and objectives must provide a clear description of what the conservation plan aims to achieve, and the monitoring program should be able to assist in determining whether these goals are being met.

Focusing monitoring on conservation and management objectives requires that data be collected on species' and natural communities' status, trends, threats, and predicted responses to management actions. More specifically, while conventional probability-designed sampling may be sufficient to detect status and trend, detecting response to management actions requires sampling across the full range of each action. Yoccoz et al. (2001) distinguished between scientific objectives that endeavor to establish knowledge about the behavior and dynamics of the system, and management objectives that attempt to determine the state of the system and provide information about its response to implemented management actions. For the MSCP, the primary focus has been on management objectives. The idea that monitoring must support practical management directives echoes Hornaday's (1914, p. 2) entreaty that "conservation is replete with urgent practical demands."

The originally stated goals of the MSCP (Ogden Environmental and Energy Services, 1998) are to (1) conserve specific species at levels that meet the take authorization issuance standards of the federal Endangered Species Act and California's Natural Community Conservation Planning Act and (2) conserve the diversity and function of the ecosystem through the preservation and adaptive management of large blocks of interconnected habitat and smaller areas that support rare ecological communities (e.g., vernal pools).

Having goals at multiple levels of biological organization require this monitoring program to include priorities, objectives, conceptual models, and protocols for species, natural vegetation communities, and ecosystem-level elements.

Prioritize covered species for monitoring—In an attempt to logically allocate scarce resources to the monitoring required for the MSCP, we adapted a risk-based species prioritization scheme from Andelman et al. (2004) and applied it to the list of 85 MSCP-covered species to prioritize them for monitoring (Regan et al., 2008). Because HCPs are intended to protect multiple species before they decline to the point of requiring listing under the ESA, prioritizing species monitoring according to high risk of decline is appropriate (see Joseph et al., 2008 for an alternative). Classifications of risk that explicitly include current and future threats have been found to be most useful in determining which species will be adversely affected by management actions (Andelman et al., 2004; Regan et al., 2008 and references therein). Ideally, this would be an early step in the design of any monitoring program. The steps were (1) apply an

Fig. 2. An area of coastal sage scrub and chaparral plant communities in the Multiple Species Conservation Program in the vicinity of Carmel Mountain (near the city of La Jolla), showing the juxtaposition of the reserve network with urban land use. Photograph by J. Franklin.

at-risk species classification (e.g., NatureServe, International Union for the Conservation of Nature); (2) ascertain threats and the degree to which each threat contributes to overall species' risk; (3) determine temporal response to threats and the spatial scale of each threat relative to the distribution of the species within the region; (4) group species according to risk levels; and (5) further prioritize according to the number of high-degree threats, and then by the total number of threats. This results in a prioritization based largely on threatening processes and highlights the most prominent threats and the habitats in which they occur. This approach to prioritization can inform management and monitoring of the threats, in addition to the species, that require the most urgent monitoring and management attention (see Regan et al., 2008 for details).

Species were grouped according to their at-risk ranking into risk groups 1–3 in descending order of risk. Temporal response was used as a tiebreaker where needed, with species having short-term responses to threats (<5 yr) ranked higher than those with long-term responses. The prioritization of MSCP covered species is reported elsewhere (Regan et al., 2008). In summary, we found that the most at-risk plant species occur in the extensive upland natural communities of chaparral and coastal sage scrub, while many at-risk animal species use less-extensive riparian, oak woodland, grassland, and several wetland habitats (Fig. 4). Threats to species are numerous in grassland and riparian woodland for covered animals and in the extensive shrublands for covered plants (Fig. 4).

Prioritize natural communities for monitoring—Multiple criteria for prioritizing species and land to be included in nature reserves have been well documented (Margules and Usher, 1981; Scott and Sullivan, 2000), but to our knowledge they have not been used to prioritize ecological communities for monitoring within an established reserve. Drawing from principles of conservation and landscape ecology, we used areal extent, representativeness, fragmentation, and endangerment as criteria to assess and prioritize communities for the MSCP (details in Hierl et al., 2008). We argued that, given the scarcity of funding for conservation monitoring and management, available resources should be focused on communities that are (1) of large extent, and therefore covering most of the reserve, supporting extensive ecosystem processes and providing habitat for many covered species; (2) underrepresented in the reserve vs. the surrounding region; (3) fragmented on the landscape and therefore at-risk for the negative effects of fragmentation; and/or (4) endangered according to published endangerment rankings (Hierl et al., 2008). We used mapped terrestrial plant species assemblages to represent natural communities, which is an approach commonly employed, e.g., in gap analysis (Scott et al., 1993; Davis et al., 1995).

We assessed each of these four criteria for communities represented in the MSCP and assigned a ranking for each criterion. A High ranking was assigned when a community was large in areal extent, proportionally underrepresented within the reserve vs. the region, fragmented, and/or had a high endangerment ranking; Moderate was assigned when a community was midlevel in extent, representativeness, fragmentation level, and/or endangerment ranking; and Low was assigned for communities small in extent, well-represented in the reserve, with low fragmentation levels, and/or a low endangerment ranking. To prioritize them, we then ordered the communities by the overall number of High, then Moderate, then Low rankings.

We found for the extensive (>10% of the region) natural communities, that chaparral is already well represented in the reserve relative to the region, while large or connected blocks of coastal sage scrub are underrepresented and should be targeted for additional acquisition (e.g., Fig. 1). As emphasized in Hierl et al. (2008) the MSCP, like many conservation plans, is actually being implemented (i.e., land acquired) over a period of decades, so analyses that help prioritize additional land acquisition are still useful this far into the process. In the MSCP, high priority communities for monitoring include coastal sage scrub (high endangerment, extensive, underrepresented, and moderately fragmented), as well as wetlands (salt marsh, freshwater marsh, vernal pool) and oak woodlands (high fragmentation, moderate endangerment and representativeness, but less extensive).

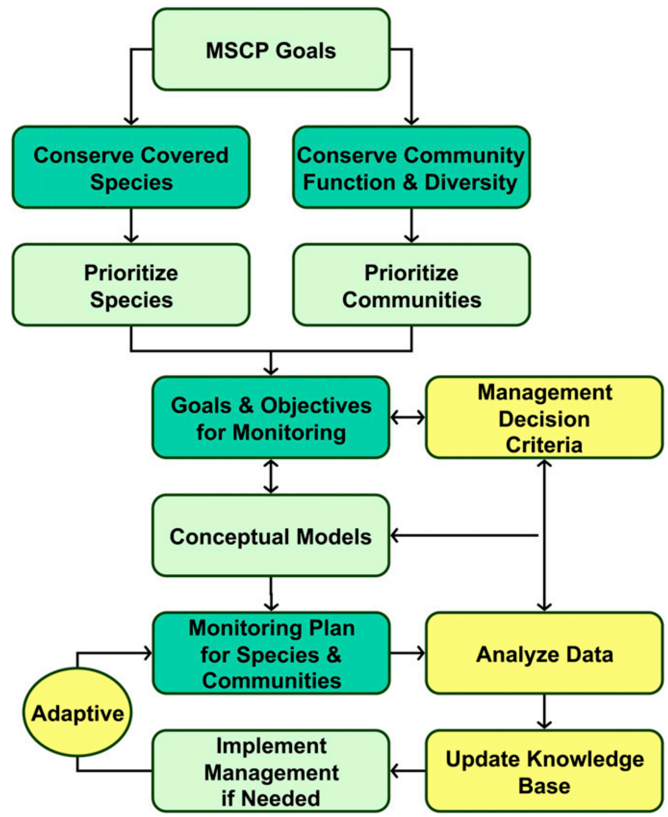

Fig. 3. Flow diagram of steps involved in designing a monitoring program for a multiple-species habitat conservation plan. MSCP: Multiple Species Conservation Program

Identify goals and objectives for priority monitoring elements—Identifying and refining conservation objectives for prioritized species and communities is an important precursor to designing the monitoring program, i.e., where, when, and how to monitor the element of concern (Parrish et al., 2003). These objectives provide more specific targets for monitoring (e.g., maintain 50% native coastal sage scrub plant cover), vs. broader goals identified at the program level (e.g., conserve ecosystem function and diversity). As indicated in the flow diagram (Fig. 3), each species or community identified as a priority monitoring element should then have specific monitoring objectives and management decision criteria identified (Mulder et al., 1999; Bisbal, 2001). For example, if monitoring aims to detect changes in the population size and geographic range of selected species, it would be important to define what constitutes the minimum significant change in the parameter before a management action should occur, e.g., a 20% or greater decline in the species' range over 25 years (Haufler, 1999). Management-oriented, or response-based, monitoring is also facilitated by creating species-specific and habitat-level conservation targets. Examples of specific conservation objectives include: (1) Threshold objectives (e.g., maintain a population of rare plant species A at 2500 individuals or greater). For example, the MSCP specified preservation of 100% of extant populations of Tecate Cypress (*Cupressus* [*Hesperocyparis*] [*Callitropsis*] *forbesii*) on Otay Mountain above 1500 feet elevation (Ogden Environmental and Energy Services, 1995); (2) Trend objectives (e.g., increase mean density of species A by 20%); (3) Response-based management objectives (e.g., decrease abundance and extent of invasive weed X by 30% at site C when the cover of plant species A declines to 1000 m^2 [National Park Service, 2005]).

In some cases, targets may also serve to guide management thresholds. For example, if site C becomes invaded with weeds X and Y, then a management action would be invoked. These targets, in tandem with conceptual model development, can also help stakeholders identify important stressors that might be the appropriate focus of monitoring and management effort. In the MSCP, species-level targets were set in the original plan, but were being revised starting in 2006. The assignment of a target or threshold can be controversial and nontrivial (as demonstrated in the specification of recovery goals for endangered species), but sometimes necessary as a mechanism for deciding if the management goal has been met or when to implement a management action (Groves, 2003). Assignment of a target is essentially a policy decision that requires scientific input. Much of the time it will be guided by expert opinion alone. However, population models could be used to set population-level management goals for species, and home range size or historic proportions of communities can be used as a guide for community-based thresholds. Finally, it is important not to lose sight of the ultimate goal, which is to ensure that biodiversity persists and not to strictly adhere to numerical or qualitative thresholds—some flexibility and open-mindedness needs to be applied.

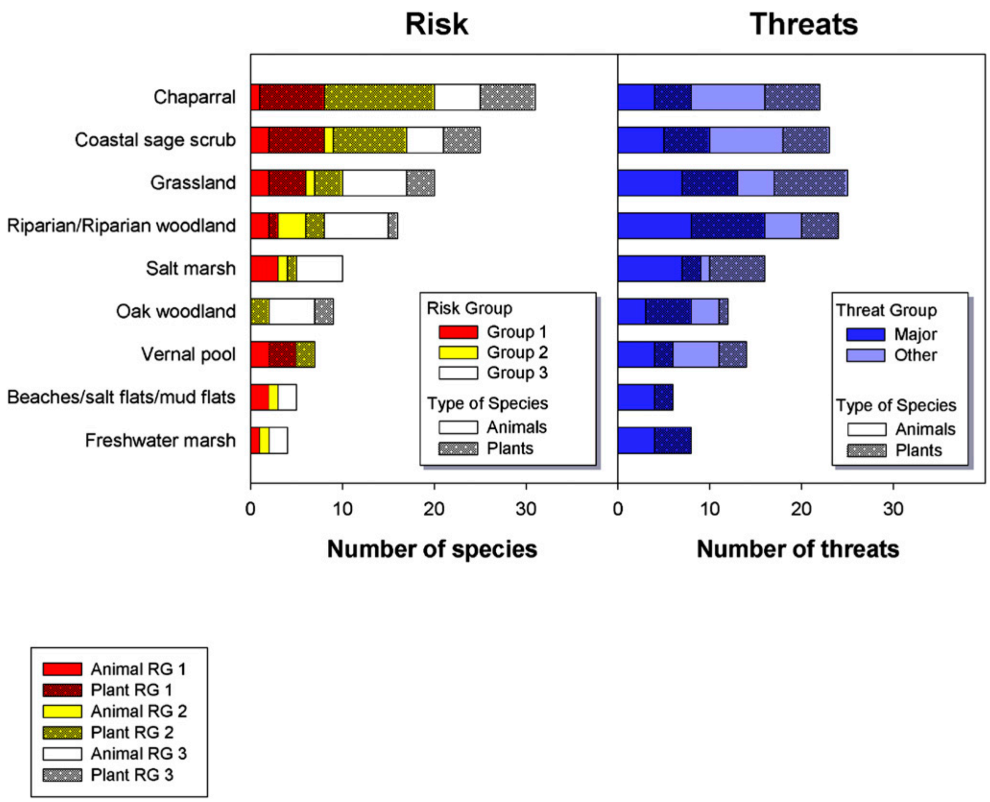

Fig. 4. Number of Multiple Species Conservation Program (MSCP) covered animal and plant species in each risk group (1–3, High to Low), and threats (Major = High-degree threats, Other = Moderate and Low-degree threats) by habitat type (natural community) in the MSCP. Shown for a subset of nine generalized habitat types that supports most of the covered species in the MSCP (for details see Regan et al., 2008). Number of Threats refers to the number of distinct threats to species occurring in the habitat type. Hence, in each bar a distinct threat only appears once.

Develop conceptual models—The development of conceptual models has been identified as a critical tool for regional habitat conservation plans (e.g., Atkinson et al., 2004). These models can be narratives or diagrams, tables or matrices, and they link causes (stressors, threats, drivers) with effects on the state of the environment or biotic responses (Noon, 2003; Burgman, 2005). While a conceptual model can describe any system and is especially valuable when designing a conservation strategy for a multispecies plan, when developed for ecological monitoring, the model should include explicit links to decision-making or management actions.

Conceptual models should include a description of the system, threats, and impacts resulting from the threats (Woodward et al., 1999). Noon et al. (1999) developed a template that can be used for each threat to identify biotic consequences at different ecological scales (landscape, community, population, genetic). The use of conceptual models has been recommended in viability assessments under the National Forest Management Act of the United States (Noon, 2003), and they have been applied to assess viability of terrestrial vertebrates in a major National Forest plan in the United States (Marcot and Heyden, 2001; Raphael et al., 2001). We have found that an important part of the process of developing a conceptual model is achieving agreement from the experts, managers, and stakeholders that the model adequately portrays the state of the system as it is currently understood.

We applied the following framework to build conceptual models for conservation elements in the MSCP. We identified (1) conservation goals for the relevant species or community; (2) major current and historical anthropogenic threats, natural drivers, and population or community parameters that dictate current or future status and trends; (3) potential responses of the relevant species or system to management; and (4) monitoring targets based on the main parameters that affect the dynamics of the species or community.

We used *Ambrosia pumila* as a case study for MSCP-covered species. This rare plant was the focus of a draft conservation strategy (McEachern et al., 2006) that described a conceptual model for this species in narrative form. It was straightforward to develop a graphical model from this (Fig. 5) given that the plan defined a management goal and linked monitoring to management (what they called “effectiveness monitoring”). For example, because competition from invasive plant species and trampling were identified as the major threats to *A. pumila*, monitoring targets should include demographic parameters of the species, estimates of invasive cover, and trampling intensity.

A second example is shown for the coastal sage scrub community. In this case, an extensive literature review and a workshop involving conservation managers were used to develop a conceptual model. We identified wildfire and precipitation as the main drivers of community dynamics. The target community parameter was identified as the balance between the abundance of native and exotic plant species, as mediated by soil type (Fig. 6). Historical threats include habitat loss (to urban and agricultural use). Current threats include increased size and frequency of anthropogenic fires and direct habitat disturbance (Diffendorfer et al., 2002), which affect the plant communities

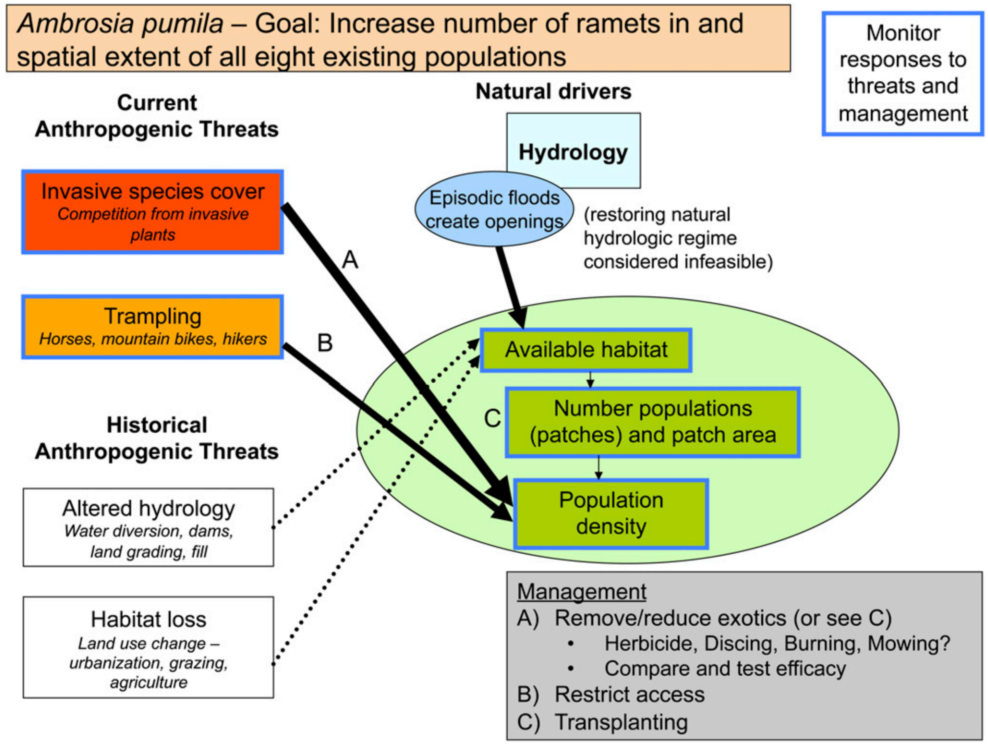

Fig. 5. Conceptual model for *Ambrosia pumila*; conservation management goal in the top left. Anthropogenic threats aligned on the left side, and natural drivers of population change along the upper middle. Threats are color-coded greatest to least (red, orange, yellow). We distinguished current anthropogenic threats, which may potentially be mitigated by management action, from historical threats, impacts that have already contributed to the rarity or endangerment of the species. The strength of interactions is indicated by arrow width. The green ellipse represents the target species (or community), and the boxes within it are variables that should be monitored (effectiveness monitoring). Boxes outlined in blue indicate variables that should be measured during monitoring and include both species and environmental attributes (natural and/or anthropogenic). The shaded box in the lower right describes potential management activities, and the letters indicate the process in the diagram each activity would affect.

via increased abundance of invasive exotic plant species. Community-level monitoring targets should include the abundance of native vs. exotic plants in periods of high and low precipitation and the occurrence of fire (Fig. 6).

Design monitoring program—The next step in the framework is to design a monitoring program (Fig. 3) for priority species and communities. Monitoring (National Research Council, 2001; Lovett et al., 2007) is a recommended element of all habitat conservation plans and is critical to assessing whether large-scale multispecies programs are meeting their stated objectives (Atkinson et al., 2004; Barrows et al., 2005). Developing effective monitoring programs for conservation plans is scientifically and logistically challenging (Greer, 2004; Knight et al., 2006b), in part because it is difficult to apply traditional statistical theory and methods to ecological monitoring (Fuller, 1999; McDonald, 2003). In ecological monitoring, the units sampled are often complex and can take many forms including habitat patches, liters of lake water, or variable-length transects flown from an aircraft. Further, processes that influence population dynamics can change across space and through time. As a result of these challenges, many monitoring programs have been criticized as inefficient or inadequate (National Research Council, 1995; Legg and Nagy, 2006).

The key to increased efficiency is to ask the right question concerning the target species or community relative to the space and scale of resources being managed in the reserve system. In highly uncertain systems, it is especially important to measure environmental drivers and stressors in addition to target species and community parameters, allowing managers to determine whether the plan's objectives are being met and at the same time update their knowledge of the system, ultimately informing adaptive management decisions (Nichols and Williams, 2006; Wilhere 2002; MacKenzie et al., 2006; Williams et al., 2007)

Monitoring programs must be designed to have adequate power to determine the status and trend of the variable of interest (Legg and Nagy, 2006; Field et al., 2007). Stevens and Urquhart (2000) distinguish two distinct aspects of monitoring: "sampling design," which they define as the process of specifying the units or points to sample, and "response design" defined as the process of deciding what to measure and how to measure it (see also Larsen et al., 2001). The sampling and response designs must address: how many and which sites should be included in the initial sample, whether and how often sites should be revisited, whether the design should be allowed to change as more data become available, and how the samples at different times should be related. These decisions depend on the relative importance of the description of status (distribution across space) vs. detection of trend (change through time), and the magnitude and scale of heterogeneity. We further define the "data design" as the implementation and quality assurance of the database as well as the statistical analysis of status and trend. All too often data collected during monitoring are inadequately analyzed due to lack of resources for data handling or insufficient quality control procedures (Vos et al., 2000).

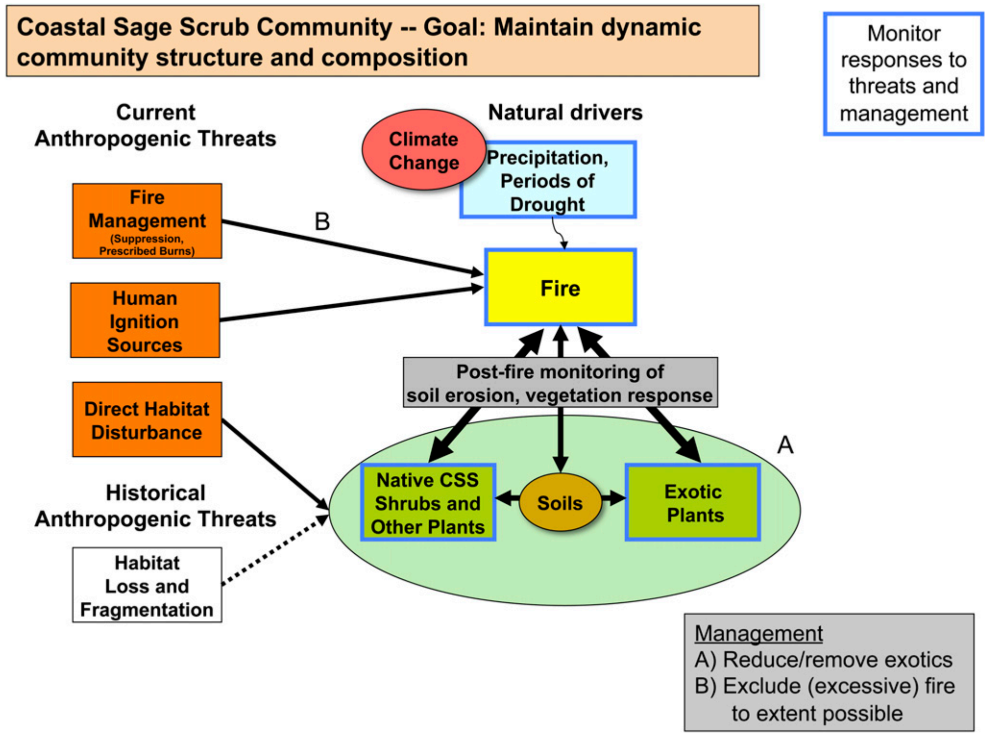

Fig. 6. Conceptual model for the coastal sage scrub plant community. See caption for Fig. 5.

In the San Diego MSCP case study, our conceptual model for the coastal sage scrub community (Fig. 6) addresses the overall conservation plan's goal of "maintaining ecosystem function and diversity" by focusing on measurement of community structure and composition. Based on the model, elements that should be monitored include the species composition of the community and the threats—exotic species cover, and some measure habitat disturbance (e.g., cover of bare soil, or some other measure of disturbance intensity).

To assess the appropriateness of these monitoring elements, we analyzed data from several recent research and monitoring studies of coastal sage scrub (Diffendorfer et al., 2004; Allen et al., 2005; Kelt et al., 2005) using a variance partitioning strategy (see Deutschman et al., 2007 for details). Variance components analysis partitions the observed variation into spatial, temporal, and unexplained variation; partial R^2 values were used as a metric to describe the relative magnitude of spatial and temporal variation of the monitoring elements (Fuller, 1999; Larsen et al., 2001). Abundance of exotic plants (mostly annual grasses and forbs) varied among sites (100s of km) and years (Fig. 7). In contrast, native shrub abundance varied little among sites or through time. Instead, native shrubs varied among plots within each site (km scale). These differences suggest that the sampling design for exotics requires that many sites be visited yearly, but that few plots need to be located within each site. In contrast, native shrubs need not be monitored every year and require fewer site visits. However, multiple plots within each site are needed to characterize the site.

This analysis of existing data suggested that both spatial and temporal variability must be addressed in monitoring the coastal sage scrub community in the MSCP, but not to the same degree for all monitoring elements. Further, it demonstrates a methodology that can be applied to preliminary or existing data to evaluate the relative importance of temporal, local, or regional spatial heterogeneity in the monitoring variables.

Conclusions—A number of well-established and tested tools are now available for systematic approaches to reserve design, and they have been used in significant conservation planning efforts (e.g., Margules and Pressey, 2000; Fernandes et al., 2005; Wilson et al., 2006). These methods emphasize optimizing the biodiversity value of the reserve, often based on criteria such as representativeness and irreplaceability of biodiversity elements, but sometimes also consider persistence, resilience, and ecosystem services, and cost and likelihood of land acquisition. Reserves are designed, however, using both systematic and ad hoc methods, owing largely to sociopolitical and economic constraints on the planning process. A specific kind of biodiversity reserve, the multiple-species Habitat Conservation Plan, is the currently legislated mechanism in the United States for protecting multiple species before they decline to the point of requiring listing under the federal Endangered Species Act. Once a reserve is designed, by whatever method, the important task is determining if it is meeting its conservation objectives through monitoring and management.

While there are scientific, logistical, and political challenges to designing and deploying an effective ecological monitoring plan for multispecies habitat conservation areas, there now exist excellent tools, and examples of their effective use in the literature, that were not available 10 years ago. We have demonstrated the use of these tools within a conceptual framework for designing a monitoring program for a regional conservation reserve network, the San Diego MSCP, which was one of the first multispecies habitat conservation plans implemented

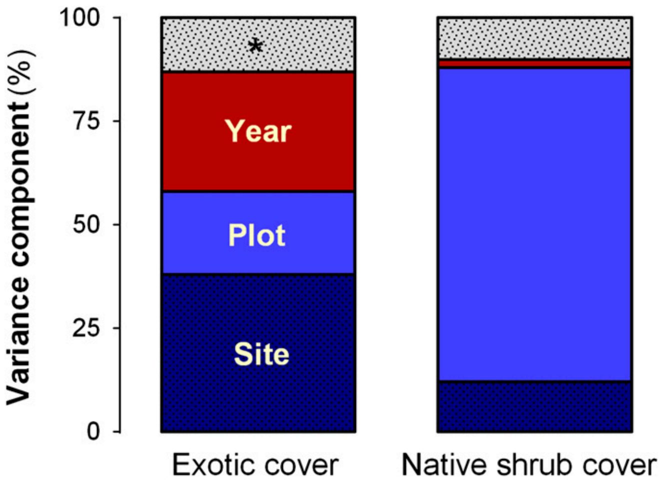

Fig. 7. Variance components analysis of the coastal sage scrub plant community showing variation in exotic (nonnative, mainly herbaceous) plant cover and native shrub cover among geographically separated sites, between years, the interaction between sites and years (*), and variation from plot to plot within sites. Data from several studies (noted in Design monitoring program).

in the USA. We emphasized certain key steps: (1) prioritizing species and communities for monitoring, (2) developing conceptual models, and (3) using existing data to assess key components of variability in monitoring targets. Identifying this variability is a useful preliminary step to defining a scientifically robust sampling, response, and data design for monitoring elements.

In an adaptive management framework (Holling 1978; Walters 1986; Walters and Holling 1990; Noon et al., 1999; Williams et al., 2007), almost every step in designing a monitoring program needs to be iterative (Fig. 3). The specific conservation goals and objectives for each monitoring element, as well as conceptual models, monitoring variables, and thresholds for management action should be revisited and revised as monitoring provides new information about the system. While most existing monitoring plans intend to be adaptive, true adaptive management and monitoring is rarely seen. Even in a well-structured monitoring program there may be a time lag between monitoring, management, and the resulting data collection and analysis that informs adaptation. Only when there is commitment to a monitoring program with well-defined goals and objectives can long-term data be collected in a way that informs adaptive management.

All these recommendations require adequate institutional and funding support (Lovett et al., 2007). Time committed to these steps before implementing a monitoring program will assure that limited resources are put toward the highest priority conservation elements and that monitoring will inform managers about the status of those elements. Even preliminary data, if analyzed properly and promptly, can help improve the monitoring program early in the process. Finally, for successful implementation of monitoring program, it is crucial that researchers, conservation managers, stakeholders, and decision-makers form partnerships to develop and implement ecological monitoring programs for habitat conservation plans (Field et al., 2007). Without appropriate communication and cooperation from vested parties, even the best designed and most well-funded monitoring program will struggle to succeed.

LITERATURE CITED

ALLEN, E. B., R. D. COX, T. TENNANT, S. N. KEE, AND D. H. DEUTSCHMAN. 2005. Landscape restoration in southern California forblands: Response of abandoned farmland to invasive annual grass control. *Israel Journal of Plant Sciences* 53: 237–245.

ANDELMAN, S. J., C. GROVES, AND H. M. REGAN. 2004. A review of protocols for selecting species at risk in the context of US Forest Service viability assessments. *Acta Oecologica* 26: 75–83.

ATKINSON, A. J., P. C. TRENHAM, R. N. FISHER, S. A. HATHAWAY, B. S. JOHNSON, S. G. TORRES, AND Y. C. MOORE. 2004. Designing monitoring programs in an adaptive management context for regional multiple species conservation plans. U. S. Geological Survey General Technical Report, Western Ecological Research Center, California Department of Fish and Game, U. S. Fish and Wildlife Service, Sacramento, California, USA.

BALL, I. R., AND H. P. POSSINGHAM. 2009. Marxan and Relatives: Software for spatial conservation prioritization. *In* A. Moilanen, K. A. Wilson, and H. P. Possingham [eds.], Spatial conservation prioritization: Quantitative methods and computational tools, 180–210. Oxford University Press, New York, New York, USA.

BARROWS, C. W., AND M. F. ALLEN. 2007. Biological monitoring and bridging the gap between land management and science. *Natural Areas Journal* 27: 194–197.

BARROWS, C. W., M. B. SWARTZ, W. L. HODGES, M. F. ALLEN, J. T. ROTENBERRY, B.-L. LI, T. A. SCOTT, AND X. CHEN. 2005. A framework for monitoring multiple-species conservation plans. *Journal of Wildlife Management* 69: 1333–1345.

BISBAL, G. A. 2001. Conceptual design of monitoring and evaluation plans for fish and wildlife in the Columbia River ecosystem. *Environmental Management* 28: 433–453.

BROOKS, T. M., R. A. MITTERMEIER, C. G. MITTERMEIER, G. A. B. DA FONSECA, A. B. RYLANDS, W. R. KONSTANT, P. FLICK, ET AL. 2002. Habitat loss and extinction in the hotspots of biodiversity. *Conservation Biology* 16: 909–923.

BURGMAN, M. A. 2005. Risks and decisions for conservation and environmental management. Cambridge University Press, Cambridge, UK.

CALIFORNIA DEPARTMENT OF FISH AND GAME CODE. 1991. Section 2800–2835, Natural Community Conservation Planning Act [online]. Website http://www.dfg.ca.gov/nccp/displaycode.html [accessed 28 February 2007].

CARROLL, C., R. E. NOSS, P. C. PAQUET, AND N. H. SCHUMAKER. 2003. Use of population viability analysis and reserve selection algorithms in regional conservation plans. *Ecological Applications* 13: 1773–1789.

CIARLEGLIO, M., J. W. BARNES, AND S. SARKAR. 2009. ConsNet: New software for the selection of conservation area networks with spatial and multi-criteria analyses. *Ecography* 32: 205–209.

COWLING, R. M., R. L. PRESSEY, A. T. LOMBARD, P. G. DESMET, AND A. G. ELLIS. 1999. From representation to persistence: Requirements for a sustainable system of conservation areas in the species-rich Mediterranean-climate desert of southern Africa. *Diversity & Distributions* 5: 51–71.

COWLING, R. M., P. W. RUNDEL, B. B. LAMONT, AND M. K. T. ARROYO. 1996. Plant diversity in Mediterranean-climate regions. *Trends in Ecology & Evolution* 11: 362–366.

DAVIS, F. W., P. A. STINE, D. M. STOMS, M. I. BORCHERT, AND A. HOLLANDER. 1995. Gap analysis of the actual vegetation of California: 1. The southwestern region. *Madroño* 42: 40–78.

DEUTSCHMAN, D. H., L. A. HIERL, J. FRANKLIN, AND H. M. REGAN. 2007. Vegetation community monitoring recommendations for the San Diego Multiple Species Conservation Program. Report to California Department of Fish and Game NCCP Local Assistance Grant #P0450009. San Diego State University, San Diego, California, USA.

DIFFENDORFER, J. E., R. E. CHAPMAN, J. M. DUGGAN, G. M. FLEMING, M. MITROVITCH, M. E. RAHN, AND R. D. ROSARIO. 2002. Coastal sage scrub response to disturbance. A literature review and annotated bibliography. San Diego State University, San Diego, California, USA.

Diffendorfer, J. R., G. Fleming, J. Duggan, R. Chapman, and D. Hogan. 2004. Creating an index of biological integrity for coastal sage scrub: A tool for habitat quality assessment and monitoring. Report to California Department of Fish & Game. San Diego State University, San Diego, California, USA.

Eken, G., L. Bennun, T. M. Brooks, W. Darwall, L. D. C. Fishpool, M. Foster, D. Knox, et al. 2004. Key biodiversity areas as site conservation targets. *BioScience* 54: 1110–1118.

Fernandes, L., J. Day, A. Lewis, S. Slegers, B. Kerrigan, D. Breen, D. Cameron, et al. 2005. Establishing representative no-take areas in the Great Barrier Reef: Large-scale implementation of theory on marine protected areas. *Conservation Biology* 19: 1733–1744.

Ferrier, S., and B. A. Wintle. 2009. Quantitative approaches to conservation prioritization: Matching the solution to the need. *In* A. Moilanen, K. A. Wilson, and H. P. Possingham [eds.], Spatial conservation prioritization: Quantitative methods and computational tools, 1–15. Oxford University Press, New York, New York, USA.

Field, S. A., P. J. O'Connor, A. J. Tyre, and H. P. Possingham. 2007. Making monitoring meaningful. *Austral Ecology* 32: 485–491.

Fuller, W. A. 1999. Environmental surveys over time. *Journal of Agricultural Biological & Environmental Statistics* 4: 331–345.

Gibbs, J. P., H. L. Snell, and C. E. Causton. 1999. Effective monitoring for adaptive wildlife management: Lessons from the Galapagos Islands. *Journal of Wildlife Management* 63: 1055–1065.

Greer, K. A. 2004. Habitat conservation planning in San Diego County, California: Lessons learned after five years of implementation. *Environmental Practice* 6: 230–239.

Groves, C. R. 2003. Drafting a conservation blueprint: A practitioner's guide to planning for biodiversity. Island Press, Washington, D.C., USA.

Hanna, S. 1997. Interior Secretary praises "Monumental Conservation Achievement" in San Diego County. Press release, U. S. Department of Interior, Washington, D.C., USA. Website http://www.sds.com/~mps/pub/wildcat/info/sand.html [accessed 21 January 2011].

Haufler, J. B. 1999. Strategies for conserving terrestrial biological diversity. *In* R. K. Baydack, H. Campa III, and J. B. Haufler [eds.], Practical approaches to the conservation of biological diversity, 17–30. Island Press, Washington, D.C., USA.

Hierl, L. A., J. Franklin, D. H. Deutschman, H. M. Regan, and B. S. Johnson. 2008. Assessing and prioritizing ecological communities for monitoring in a multi-species regional habitat preserve. *Environmental Management* 42: 165–179.

Holling, C. S. [ed.]. 1978. Adaptive environmental assessment and management. John Wiley, New York, New York, USA.

Hornaday, W. T. 1914. Wildlife conservation in theory and practice. Yale University Press, New Haven, Connecticut, USA.

Hoyer, M. V., J. Winn, and D. E. Canfield Jr. 2001. Citizen monitoring of aquatic bird populations using a Florida lake. *Lake and Reservoir Management* 17: 82–89.

Joseph, L. N., R. F. Maloney, S. M. O'Connor, P. Cromarty, P. Jansen, T. Stephens, and H. P. Possingham. 2008. Improving methods for allocating resources among threatened species: The case for a new national approach in New Zealand. *Pacific Conservation Biology* 14: 154–158.

Kelt, D. A., J. A. Wilson, and E. S. Konno. 2005. Differential response of two kangaroo rats (*Dipodomys*) to the 1997–1998 El Niño southern oscillation event. *Journal of Mammalogy* 86: 265–274.

Kiesecker, J. M., T. Comendant, T. Grandmason, E. Gray, C. Hall, R. Hilsenback, P. Kareiva, et al. 2007. Conservation easements in context: A quantitative analysis of their use by The Nature Conservancy. *Frontiers in Ecology and the Environment* 5: 125–130.

Knight, A. T., R. M. Cowling, and B. M. Campbell. 2006a. An operational model for implementing conservation action. *Conservation Biology* 20: 408–419.

Knight, A. T., A. Driver, R. M. Cowling, K. Maze, P. G. Desmet, A. T. Lombard, M. Rouget, et al. 2006b. Designing systematic conservation assessments that promote effective implementation: Best practice from South Africa. *Conservation Biology* 20: 739–750.

Lambeck, R. J. 1997. Focal species: A multi-species umbrella for nature conservation. *Conservation Biology* 11: 849–856.

Landres, P. B., J. Verner, and J. W. Thomas. 1988. Ecological use of vertebrate indicator species: A critique. *Conservation Biology* 2: 316–328.

Larsen, D. P., T. M. Kincaid, S. E. Jacobs, and N. S. Urquhart. 2001. Designs for evaluating local and regional scale trends. *Bioscience* 51: 1069–1078.

Legg, C. J., and L. Nagy. 2006. Why most conservation monitoring is, but need not be, a waste of time. *Journal of Environmental Management* 78: 194–199.

Lovett, J. M., D. A. Burns, C. T. Driscoll, J. C. Jenkins, M. J. Mitchell, L. Rustad, J. B. Shanley, et al. 2007. Who needs environmental monitoring? *Frontiers in Ecology and the Environment* 5: 253–260.

MacKenzie, D. I., J. D. Nichols, J. A. Royle, K. H. Pollock, L. A. Bailey, and J. E. Hines. 2006. Occupancy modeling and estimation. Academic Press, San Diego, California, USA.

Marcot, B. G., and M. V. Heyden. 2001. Key ecological functions of wildlife species. *In* D. Johnson and T. O'Neil [eds.], Wildlife–habitat relationships in Oregon and Washington, 168–186. Oregon State University Press, Corvallis, Oregon, USA.

Margules, C., and M. B. Usher. 1981. Criteria used in assessing wildlife conservation potential: A review. *Biological Conservation* 21: 79–109.

Margules, C. R., and R. L. Pressey. 2000. Systematic conservation planning. *Nature* 405: 243–253.

Margules, C. R., and S. Sarkar. 2007. Systematic conservation planning. Cambridge University Press, Cambridge, UK.

McDonald, T. L. 2003. Review of environmental monitoring methods: Survey designs. *Environmental Monitoring and Assessment* 85: 277–292.

McEachern, K., B. M. Pavlik, J. Rebman, and R. Sutter. 2006. San Diego Multiple Species Conservation Program (MSCP) rare plant monitoring review and revision. Scientific Investigations Report 2007-5016, U. S. Geological Survey, Western Ecological Research Center, Sacramento, California, USA.

Mittermeier, R. A., N. Myers, J. B. Thomsen, G. A. B. Da Fonseca, and S. Olivieri. 1998. Biodiversity hotspots and major tropical wilderness areas: Approaches to setting conservation priorities. *Conservation Biology* 12: 516–520.

Moilanen, A., A. M. A. Franco, R. Early, R. Fox, B. A. Wintle, and C. D. Thomas. 2005. Prioritising multiple-use landscapes for conservation: Methods for large multi-species planning problems. *Proceedings of the Royal Society of London, B, Biological Sciences* 272: 1885–1891.

Moilanen, A., and H. Kujala. 2006. The Zonation conservation planning framework and software v. 1.0: User Manual. Edita, Helsinki, Finland [online]. Website www.helsinki.fi/Bioscience/ConsPlan [accessed 1 August 2010].

Moilanen, A., K. A. Wilson, and H. P. Possingham. [eds.]. 2009. Spatial conservation prioritization: Quantitative methods and computational tools. Oxford University Press, New York, New York, USA.

Mulder, B. S., B. R. Noon, T. A. Spies, M. G. Raphael, C. J. Palmer, A. R. Olsen, G. H. Reeves, and H. H. Welsh. 1999. The strategy and design of the effectiveness monitoring program for the Northwest Forest Plan. U. S. Department of Agriculture General Technical Report PNW-GTR-437, Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Myers, N., R. A. Mittermeier, C. G. Mittermeier, G. A. B. Da Fonseca, and J. Kent. 2000. Biodiversity hotspots for conservation priorities. *Nature* 403: 853–858.

National Park Service. 2005. Guidance for designing an integrated monitoring program [online]. Website http://science.nature.nps.gov/im/monitor/ [accessed 28 February 2007].

National Research Council. 1995. Review of EPA's environmental monitoring and assessment program: Overall evaluation. National Academy Press, Washington, D.C., USA

National Research Council. 2001. Grand challenges in environmental sciences. National Academy Press, Washington, D.C., USA.

Nichols, J. D., and B. K. Williams. 2006. Monitoring for conservation. *Trends in Ecology & Evolution* 21: 668–673.

Noon, B. R. 2003. Conceptual issues in monitoring ecological resources. *In* D. E. Busch and J. C. Trexler [eds.], Monitoring ecosystems: Interdisciplinary approaches for evaluating ecoregional initiatives, 27–72. Island Press, Washington, D.C., USA.

Noon, B. R., T. A. Spies, and M. G. Raphael. 1999. Conceptual basis for designing an effectiveness monitoring program. U. S. Department of Agriculture General Technical Report PNW-GTR-437, Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

Noss, R. F., and A. Y. Cooperrider. 1994. Saving nature's legacy: Protecting and restoring biodiversity. Island Press, Washington, D.C., USA.

NSWNPWS. 2001. C-Plan, conservation software, user manual, version 3.06, New South Wales, National Parks and Wildlife Service (NSWNPWS), Armidale, New South Wales, Australia.

Ogden Environmental and Energy Services. 1995. Multiple Species Conservation Program (MSCP) Administrative Record. San Diego, California, USA.

Ogden Environmental and Energy Services. 1998. Final Multiple Species Conservation Program: MSCP Plan [online]. Website http://www.sandiego.gov/planning/mscp/ [accessed 1 August 2010].

Olsen, A. R., J. Sedransk, D. Edwards, C. A. Gotway, W. Liggett, S. Rathbun, K. H. Reckhow, and L. J. Young. 1999. Statistical issues for monitoring ecological and natural resources in the United States. *Environmental Monitoring and Assessment* 54: 1–45.

Parrish, J. D., D. P. Braun, and R. S. Unnasch. 2003. Are we conserving what we say we are? Measuring ecological integrity within protected areas. *Bioscience* 53: 851–860.

Pressey, R. L., H. P. Possingham, V. S. Logan, J. R. Day, and P. H. Williams. 1999. Effects of data characteristics on the results of reserve selection algorithms. *Journal of Biogeography* 26: 179–191.

Pressey, R. L., M. Watts, M. Ridges, and T. Barrett. 2005. C-Plan conservation planning software, user manual. New South Wales Department of Environment and Conservation, Armidale, New South Wales, Australia.

Raphael, M. G., M. J. Wisdom, M. M. Rowland, R. S. Holthausen, B. C. Wales, B. G. Marcot, and T. D. Rich. 2001. Status and trends of habitats of terrestrial vertebrates in relation to land management in the interior Columbia River Basin. *Forest Ecology and Management* 153: 63–87.

Regan, H. M., F. W. Davis, S. J. Andelman, A. Widyanata, and M. Freese. 2007. Comprehensive criteria for biodiversity evaluation in conservation planning. *Biodiversity and Conservation* 16: 2715–2728.

Regan, H. M., L. A. Hierl, J. Franklin, D. H. Deutschman, H. L. Schmalbach, C. S. Winchell, and B. S. Johnson. 2008. Species prioritization for monitoring and management in regional multiple species conservation plans. *Diversity & Distributions* 14: 462–471.

Scott, J. M., F. W. Davis, B. Csuti, R. Noss, B. Butterfield, C. Groves, H. Anderson, et al. 1993. Gap analysis: A geographical approach to protection of biological diversity. *Wildlife Monographs* 123: 1–41.

Scott, T. A., and J. E. Sullivan. 2000. The selection and design of multiple-species habitat preserves. *Environmental Management* 26: S37–S53.

Smallwood, K. 2000. A crosswalk from the Endangered Species Act to the HCP handbook and real HCPs. *Environmental Management* 26 (supplement 1): 23–35.

Stevens, D. L., and N. S. Urquhart. 2000. Response designs and support regions in sampling continuous domains. *Environmetrics* 11: 13–41.

Underwood, E. C., J. H. Viers, K. R. Klausmeyer, R. L. Cox, and M. R. Shaw. 2009. Threats and biodiversity in the Mediterranean biome. *Diversity & Distributions* 15: 188–197.

U. S. Fish and Wildlife Service. 1996. Habitat conservation planning and incidental take permit processing handbook [online]. U.S. Department of the Interior Fish and Wildlife Service and U. S. Department of Commerce National Oceanic and Atmospheric Administration National Marine Fisheries Service. Website http://www.fws.gov/endangered/hcp/hcpbook.html [accessed 3 April 2007]

U. S. Fish and Wildlife Service. 2007. Conservation Plans and Agreement Database [online]. Website http://ecos.fws.gov/conserv_plans/index.jsp [accessed 28 February 2007].

Vos, P., E. Meelis, and W. J. T. Keurs. 2000. A framework for the design of ecological monitoring programs as a tool for environmental and nature management. *Environmental Monitoring and Assessment* 61: 317–344.

Walters, C. J. 1986. Adaptive management of renewable resources. McMillan, New York, New York, USA.

Walters, C. J., and C. S. Holling. 1990. Large-scale management experiments and learning by doing. *Ecology* 71: 2060–2068.

Wilhere, G. F. 2002. Adaptive management in habitat conservation plans. *Conservation Biology* 16: 20–29.

Williams, B. K., R. C. Szaro, and C. D. Shapiro. 2007. Adaptive management: The U.S. Department of Interior technical guide [online]. Adaptive Management Working Group, U. S. Department of the Interior, Washington, D.C., USA. Website http://www.doi.gov/initiatives/AdaptiveManagement/documents.html [accessed 1 August 2010].

Williams, P. H. 2000. WORLDMAP IV for Windows: Software and help document 4.2, Computer program and documentation distributed by the author [online]. Website http://www.nhm.ac.uk/research-curation/research/projects/worldmap/worldmap/demo2.htm [accessed 1 August 2007].

Williams, P. H., and K. J. Gaston. 1994. Measuring more of biodiversity: Can higher-taxon richness predict wholesale species richness? *Biological Conservation* 67: 211–217.

Wilson, E. O., and E. O. Willis. 1975. Applied biogeography. *In* M. L. Cody and J. Diamond [eds.], Ecology and evolution of communities, 522–534. Belknap Press of Harvard University Press, Cambridge, Massachusetts, USA.

Wilson, K. A., M. F. McBride, M. Bode, and H. P. Possingham. 2006. Prioritizing global conservation efforts. *Nature* 440: 337–340.

Woodward, A. K., K. J. Jenkins, and E. G. Schreiner. 1999. The role of ecological theory in long-term ecological monitoring: Report on a workshop. *Natural Areas Journal* 19: 223–233.

Yoccoz, N. G., J. D. Nichols, and T. Boulinier. 2001. Monitoring of biological diversity in space and time. *Trends in Ecology & Evolution* 16: 446–453.