The need for maintaining connectivity arose as one of the highest conservation priorities in the Mayacamas Mountain Plan assembled in 2008. The District recognizes the importance of preserving landscape connectivity before widespread development occurs. Implementation costs can be greatly reduced by securing early protection for connectivity, rather than waiting until options are constrained by development and land values become astronomical. This follow-up project was designed to capitalize on a method that members of the Mayacamas planning team\(^1\) developed to identify and prioritize linkages within the Mayacamas and among neighboring habitat patches. Here we address this need by generating and testing a landscape connectivity model for this region. We hope that this information will help the District and its partners target cost-effective conservation of habitat linkages that will provide the greatest biological benefit and provide increased resilience to future alterations associated with climate change.

In many places, the landscape surrounding habitat patches, often referred to as the ‘matrix,’ can reduce opportunities for the movement of organisms among fragments of suitable habitat. It can be useful to think of the matrix as a gradient with respect to its permeability or traversability for animal and plant movement. The level of permeability can be influenced by physical features such as topography, substrate types, and the built environment, including roads, structures, and other types of development. Permeability will also depend on a species’ access to the matrix, the quality of the matrix with respect to survival and ease of movement, and the distance to neighboring patches. The variables of distance and quality interact in such a way that if distances are short and quality is good, individuals may be able to move more regularly among habitat patches. Our understanding of how species respond to the matrix is limited due to the complexity of the types of matrix that exist and the difficulties inherent in studying large-scale ecology.

It is important to note that the optimal strategy for conserving connectivity depends on the habitat characteristics and human land uses of the local landscape. Identifying priority linkages in a densely-settled urban environment can be a simple exercise of selecting the last remnant habitat corridors across the landscape. On the other hand, when large areas of natural habitat remain surrounded by landscapes composed of different types and intensities of development, then prioritizing protection among the many possible landscape linkages can be more challenging. In landscapes such as the Mayacamas Mountains, confined, linear corridors may not offer the best solution for maintaining connectivity among oak woodland habitats. Instead, protecting large swaths of habitat through a combination of protecting remnant undeveloped lands and adjacent diverse agroecosystems may better ensure a permeable landscape for wildlife movement.

\(^{1}\) Tom Robinson, Mark Reynolds, Sarah Reed, Shane Feirer, Adina Merenlender
Background on methodological approach

The primary approach to identifying linkages relies on cost-distance analysis to identify specific areas that may present preferred pathways for movement across the matrix (Knaapen et al. 1992; Adriaensen et al. 2003, Beier et al. 2008). The least-cost path method identifies a string of grid cells with the lowest travel cost between two points. ‘Cost’ in this case is a function of distance and habitat permeability along the movement path. Linkages comprised of the least-cost path plus additional neighboring grid cells with relatively low cost values are often mapped when wider wildlife corridors are desirable for planning purposes (Beier et al. 2006). The habitat permeability values used to inform the cost of traversing each grid cell are sometimes empirically derived from species occurrence data or resource selection functions, or may be based on rule sets generated by expert opinion (Hirtzel et al. 2001; Hirtzel et al. 2002; Chetkiewicz et al. 2006; Penrod et al. 2006; Beier et al. 2008). Linkages for more than one species are generated by a union or overlay of several single species models. It is recognized that the combination of habitat maps and linkages for dozens of species can compound error and result in a complex model with behavior that is hard to understand through sensitivity analysis (Beier et al. 2008).

Recent methods have been developed to help prioritize conservation of habitat connectivity across a landscape or region by comparing the relative importance of different patches and linkages in a landscape network. One approach is the ecological application of graph theory to reserve network analysis (Urban and Keitt 2001; Minor and Urban 2008). Graph theory represents a reserve network as a series of linkages (‘edges’) between habitat patches (‘nodes’). Several types of operations for ranking the importance of edges, such as the identification of a minimum spanning tree, can then be applied across the network. The work we present in this report employs and compares the results from both of these approaches to landscape connectivity analysis.

The key to our new approach to connectivity analysis for the Mayacamas region lies in our efforts to estimate permeability for the entire mixed oak woodland community in a continuous manner. How species respond to landscape structure can influence habitat suitability and connectivity (Tichendorf and Fahrig 2000). In particular, there is often a strong influence of the built environment (e.g. roads and buildings) for a large number of species of conservation concern, especially as they present barriers to movement (e.g. Forman 2000). The strong influence of the built environment on community composition and species abundance is often easier to detect than more subtle effects of habitat degradation. Fortunately, accurate mapped information is also available for the built environment, while fine-grain differences in natural habitat quality related to vegetation structure and composition is harder to obtain at a landscape scale. Therefore, we believe examining the relationships between various aspects of the built environment and the configuration of habitat that emerges is the most promising approach to determine the permeability of the matrix for communities of species. Estimates of permeability are be based on regression models of the relative abundance of species assemblages (e.g. abundance of native carnivores, temperate migrant birds, urban avoiding birds, and conservation targets = dependent variables) on development intensity and other landscape variables that may be important independent variables to examine. The observed statistical relationships revealed from regression analysis of the field and landscape data were then used to estimate travel cost
functions for the entire matrix. The travel cost functions are used to map permeability across the matrix and to derive linkages or the most likely dispersal pathways between core patches.

In sum, this approach provides estimates of permeability for the matrix between patches in a network based on the influence of development on community composition and abundance. While permeability is defined in this case as the ability of an individual or its propagules to pass through the matrix, we recognize that some species will move over generations, with individuals living and reproducing in the matrix, while others may move between patches in less than one day. Here we use information from empirically derived statistical models of the likelihood a site will support certain assemblages to a greater and lesser extent to estimate matrix permeability. These approximations are necessary since documenting individual movement through the matrix is currently too difficult to apply to whole communities.

Project goals

This project included five phases, four of which were completed and summarized here. The first phase was to refine the landscape approach to connectivity analysis discussed by the Mayacamas Mountains planning team that relied on estimating habitat permeability continuously across the matrix, in what we are terming a ‘biologically-informed structural habitat connectivity model.’ In addition, we tested the sensitivity of our choices of input data and uncertainty in variable relationships in the habitat connectivity model. The planning team felt that this new approach better addresses the habitat distribution and land ownership pattern in Sonoma County than the more traditional focal species-based approaches to modeling habitat connectivity. However, the two approaches should be compared (phase 2). Graph theory, with the advantages discussed above, was then employed to identify high-priority linkages among those delineated using the biologically-informed structural connectivity model developed in phase 1. The method of prioritization presented here is based on reserve network geometry, with the objective being the protection of the most efficient number of linkages required to connect all mapped habitat patches (phase 3). And finally, the average probability of habitat conversion to vineyard, suburban, and rural residential development (three primary drivers of land use change in Sonoma County) was estimated using published land-use change models for all linkages identified within Sonoma County, as well as the number of land parcels that make up each corridor and their average cost (Newburn et al 2006). This information is critical to identify which linkages are most in need of protection and will be cost effective to address. With this analysis completed, we are now in a position to work collaboratively with District staff and their partners to develop an action plan for conservation and restoration of habitat connectivity for the Mayacamas mountains within Sonoma County.

Methods

Identifying patches

First we delineated the study area which includes the Mayacamas mountains and areas required to connect to adjacent ranges (Figure 1). We then developed a map of remnant patches of natural land cover by selecting terrestrial vegetation cover categories from existing land cover data (CalVeg 2003) and removing roads (BTS 2001), intensive agricultural areas (primarily vineyards, but also other row crops and orchards), water bodies, and all land parcels less than 2 ha. We eliminated the boundaries between adjacent parcels of remnant vegetation and defined a habitat ‘patch’ to be any contiguous area of remnant natural vegetation that was greater than or
equal to 4 ha in area. We selected this minimum patch size to represent the minimum exurban lot size likely to be managed by a single owner in the study system.

Selecting which patches should be analyzed as part of a habitat connectivity network is very difficult in a landscape where the majority of the area is privately-owned and public lands do not represent a significant portion of the core wildlands. We began by selecting the largest patches in the landscape, which we defined to be any patch greater than 100 km² (10,000 ha), which is the order of magnitude of the minimum area reported for mountain lion and black bear home ranges (Crooks 2002). These patches are considered core wildlands because they are likely to be large enough to support the full complement of species found in the Mayacamas mountains. In addition to these larger patches, smaller patches found in the more fragmented parts of the study area were included if they were the largest patch within a fixed kernel distance ranging between 1 and 10 km from any given point in the landscape – a range of median dispersal distances expected for terrestrial vertebrates found in the area. Ultimately, it is likely that decision-makers will set the criteria for habitat patches that should be considered as part of a land conservation strategy. In some cases these patches are protected lands such as publically-owned lands and conservation easements while in other cases they include working landscapes that make up large continuous areas of habitat unlikely to be converted to more intensive land use in the near future.

Using the cost-distance methods described above, the total number and distribution of patches does greatly influence the final reserve network and linkage locations. Therefore comparisons were made using several different patch networks including; 1) all habitat patches, 2) large patches and those included as the “locally” largest patch using a kernel size of 1km, 2km, 5km, and 10km; and 3) protected areas only (wildland easements, public lands managed as natural parks). After comparing the resulting habitat networks and through discussions with SCAOSD staff, we restricted further comparisons with focal species-based models and sensitivity analyses to the patch layer using a 2km kernel for determining locally-important core areas and the protected area data set (Figure 2).

Permeability analysis

Based on previous large-scale field studies across land-use gradients, and preliminary analysis of the study region, we examined three landscape layers to characterize the permeability of the matrix. First, there is overwhelming evidence of the effects of roads on natural communities, and thus we included distance to nearest road as one of the core disturbance layers. We scaled the effect of distance to roads by traffic volume, based on published research documenting the relationship between bird species composition and abundance (Forman 2000). Second, we used parcel size as a small-scale, inverse measure of the intensity of human use, which may include the size and number of structures and frequency of human presence. Empirically, our prior research shows a substantial influence of parcel size on certain bird assemblages, such as urban adapters and temperate migrants and guilds such as ground feeders and shrub nesters (Merenlender et al. 2009). We believe that parcel maps are a useful surrogate to measure development density and patterns. This surrogate is needed because land cover has been shown to be a poor predictor of land use intensity for low-density residential development, which is the dominant development pattern in our study area and, by some accounts, the fastest growing land use type in the United States (Theobald 2005).
Third, we used a landscape metric, of which many types are regularly employed to quantify fragmentation or map the distribution and amount of habitat around a given patch or protected area (Calabrese and Fagan 2004). The simplest metric commonly used is the ‘nearest neighbor’ distance, which calculates the Euclidean, inter-patch distance between the protected area and the nearest comparison patch. A second group of measures are sometimes referred to as ‘buffer area’ metrics. A buffer area metric sums the total amount or proportion of suitable habitat within a given buffer radius of a protected area (Moilanen and Neiminen 2002). Lastly, a ‘proximity index’ sums the ratios of patch areas to inter-patch distances for all habitat patches that fall within a specified distance of the focal patch (Bender et al. 2003). There is increasing recognition that area informed metrics are useful to explain variation in wildlife abundances and movement capacity (Bender et al. 2003; Magel et al. 2009), and area-informed metrics generally perform well in analyses of landscape connectivity (Bender et al. 2003). We use a metric that we call ‘median patch size’ which is calculated from the median area of habitat patches within a fixed buffer radius; it is an area-informed measure of the size and contiguity of habitat patch size. In exploratory analyses, we found that calculating the median rather than the mean of patch sizes within the buffer radius was preferable because the value was not skewed by the presence of a few very large patches in the landscape. The median area of habitat patches within 2500 m of each site was an important predictor of the relative densities of all three commonly found meso-carnivores in the study area (Reed 2007). This work also revealed ‘median patch size’ to be a better predictor than buffered radius indices or proximity metrics (Reed 2007).

We assembled the data necessary to map all three metrics and modeled species responses to them based on earlier studies of birds and carnivores (Merenlender et al. 2009; Reed 2007; Forman 2000). These models were extrapolated across the entire matrix and ultimately combined to estimate a continuous surface of travel cost between patches within the network (Figure 3). Landscape response models for these species assemblages were ultimately used to construct permeability layers (or combined cost layers) to identify least cost pathways between existing protected areas in an ArcGIS toolbox called Funconn (Theobald 2006). A continuous map resulting from combined permeability layers is perhaps the most useful for examining how habitat connectivity varies continuously across the matrix (Figure 4).

Species models

To complement our modeling approach described above, we also created maps of permeability for ten focal species of conservation concern found in the study region (Table 1). These species were selected in a collaborative process between SCAOSD staff and the report authors.

To construct the permeability maps for each species, we recreated as closely as possible the process used by Paul Beier of Northern Arizona University and the non-profit South Coast Wildlands to map habitat corridors for focal species in southern California and Arizona (Appendix A). For each species, we constructed two initial suitability layers based on vegetation cover and distance from roads. The suitability for each vegetation type was ranked between 0 and 1 based on data extracted from the California Wildlife Habitat Requirements (C WHR) database and linked to CalVeg land cover maps. The suitability for each distance from a road was taken directly from the values reported in previous reports for Southern California and Arizona. These two initial suitability layers were combined into a single permeability map using weights that were drawn from these same reports. Where data for our particular focal species was
not available, we applied data from the most similar species included in these prior reports and the CorridorDesigner tool.

The southern California and Arizona reports additionally considered the influence of elevation and topography on permeability. Our literature review indicated that none of our twelve focal species was elevation restricted within the study area, and thus we did not include elevation as a factor in our analysis. After internal review, we also concluded that our knowledge of these focal species responses to topography was very uncertain, and thus we chose to exclude the influence of topography as well. As the southern California and Arizona reports assigned very little relative weight to elevation and topography in creating final permeability maps, we do not believe that this omission significantly biases our conclusions.

The corridors selected for these twelve focal species models were then compared to the corridors selected by our previously described biologically-informed structural connectivity model. For this comparison, we used the same base core patch network as the other model comparisons. For each of the twelve focal species permeability layers, we used FunnConn to delineate discrete corridors between each patch. We masked out any corridor with lower than 0.3 average permeability as unsuitable for use, and then created a corridor-count map, with values ranging from 0 to 12, showing how many focal species corridors overlapped each grid cell (i.e., if every focal species in the analysis had a corridor overlapping a particular cell, that cell was assigned a score of 12).

We then created a similar discrete corridor network based on our biologically-informed structural connectivity model and calculated a “score” for our model based on the percentage of cells with each corridor-count level that fell within our corridors. If our model is successful in capturing the needs of the twelve focal species, we would expect the scores to be high for corridor-count cells with values of 12 or higher, 11 or higher, 10 or higher, and so on. We would expect the score to decrease as we approach counting all corridor-count cells scoring 0 or higher (i.e., the entire landscape).

Sensitivity analysis

One of the primary reasons we chose to use a biologically-informed, structural approach to model the relative permeability of the matrix for species movement is that it allowed us to effectively test the sensitivity of the model outcomes to our choices of input data and the uncertainty in the input data relationships. Our methods contrast with approaches that combine a large number of individual species models. These composite models have the potential to compound uncertainty through many nested equations, such that testing the sensitivity of input parameters becomes difficult or impossible. This is particularly problematic given further uncertainty regarding the variable relationships representing species behaviors or habitat preferences, which may not be known or correctly estimated for a particular landscape.

Although it is important to pursue connectivity analysis and conservation despite limited knowledge regarding species biology, it is also important to understand how the guesses that we make influence the model outcomes. Sensitivity analysis will help us to determine how resilient our priority linkages are likely to be to changes in the input parameters and identify places where additional research or field monitoring could help to reduce uncertainty in linkage selection.

We compared the permeability surfaces resulting from our model analyses to assess the underlying factors that determine the outputs of our connectivity model. The first is the extent to
which each cost layer (road effect, parcel size, and median patch size) contribute to the overall permeability of the landscape. Therefore we evaluated the relative contribution of each cost input layer (road effect, parcel size, and median patch size) to the combined connectivity model. We removed each cost input layer sequentially, and we compared the resulting models to the full, combined model incorporating all three layers. We also compared models using single cost input layers to evaluate the redundancy between the removed layer and the remaining layers.

Another important issue is to examine how uncertainty in the estimates used to quantify relationships between species responses to the built environment impacts the results. We based the ‘cost’ relationships for road effect, parcel size, and median patch size on prior landscape-level empirical studies of the responses of bird and mammalian carnivore communities to these three factors (Forman 2000; Reed 2007; Merenlender et al. 2009). We evaluated the models’ sensitivity to uncertainty in the relationships with the cost input values. Specifically, we compared models using the minimum and maximum values for the three principal cost input layers (road effect, parcel size, and median patch size) with the mean values of our combined model. We used variable values reported for the traffic volume thresholds associated with different road classes to estimate minimum and maximum values of the distance and magnitude of road effect. For parcel size and median patch size, we used minimum and maximum values from the 95% confidence intervals around the linear regression relationships reported in the prior studies.

For each pair of models, we compared linkage locations at decreasing levels of permeability for the same network of core habitat patches. We used the 2 km ‘local cores’ network for the sensitivity analysis and cost input comparisons, and both the 2 km local cores and protected area networks for the focal species comparisons. We converted the continuous connectivity surface resulting from the FunConn model runs into 10 cumulative quantiles of linkage permeability. For example, the 0.1 quantile corresponded to the locations of the 10% most permeable linkages, and the 0.5 quantile corresponded to the locations of the 50% most permeable linkages. For each quantile, we calculated the total proportion of linkage locations that overlapped between the two models, and we plotted the relationship between linkage permeability and proportion of model overlap. This relationship illustrates the degree of redundancy between pairs of models; a high degree of redundancy is indicated by models that rapidly approach a model overlap of 1.0 and low quantiles of permeability.

Priority linkages based on network geometry

In addition to the continuous landscape permeability surfaces described above, FunConn represents the habitat connectivity models resulting from our analyses as a network (or ‘graph’) of patches and linkages. Specifically, the network of patches is represented as a set of ‘nodes,’ symbolized by points that incorporate the attribute data associated with the patches. The nodes are connected to one another by a set of ‘edges,’ symbolized by polylines that incorporate attribute data associated with the linkages. Graph analyses of the network relationships among the nodes and edges allows for estimation of the relative contributions of different patches and linkages to the overall connectivity of the landscape.

A path through the graph network is a sequence of nodes connected by edges, and a tree is a path that includes every node in the graph (only once). The minimum spanning tree is a path including every node with the shortest possible length. Researchers have suggested that the minimum spanning tree can serve as a useful guide for establishing connectivity conservation
priorities because it identifies the ‘backbone’ of the habitat network – or the set of key linkages that are most connected to all patches in the network (Urban and Keitt 2001). We used FunConn v. 1.9 to calculate the minimum spanning tree of the 2 km ‘local cores’ patch network. We ranked the resulting patches and linkages as a function of their area and permeability, respectively. Linkages that are central to the graph, offer unique connections between large components of the network, and/or currently have relatively low levels of permeability could all be priorities for protection.

Additional graph analyses could provide a more refined ranking of the relative importance of habitat patches and linkages to the overall connectivity of the landscape. For example, in an edge-removal analysis, edges are systematically removed from the network – representing the sequential loss of linkages – and indices of overall graph connectivity are recorded. The graph indices can then be plotted against different attributes of edges (e.g., linkage permeability or development threat) to identify critical thresholds of declines in network connectivity. We would like to apply these additional analyses to the Mayacamas mountains models, and we are actively collaborating with the developers of FunConn (Theobald et al. 2006) to update and refine tools that can make edge- and node-removal calculations. We will share the results of these further analyses when they are available.

**Average threat and cost analysis**

These methods were focused on determining the average probability that parcels within an identified corridor may be converted to vineyard, suburban or rural residential development. To do this we estimated these probabilities from existing land use change models described here and then calculated the average value for each corridor identified using the patch layer based on a 2km kernel to demonstrate the utility of using these values to help prioritize corridor conservation. These spatially-explicit parcel-scale models have been previously transferred to the SCAOSD along with meta-data detailing their development as part of ReVision 2005 and are fully described in Newburn et al. (2006). Briefly, parcel-level land-use change models were constructed for the period 1994-2002 using the tax-assessment data to determine residential development and aerial photos to determine vineyard development. The data were initially compiled to determine the set of developable parcels in 1994 and then used to assess whether the developable parcels were converted to either vineyard or one of several housing densities from 1994–2002. A parcel was considered developable if there was no vineyard use in 1994 and the existing housing density in 1994 was <1 structure per 40 acres. Hence, the set of developable parcels excluded those parcels protected in parks and reserves and already converted to vineyard or residential development before 1994. Land-use conversion was defined as transitions from developable parcels into vineyard development or one of the residential density classes during the period 1994–2002.

A multinomial logit model was developed to explain land-use transitions as a function of parcel-site characteristics, including average slope, growing degree days (microclimate), 100–year floodplain, accessibility to major employment centers, designated sewer and water services, and minimum lot-size zoning. In a similar fashion, we used a hedonic price model to determine the market value for developable land as a function of the site-specific characteristics. Specifically, recent property transactions of developable parcels were used to estimate the actual sales price as a function of the parcel land characteristics. The Sonoma County Tax Assessor’s database provides the necessary information on individual parcels for the land value, current land
use, and other property characteristics. Using the GIS, we used a similar set of explanatory variables for each parcel, including characteristics for land quality (slope, elevation, microclimate, 100-year floodplain), accessibility (travel times to urban centers, sewer and water service), neighboring land-use externalities (percent protected open space and urban), and zoning (land use designations, minimum lot size).

**Results**

The methods necessary to accomplish corridor analysis were fully developed and the results for four of the tasks are complete. However, it is important to mention that this does not mean that this report contains a final selection of priority corridors for conservation planning purposes. Rather, with this analysis completed, we are now in a position to work collaboratively with District staff and their partners to develop an action plan for conservation and restoration of habitat connectivity for the Mayacamas mountains within Sonoma County. However, to demonstrate how the resulting analysis can be used to identify linkages and how these linkages compare with those resulting from other methods we used the corridors that result from Funconn analysis (e.g. Figure 6). However, we want to stress that Figure 4 actually represents our understanding of connectivity across the study area. We can use this information to designate corridors that are likely to be functional for wildlife and plant communities. Also, considering the relative development threat that each corridor may be facing as well as the implementation costs for protection should help set District priorities. We look forward to collaborating with the District on a linkage plan for Sonoma County that will improve climate change resilience and the overall value of the existing protected lands network.

**Patches**

There was good consensus among our team that using a patch network that includes a large number of patches was a good approach for protecting corridors on the ground (all green and orange patches mapped in figure 2). This is in part because the resulting corridors are smaller and represent useful conservation units for on the ground project planning and in part because other efforts are underway to protect core habitat patches which represent important stepping-stones for wildlife movement throughout the region. However, further work is underway to explore methods of least-cost-path analysis that will not be as sensitive to the patch network being used.

**Species model comparisons**

Sample permeability layers for three of our twelve focal species, the black bear, ringtail, and spotted owl, can be found in Figure 5. Figure 6 shows our base patch network, corridors as delineated by our community-wide model, and the focal species corridor-counts, which range from 0 to 12.

The results of the community-wide and focal species model comparison are shown in Figure 7. The overlap between the community-wide corridors and the specific areas of the landscape used by all 12 focal species is approximately 75%, but this overlap grows to above 90% for corridors used by 11 or more species and 10 or more species and then decreases slowly. Although the low overlap score for corridor-count 12 areas is initially surprising, visual inspection shows that only five very small areas (too small to display visually on a landscape-
wide map) were assigned a corridor-count of 12. The dip in overlap observed with 12 focal species is due to a single outlier species (e.g. pond turtle) whose habitat suitability was dramatically different from the other 11 species. When averaged among the 11-species models, it did not have the effect that it does in the sole 12-species model comparison. All of these corridor-count 12 areas were within approximately 150 m of a corridor delineated by the community-wide model. Also, it is important to note that the corridors outlined in Figure 6 are resulting from Funconn output that forces each two patches to be connected by a corridor, regardless of their permeability – hence, the need for further examination of the suite of factors that will help provide a priority corridor map for Sonoma County specifically. We can consider these results to confirm our hypothesis that a union of individual focal species corridors can be closely approximated by a simple, community-wide corridor model that considers only the effects of the built environment.

**Sensitivity analysis of the new model techniques employed**

What we observe by comparing our full model (roads, parcels, median patch size) with a reduced model is how the overlap in linkages compares with and without each of these variables (Figure 8). Among the three cost input layers, the model excluding median patch size had the least redundancy with the full, combined model. Our results show that the overlap between a reduced model without median patch size (MPS) and the full model is closer to 80% when examining corridors delineated using the 20% most permeable linkages as compared to almost 90% when the other layers are removed. The estimate of MPS operates at a larger scale than the other two parameters and a scale that we believe large ranging species are operating at. Therefore, we recommend retaining this large-scale habitat metric in estimating the suitability of the matrix for species. There was also limited redundancy between models using single cost inputs of road effect and parcel size. Together, these results indicate that all three cost layers contributed unique information towards determining landscape permeability in the study system.

Models for all three cost input layers were most sensitive to increases above the mean values of the cost input variable (Figure 9). This is likely due to the fact that the distributions of all three input layers were concentrated at lower cost values. This means that a higher cost for one variable had the potential to overwhelm the other two in the calculation of the mean cost value and shift the locations of the resulting linkages. On the other hand, a lower cost for one variable would have been balanced by the other two variables, resulting in less of a shift in linkage locations. Uncertainty in the road effect cost relationship had the least influence on the resulting linkage locations. The relatively high redundancy for the road effect models indicated that our choice of traffic volume thresholds to assign the magnitude and distance of effects for different road classes had little effect on the model outcomes. On the other hand, models for the maximum values of median patch size and parcel size had the least redundancy with the mean, combined model. This suggests that the model results are sensitive to uncertainty in the cost variable relationships given by the underlying empirical data. Additional field surveys or further model analyses could help to reduce the uncertainty in the variable relationships and model outcomes.
Example of minimum spanning tree analysis to generate a priority network geometry

Out of 1,700 total edges, or linkages, in the habitat network for the 2 km ‘local cores’ patch network, 293 were selected as belonging to the graph’s minimum spanning tree (Figure 10). These linkages provide the shortest path between all patches in the network and represent the ‘backbone’ of the habitat network. We also ranked the linkages according to their current level of permeability. Those linkages that connect large components of the graph and which have relatively low levels of current permeability (or relatively high levels of development threat) could be identified as priorities for conservation.

In Sonoma County, for example, the network of habitat patches along Sonoma Mountain is isolated from the larger network in the Mayacamas Mountains to the west by vineyard and residential development in the valley along Highway 12. A single habitat linkage is identified by the minimum spanning tree connecting these two network components through an area of lower-density development towards the northern end of Annadel State Park (circled in Figure 10). Although the full model identifies many other potential linkages across the valley, the linkage selected by the minimum spanning tree currently has a greater level of permeability and provides greater benefits for the overall connectivity of the network.

Example of threat and cost values for mapped linkages

First we present the total number of individual parcels that intersect with each corridor identified by the Funconn program using all patches (nodes) and linkages (edges) in the 2km ‘local cores’ patch network (Figure 11). It is worth noting that the number of parcels ranges greatly from 1 to over 4,000. This type of information may be useful as working with fewer landowners is more feasible when developing coordinated conservation planning and land management. Of course the protection of corridor function may not require that all parcels falling within the corridor be addressed through private land conservation tools. These numbers can be generated for any suite of potential corridors of interest to the District using existing GIS capabilities.

Maps of how susceptible the parcels within each funconn corridor, based on the average value across all parcels that fall within the corridors, to suburban, rural residential, and vineyard development are represented in Figures 12-14. We see that the corridors important to maintaining connectivity along the spine of the Mayacamas are threatened by rural residential development as compared to suburban higher density uses as they generally fall outside of incorporated areas. The demand for rural residential housing on Sonoma County is quite high because of the rural character and natural beauty of these areas while remaining relatively accessible. We also see that all valley bottom corridors have the highest probability of being converted into intensive agriculture, where as this threat declines in hillside locations it is not eliminated entirely. Depending on site characteristics and transportation opportunities, hillside vineyard planting can be a possibility throughout the Mayacamas. The absolute average land values are likely to differ given recent changes in the real-estate market, however, these values are still provide a useful relative estimate of cost for comparative purposes (Figure 15). We see here that land values to vary across the county and need to be considered in addition to the number of land parcels that need to be considered within a corridor of interest.
Summary

We were able to refine our analysis approach to habitat connectivity for the Mayacamas Mountains initially proposed by the team that worked on the first Mayacamas Mountain plan. Our work resulted in a ‘biologically-informed structural habitat connectivity model’ that resulted in linkages that are very similar to those derived from the union of 12 species models, but allows for sensitivity analysis and is more uniformly applicable to the entire study area and beyond. Graph theory was employed to examine the minimum number of corridors needed to connect identified core patches of habitat across the study landscape. Also, the probability of habitat conversion to suburban, rural residential and vineyard development was assessed for a larger suite of corridors to provide important threat and cost estimates that need to be considered when prioritizing land conservation actions. The analyses completed through this project is critical to identify which linkages are most in need of protection in the Mayacamas mountains region. This information will be incorporated into existing District GIS databases so that it can be more closely examined by District and be combined with other landscape characteristics to improve land conservation decision-making. With the four phases of analysis completed, we are now in a position to work collaboratively with District staff and their partners to develop an action plan for conservation and restoration of habitat connectivity for the Mayacamas mountains within Sonoma County.
References


Reed, S. E. and A. M. Merenlender 2008 Quiet, Non-Consumptive Recreation Reduces Protected Area Effectiveness. Conservation Letters 1(3):146-154


Table 1. List of 12 target species by common name selected by SCAOSD staff and authors.

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<thead>
<tr>
<th>Species</th>
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<tr>
<td>Black bear</td>
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<td>Mountain lion</td>
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<td>Grey fox</td>
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<td>Ringtail</td>
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<td>California ground squirrel</td>
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<td>Pallid bat</td>
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<td>Townsend’s big-eared bat</td>
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<td>Spotted owl</td>
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<td>Purple martin</td>
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<td>Orange-crowned warbler</td>
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<td>Acorn woodpecker</td>
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<td>Western pond turtle</td>
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Figure 1. Map of the study area analyzed for this project.
Figure 2. This maps shows all of the natural habitat patches (all colors together) identified and designated according to which ones were selected using a local neighbor analysis with different diameter kernels (1km yellow, 2km orange, & 5km green).
Figure 3. This is a combined ‘cost’ layer that includes the influence of roads, parcel size, and median patch size for the entire study site used in the connectivity analysis to identify linkages between the patches.
Figure 4. Continuous map of permeability across the matrix surrounding patches (dark blue, 2km kernel cores) with the most resistant paths (low connectivity) mapped in yellow and increasing connectivity in green through to dark blue.
Figure 5. Sample permeability maps for the Mayacamas study area for (a) black bear, (b) ringtail, and (c) spotted owl. Red areas indicate low permeability, and green areas indicate high permeability.
Figure 6. Base patch layer, community-wide model corridors, and corridor-count layer
Figure 7. Comparison of community-wide corridor model to twelve focal species models. The x-axis shows a cumulative count of the number of focal species corridor overlaps (e.g., the 11 value indicates all areas in the landscape that were selected as a corridor by 11 or more individual focal species models). The y-axis indicates the percentage of the landscape selected at each corridor-count level that was contained within a corridor defined by the community-wide built-environment model (e.g., a score of 92% for corridor-count 11 indicates that 92% of all areas in the landscape used as a corridor by 11 or more focal species was located within a corridor identified by the community-wide built-environment model).
Figure 8. Relative importance of cost input layers used to derive connectivity model. Comparisons are shown for models excluding: (a) road effect (distance to roads, scaled by traffic volume), (b) parcel size, and (c) median patch size.
Figure 9. Sensitivity analysis comparing minimum and maximum values for model relationships with cost input layers: (a) road effect (distance to roads, scaled by traffic volume), (b) parcel size, and (c) median patch size.
Figure 10. Minimum spanning tree model for all patches (nodes) and linkages (edges) in the 2km ‘local cores’ patch network. Patches are shown scaled by their area (larger patches are represented by larger points) and linkages are shown scaled by their permeability (more permeable linkages are represented by thicker lines). The minimum spanning tree is the path that connects all of the patches in the network by the shortest possible cost-distance. Linkages that are central to the graph, offer unique connections between large components of the network, and/or have relatively low levels of permeability could be priorities for protection – for example, the linkage connecting habitat patches on Sonoma Mountain to the larger Mayacamas mountain chain is circled below. The circle is in an area of lower-density development towards the northern end of Annadel State Park.
Figure 11. Total number of parcels that intersect with each corridor identified by the Funconn program using all patches (nodes) and linkages (edges) in the 2km ‘local cores’ patch network.
Figure 12. The relative threat level represented mapped according to the probability of conversion to suburban development for each corridor in Sonoma County designated using the final Funconn analysis with a core patch layer (2km kernel).
Figure 13. The relative threat level represented mapped according to the probability of conversion to rural residential development for each corridor in Sonoma County designated using the final Funconn analysis with a core patch layer (2km kernel).
Figure 14. The relative threat level represented mapped according to the probability of conversion to vineyard for each corridor in Sonoma County designated using the final Funconn analysis with a core patch layer (2km kernel).
Figure 15. The relative average parcel cost for each corridor in Sonoma County designated using the final Funconn analysis with a core patch layer (2km kernel).